FINAL REPORT

Evaluating the Effectiveness of Restricted Fishing Areas for Improving the Bottomfish Fishery in the Main Hawaiian Islands

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EXECUTIVE SUMMARY

The goal of this project was to evaluate the effectiveness of DAR's bottomfish management plan by determining how bottomfish restricted fishing areas (BRFAs) effect bottomfish populations inside and outside of their boundaries and to refine species-habitat relationships. We used a non-lethal video assessment tool, the BotCam stereo-video baited camera system, to collect data for determining

1) *if bottomfish populations on habitats inside the BRFAs improve (i.e., increase in numbers and average size of fish) over time, and if so,*
2) *if bottomfish populations on adjacent habitats outside the BFRAs also improve (as a result of spillover), or*
3) *if bottomfish populations on adjacent habitats outside the BRFAs diminish (decrease in numbers and average size of fish) as a result of displaced fishing effort and*
4) *to quantify species specific habitat associations to assist in refining BRFA placement and design.*

The goal was to monitor 6 of the 12 BRFAs as representative of the entire suite (BRFAs B, D, E, F, H, L). We adopted a split-panel design that balanced regular sampling at one site (BRFA F) to control for interannual shifts in conditions with periodic monitoring at the other sites. For each of the 6 BRFAs, areas inside and equal neighboring fished areas were sampled to count and measure deep 7 bottomfish. Samples were randomly chosen and stratified by habitat type (using multibeam bathymetry). Data was collected beginning in 2007 and continued for a total of 6 years.

1474 successful BotCam deployments were conducted in areas both inside and outside of six of the twelve BRFAs and in the Kahoolawe Island Reserve. In addition we performed 35 deployments identifying a nursery ground for opakapaka off of Waikiki and performed another 629 deployments of the system as part of the NOAA led gear intercalibration experiment. For the BRFA project 6801 fish (4526 individuals of deep7 species) were measured.

Species specific habitat associations were evident and change ontogenetically in some species. Opakapaka occurred at depths shallower than the depths at which ehu, onaga and gindai were observed, and this species showed an ontogenetic shift to deeper water with increasing size. Opakapaka, kalekale, and onaga exhibited size-related shifts with habitat type. The results also suggest that opakapaka has widespread juvenile grounds around the State. Adult habitat association data is being used as the foundation for models that project habitat association information across BRFAs and the entire domain of the Hawaiian Islands. Finally our results contributed to a refinement of the EFH designations of deep7 species.

The results repeatedly suggest that BRFAs have positive effects on deep7 populations. These results are clear after taking habitat associations into consideration statistically. An examination of bottomfish populations in BRFAs B and L in the first project year (having been closed for 9 years) shows that protection resulted in greater size of onaga and opakapaka. At Niihau the difference in opakapaka size inside to outside of the reserve is approximately equal to 10 years of growth. Kahoolawe Island Reserve, which is not a BRFA but has been protected from fishing through Navy bombardment or State management since the early 1990's, has a greater diversity of deep7 bottomfish and most species are larger with greater proportions of sexually mature fish than other nearby regions in which active fishing occurs. Most importantly, the analysis of bottomfish sizes over time in BRFAs B, E, F and H (4 years of data) clearly show increases within BRFAs and no change or declines outside the BRFAs for the most commercially important species. Trends for abundance are more difficult to discern given the hyperdispersed nature of the count data but onaga and opakapaka abundance inside BRFA F and E increased while there were no changes outside these BRFAs over time. Analysis of all 6 years of data which was available for BRFA E and F, provided evidence of spillover for at least some of the deep7 and benefits to fishery yield. Relative abundance, fish size, and species richness declined with distance from BRFAs, signifying that the recovering Deep 7 community inside these reserves had begun to spillover the boundary of the BRFAs and that BRFAs were a source of more and larger fish to fished areas. In addition, changes in fish size over time suggested both density independent and dependent processes contributed to spillover. Displaced fishing effort also likely caused initial declines in onaga size and catch data that increased in later years.

Our results mirror those of other studies around the world and in Hawaii suggesting that despite a lack of rigorous enforcement, the BRFAs protect bottomfish populations from fishing mortality. The predominant finding of larger more mature fishes inside the BRFAs and increases in abundance and size inside versus outside of these zones strongly suggests that the BRFAs can benefit Hawaiʻi's deepwater fish populations by allowing populations of large spawning fish to develop. Further, data from the last portion of the monitoring period is suggesting that catch is increasing in fished zones that border the BRFAs.

There are several other important studies conducted recently by other scientists that are relevant to bottomfish management. First, a larval dispersal model was developed for deep 7 species in the Main Hawaiian Islands and it can be used to inform management by evaluating the connectivity of BRFAs to fished regions. Second, NOAA-PIFSC is leading an effort, and we are collaborating, to develop a fishery independent stock assessment survey. This project is evaluating different sampling methodologies (i.e. fishing, BotCam, acoustics) to intercalibrate each technique. Third, a separate way to evaluate BRFA efficacy and design is through tracking of deep 7 fish movements. Dr. Kevin Weng has done this in BRFA B (Niihau) and has begun work around BRFA F (Penguin Bank).

BACKGROUND AND PROJECT JUSTIFICATION

The most important members of the Hawaiian bottomfish fishery are four species of eteline snappers, the onaga, *Etelis coruscans*, the ehu, *Etelis carbunculus*, the ʻōpakapaka, *Pristipomoides filamentosus*, the uku, *Aprion virescens*, and one endemic species of grouper, the hāpuʻupuʻu, *Epinephelus quernus.* Four of these, the onaga, ehu, ʻōpakapaka, and hāpuʻupuʻu are considered to be deeper complex species whose essential fish habitat (EFH) is presently defined as the 0-400 m depth range around each island and bank in the Hawaiian archipelago. From 1986-2004, DAR and WPFMC assessed the stocks of these species in the main Hawaiian Islands (MHI), as well as the Northwestern Hawaiian Islands (NWHI) by, among other ways, calculating their estimated Spawning Potential Ratios (SPRs) from annual commercial catch data. An SPR of 20% was established as the critical threshold for designating a stock as recruitment overfished. In the NWHI, SPRs for all bottomfish species have consistently been above this critical level however in the MHI, the onaga and the ehu have had SPRs below 20% for well over a decade. Since the data from these two regions were reported separately until 1999, MHI onaga and ehu were considered to be separate stocks from NWHI onaga and ehu. Therefore, when the Magnuson Fisheries Act was revised in 1996, they were federally listed as recruitment overfished.

The amended Magnuson Act, now referred to as the Magnuson-Stevens Act, imposed a mandate on WPRFMC to restore the stocks of species listed as overfished to healthy levels (i.e., SPR > 20%) within a ten-year time period. Since most of the MHI bottomfishing grounds are within state rather than federal waters, WPRFMC turned to DAR to address this problem. In 1997, DAR responded by creating a new bottomfish management plan and funding research on bottomfish to provide additional information on these species. A key element in the plan (Hawaiʻi Administrative Rules, Chapter 13-94, Bottomfish Management) was the creation of nineteen bottomfish restricted fishing areas (BRFAs) where bottomfishing was prohibited. The BRFAs were spread throughout the MHI and were designed to protect 20% of the designated 0-400 m essential fish habitat (EFH) for onaga and ehu. The closure of these areas took effect on June 1, 1998 and their effectiveness, in terms of the quantity and type of habitat protected and their effect on commercial landings, was subsequently reviewed in 2005. It was concluded that the system did not protect an adequate amount of preferred habitat. For example, onaga and ehu appear to aggregate over hard, high relief, structurally complex substrates (WPRFMC 1998). Only 5% of this type of habitat was believed to occur within the boundaries of the BRFAs. DAR's commercial catch data analysis furthermore indicated that modifications to the BRFA system were warranted.

A new BRFA system was therefore created, this time with a much greater understanding of the distribution of MHI bottomfish habitat as a result of the multibeam sonar mapping which has taken place throughout much of the Main Hawaiian Islands during the last 8 years. The number was reduced from 19 to 12 and their boundaries were designed to protect selected habitats but also to facilitate spillover and thereby sustain adjacent habitats open to fishing. The new BRFAs were established for three purposes:

1) reduce fishing mortality of MHI bottomfish stocks by 15%
2) rebuild bottomfish populations on habitats inside the BRFAs
3) improve bottomfish populations in adjacent open fishing areas via larval export and/or adult spillover from the BRFAs

The new system took effect on July 1, 2007 and both fisheries biologists at DAR and NOAA's Pacific Islands Fisheries Science Center (PIFSC) stressed the importance of obtaining baseline data as well as population monitoring, which were lacking for the areas in the original system. Furthermore, some PIFSC biologists were concerned about the possibility of displaced fishing effort causing greater depletion of the areas surrounding the BRFAs.

PROJECT OBJECTIVES

The goal of this project was to evaluate the effectiveness of DAR's bottomfish management plan by determining how BRFAs effect bottomfish populations inside and outside of their boundaries and to refine species-habitat relationships. We used a non-lethal video assessment tool, the BotCam stereo-video baited camera system, to collect data for determining

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4) *to quantify species specific habitat associations to assist in refining BRFA placement and design.*

DESIGN AND METHODS

To evaluate BRFA efficacy a sampling strategy was developed in consultation with DAR, NMFS, and U. of Miami. In short, the goal was to monitor 6 of the 12 BRFAs as representative of the entire suite. Each of the 6 areas and neighboring fished areas were sampled using a non-lethal baited stereo-video camera system to count and measure bottomfish. Samples were randomly chosen and stratified by habitat type (using multibeam bathymetry) so that most samples occurred in steep rocky terrain where previous studies have found the highest concentrations of bottomfish to live (Ralston and Polovina 1982; Kelley et al. 2006). Data was collected beginning in 2007 and continued for a total of 6 years.

BotCam Description

The Bottom Camera bait station (BotCam) is a stereo video system that is deployed on the seafloor to monitor commercially important bottomfish (Fig. 1). It was originally developed by the PIFSC Coral Reef Ecosystem Division (Merritt 2005; Merritt et al. 2011) and has since been modified and upgraded by our team. It has an operating depth of 330m and provides a non-extractive, and hence very attractive, method to monitor fish populations within restricted fishing areas. The system consists of two ultra-low light video cameras and a digital video recorder which are programmed to record video once the system is in the water. Numbers of fishes and their identity are determined in addition to habitat variables as discussed below. The stereo camera arrangement facilitates size and distance determinations of fish and habitat features in the field of view. A temperature depth recorder (CTD) and a current meter (only available for some deployments) help with characterization of the physical environment where observations are made.

In the second year of this project we completely redesigned the video recorder component of the BotCam, building 5 new reliable units. The original video recorder unit built by Deep Development Inc. was regularly faulty, did not include any indicator lights to monitor function and resulted in many failed deployments in the first year (though we still achieved our goals). The new system was customized to provide an infinitely easier user interface and gave us the capability to repair the units in house. This greatly improved our field efficiency.

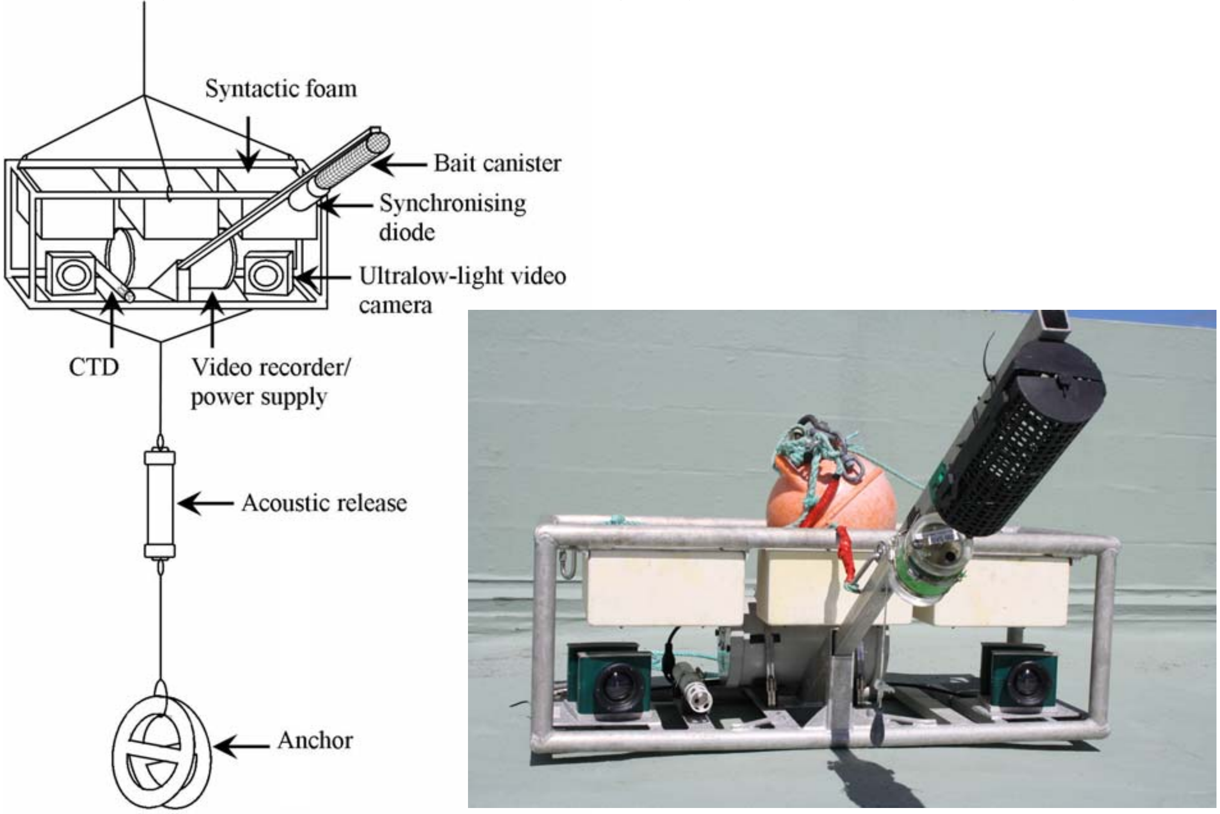


***Fig. 1**. Schematic of the BotCam (Bottom Camera bait station) system as it would appear on the seafloor (left) and photograph showing the actual system (right).*

BotCam Standardized Site Selection and Deployment Protocols

After consultation with the Pacific Islands Fisheries Science Center (M. Parke, G. DiNardo, J. Brodziak) and the University of Miami (J. Ault), a standardized BotCam sampling protocol was developed for this project. The design of the protocol was based on the assumption that sampling effort should be correlated to habitat suitability because it, in turn, is correlated to bottomfish abundance. As mentioned above, previous studies have found that adult bottomfish often associate with higher relief and/or rocky substrate (Polovina et al. 1985; WPRFMC 1998). Based on a study on Penguin Banks, this type of substrate is patchy, and may make up only a fraction of the total substrate within their depth range (Haight 1989). Onaga and ehu may be attracted to these areas because they provide suitable habitat for their prey. Both species are primarily piscivorous (Haight 1990; Haight et al. 1993) and recent studies found a positive correlation between their presence and the presence of several species of small fish they may be feeding on (Kelley et al. 1997; Conklin et al. 2000). Furthermore ehu, the smaller of the two species, seem to require shelter (Kelley et al. 1997; Kelley et al. 2000b) which this type of substrate also provides. Onaga adults appear to feed on fish in the water column and therefore their attraction to these areas could also be related to current flow and its effect on prey density (Ralston et al. 1986; Haight et al. 1993).

Multibeam data now exists for almost all of the bottomfish EFH in the main Hawaiian Islands. Depth, substrate hardness, and slope are all habitat variables that can be filtered from the bathymetry and backscatter data in ArcGIS. Furthermore, intercepts of these variables can also be extracted to provide combinations that represent different habitat types. Given that backscatter values can typically range from 0-255 while slopes can range from 0-90°, an enormous number of combinations could theoretically be generated. This process was therefore simplified by first, filtering the bathymetry data for the 100-300 m range, selecting a single "boundary value" of 20° for high/low slopes, and selecting a single boundary value for hard/soft substrates which varied according to the multibeam sonar system used and how the raw data were initially processed. Filtering and creating intercepts of the values above and below these boundaries yielded four simplified habitat types labeled as hard substrate/high slope, hard substrate/low slope, soft substrate/high slope, and soft substrate/low slope (Fig. 2). Each type was then assigned a habitat suitability index value ranging between 1 for "soft/low" to 3 for "hard/high". The amount of sampling effort in each habitat type could then be varied proportionally by establishing BotCam "units of effort" that would be multiplied by the suitability index values. Originally each BotCam unit of effort was selected to be 4 deployments but it is now clear that twice that effort is required to provide the necessary statistical power to evaluate protection effects after controlling for substrate interactions. Thus the total number of samples made in a BRFA area (both inside and outside) went from 64 to 128 in year 5.

Table 1 summarizes the outcome of this process, which generated a sampling requirement of 128 deployments (64 with the original scheme) in and around each of the selected BRFAs. The extent of the 100-300 m depth range to be sampled outside of the boundaries was based on equal area to either side of the BRFA.

***Table 1**: Summary of the Standardized BotCam Sampling Protocol for each RFA. The numbers in columns 3-5 represent the number of BotCam deployments.*

Habitat Type	Suitability Index	Inside RFA	Outside RFA	Combined
Hard substrate/high slope	3	24	24	**48**
Hard substrate/low slope	2	16	16	**32**
Soft substrate/high slope	2	16	16	**32**
Soft substrate/low slope	1	8	8	**16**
Total Deployments		**64**	**64**	**128**

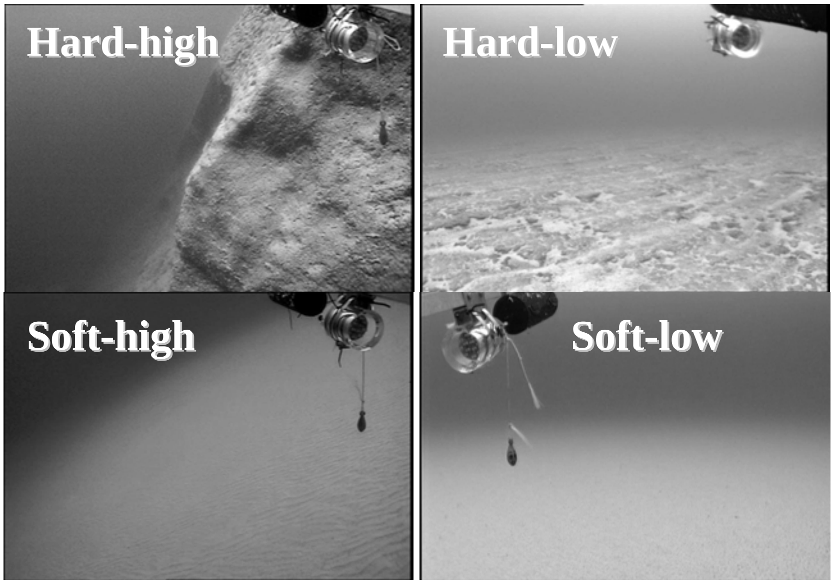


***Fig. 2**. Each of the four main habitat types is shown with a representative image from the BotCam.*

To select deployment locations, the sampling areas were gridded and assigned one of the 4 habitat types. A grid cell size of 200x200m was chosen to reflect an area from which the fish would likely be drawn to the bait and large enough to provide an adequate target for deployment. The area of each of the 4 habitat types present in each grid cell was determined using the ArcGIS Spatial Analyst extension. The habitat type with the highest percentage of the cell's area was chosen to represent the cell. The majority habitat type was >50% of the area in 95% of the grid cells. Seventy-five percent of the grids contained >70% coverage of its majority habitat type. Grid cells were chosen for sampling using a random selection procedure contained within the Hawth's Analysis Tools for ArcGIS extension (Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS. Available at http://www.spatialecology.com/htools).

In the field, the center of each grid was targeted with a deployment of one of several BotCam units. 800g of a pre-made ground, raw fish/squid mix was loaded into a lobster trap bait-canister that was fixed onto the BotCam arm near the location of the synchronization device. The camera system was triggered on the back deck just prior to deployment so that the unit was recording when it arrived on the bottom. The system recorded for ~45 minutes to ensure that a minimum of 30 minutes of video was recorded on the seafloor.

Description of BRFA selection for field work

During the first 6 years of this project, six of the 12 BRFAs in the new management plan were assessed (BRFAs B, D, E, F, H, L; Fig. 3). The goal was to sample all six in the first year to create a baseline, and then sample each one every other year to create a time series of observations. The BRFAs were selected on the basis of logistics as well as other factors such as habitat type, topography, and whether they were new or continuing closed areas. The number of sampled BRFAs was reduced over the time series because a) increased sampling intensity was required representing a tradeoff between intensity and BRFA coverage, b) there was a relatively flat budget and ship/fuel costs increased greatly and c) initial results in at least one BRFA (D) revealed little suitable habitat and few bottomfish. Thus the sampling plan shifted to a split panel design in year 3. This design balances regular sampling at one site (BRFA F) to control for interannual shifts in conditions with periodic monitoring at the other sites. Periodic sampling was justified because many of the deep 7 though they have fast initial growth have high longevities (20-40 years; Everson et al. 1989; Andrews et al. 2011). Thus periodic sampling of other locations should be adequate to capture changes in lengths and abundances statistically. The resulting field sampling, by year, is shown in Table 2 and a brief synopsis of the initial rational for study is given for each BRFA in the paragraphs below.

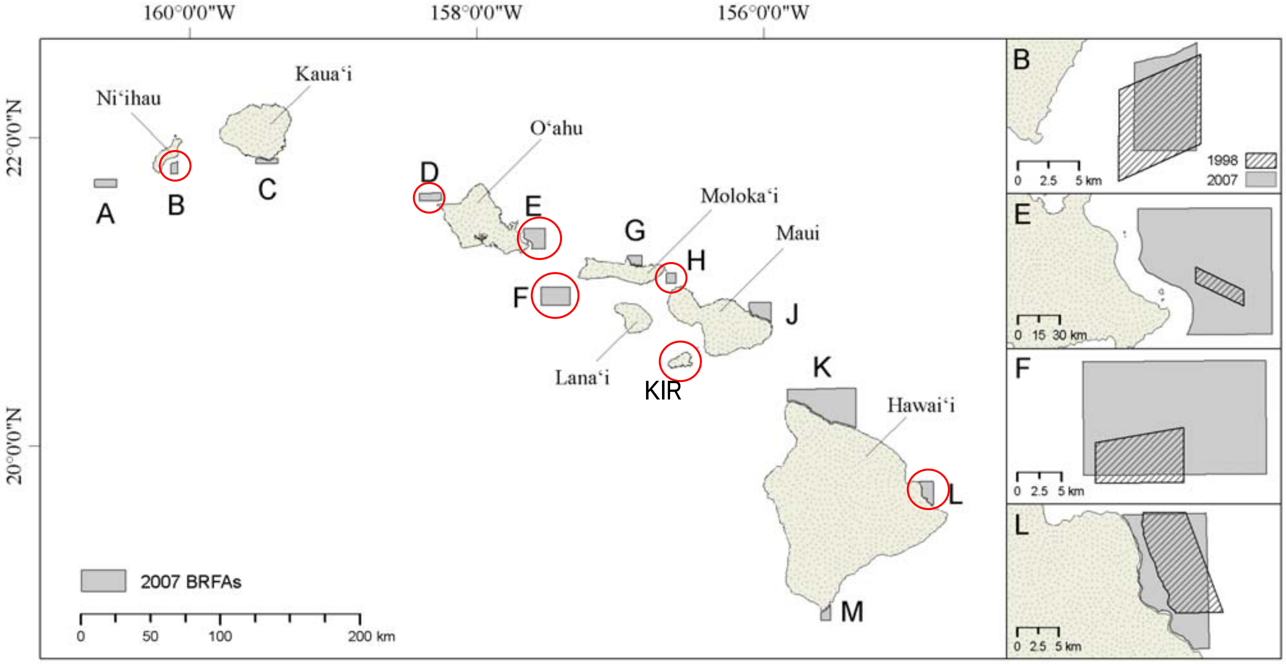


Fig. 3 *Map of the main Hawaiian Islands showing location of Bottomfish Restricted Fishing Areas (BRFAs) implemented in 2007. Each BRFA has been coded by the State of Hawai'i with a letter, proceeding from west to east. Those circled were sampled using the BotCam. Inserted to the right are enlarged maps of the 2 ongoing BRFAs (B and L) and the 2 BRFAs encompassing smaller pre-existing closed areas (E and F). Diagonal hatching indicates location of old 1998 BRFAs. KIR is the independently managed reserve off Kahoolawe Island that protects deep7 bottomfish within its borders but is not a BRFA per se.*

***Table 2**. BRFA sampling completed (2007-2014, accepted deployments only). Letters correspond to BRFA designations in Fig. 3. Intercalibration study was done in the Maui Triangle area. BRFA type indicates whether the BRFA was new in 2007, continued relatively unchanged since BRFA inception in 1998, or was partially new, meaning that the 2007 BRFA enclosed a smaller BRFA from the 1998 system. *Deployments were part of a PIRO funded study of pinnacle features around Niihau and don't conform to the BRFA stratified random sampling plan*

		Project year						
BRFA	BRFA type	07/08	08/09	09/10	10/11	11/12	12/13	13/14
B	continuing	64	62	32*	72			
D	new	64		63				
E	partial new	64	64		72		128	
F	partial new	63	57	64	72	127	120	
H	new	62	64		72			
L	continuing	61						
KIR	continuing		34	25				
Waikiki nursery	n/a		4	28		3		
Intercal study	n/a				84	79	208	258

BRFA B is a continuing closed area from 1998 that was called RFA 1 (Fig. 4). It protects two important types of habitat: a very large integrated pinnacle (guyot) and a major terrace. This RFA has potential for bi-directional spillover along the major terrace and unidirectional spillover from the guyot to Pueo Point Pinnacle just outside of the eastern boundary. Interestingly tagging and tracking data (Weng 2013) suggest residence of fish in the BRFA with some movements to Pueo Point Pinnacle. This site also has excellent habitat and a relatively high density of fish compared to several other BRFAs (Moore et al. 2013).

BRFA D is a new (as of 2007) RFA that closed an area off Ka‘ena Point, O‘ahu previously open to fishing. The original RFA 4 on this end of O‘ahu was located further south on the Wai‘anae side of the point (Fig. 4). The predominant bottomfish habitat type is a major terrace. However, a landslide escarpment also can be found inside the eastern boundary where ‘ōpakapaka, ehu, and small onaga have been caught. The presence of larger onaga has been confirmed near the southwestern boundary. This BRFA has potential for bi-directional spillover however, its suitability for examining objectives 1 (population recovery) and 3 (displaced fishing effort) as well as logistics (day charters can be arranged from Hale‘iwa) were the main reasons this RFA was selected. We discontinued sampling the BRFA in year 3 because despite initial positive habitat assessments, little suitable habitat was found visually in the area and very few fish were found.

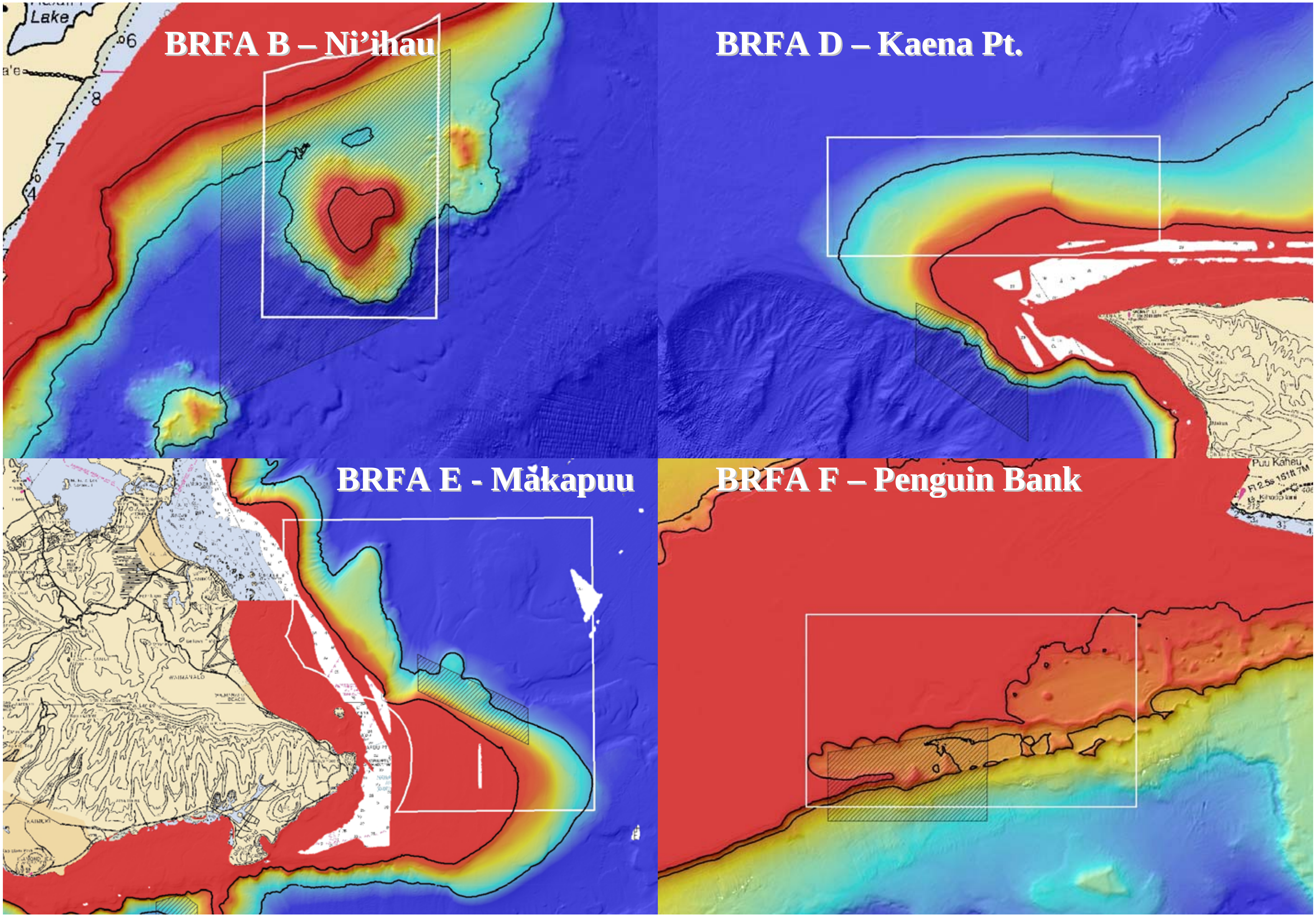
BRFA E is located off East O‘ahu between Lanikai and Makapu‘u Pt, a very logistically accessible location. The 2007 boundaries completely enclosed the previously established RFA 6 (Fig. 4) which anecdotally was an onaga habitat but only confirmed by submersible surveys to contain small ehu. This is the more important of the two RFAs located off O‘ahu because the northwestern corner encloses the Lanikai promontory which is 1 of only 3 onaga nursery grounds so far identified in the main Hawaiian Islands. This has probably been the most heavily fished site on O‘ahu and based on interviews with retired fishermen, is believed to have been severely depleted over the last 30-50 years. A canyon feature in the middle of the RFA is a known adult ‘ōpakapaka habitat whereas Makapu‘u promontory near the southern boundary is habitat for juvenile and adult ʻōpakapaka and kale. Furthermore, the boundaries enclose a substantial portion of the most productive precious coral beds in the main islands. This BRFA was chosen as an HAPC in 2012. Bidirectional spillover was expected from this BRFA particularly for ʻōpakapaka.

BRFA F, similar to BRFA E, completely encloses an older RFA, which in this case is RFA 10 (Fig. 4). The boundaries were designed to protect two very important bottomfish habitats, the second and third "fingers" of Penguin Bank, Molokaʻi. The tip of the second finger along with a narrow ridge extending to the base of the first "finger" were purposely left open to facilitate bi-directional spillover from the second to the first finger and from the third finger toward the tip of the bank. BRFA F was chosen as the continuously sampled site because Penguin Bank is heavily fished and it represents a significant portion of the total bottomfish EFH in the main Hawaiian Islands. In addition it is logistically accessible from the island of Oahu.

BRFA H is located squarely in the northern mouth of the Pailolo Channel (Fig. 4) and it encloses an area that was not previously protected from fishing in the 1998 system. The boundaries enclose two well-known and heavily fished bottomfish habitats: "Pinnacle 88" and the "119 Wall". These two features are connected via hard relatively flat substrate forming a larger complex that has subsequently been found to harbor juvenile onaga and ehu (Misa et al. 2013). This area is also suspected of being a potentially important coral habitat. It has recently been designated as a HAPC by WESTPAC (Kelley and Moriwake 2011).

BRFA L is essentially old RFA 18 with an extended southern boundary and enclosing shallower waters (Fig. 4). The northern boundary is almost identical and therefore much of the habitat enclosed has been protected since 1998. The primary habitat type is a major terrace with spillover routes present from both the northern and southern boundaries. Given the young age of the flows around the big island, the expectation was that the habitat would be somewhat different than other areas. Initially this site was chosen because of the ease of logistics out of Hilo harbor. However, we were unable to find suitable vessels with overhead lifting capacity out of Hilo so vessels had to be chartered from Oahu at great expense. As a result, monitoring of this BRFA ended after the first year.

Kahoolawe Island Reserve (KIR), though not a BRFA, does protect deep7 bottomfish. It has had a checkered past as a Navy bombing range and finally transferring to the State in 1994. It has been protected from fishing since the 1990's when the Navy was actively bombing the site, with some possible breaks between Navy occupation and KIRC management. Regular enforcement patrols began about 2005 by the management of the KIR Commission. Along its south coast is a steep terrace and two canyon features (Fig. 4). It has the potential for spillover both to the west into the Maui Triangle fishing grounds and to the east towards Maui.


BRFA B – Ni'ihau
BRFA D – Kaena Pt.
BRFA E - Mākapuu
BRFA F – Penguin Bank
Lake
Puu Kaheu

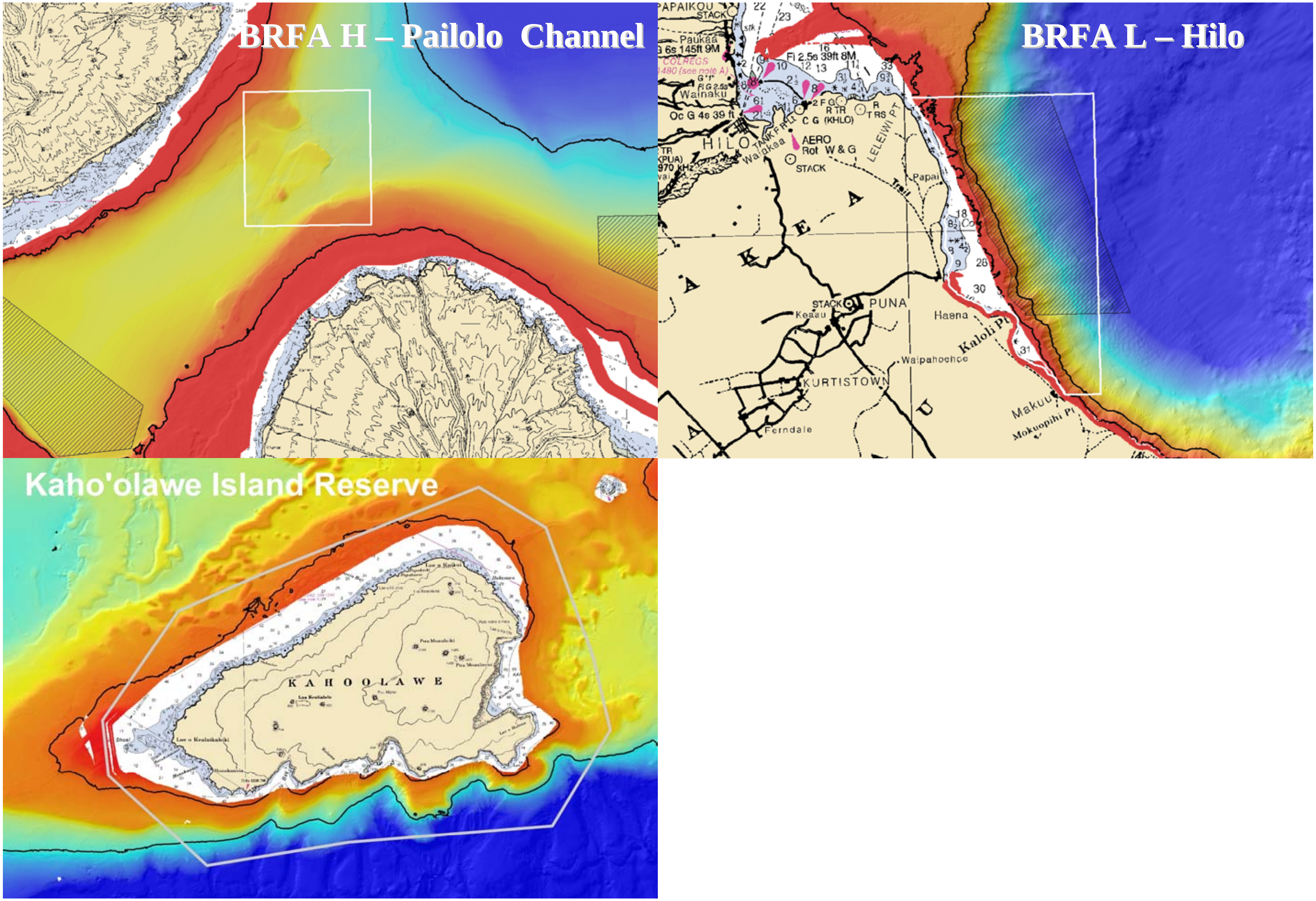


Fig. 4. *2007 BRFAs (white polygons) and 1998 BRFAs (Hatched polygons). Black contour lines are the 100 m and 400 m bottomfish EFH boundaries.*

BotCam Data extraction and analysis

The data extraction and processing protocol was formalized in conjunction with NOAA and HURL. An MS Access database was developed to assist in archiving and analyzing the data. An updated copy of this database will be provided as part of this report in electronic form. For each deployment of the BotCam the following information was extracted:

Habitat classification
1) general substrate type – (hard or soft)
2) general slope - (high or low)
3) specific substrate type – bedrock, boulders, cobbles/pebbles, sediment
4) bottom relief
5) presence of cavities/caves

Bottomfish information
1) Presence/absence for all fishes including major bottomfish species.
2) Overall Nmax: The maximum number of fish of each deep 7 and kahala species (for a portion of the data there is Nmax for all species) observed in a single frame during the 30-minute deployment.
3) Time of first arrival: The time between the start of the deployment (when BotCam touches down on the bottom) and when the first individual of each species was observed.
4) The lengths of all eteline snappers, groupers, and kahala (deep 7 and other large commercially caught species) at the time when the maximum number could be measured (often but not always at the time of Nmax).

Fish lengths were obtained using the software Visual Measurement System (Geomsoft), Photomeasure and later Eventmeasure (SeaGIS Inc.). The software allows for a calibrated stereogrammetric analysis of the synchronized output from the two cameras. System calibrations were performed in a pool prior to each major field effort.

Relative abundances of each fish species were estimated using Nmax which positively correlates with fish density. Existing models relating both Nmax and time of first arrival to abundance have been generated primarily in deep abyssal plain habitats (Priede and Merrett 1998; Yau et al. 2001; Bailey and Priede 2002). Most recent work in shallower habitats with complex bathymetry and uneven fish distributions find that Nmax best correlated to independent estimates of fish density such as SCUBA transect, beach seines and fishing (Willis et al. 2000; Cappo et al. 2004; Stoner et al. 2008; Colton and Swearer 2010).

To address our objectives, the relative abundance variables and size frequency data were used to assess changes over time, differences between BRFAs and adjacent areas, and differences between habitat types. Length data were normally distributed permitting conventional parametric statistical analysis. Abundance data are distributed as a negative binomial. In other words they are hyperdispersed due to the schooling nature of most of the deep7 species and our analysis of the relative abundance in the first year suggests a large amount of variance. Therefore the assumptions of conventional parametric statistics are violated. We took a non-parametric approach to data analysis using permutational ANOVA (PERMANOVA) (Anderson et al. 2008). Latter analysis, using more data, employed general linear models (GLM) with the negative binomial distribution (Krebs 1999).

RESULTS AND DISCUSSION

During the 6 years of sampling of the BRFA efficacy project we performed 1474 successful BotCam deployments in areas both inside and outside of six of the twelve BRFAs and in the Kahoolawe Island Reserve (Table 2). In addition we performed 35 deployments identifying a nursery ground for opakapaka off of Waikiki and performed another 629 deployments of the system as part of the NOAA led gear intercalibration experiment (described below). For the BRFA project 6801 fish (4526 individuals of deep7 species) were measured.

Most of the main findings have been published in international peer-reviewed scientific journals all of which have been made available to DLNR/DAR and are also available online. They are attached here as the best representation of our results available and to provide a single coherent reference document. Their citations are organized by main finding. In a few instances, results have not yet been published and these are described below main findings.

1. Species specific habitat associations are evident and change ontogenetically in some species (Merritt et al. 2011; Misa et al. 2013).

As yet unpublished are our findings of nursery grounds for very small juvenile bottomfish. Earlier surveys found aggregations of juvenile opakapaka off Kaneohe Bay and south Molokai (Moffitt and Parrish 1996; Parrish et al. 1997). Subsequent submersible and fishing surveys conducted for DAR from 1998 to 2007 identified additional juvenile opakapaka sites off east (Kahana Bay) and north (Haleiwa) Oahu as well as onaga and/or ehu juveniles off east Oahu, in Pailolo Channel, and in the Maui Triangle (Kelley et al. 2000a; Kelley and Moriwake 2012). Our results found juvenile opakapaka (<20cm and 1 year old) in several locations. We documented an aggregation of juvenile opakapaka off Waikiki beach which were present between 40 and 42m in May 2009 and March 2012 suggesting that this location may be a regular nursery ground. A 28.6 cm individual was also observed in February 2010 and this is around the size where this species migrates to deeper waters. In addition we found juveniles along the east coast of Hawaii (in and around BRFA L) and within the BRFA at Makapuu at depths of 92-122m. This suggests that opakapaka has widespread juvenile grounds around the State.

Similar to previous submersible and fishing surveys, juvenile onaga (<30cm) were found using BotCam in Pailolo Channel within and in areas adjacent to BRFA H. All were found on primarily hard-low habitat. The smallest individual measured was 16 cm. Small kalekale (<14.5 cm) were observed in areas adjacent to BRFA at Niihau, Penguin Bank, and Pailolo in the first or second years of sampling.

Project data have been used as the foundation for models that map the likely habitat of deep7 bottomfish species across BRFAs and the main Hawaiian Islands. This is work in progress and has not yet been published (Moore, unpublished data). Species distribution modeling (boosted regression trees) was used to define and predict individual species-environment relationships. Each species model identifies major environmental determinants of their distributions and can also be used to predict and map occurrence. In addition the approach could be used with the comprehensive multibeam mapping available to identify the area of preferred habitat protected within each of the BRFAs and compare our results with the current understanding of bottomfish EFH. As an example, the outcome of this approach is given below for onaga around Penguin Bank (Fig. 5). This aspect of the project did not advance beyond preliminary analysis due to personnel changes (postdoc C. Moore returned to her home in

Australia), however we are seeking ways to collaborate with NOAA to update and finalize the work.

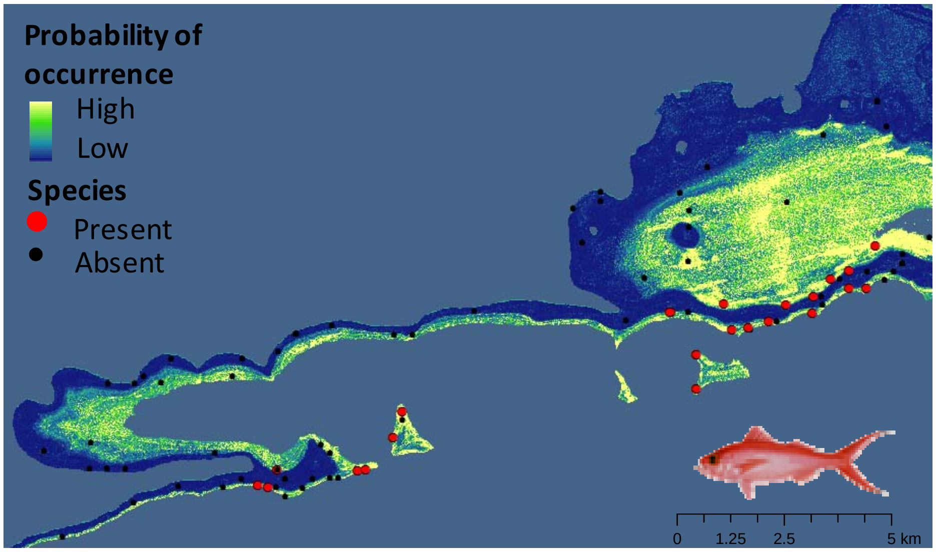


Fig. 5. *Map showing the probability of occurrence of onaga within BRFA F (Penguin Bank). Red (presence) and black (absence) dots show locations of actual BotCam deployments withheld from the model construction and indicate good prediction of presence by the model but less accurate prediction of absences of onaga (note black dots in yellow fields).*

Chris Kelley of HURL led an effort (separately funded) to combine the BotCam information with that collected by submersibles and fishing surveys to propose refinements to the EFH definitions of the Hawaiian Bottomfish Management Unit Species (BMUS) of which the deep7 are a part. Data were presented in April of 2011 as part of a WSPAR workshop to reevaluate Hawaiian bottomfish essential fish habitat definitions. This workshop resulted in the legal reclassification of Hawaiian bottomfish EFH into three guilds based principally on depth distributions. EFH for the deep 7 are now divided into an intermediate depth complex (0-320m) including lehi, opakapaka, hapu'upu and a deep complex (0-400m) including ehu, onaga, gindai, and kalekale.

2. An examination of bottomfish populations in BRFAs B and L in their first year (having been closed for 9 years) shows that protection from fishing resulted in greater size of several deep 7 species (Moore et al. 2013).

3. KIR has a greater diversity of deep7 bottomfish and most species are larger with greater proportions of sexually mature fish than other nearby regions in which active fishing occurs (Drazen et al. 2010).

We are currently working to improve our analysis of the KIR dataset. Since the original report (Drazen et al. 2010) was written, full multibeam habitat assessment has been made within the KIR boundaries. This was not available until 2012. It is now possible to categorize the habitats sampled as for other regions in the Main Hawaiian Islands. In addition, the NOAA led deep7 intercalibration study (see below), has conducted 6 field efforts in the Maui triangle area, including fished areas adjacent to KIR. Though these samples were taken about 2 years after our KIR sampling they provide a useful comparative dataset that we will explore in FY14/15.

4. The analysis of bottomfish sizes over time in BRFAs B, E, F and H show increases within BRFAs and no change or declines outside the BRFAs which clearly indicates that BRFAs protect deep7 species allowing populations to recover from fishing (Sackett et al. 2014).

5. The analysis of bottomfish abundance over time in BRFAs B, E, F and H show a few increases within BRFAs and no change outside BRFAs again suggesting that BRFAs allow deep7 populations to recover from fishing (Sackett et al. 2014).

6. Data from 6 years of monitoring around BRFAs E and F suggest that for a few species of deep7 bottomfish spillover from protected areas may be occurring (Sackett et al. submitted).

This manuscript has just been submitted to a peer-reviewed scientific journal and is attached. Briefly, BotCam and catch data were evaluated over time to examine whether spillover had begun to occur around BRFA E and F. Botcam data were also temporally examined with distance from the BRFAs, as this analysis is often used to evaluate spillover, and indicates whether the protected area is a source of more and larger fish to fished areas (McClanahan and Mangi 2000; Russ et al. 2004; Abesamis et al. 2006; Harmelin-Vivien et al. 2008; Stamoulis and Friedlander 2013). This analysis indicated that relative abundance, fish size and species richness declined with distance from both BRFAs for several deep 7 species and that declines with distance only developed in the most recent years. Analysis of fish size over time also supported our results, demonstrating that fish size had begun to level-off inside the BRFAs in the most recent years, while outside fish sizes had begun to increase. Lastly catch data also supported these results, showing similar trends to BotCam data over time, with increases in the size and number of fish caught per trip in recent years around the protected areas.

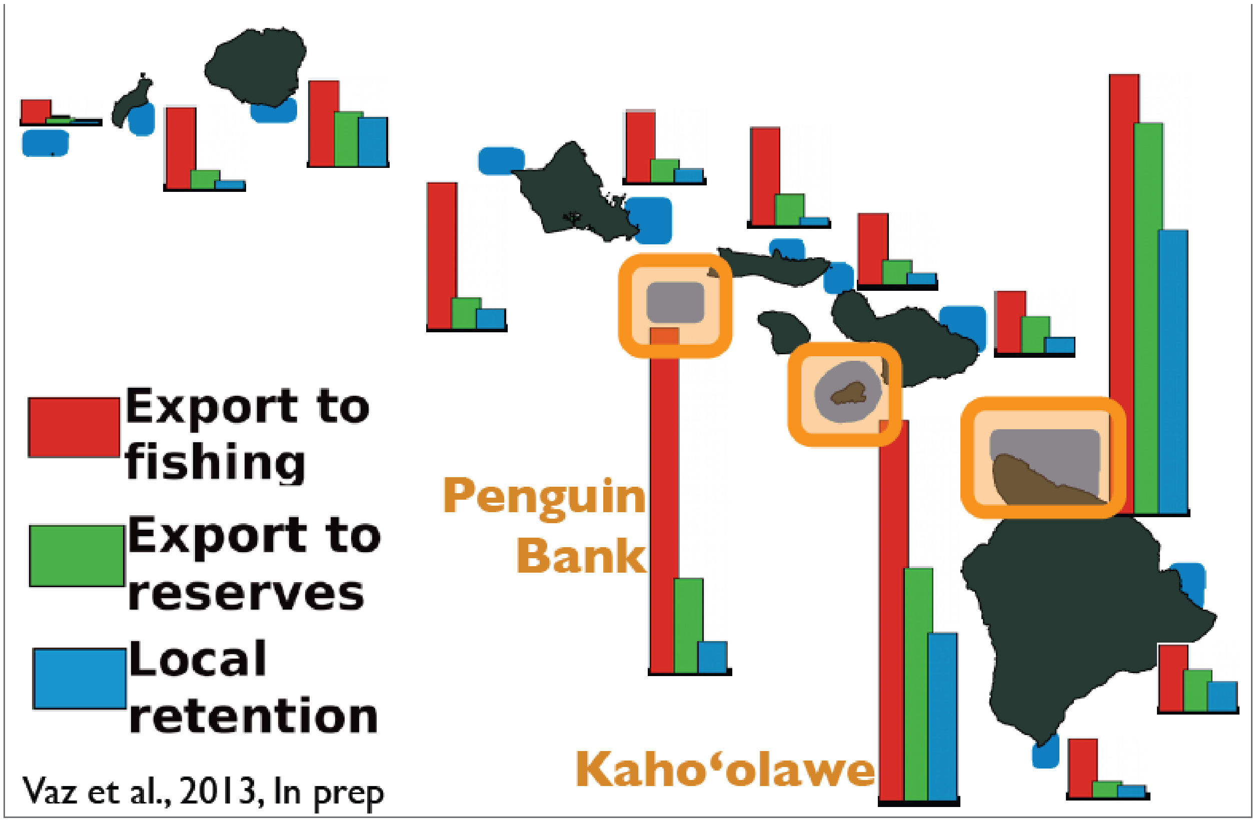
SUMMARY OF OTHER'S PROJECTS RELEVANT TO BRFA EFFICACY

At the request of DAR staff we are including a brief summary of ongoing projects, conducted by other scientists, that are relevant to deep7 bottomfish management in the Hawaiian Islands.

1. A larval dispersal model has been developed for deep 7 species in the Main Hawaiian Islands and it can be used to inform management by evaluating the connectivity of BRFAs to fished regions.

There is a hydrodynamic ocean circulation model (HYCOM) of the flow around the Hawaiian Islands (http://apdrc.soest.hawaii.edu/datadoc/hycom_iprc.php). Vaz et al (in review) used output from HYCOM and coupled it offline with the BOLTS biological model (Paris et al. 2007) using adult spawning strategy, larval development, displacement and mortality. Most of these variables were determined from limited data on bottomfish larvae (Leis 1987; Leis and Lee 1994) and the results of egg and larval development in aquaculture settings (Kelley et al. 2000a). Bottomfish larvae are pelagic, but unlike eggs, are active swimmers during most of this stage which can last from 25 to 180 days post hatch. Their swimming proficiency improves dramatically from hatching to metamorphosis. In addition to diel vertical movements, bottomfish larvae acquire the ability to move effectively in the horizontal plane in response to current flow, prey detection, and possibly sound and magnetic fields, which have been documented in other species (Fuiman 2002; Simpson et al. 2005; Cowen et al. 2006).

This model was used to predict for each BRFA the relative number of larvae exported to fished areas, other reserves or retained locally. The greatest larval export was from Penguin Bank, Kohala and KIR. Very poor larval transport came from Kaula Rock and South Point.



Friday, March 15, 2013

Fig. 15*. Predicted export of larvae from each BRFA and KIR to either fished areas (red bars), other reserves/BRFAs (green bars) and retained locally within the BRFA (blue bars) from Vaz et al (in review).*

2. BotCam sampling is one of several sampling methodologies being evaluated by NOAA for a long term fishery independent stock assessment.

There are many recognized problems with using fishery dependant data in stock assessment and management. Therefore there is a push at both federal and state levels to develop fishery independent survey approaches. NMFS-PIFSC is leading an effort, which we have been integrally involved with, to evaluate different sampling techniques in order to establish a fishery independent survey for deep7 bottomfish in the main Hawaiian Islands. The field effort and data annotation are largely paid for by NOAA though these efforts do leverage DAR funding of the BRFA efficacy project.

Four principle sampling methodologies have been evaluated. First, commercial line fishing has been used to generate CPUE metrics and this data will facilitate comparison to DARs historical fishery dependant data also. Second, BotCam, also a baited but a non-extractive technique is being used. This experiment will facilitate comparison of the BotCam data to conventional fishing metrics. Third, an autonomous underwater vehicle (AUV) with a pair of stereo cameras identical to those on BotCam has been utilized for line transects. The AUV is very expensive and requires a large ship to deploy and recovery but line transect data can be easier to interpret than point counts with either fishing or BotCam. Fourth, ship mounted acoustics (EK-60 at 38, 70 and 120 kHz) has been used to assess total fish biomass. This technique has the advantage of rapidly assessing fish biomass with excellent temporal and spatial resolution. However, there have been difficulties in determining the source of sound scattering as many species live in association with the deep7. Different survey methods (i.e. active acoustics, visual AUV transects, fishing, BotCam) are likely to have different strengths and weaknesses when it comes to assessing deep7 bottomfish populations.

Beginning in 2011 there have been 6 field sampling efforts all in the Maui Triangle region. In total the BotCam has sampled 629 times as part of this project. The results are being analyzed by PIFSC and U. of Miami statisticians and will be available late in 2014.

3. A separate way to evaluate BRFA efficacy and design is through tracking of deep7 fish movements as has been done in BRFA B (Niihau).

Dr. Kevin Weng (UH, Manoa) has carried out a pilot project to tag and track onaga and ehu in and around BRFA B (Weng 2013). Despite potential problems with gas bladders and barotraumas, even deep living bottomfish such as onaga and ehu were successfully tagged and tracked. Both fishes were tagged inside BRFA B on the central guyot and to the north on Pueo Pt. pinnacle. 39 onaga and 14 ehu were detected on a grid of acoustic receivers that spanned the BRFA area and the adjacent pinnacles and slope. After accounting for fish that disappeared or died, there were 15 good tracks for onaga and 6 for ehu. Onaga moved, on average greater distances (max 8.9km) and more frequently than the ehu (max 4.3 km) suggesting that BRFA B (5.2 km in meriodonal dimension) will protect both species but ehu to a greater extent. In addition, despite differences in movement frequency and scale, most of both species remained inside the BRFA. Overall, these results support the use of BRFAs as a management tool and they stand in contrast to some previous assumptions of regular interisland movements of deep7 based on conventional tagging data. However, these results are preliminary and it is very important that deep7 tagging work continue to better develop an understanding of their movement patterns in relation to the BRFAs and in general.

PUBLICATIONS RESULTING FROM THIS PROJECT
All are appended to this report

Drazen, J.C., Moriwake, V., Demarke, C., Alexander, B., Misa, W., Yeh, J., 2010. Assessing Kaho'olawe Island Reserve's bottomfish populations: a potential benchmark for main Hawaiian Island restricted fishing areas. prepared for the Kahoolawe Island Reserve Commission, pp. 1-33.

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Sackett, D., K., Drazen, J.C., Moriwake, V.M., Kelley, C.D., Schumacher, B.D., Misa, W.F.X.E. 2014. Marine protected areas for deepwater fish populations: an evaluation of their effects in Hawaii. Marine Biology 161: 411-425.

Sackett, D., Drazen, J.C., Moriwake, V. 2014. Bottomfish Restricted Fishing Areas and the deep 7: an update on monitoring results. Bottomfish News 17: 3-4.

Sackett, D., Moriwake, V., Kelley, C.D., Drazen, J.C. submitted. Spilling over deepwater boundaries; evidence of spillover from two deepwater protected areas in Hawaii. Ecological Applications.

PUBLIC PRESENTATIONS RESULTING FROM THIS PROJECT

7/08 Technique to monitor and study fish and their habitat below scientific diving depths. D. Merritt, K. Wong, M. Parke, C. Kelley, J. Drazen. 11th International Coral Reef Symposium, Ft. Lauderdale, FL.

10/08 *Invited*. Evaluating the Effectiveness of Restricted Fishing Areas for Improving the Bottomfish Fishery in the Main Hawaiian Islands. J. C. Drazen and C. Kelley. Western Pacific Regional Fishery Management Council Annual Meeting, Honolulu, HI

9/09 *Invited*. Baited Cameras to Evaluate the Effectiveness of Restricted Fishing Areas for Improving the Bottomfish Fishery in the Main Hawaiian Islands. **J. C. Drazen**. Deepslope Bottomfish Ecosystem and Monitoring Workshop, Honolulu, HI

9/09 Deep Bottomfish Predator/Prey Relationships & Competition. **J. C. Drazen**. Deepslope Bottomfish Ecosystem and Monitoring Workshop, Honolulu, HI

4/11 *Invited.* Baited cameras as a tool to evaluate essential fish habitat for the Deep 7 bottomfish. **Jeff Drazen** and Chris Kelley. WSPAR workshop to reevaluate Hawaiian bottomfish legal essential fish habitat definitions. Honolulu, HI

9/11 Hitting Rock Bottom? Modelling Essential Fish Habitat for Commercially Important Hawaiian Bottomfish. Cordelia Moore, Jeffrey Drazen, and William Misa. American Fisheries Society Meeting, Seattle, WA

2/12 Establishing species-habitat associations for 4 eteline snappers using a baited stereo-video camera system. W. F. Misa, **J. C. Drazen**, V. Moriwake, C. Kelley, and C. Moore. JIMAR symposium. Honolulu, HI

5/12 Investigation of bottomfish biomass and spatiotemporal distribution in the Hawaiian Archipelago: Results from baited video camera and charter fishing surveys. Don Kobayashi, Benjamin Richards, **Jeffrey Drazen**, Audrey Rollo, Clayward Tam. 5th Annual Hawaiian Islands Symposium. Honolulu, HI

4/12 Invited. Baited Cameras to Evaluate the Effectiveness of Restricted Fishing Areas for Improving the Bottomfish Fishery in the Main Hawaiian Islands. **J. C. Drazen**, C. Moore, V. Moriwake, C. Demarke, J. Friedman, W. Misa, and M. Waterhouse. Hawaii Plan Team meeting of Western Pacific Regional Fisheries Management Council, Honolulu, HI

7/12 Invited. The biology of deep-sea animals revealed by cameras. Jeffrey Drazen. SPAWAR Systems Center Pacific, San Diego, CA

6/12 Invited. Assessing the Effectiveness of Restricted Fishing Areas in the Main Hawaiian Islands. **Jeffrey Drazen**, Cordelia Moore, Dana Sackett. 154th Meeting of the Western Pacific Regional Fishery Management Council. Honolulu, HI

6/12 Testing BRFA efficacy in the main Hawaiian Islands. Cordelia Moore, **Jeffrey Drazen**, Chris Kelly, and William Misa. Science and Statistical Committee meeting of Western Pacific Regional Fishery Management Council. Honolulu, HI

6/12 Invited. Using Botcam to study the ecology and fisheries of Hawaii's Deep 7 bottomfish. **Jeffrey Drazen**, Cordelia Moore, Dana Sackett, Virginia Moriwake, Chris Demarke, Jason Friedman, William Misa, and Matt Waterhouse. NOAA PIFSC Modular Stereo-Video System Development Workshop. Honolulu, HI

5/13 Invited. Evaluating the Effectiveness of Bottomfish Restricted Fishing Areas in the Main Hawaiian Islands. **Jeffrey Drazen**, Dana Sackett, Cordelia Moore, William Misa, Virginia Moriwake, Chris Kelley and the bottomfish team. Department of Aquatic Resources: Fisher's Talk Story. Honolulu, HI

6/13 Invited. Evaluating the Effectiveness of Bottomfish Restricted Fishing Areas in the Main Hawaiian Islands. **Jeffrey Drazen**, Dana Sackett, Cordelia Moore, William Misa, Virginia Moriwake, Chris Kelley and the bottomfish team 155th Meeting of the Western Pacific Regional Fishery Management Council. Honolulu, HI

6/13 Invited. Evaluating the effectiveness of restricted fishing areas in the main Hawaiian islands. Dana K. Sackett, **Jeffrey C. Drazen**, Virginia Moriwake, Cordelia Moore, Chris Kelley, William Misa, and the bottomfish team. Science and Statistical Committee meeting of Western Pacific Regional Fishery Management Council. Honolulu, HI

ONLINE PRESENTATION ON OLELO

http://olelo.granicus.com/MediaPlayer.php?view_id=30&clip_id=38082

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ORIGINAL PAPER

Marine protected areas for deepwater fish populations: an evaluation of their effects in Hawai'i

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Abstract The success of marine protected areas (MPAs) as a tool for conservation and fisheries management has been well documented. However, these results have typically been seen in shallow water systems and questions remain whether this management strategy could be successfully applied to deepwater ecosystems. Our objectives were to determine the efficacy of four deepwater MPAs called bottomfish restricted fishing areas (BRFAs), with various time spans of protection, monitored at depths between 90 and 310 m from 2007 to 2011 for six species of deepwater snapper and one grouper harvested in the Main Hawaiian Islands. Our results suggested that the duration of protection influenced reserve effects, particularly for target species. Mean fish length, and in some cases abundance, increased for one or more of the most economically important target species inside nearly all tested BRFAs. In addition, more mature fish were seen inside the BRFA with the longest duration of protection (~14 years); species richness increased outside this area while inside it remained the same. Here, we provide the first evidence that deepwater MPAs can have positive effects on deepwater species and that many protection effects were consistent with results found in shallow water ecosystems. While these findings are novel, additional data over greater temporal scales will be necessary to determine whether these trends will continue and if others will become important over time.

Introduction

Marine protected areas (MPAs; marine areas that exclude some or all forms of harvest) have been used worldwide as a conservation tool and long-term fishery management strategy to benefit fish stocks (Roberts and Polunin 1991; Babcock et al. 2010; Gaines et al. 2010). While there are numerous potential benefits to employing MPAs (e.g., protect biodiversity, habitat, genetic diversity), one of the primary benefits is linked to the exponential increase in fish fecundity with body size (Bohnsack 1994; Roberts and Polunin 1991; Bohnsack 2011). For example, a large female red snapper (*Lutjanus campechanus*; 61 cm) can produce the same number of eggs as 212 smaller female snapper (42 cm; Bohnsack 1994). Ideally, protection of important marine habitats would, therefore, lead to increased fish size inside a protected area, followed by increased recruitment to the whole population (Bohnsack 1994; Pelc et al. 2010). Fish abundance would also ideally increase inside MPAs as fish populations rebuild to unfished levels and density-dependent processes cause adults to emigrate to fished areas (spillover; e.g., Harmelin-Vivien et al. 2008; Stobart et al. 2009; Bohnsack 2011). Previous research has demonstrated these benefits for various exploited species when MPAs are well designed and managed (Halpern and Warner 2003; White and Kendall 2007; White et al. 2008; Lester et al. 2009; Gaines et al. 2010). However, studies on MPAs primarily focus on shallow water reef systems and

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questions remain regarding whether MPAs could be successful in other environments, specifically deepwater habitats and ecosystems.

Understanding whether MPAs could be a useful fishery management tool in deepwater environments is particularly important because fishers have targeted fish stocks in increasingly deeper waters over the last several decades as many shallow water stocks have become depleted (Haedrich et al. 2001; Morato et al. 2006). For example, global trends since the 1950s suggest mean fishing depth has increased from approximately 40–150 m, with an increasing mean rate of 13 m decade^{-1} in more recent years (Morato et al. 2006). Management measures might therefore be required to replace the natural refuge that depth previously provided. In addition, information on deepwater species ecology suggests that many targeted species are characterized by extended longevity, slow growth rates, late maturity and low rates of natural mortality (Drazen and Haedrich 2012; Williams et al. 2013). As such, deepwater species often have exceptionally low production potential and are, therefore, highly vulnerable to overfishing (Cheung et al. 2005; Morato et al. 2006). These traits suggest deepwater stocks can be rapidly depleted and very slow to recover. Consequently, there is a critical need to apply successful fishery management strategies to deepwater species in a timely manner (Haedrich et al. 2001; Williams et al. 2013). Indeed, fisheries managers have turned to protected area management in deep-sea areas in recent years, often to protect fragile corals and in a few instances to protect fish species (e.g., the protection of a deepwater snapper–grouper complex in the south Atlantic, SAFMC 2013). With the exception of deeper shelf waters (to ~150 m; Harter et al. 2009; Rudershausen et al. 2010), there are no other studies, to our knowledge, which directly evaluate protected area effects on deepwater fished species.

Deepwater fisheries have existed in the Hawaiian Islands for several decades (Grigg 2001; Williams et al. 2013). For instance, deepwater snappers were historically fished by hand by native Hawaiians and have been reported as a commercial fishery since the late 1950s (Hospital and Beavers 2012; Williams et al. 2012). However, the advent of electric reels, advanced fish finders and GPS has increased catch to substantial levels in recent years (Dalzell and Preston 1992; Williams et al. 2012). Further, the spread of these technological advances in the Pacific region suggests that exploitation will grow steadily throughout the range of these species (Dalzell and Preston 1992; Williams et al. 2012). In accordance, there has been a recent request for stock assessments and an evaluation of management strategies for the Pacific region's deepwater snapper stocks (Williams et al. 2012; Williams et al. 2013). Our aim was to evaluate four deepwater MPAs in the Main Hawaiian Islands to determine whether this management strategy could be successfully applied to deepwater snappers.

A particularly important complex of exploited deepwater bottomfish species in Hawai'i is known as the "Deep 7" and comprises six snappers (Lutjanidae) in the subfamily Etelinae: deepwater red snapper *Etelis carbunculus*, deepwater long-tail red snapper *E. coruscans*, crimson jobfish *Pristipomoides filamentosus*, lavender jobfish *P. sieboldii*, oblique-banded snapper *P. zonatus*, rusty jobfish (*Aphareus rutilans*), and one grouper (Serranidae): Hawaiian grouper *Hyporthodus quernus*. Of these, *E. carbunculus*, *E. coruscans and P. filamentosus* are the most economically and commercially important in the Main Hawaiian Islands (Haight et al. 1993; Kelley et al. 2006). Although limited, data on life history characteristics for these species indicate that they are generally long lived (20 to 40+ years for some species) and relatively late maturing ($\geq$ 6 years for some species; Andrews et al. 2011, 2012; Kelley and Moriwake 2012). These traits are intermediate to the rapid growth and maturity of many reef fishes and the extreme slow growth and maturity of deep slope or seamount associated fishes (Drazen and Haedrich 2012).

During the 1990s, catch rates and spawning potential ratios of the Deep 7 indicated that their populations had declined, with metrics for *E. carbunculus* and *E. coruscans* generating particular concern. Also, because Deep 7 species are relatively site attached, often forming aggregations around high relief structures such as pinnacles, it was believed that a spatially based management strategy such as a network of MPAs would benefit these fish stocks (Ralston et al. 1986; Haight et al. 1993; Kelley et al. 2006; Parke 2007; Merritt et al. 2011). Therefore, in 1998, the State of Hawai'i, Department of Land and Natural Resources implemented a system of 19 MPAs labeled bottomfish restricted fishing areas (and hereafter referred to as BRFAs) throughout the Main Hawaiian Islands. These BRFAs excluded bottomfish harvest, protecting the deepwater environment and species that reside there while leaving surface waters open to fishing for pelagic species. On June 1, 2007, the system was revised to reduce the overall number to 12, but increased the area protected to include more essential fish habitat (EFH; Rosenberg et al. 2000; Moffitt et al. 2006; Parke 2007; Kelley and Moriwake 2012; Moore et al. 2013; Fig. 1).

Here, we examined whether relative abundance, mean length and species richness of the Deep 7 complex increased inside BRFAs compared to adjacent fished areas using data acquired from a baited camera system from 2007, when the revised BRFAs were established, to 2011. Further, because some BRFAs remained unchanged after they were revised in 2007 while others were expanded or newly created, we were able to compare BRFAs with different time spans of protection to examine the potential

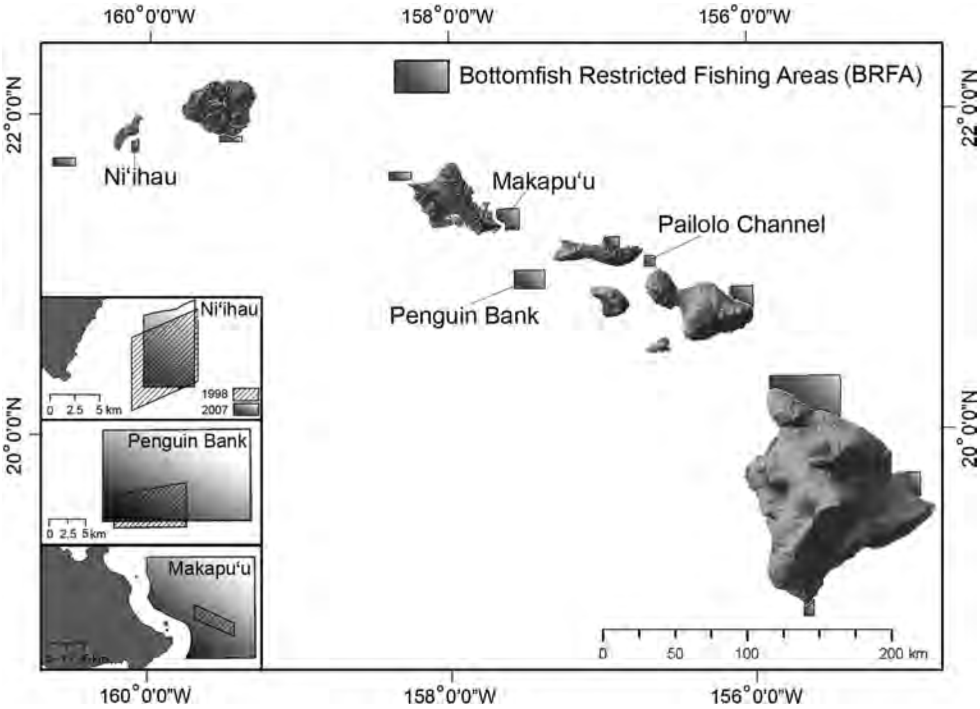


Fig. 1 In 1998, 19 deepwater marine protected areas called bottomfish restricted fishing areas (BRFAs) were implemented throughout the Main Hawaiian Islands and later revised on June 1, 2007, reducing the overall number to 12 (revised BRFAs depicted with gradient fill). Our study sites included four BRFAs, one had boundaries similar to the original 1998 BRFAs (original BRFAs depicted with diagonal hatching in *insets*) and thus had been protected for approximately 14 years (Ni'ihau), two encompassed smaller pre-existing closed areas (Penguin Bank and Makapu'u), and one was newly closed in 2007 (Pailolo Channel)

progress of protection effects beyond when monitoring began.

Materials and methods

Data collection

The sampling design and technique used here have been described previously by Moore et al. (2013). Briefly, our baited stereo-video camera system (BotCam) was specifically designed as a fishery-independent tool to monitor Hawaiian deepwater bottomfish and their habitat (Merritt et al. 2011). The system employed two ultralow-light video cameras that recorded under ambient light to a depth of 310 m, used a light diode to synchronize the stereo-video pair and enabled accurate fish length measurements (Harvey and Shortis 1995; Shortis et al. 2008; Watson et al. 2010). BotCam floated approximately 3 m above the seafloor to optimize the field of view for our targeted species. For instance, those Deep 7 species closely associated with the seafloor (e.g., *P. zonatus*) and those that resided higher in the water column (e.g., *E. coruscans*) were both visible in the field of view. Bait was kept in a plastic mesh bait canister in view of both cameras and consisted of approximately 800 g of chopped and frozen anchovy (*Engraulis mordax*) and squid (*Loligo opalescens*). Local commercial anglers also use this bait on hooks and in chum bags while fishing.

We used BotCam to conduct paired sampling inside and outside of four BRFAs in the Main Hawaiian Islands from approximately 90 to 310 m (Fig. 1). Of these four BRFAs, one had boundaries similar to the original 1998 BRFAs (Ni'ihau) and thus had been protected for approximately 14 years, two encompassed smaller pre-existing closed areas (Penguin Bank and Makapu'u), and one was newly closed in 2007 (Pailolo Channel; Fig. 1). Data were collected from May 2007 to June 2011. Sample sites were selected using a stratified random sampling protocol with strata based on protection and habitat. An equal number of samples were targeted inside and outside but adjacent to each BRFA with the same habitat designations. Habitat designations were classified as high slope (≥20°) or low slope (<20°) and as consolidated hard substrate or unconsolidated soft substrate for every 200 m^2 area based on multibeam bathymetry and backscatter data. These habitat classifications resulted in four possible designations: hard-high, hard-low, soft-high and soft-low (Moore et al. 2013; Misa et al. 2013). The 200 m^2 grid-cell size was chosen to reflect the area where fish would likely be drawn by bait and large enough to provide an adequate target for the deployment of BotCam. At each sample site, BotCam was deployed and left to record for 45 min before being retrieved, a time previously noted for optimizing peak feeding activity using bait (Harvey and Cappo 2001). In addition, BotCam units deployed concurrently were placed at a minimum of 400 m apart to reduce if not prevent bait plume overlap and sampling the same fish by both systems (Moore et al. 2013).

Video analysis

All fish within a single video were identified to the lowest taxonomic unit, commonly species. Relative abundance was recorded as the maximum number of each species observed in a single frame of video (MaxN; Parrish 1989; Priede et al. 1994; Cappo et al. 2003). Species not seen in the video were given a value of zero. Because of the high number of zeros in the MaxN datasets for each species and because not all species occupy the entire depth range sampled, we included only those data from a preferred depth range for each of our target species. These preferred depth ranges were previously determined by Misa et al. (2013) using a Euclidean distance matrix and pair-wise PERMANOVA of MaxN data in 30-m depth bins for four of the Deep 7 and are as follows: 210–310 m for *E. carbunculus* and *E. coruscans*, 90–210 m for *P. filamentosus* and 180–270 m for *P. sieboldii*. We calculated the preferred depth ranges for the remaining three species using the same method as Misa et al. (2013): 150–270 m for *P. zonatus*, 120–240 m for *H. quernus* and 90–240 m for *A. rutilans*. More than 90 % of the mean relative abundance for each species was within the specified depth ranges and allowed for more robust statistical analyses by reducing the number of zeros in each species' dataset. Fork lengths (mm) were also taken only one time in a single video to avoid measuring the same fish more than once. These measurements were recorded when the highest number of measurable fish (the entire fish was visible in both cameras) was seen and computed using one of three stereo-photometric programs (Visual Measurement System version 7.5, Geomsoft, Victoria, Australia; PhotoMeasure version 1.74, SeaGIS Pty Ltd; EventMeasure Stereo version 3.32, SeaGIS Pty Ltd). Approximately, five replicate measurements were also taken for each individual to increase precision and accuracy. In addition, we used species richness, a count of the number of Deep 7 species that were present in a single video, as a measure of biodiversity (e.g., if all Deep 7 species were viewed in a single 45 min video the species richness value would be seven).

Statistical analysis

To analyze factors that affected relative fish abundance and size structure for all seven species from each BRFA, we ran a series of seven candidate models for both MaxN and fish length for each species and BRFA and ranked them with Akaike's information criterion ($\mathrm{AIC_c}$; Burnham and Anderson 1998; Table 1). Factors included sampling year, protection (inside or outside the BRFA), habitat designations (hard-high, hard-low, soft-high and soft-low), and the interaction between protection and sampling year (year*protection). Model selection was based on available factors, the sample size of the dataset and to answer specific questions about the data. For instance, the interaction between protection and sampling year was tested to determine whether there were different trends in fish length or abundance over time inside versus outside each BRFA (i.e., the slopes of the regression lines inside and outside were significantly different). In all cases, sample depth was treated as a random effect to address potential bias in our model results because samples were not stratified by depth until year 4, and then, stratification was very broad (above and below 200 m depth). MaxN data were analyzed using generalized linear mixed models (GLMM) with a negative binomial distribution to account for the hyperdispersed nature of count data. This method has been successfully used in previous studies with similar datasets (Martinez et al. 2011; Smith et al. 2012). Because length data met assumptions of normality, we used standard least squares multiple regression models for these analyses. Model probability weights (W_i) were used to examine the strength of evidence for each model (W_i indicates the probability that a model is the best of the set of models tested; Burnham and Anderson 1998). Here, we display only the top ranked model, using $\mathrm{AIC_c}$ for each species, BRFA and analysis (MaxN and fish length; Tables 2, 3).

Table 1 Seven candidate models tested to explain MaxN and fish length data for each species in each BRFA

Candidate models
habitat
protection
year
year*protection
year*protection, habitat
year*protection, protection
year*protection, protection, habitat

Candidate models were also tested to explain species richness data in each BRFA. The best of the seven models tested for each analysis was determined using Akaike's information criterion ($\mathrm{AIC_c}$; Burnham and Anderson 1998)

* Represents an interaction between the two parameters

To examine factors that best explained the variation in species richness data, we used the same series of seven candidate models and ranked them with $\mathrm{AIC_c}$ (Table 1). All significant ($P < 0.05$) factor effects in our weighted models were further investigated using adjusted means and mean predicted values from model output because these measures take other model effects into account. The percent of mature fish inside and outside each BRFA was also compared (sizes at maturity were *E. carbunculus* = 279 mm (DeMartini and Lau 1999), *E. coruscans* = 700 mm (Everson et al. 1989), *P. filamentosus* = 450 mm (Ralston and Miyamoto 1983), *P. sieboldii* = 290 mm (DeMartini and

Table 2 Top ranked generalized linear mixed models for explaining MaxN data for each species in each BRFA using Akaike's information criterion (AIC_c; Burnham and Anderson 1998)

BRFA	Species	Top model	P	N	W_i	Models tested
Ni'ihau	*E. carbunculus*	year*protection, protection, habitat	0.73	90	0.90	7
	E. coruscans	year*protection, protection, habitat	0.87	90	0.83	7
	P. filamentosus	year*protection, protection	0.35	102	1.00	7
	P. sieboldii	year*protection, protection, habitat	0.22	90	0.92	7
	P. zonatus	year*protection, protection, habitat	0.63	123	0.99	7
	H. quernus	protection	0.27	116	0.99	7
	A. rutilans	–	–	128	–	0
Penguin Bank	*E. carbunculus*	protection	0.37	140	0.62	7
	E. coruscans	year*protection[#], protection[#], habitat[##]	0.01	140	0.99	7
	P. filamentosus	year*protection, protection, habitat	0.22	105	0.96	7
	P. sieboldii	habitat	0.51	145	0.81	7
	P. zonatus	year*protection, protection, habitat	0.76	175	1.00	3
	H. quernus	protection	0.27	153	1.00	7
	A. rutilans	year*protection, protection, habitat	0.99	168	0.89	6
Makapu'u	*E. carbunculus*	year*protection, protection, habitat	0.71	72	0.82	7
	E. coruscans	year*protection, protection, habitat	0.19	72	0.93	5
	P. filamentosus	year*protection[##], habitat[##]	0.00	123	1.00	6
	P. sieboldii	protection	0.99	76	1.00	1
	P. zonatus	habitat	0.99	125	0.93	7
	H. quernus	habitat	0.45	131	0.93	7
	A. rutilans	year*protection	0.23	148	0.88	4
Pailolo Channel	*E. carbunculus*	year	0.76	142	0.47	7
	E. coruscans	year*protection, protection[##], habitat[##]	0.07	142	1.00	7
	P. filamentosus	habitat	0.41	51	0.94	2
	P. sieboldii	year*protection, protection, habitat	0.61	122	1.00	6
	P. zonatus	–	–	135	–	0
	H. quernus	year	0.34	111	1.00	7
	A. rutilans	–	–	113	–	0

Model probability weights (W_i) indicated the probability that a model is the best of the set of models tested. The full scientific name for each species is *Etelis carbunculus, E. coruscans, Pristipomoides filamentosus, P. sieboldii, P. zonatus, Hyporthodus quernus* and *Aphareus rutilans*

[#] indicates a marginally significant model effect ($0.05 < P < 0.10$); [##] indicates a significant model effect ($P < 0.05$)

Lau 1999) and *H. quernus* = 580 mm (DeMartini et al. 2011)). *H. quernus* are protogynous hermaphrodites, and while we used the 580 mm size at which females reach maturity for our analyses, we also summed the number of fish sampled that were large enough to be male (895 mm; DeMartini et al. 2011). All analyses were conducted using JMP 9.0.2 (2010 SAS Institute Inc.) and SPSS 21 (2012 IMB Corp.).

Results

Our results showed that the same model was ranked as best among nearly all BRFAs and species for explaining the variation in relative abundance (MaxN) and fish length data (Tables 2, 3). Factors in this model included year*protection, protection and habitat. The strength (W_i) of the best and significant ($P < 0.05$) models in explaining MaxN and fish length data ranged from 0.90 to 1.00. Of the significant ($P < 0.05$) weighted models, not all factors included in the models had significant effects. For instance, although year*protection, protection and habitat were all in the top model for *P. filamentosus* length data in Makapu'u BRFA, the protection factor (comparing mean length inside to outside with years pooled; adjusted mean length inside = 495 mm, outside = 479 mm) was not significant ($P = 0.22$) while year*protection and habitat factors were significant ($P < 0.01$). Also, among BRFAs and species, small sample sizes, especially among explanatory variable categories, resulted in a loss of power to run all seven models. At least six models could be run for 20 of the 28 tests for MaxN data (Table 2) and 13 of the 28 tests for length data (Table 3). This limitation reduced our ability to examine protection effects for all species in all BRFAs. Overall, however, changes in length and relative abundance over time occurred for one or more of the most economically important and abundant target species (*E. carbunculus*, *E. coruscans*, *P. filamentosus*; Table 4) in nearly all tested BRFAs. Results for other species

Table 3 Top ranked standard least squares models for explaining fish length data for each species in each BRFA using Akaike's information criterion ($\mathrm{AIC_c}$; Burnham and Anderson 1998)

BRFA	Species	Top model	*P*	R^2	*N*	W_i	Models tested
Ni'ihau	*E. carbunculus*	year*protection, protection, habitat$^{\#\#}$	0.01	0.34	42	0.95	7
	E. coruscans	year*protection, protection$^{\#\#}$, habitat	0.04	0.13	92	1.00	4
	P. filamentosus	year*protection, protection$^{\#\#}$, habitat	0.01	0.31	61	1.00	7
	P. sieboldii	year*protection$^{\#\#}$, protection$^{\#\#}$, habitat$^{\#\#}$	0.00	0.18	244	1.00	7
	P. zonatus	–	–	–	9	–	0
	H. quernus	year*protection, protection, habitat	0.17	0.31	31	0.93	5
	A. rutilans	–	–	–	2	–	0
Penguin Bank	*E. carbunculus*	year*protection$^{\#\#}$, protection$^{\#\#}$, habitat	0.03	0.13	158	0.98	7
	E. coruscans	year*protection$^{\#\#}$, protection$^{\#\#}$, habitat	0.00	0.21	118	1.00	5
	P. filamentosus	year*protection$^{\#\#}$, protection$^{\#\#}$, habitat$^{\#\#}$	0.00	0.42	230	0.95	7
	P. sieboldii	year*protection$^{\#\#}$, protection$^{\#\#}$, habitat$^{\#\#}$	0.00	0.21	312	1.00	7
	P. zonatus	year*protection, protection, habitat	0.07	0.27	28	0.90	6
	H. quernus	year*protection, protection, habitat	0.69	0.33	17	0.96	7
	A. rutilans	year*protection, protection$^{\#\#}$, habitat	0.02	0.41	29	1.00	6
Makapu'u	*E. carbunculus*	year*protection, protection, habitat	0.31	0.42	37	0.96	7
	E. coruscans	year*protection, protection	0.86	0.70	10	0.46	6
	P. filamentosus	year*protection$^{\#\#}$, protection, habitat$^{\#\#}$	0.00	0.44	215	0.92	7
	P.sieboldii	habitat	0.12	0.88	5	1.00	1
	P. zonatus	year	0.98	0.40	3	1.00	1
	H. quernus	–	–	–	4	–	0
	A. rutilans	year	0.41	0.65	3	1.00	1
Pailolo Channel	*E. carbunculus*	year*protection, protection	0.09	0.06	289	0.48	4
	E. coruscans	year*protection$^{\#}$, protection$^{\#\#}$	0.00	0.22	166	0.97	4
	P. filamentosus	year*protection, protection$^{\#\#}$, habitat	0.00	0.49	76	0.95	7
	P. sieboldii	year*protection, protection$^{\#\#}$	0.00	0.12	95	0.90	3
	P. zonatus	–	–	–	1	–	0
	H. quernus	year*protection, protection	0.15	0.07	34	0.95	4
	A. rutilans	–	–	–	0	–	0

Model probability weights (W_i) indicate the probability that a model is the best of the set of models tested. The full scientific name for each species is *Etelis carbunculus, E. coruscans, Pristipomoides filamentosus, P. sieboldii, P. zonatus, Hyporthodus quernus*, and *Aphareus rutilans*

$^{\#}$ indicates a marginally significant model effect ($0.05 < P < 0.10$); $^{\#\#}$ indicates a significant model effect ($P < 0.05$)

were limited due to sample size. Indeed, significant results were only found when samples sizes were greater than approximately 100 for length data, and the percent of samples in which a particular species was present was greater than 40 % (Table 4).

Site with the longest duration of protection

The BRFA with the longest duration of protection, Ni'ihau BRFA, showed no significant effects for MaxN data for any of the models tested (Table 2). This may have been the result of the smaller fraction of deployments in which a particular species was observed (nonzero MaxN data <35 %) in this BRFA (Table 4). Conversely, our length analysis showed significant results for three species. *P. filamentosus* inside Ni'ihau BRFA were larger compared to outside, while the opposite was seen for *E. coruscans* and *P. sieboldii* (Fig. 2a). Over time mean predicted lengths decreased significantly inside for *P. sieboldii* and increased significantly outside ($P < 0.05$; Fig. 3a). Similar trends for *E. coruscans* and *P.filamentosus* were not significant ($P_{\text{year*protection}} = 0.17$; $P_{\text{year*protection}} = 0.14$). Diversity (species richness) of our target species increased outside this BRFA ($P_{\text{year*protection}} = 0.01$; $P_{\text{out}} = 0.01$), although it remained unchanged inside ($P_{\text{in}} = 0.33$; Table 4c). There were also higher percentages of mature fish inside Ni'ihau BRFA than outside for each species examined, with the exception of *P. filamentosus* where 100 % of the fish seen inside and outside the BRFA were mature (Fig. 4a). In addition, for *H. quernus*, which undergo a sex change from female to male at approximately 895 mm (DeMartini et al. 2011), none of the fish

Table 4 A summary of significant ($P < 0.05$, bold) and marginally significant ($0.05 < P < 0.10$, italics) trends in (a) length (b) relative abundance and (c) species richness data over time inside and outside protected areas (year*protection)

a. Length	Ni'ihau			Penguin Bank			Makapu'u			Pailolo Channel		
Species	N	In	Out	N	In	Out	N	In	Out	N	In	Out
E. carbunculus	42			**158**	↑	↓	37			289		
E. coruscans	92			**118**	↑	↓	10			*166*	↑	↓
P. filamentosus	61			**230**	↑	–	**215**	↑	–	76		
P. sieboldii	**244**	↓	↑	**312**	↓	↑	5			95		
P. zonatus	9			28			3			1		
H. quernus	31			17			4			34		
A. rutilans	2			29			3			0		

b. MaxN	Ni'ihau			Penguin Bank			Makapu'u			Pailolo Channel		
Species	Presence*	In	Out	Presence*	In	Out	Presence*	In	Out	Presence*	In	Out
E. carbunculus	0.20 (90)			0.48 (140)			0.19 (72)			0.77 (142)		
E. coruscans	0.31 (90)			**0.44 (140)**	↑	–	0.13 (72)			0.48 (142)		
P. filamentosus	0.24 (102)			0.53 (105)			**0.47 (123)**	↑	↓	0.43 (51)		
P. sieboldii	0.32 (90)			0.34 (145)			0.04 (76)			0.23 (122)		
P. zonatus	0.12 (123)			0.16 (175)			0.07 (125)			0.01 (135)		
H. quernus	0.14 (116)			0.10 (153)			0.04 (131)			0.17 (111)		
A. rutilans	0.01 (128)			0.14 (168)			0.01 (148)			0.00 (113)		

c. Species richness	Ni'ihau			Penguin Bank			Makapu'u			Pailolo Channel		
	Presence*	In	Out	Presence*	In	Out	Presence*	In	Out	Presence*	In	Out
Deep 7	*0.45 (190)*	–	↑	0.70 (244)			0.44 (192)			0.75 (190)		

↑ signifies an increase over time, – signifies no change over time and ↓ signifies a decrease over time. The full scientific names for each species that make up the Deep 7 are *Etelis carbunculus, E. coruscans, Pristipomoides filamentosus, P. sieboldii, P. zonatus, Hyporthodus quernus* and *Aphareus rutilans*

* The percent of nonzero data in each database with the sample size in parentheses

measured were large enough to be male inside the BRFA, and only one of 15 was large enough to be male outside the BRFA.

Significant habitat associations occurred for only two species in Ni'ihau BRFA, *E. carbunculus* and *P. sieboldii*, and only with length ($P \leq 0.01$) and species richness data ($P < 0.01$). Mean predicted lengths of *E. carbunculus* were largest in soft-low habitats (532 ± 43 mm SE) and smaller in hard-high and hard-low habitats (392 ± 15 mm, 385 ± 17 mm). There were no length measurements taken in soft-high habitats for this species. The largest *P. sieboldii* were in soft-high habitats (379 ± 12 mm), followed by hard-high and hard-low (351 ± 3 mm, 342 ± 6 mm), with the smallest fish found in soft-low habitats (289 ± 12 mm). Habitat associations with species richness in this BRFA indicated that hard-high and hard-low habitat types had the highest mean number of target species in a single deployment (1.14 ± 0.06, 0.91 ± 0.04) followed by soft-low (0.52 ± 0.04) and then soft-high habitats (0.28 ± 0.01).

Sites with an intermediate duration of protection

The Penguin Bank and Makapu'u BRFAs were expanded from their original 1998 boundaries in 2007 to include previously unprotected areas. As a consequence of the blend of newly protected sections and those sections protected since 1998 inside these BRFAs, we defined the time span of protection intermediate compared to the others we tested. Protection of the area influenced relative fish abundance and fish length in both BRFAs. At Penguin Bank mean predicted MaxN for *E. coruscans* was higher inside the BRFA compared to outside, though this result was only marginally significant ($0.05 < P < 0.10$; Fig. 5a). Adjusted mean length was also higher in this BRFA for four species (*E. carbunculus, E. coruscans, P. filamentosus, P. sieboldii*) while the opposite was seen for *A. rutilans* (Fig. 2b). Higher percentages of mature fish were also noted inside compared to outside Penguin Bank BRFA for *E. coruscans, P. sieboldii* and *H. quernus*, while results were approximately even for *E. carbunculus* and opposite for *P. filamentosus*

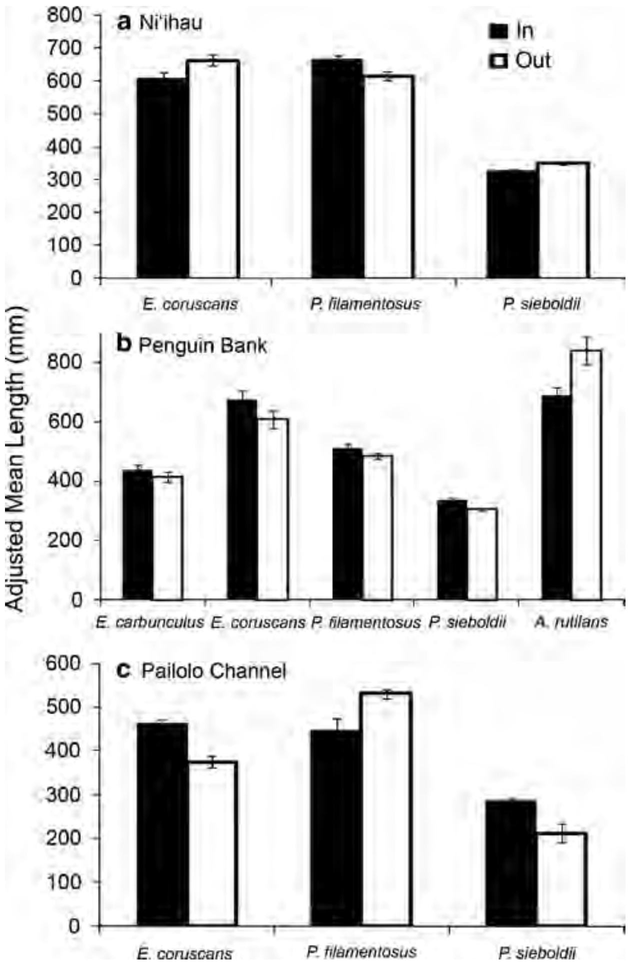


Fig. 2 Significant ($P < 0.05$) fish length model effects for the factor protection (differences in length inside and outside BRFAs; see Table 3). The genera for displayed species are *Etelis*, *Pristipomoides* and *Aphareus*

(Fig. 4b). Maturity results were also varied in Makapu'u BRFA, where more mature *E. coruscans* were inside the BRFA and more mature *E. carbunculus* were outside the BRFA (Fig. 4c). Furthermore, none of the measured *H. quernus* inside Penguin Bank BRFA or outside Makapu'u BRFA (*H. quernus* were only measured outside Makapu'u BRFA) were large enough to be male. Outside Penguin Bank BRFA, two of nine measured *H. quernus* were large enough to be male.

In both the Penguin Bank and Makapu'u BRFAs protection also influenced relative fish abundance and fish length for a few species over time. As such, there was a significant increase in *E. coruscans* mean predicted MaxN inside Penguin Bank BRFA, while outside MaxN remained unchanged (Fig. 6a). Although the difference in slope of these regressions (year*protection) was only marginally significant ($0.05 < P < 0.10$) in the top ranked model, in the second best model for the same species and BRFA, the slopes of these regressions were significantly different ($P_{\text{year*protection}} = 0.02$, $P_{\text{in}} = 0.03$, $P_{\text{out}} = 0.97$) and showed the same trends over time. For *P. filamentosus* in Makapu'u BRFA, the slopes of the MaxN regressions over time inside and outside the BRFA were significantly different with an increasing trend occurring inside and a decreasing trend occurring outside the BRFA (Fig. 6b). However, the individual regressions were not significant. Mean predicted lengths increased inside and decreased or showed no change over time outside Penguin Bank BRFA for *E. carbunculus*, *E. coruscans* and *P. filamentosus* (Fig. 3b–d). *P. sieboldii* displayed an opposite trend in Penguin Bank BRFA (Fig. 3e). In Makapu'u BRFA, mean fish length increased inside and showed no significant change outside for *P. filamentosus* over time (Fig. 3f).

Habitat associations with MaxN data varied by species and were only significant in Penguin Bank BRFA for *E. coruscans* ($P = 0.01$) and *E. carbunculus* ($P < 0.01$) and in Makapu'u BRFA for *P. filamentosus* ($P < 0.01$). *E. carbunculus* had the highest mean predicted MaxN in hard-high and soft-high habitats (2.99 ± 0.15 SE, 2.81 ± 0.21) followed by hard-low (1.78 ± 0.15) and then soft-low (0.12 ± 0.01), which had the lowest predicted MaxN. *E. coruscans* had the highest predicted mean MaxN in hard-high and hard-low habitats (3.89 ± 0.32, 4.13 ± 0.57) followed by soft-high (2.11 ± 0.29), then soft-low (0.64 ± 0.10). *P. filamentosus* had the highest predicted MaxN in hard-low and soft-low habitats (5.45 ± 0.52, 6.98 ± 1.07) followed by hard-high (2.78 ± 0.23) and then soft-high (1.02 ± 0.07). Habitat associations with length data were only significant for *P. filamentosus* in Penguin Bank and Makapu'u BRFAs and *P. sieboldii* in Penguin Bank BRFA. In both Penguin Bank and Makapu'u BRFAs, the largest *P. filamentosus* were in soft-high habitats (569 ± 11 mm, 582 ± 19 mm) followed by hard-high habitats (508 ± 11 mm, 523 ± 10 mm), while smaller *P. filamentosus* were in hard-low (470 ± 10 mm, 415 ± 8 mm) and soft-low habitats (441 ± 35 mm, 428 ± 26 mm). The largest *P. sieboldii* in Penguin Bank BRFA were in hard-high and hard-low habitats (330 ± 5 mm, 333 ± 8 mm) followed by soft-high habitats (301 ± 6 mm).

Site with the shortest duration of protection

Pailolo Channel BRFA was newly created in 2007 and thus had the shortest duration of protection compared to the others tested. In this BRFA, larger and more *E. coruscans* were found inside compared to outside the reserve (Figs. 2c, 5b). Larger *P. sieboldii* were also found inside this BRFA compared to outside, though the opposite was seen for *P. filamentosus* (Fig. 2c). Maturity results varied among species in Pailolo Channel BRFA, with more mature *E. coruscans*

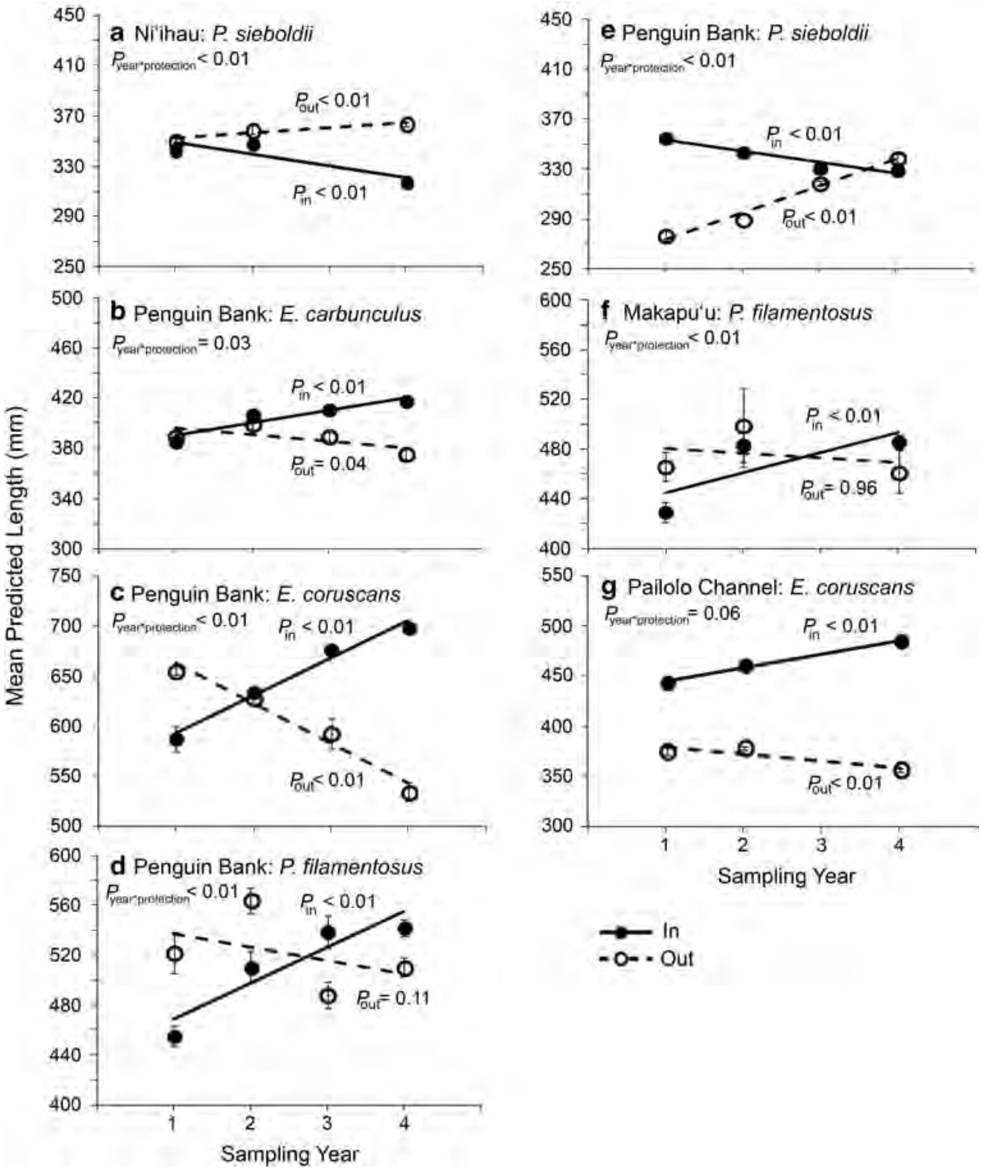


Fig. 3 Significant ($P < 0.05$) and marginally significant ($0.05 < P < 0.10$) fish length model effects for the interaction between year and protection (differences in the trends seen over time inside and outside BRFAs; see Table 3). The genera for displayed species are *Etelis* and *Pristipomoides*

and *P. sieboldii* inside the BRFA and more mature *E. carbunculus*, *P. filamentosus* and *H. quernus* outside (Fig. 4d). In addition, only one of eight measured *H. quernus* inside and only two of 26 outside this BRFA were large enough to be male. Over time, there was a slight trend of increasing fish length for *E. coruscans* inside the BRFA; however, the difference in the slopes of these regressions inside and outside the BRFA was only marginally significant ($0.05 < P < 0.10$; Fig. 3g).

Species richness in Pailolo Channel BRFA was significantly ($P < 0.01$) associated with habitat; hard-low habitats had higher species richness (1.59 ± 0.01 SE) than soft-low habitats (0.38 ± 0.01). Fish length and relative abundance were not significantly associated with habitat for any species in this BRFA, though only two habitat types were present (hard-low and soft-low).

Discussion

Our study used data collected inside and outside of four deepwater MPAs (BRFAs) in the Main Hawaiian Islands and provided evidence that this strategy can benefit deepwater fish populations similar to shallow water MPAs. For instance, studies on the efficacy of shallow water MPAs have demonstrated increased length and abundance of targeted fish inside protected areas relative to areas that remained open to fishing (Russ and Alcala 1996; Friedlander et al. 2003). We show that mean fish length, and in some cases abundance, increased for one or more of the most economically important deepwater bottomfish species (*E. coruscans*, *E. carbunculus* and *P. filamentosus*) inside nearly all tested deepwater BRFAs (Table 4). However, the strength and number of significant protection effects

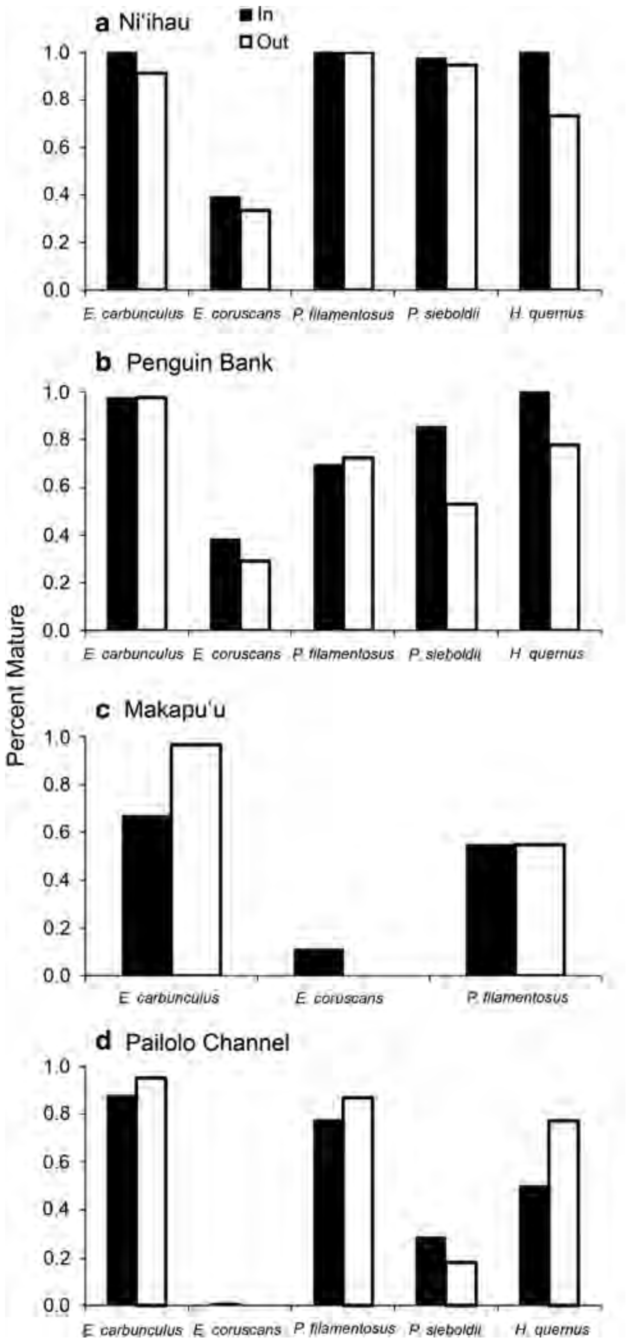


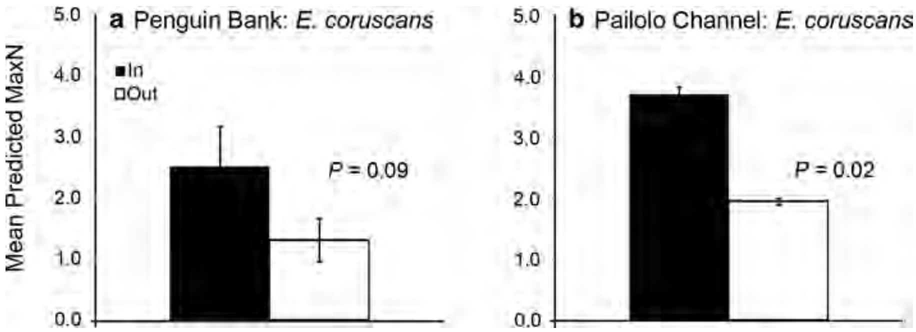
Fig. 4 The percentage of mature fish inside and outside BRFAs for each species with enough data to make a comparison. Sizes at maturity were *Etelis carbunculus* = 279 mm (DeMartini and Lau 1999), *E. coruscans* = 700 mm (Everson et al. 1989), *Pristipomoides filamentosus* = 450 mm (Ralston and Miyamoto 1983), *P. sieboldii* = 290 mm (DeMartini and Lau 1999) and *Hyporthodus quernus* = 580 mm (DeMartini et al. 2011)

varied among BRFAs and were likely a consequence of the time span of protection, potential poaching inside some BRFAs, small sample sizes for length and MaxN data and the inherent variability of MaxN data (i.e., hyperdispersed count data with excess zeros; Martinez et al. 2011; Smith et al. 2012). For instance, only 86 *H. quernus* were measured in all four BRFAs over 4 years of data collection and even more, only six of those were large enough to be male (>895 mm; DeMartini et al. 2011)). Despite these effects, our results suggest that a spatial management strategy such as a network of MPAs can benefit deepwater fish populations, many of which are in need of sustainable fisheries management (Haedrich et al. 2001; Morato et al. 2006; Baker et al. 2009; Williams et al. 2012, 2013).

Typical reserve effects (e.g., increased fish length and abundance inside the MPA; White and Kendall 2007; White et al. 2008; Lester et al. 2009; Gaines et al. 2010) were most often present in Penguin Bank and Makapu'u BRFAs. These BRFAs were intermediate in the duration of protection compared to the others tested because they included a blend of newly protected habitat (starting in 2007) and habitat that had been protected since 1998 (Moffitt et al. 2006; Kelley and Moriwake 2012). For instance, the three most economically important species in the fishery (*E. carbunculus*, *E. coruscans* and *P. filamentosus*; Haight et al. 1993; Kelley et al. 2006) were larger inside Penguin Bank BRFA compared to outside the reserve, with increases in length occurring over time inside and decreases or no change observed outside the BRFA. Further, increases in fish size seen over time were equivalent to approximately 1–3 years of growth (Smith and Kostlan 1991; Williams and Lowe 1997; Andrews et al. 2012). *E. coruscans* relative abundance also increased inside and showed no change over time outside the BRFA. Similar results were evident in Makapu'u BRFA, but only for a single species, *P. filamentosus*. The lack of significant length and abundance results for other species in Makapu'u BRFA was likely due to very low sample sizes (Table 4). This area has been anecdotally labeled as a fishing ground for *P. filamentosus* and *E. coruscans* by local anglers and, as expected, *P. filamentosus* was the most abundant species in this area while unexpectedly *E. coruscans* were rarely sampled. Conversely, *P. sieboldii*, which are generally not targeted by commercial fishers because of their small body size (Kelley et al. 2006), showed the opposite trend in Penguin Bank BRFA, increases in fish length outside and decreases inside the BRFA over time. The decline in mean length seen inside this BRFA may be the result of larger target species outcompeting this smaller non-target species inside the BRFA (Sanchez Lizaso et al. 2000).

While decreased fish length in fished areas adjacent to Penguin Bank BRFA for *E. carbunculus* and *E. coruscans* may indicate displaced fishing effort, fish abundance did not change over time outside the reserve. In previous studies, evidence of displaced fishing effort included decreases in fish catch and abundance adjacent to the MPA (Greenstreet et al. 2009; Halpern et al. 2004). Decreased mean

Fig. 5 Significant ($P < 0.05$) and marginally significant ($0.05 < P < 0.10$) MaxN model effects for the factor protection (differences in MaxN inside and outside BRFAs; see Table 2). The genus for displayed species is *Etelis*



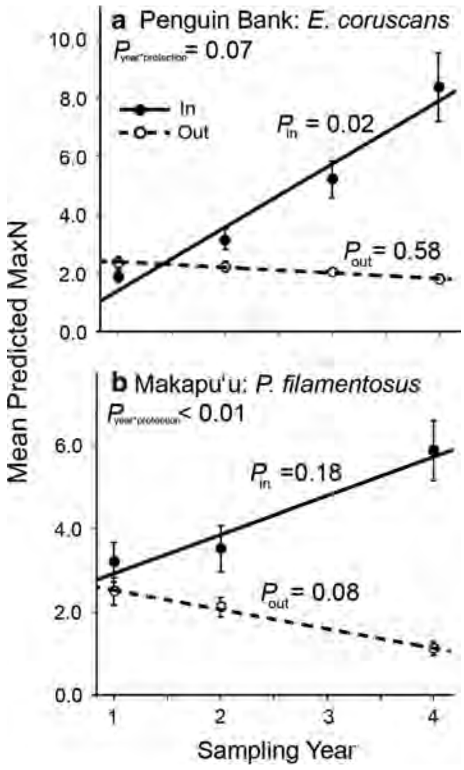


Fig. 6 Significant ($P < 0.05$) and marginally significant ($0.05 < P < 0.10$) MaxN model effects for the interaction between year and protection (differences in the trends seen over time inside and outside BRFAs; see Table 2). The genera for displayed species are *Etelis* and *Pristipomoides*

fish length in fished areas could also be from the larger and more productive spawning stock inside the reserve increasing recruitment to fished areas along with continual fishing pressure selectively removing large individuals from the population (Bohnsack 1994; Halpern and Warner 2003; Pelc et al. 2010). In keeping with this hypothesis, Vaz et al. (in review) modeled egg dispersal of Deep 7 species inside the BRFAs among the Main Hawaiian Islands and showed the majority of eggs spawned in deepwater BRFAs would disperse to fished areas. In addition, Halpern et al. (2004) evaluated the affects of displaced fishing effort and reported that exported production from reserves can supply and sustain fisheries at current or higher levels, compensating anglers for the closure of fishing grounds (Pelc et al. 2010). While this is clearly an objective of using MPAs for fisheries management, we currently do not have the data necessary to evaluate exported production from the BRFAs.

Consistent and significant positive effects from protection were seen less often in Pailolo Channel BRFA, as expected, given that this BRFA had the shortest period of protection in our study (from 2007). Although significant differences in mean fish length and relative abundance occurred inside compared to outside, results were mixed and many did not change significantly over time. Previous research has indicated that direct effects on target species as a result of protection first appear, on average, within 5 years (Babcock et al. 2010). Although life history data for our target species are limited, studies suggest they are generally long lived (20 to 40+ years for some species; Andrews et al. 2011, 2012; Kelley and Moriwake 2012), with slower growth than many reef fishes, such as those studied in Babcock et al. (2010). Consequently, it would likely take more than 4 years for protection effects demonstrated in other shallow water MPAs to develop for deepwater species (Haedrich et al. 2001; Morato et al. 2006; Baker et al. 2009). Another possible reason for inconsistent results among species in this BRFA compared to others was that only two types of habitat were available, hard-low and soft-low. The limited available habitat in this reserve could affect species composition compared to other BRFAs, particularly because both *E. carbunculus* and small *E. coruscans* prefer hard-low habitat types (Misa et al. 2013). Additionally, previous research has indicated that this area may be a nursery ground for *E. coruscans*, in which case changes in fish lengths may not be expected (Misa et al. 2013). As such, the slight increase in mean predicted fish length seen in this BRFA for *E. coruscans* did not exceed the estimated size at maturity (700 mm, Everson et al. 1989) and may indicate that fish leave this area when they reach maturity. Because there are no minimum size regulations for non-commercial fishers for any of the Deep 7, protecting nursery habitat should still provide benefits to the population and fishery.

The boundaries of Ni'ihau BRFA have changed very little since 1998, providing nearly 14 years of protection to this area. In accordance, our maturity results suggested that long-term protection in this BRFA had benefits not seen in other BRFAs. For instance, across all years each species in Ni'ihau BRFA had a higher percent of mature fish inside the reserve compared to outside (with the exception of *P. filamentosus* where all of the fish sampled inside and out were mature), a result not yet seen in the other analyzed BRFAs, which had shorter durations of protection. In addition, Niihau BRFA was the only area where four of the five species tested had >98 % maturity inside the BRFA. Further, this BRFA was the only one tested to show any changes in species richness over time, with increases occurring in adjacent fished habitats and no changes occurring inside Ni'ihau BRFA; a result that may indicate a spillover effect. A study by Russ and Alcala (1996) indicated that an increase in species richness of large predators, including Lutjanid and Serranid species, also increased with reserve age and suggested that the increase seen outside the reserve was due to spillover. However, they also hypothesized that increased species richness outside the reserve could be caused by successful recruitment. Other studies have indicated that initial signs of spillover are generally evident after approximately 15 years of protection (Abesamis and Russ 2005; Molloy et al. 2009). In addition, a previous examination of this BRFA to establish baseline data in the first year of monitoring (9 years after reserve creation) demonstrated that mean *E.coruscans* and *P. filamentosus* lengths were significantly larger inside the reserve; this difference was equivalent to ~10 years of growth for *P. filamentosus* (Moore et al. 2013). These results suggest that an increase in fish length had occurred for these species over the first 10 years of protection inside the reserve. Our results were consistent with this analysis, demonstrating that in the first year of monitoring there were larger *E. coruscans* and *P. filamentosus* inside the BRFA compared to outside; however, for *E. coruscans*, the opposite trend was seen when all 4 years were averaged together. This difference was likely due to a decline in mean fish length that seemed to occur inside and an increase that seemed to occur outside the reserve for *E. coruscans*. However, these trends over time were not significant ($P = 0.17$). Indeed, no changes in fish size or abundance over time were noted in this BRFA for any tested species, with the exception of *P. sieboldii*, a predominantly schooling nontarget species that many commercial anglers avoid due to their small body size (Kelley et al. 2006).

The lack of significant changes in fish size and relative abundance over time inside Ni'ihau BRFA may be the result of small sample sizes, and high variance in the data that increases in fish length and abundance has reached an asymptote, or due to an increase in poaching inside the BRFA in recent years. While other studies have indicated that an asymptote in fish length and abundance is often seen around 15 years of protection (Abesamis and Russ 2005; Molloy et al. 2009), small sample sizes may be a likely explanation for our lack of significant results. There were fewer length data for all but *P. sieboldii*, in Ni'ihau BRFA in comparison with situations where significant trends were found (N ~ 100; Table 4). In addition, the percent of nonzero data in datasets from Ni'ihau BRFA were <35 %, also lower than those that had significant results (~40 %; Table 4). However, previous research has demonstrated that using a GLMM with a negative binomial distribution, as we did, is likely to have reduced this possibility for our relative abundance data (Martinez et al. 2011; Smith et al. 2012). The lack of changes with time in Ni'ihau BRFA could also be due to a recent increase in anglers disregarding the boundaries of the BRFA and bottomfishing inside the reserve. While anecdotal evidence and a survey of fishers have suggested that poaching in BRFAs has occurred (Hospital and Beavers 2011), the local population near this BRFA is small and unlikely to have had a major impact on the bottomfish populations inside the BRFA. Ni'ihau is a small private island with a very small population of approximately 170 individuals living as ancestral Hawaiians did, without power boats or most modern technology (Hawaii State Data Center 2011). Instead, there is a relatively small community of non-local fishers using this area from neighboring islands. This community has changed during the years we monitored Ni'ihau BRFA with at least one commercial fisher that respected the boundaries of the BRFA retiring during this period. This BRFA is the most remote and therefore most difficult to monitor, so community enforcement is important. Further, previous research has demonstrated that even a small degree of fishing inside a MPA can result in little to no positive reserve effects (Denny and Babcock 2004; Shears et al. 2006). Despite this, positive effects of protection were demonstrated in Penguin Bank, Makapu'u and Pailolo Channel BRFAs, where poaching has been reported (Hospital and Beavers 2011).

Previously reported habitat associations for Hawaiian bottomfish have suggested they are generally found in hard bottom habitat types, particularly hard-high habitat (Parke 2007). In accordance with these finding, species richness was highest in hard habitat types in two of our research areas (Ni'ihau and Pailolo Channel BRFAs). However, significant relative abundance and mean fish length results were inconsistent with previously reported habitat associations in a few instances, and varied among BRFAs and species. For example, both juvenile and adult *E. carbunculus* have been reported to inhabit hard habitat types because they are smaller than most bottomfish and use rocky substrate as cover from predation (Kelley et al. 2006; Misa et al. 2013). While the smallest *E. carbunculus* were found in hard habitat types in Ni'ihau BRFA, the largest individuals were found in soft-low habitats. A previous study by Misa et al. (2013) into habitat associations for Hawaiian bottomfish showed no ontogenetic shifts in habitat preference for *E. carbunculus*, however, very few juveniles were observed. Conversely, habitat associations for *P. sieboldii* were generally in agreement with previous research. Misa et al. (2013) suggested *P. sieboldii* have no significant habitat preferences, possibly as a result of their reliance on schooling rather than habitat for protection against predators. Our results were in agreement as *P. sieboldii* had different habitat preferences among Ni'ihau and Penguin Bank BRFAs. Lastly, most *P. filamentosus* in Makapu'u BRFA were found in low habitat types with the largest fish in soft-high habitats followed by hard-high habitats in both Penguin Bank and Makapu'u BRFAs. Soft-low habitat preferences have been reported for juvenile *P. filamentosus* (Moffitt and Parrish 1996, Parrish et al. 1997) and an ontogenetic shift from soft-low to hard-low (transition) and finally hard-high habitat was also recently reported to occur as fish size increases (Misa et al. 2013). Our results, therefore, suggested that many *P. filamentosus* from these areas (Penguin Bank and Makapu'u BRFAs) were juveniles. However, the largest *P. filamentosus* were associated with soft-high habitats, which was inconsistent with previously reported patterns. Overall, our results indicate that these species are responding to their habitat in a more complex manner than previously thought, that there are species-specific differences in habitat preferences and that ontogenetic shifts in habitat preferences are occurring for many species as proposed (Misa et al. 2013). Habitat classifications used here were broad, based on the dominant habitat within a 200 m^2 grid, to classify the area of bait attraction for target species (Moore et al. 2013). While this approach was the most appropriate with our current understanding of species–habitat associations and proved useful in establishing protection effects, a more detailed understanding of species–habitat associations is needed.

Conclusions

Differences among our BRFAs were likely influenced by the age of the BRFA. For instance, the oldest BRFA (Ni'ihau, protected approximately 14 years) showed more mature fish inside compared to outside the reserve for each species examined, and species richness in adjacent fished habitats increased while remaining unchanged inside the reserve, possibly due to spillover. Those with an intermediate duration of protection (Penguin Bank and Makapu'u) had positive protection effects (i.e., increases in mean fish lengths and relative abundance), and the youngest BRFA (Pailolo Channel, protected approximately 4 years) showed little change over the duration of protection. Similarly, Molloy et al. (2009), reported that protection effects were positive but less reliable in "new" reserves (<5 years), young reserves (5 and 10 years) showed positive effects, established reserves (10–15 years) showed no change in relative fish density, and old reserves (>15 years) showed the most benefits to protection (consistently higher fish densities inside the reserve and overall relative fish densities reliably increased at ~5 % per annum). Our results follow this trend assuming Penguin Bank and Ni'ihau BRFAs were equivalent to approximately 5–10 years of protection. These authors and others also suggest that at least 15 years of protection are necessary to see reliable benefits of protection (Molloy et al. 2009; Russ and Alcala 2010). Accordingly, our data suggested that it may take a decade or more for target species to reach an equilibrium and spillover into adjacent fished areas. Importantly, though differences among BRFAs may be due to differences in the age of each BRFA, the degree of compliance among local anglers, initial fish population size inside the reserve, degree of fishing mortality reduction inside the reserve, local human population size, the extent of nearby fishing pressure, differences in fish biology, differences in habitat and smaller-scale habitat preferences may have also influenced our results (Mosqueira et al. 2000; Tetreault and Ambrose 2007; Gaines et al. 2010). Nonetheless, the predominant finding of larger more mature fishes inside the BRFAs and increases in abundance and size inside versus outside of these zones strongly suggests that the BRFAs can benefit Hawai'i's deepwater fish populations. While the results of this study are unique and provide evidence of the potential success of MPAs for deepwater species, additional data over greater temporal scales will be necessary to determine whether these trends will continue and if others will become important over time (Molloy et al. 2009). Deep water habitats and species are notoriously difficult to study and scientists' knowledge of deep-sea fish stocks frequently lags far behind fisheries exploitation (Haedrich et al. 2001). Using the precautionary principle, Lauck et al. (1998) suggested that MPAs are the best solution to protect fishery resources and enhance long-term sustainability in the face of data deficient and uncertain traditional management approaches. A network of moderately sized reserves that protect a diverse complex of species, as seen here and in other studies, may be the best strategy for deepwater species (Halpern and Warner 2003; Gaines et al. 2010).

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Conflict of interest The authors declare that they have no conflict of interest.

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Deepwater marine protected areas of the main Hawaiian Islands: establishing baselines for commercially valuable bottomfish populations

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ABSTRACT: This study provides the first comprehensive fishery-independent baseline assessment of commercially important deepwater bottomfish populations across the main Hawaiian Islands. Differences in bottomfish relative abundance and size distribution were evaluated for 6 deepwater Bottomfish Restricted Fishing Areas (BRFAs). While no differences were detected in species relative abundance, evaluation of size-frequency distributions found the 2 most commercially valuable species (*Etelis coruscans* and *Pristipomoides filamentosus*) to be significantly larger inside the BRFA at Ni'ihau, located off the most remote of the main Hawaiian Islands. This BRFA is 1 of 2 ongoing BRFAs offering 10 yr of protection. This result highlighted the time it may take a long-lived and slow-growing species to show a detectable response to protection and that size distribution analyses can detect these more subtle changes. No positive effects of protection were detected for the second ongoing BRFA located off Hawai'i. Instead, 2 species (*P. filamentosus* and *P. sieboldii*) were significantly larger outside the BRFA. In contrast to Ni'ihau, the second BRFA established in 1998 originally included less preferred habitat and is next to the second largest port in Hawai'i, offering greater access, higher population pressure and more problematic enforcement. This study demonstrates that biological, sociological and environmental context must also be considered when interpreting the effectiveness of marine protected areas.

KEY WORDS: Hawaiian bottomfish fishery · Fishing · Fish length · Stereo video · Marine protected area

INTRODUCTION

Marine protected areas (MPAs) have been widely used as a tool to reduce overfishing and conserve biodiversity. Heavily exploited fish populations not only experience reduced abundance but also reduced age and size (Willis et al. 2000, Longhurst 2002, Stewart 2011). With increasing evidence that older, larger individuals produce more eggs and possibly better conditioned larvae than smaller individuals, removal is likely to influence population size and resilience through decreased reproductive potential (Berkeley et al. 2004, Green 2008). No-take MPAs can be used to protect targeted species by rebuilding reserves of individuals, particularly large individuals, to increase reproductive potential and resilience to fishing pressure. While there is growing evidence that MPAs can support higher abundance, biomass and size of harvested species (Russ & Alcala 1996, Halpern & Warner 2002, Willis et al. 2003), the ef-

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fectiveness of MPAs has also been found to vary considerably, both in direction and magnitude of response (Micheli et al. 2004, Russ et al. 2005). Recent reviews examining assessment procedures for MPAs have highlighted some important considerations often overlooked when analysing and interpreting results. These include a species' economic value, size and growth rate, time to maturity, habitat preference, depth range and behaviour (Claudet & Guidetti 2010, Claudet et al. 2010). Species with different life histories and ecological traits have been found to respond differently to reserve design and period of protection. For example, Claudet et al. (2010) found that period of protection strongly affected larger species but not small or medium-sized species. They attributed this difference to the longer period of time larger species often require to grow and mature. Another important consideration is the effect of habitat structure (Claudet & Guidetti 2010). MPAs often include complex and heterogeneous habitats, and the effect of these must be clearly separated from the effects of protection (García-Charton et al. 2004). Factors such as accessibility, population density and level of enforcement and compliance within an MPA will influence the effectiveness of the MPA (Lundquist & Granek 2005). The choice of sampling method and technique will also have a profound influence on the accuracy of the results and ecological questions answered (Thresher & Gunn 1986, Willis et al. 2000, Watson et al. 2010)

The Hawaiian Islands bottomfish fishery is the second most important fishery in Hawai'i and targets a multispecies group of deep-sea fish (down to 400 m) including snappers (Lutjanidae), jacks (Carangidae) and an endemic grouper, *Hyporthodus quernus* (Epinephelidae). Most of the commercially important species have a relatively high age at maturity, long life span and slow growth rate, making them particularly susceptible to overfishing (Ralston & Polovina 1982, Haight et al. 1993). In 1998, following a steady decline in bottomfish catch rates and evidence that the 2 commercially valuable species *Etelis carbunculus* and *E. coruscans* may be overfished, the State of Hawai'i Department of Land and Natural Resources implemented 19 Bottomfish Restricted Fishing Areas (BRFAs) throughout the main Hawaiian Islands (State of Hawai'i 2006). These BRFAs were designed to protect 20 % of the essential fish habitat (EFH) for *E. carbunculus* and *E. coruscans* and to help replenish depleted bottomfish stocks by prohibiting bottom fishing within them, thus ensuring the long-term sustainability of the fishery (Parke 2007). EFH is defined as those waters and substrate necessary for fish spawning, feeding or growth to maturity (Rosenberg et al. 2000). Currently, the EFH definition is one that encompasses the suite of targeted bottomfish species and is defined as 'all bottom waters between 0 and 400 m'. Submersible observations and fishing effort have demonstrated that adult bottomfish often associate with higher profile substrates and/or rocky substrates (Polovina et al. 1985, WPRFMC 1998, Kelley et al. 2006). These results have since been used to further constrain preferred adult bottomfish habitat to hard bottom between 100 and 400 m with a slope of >20° (Kelley et al. 2006, Parke 2007). In 2005, with this increased understanding of bottomfish habitat requirements and the availability of multibeam and sidescan sonar mapping of the Hawaiian Islands, it was determined that only about 5 % of this type of habitat occurred within the boundaries of the BRFAs (Parke 2007). In addition, the Pacific Islands Fisheries Science Centre (PIFSC) established that bottomfish continued to be overfished in the main Hawaiian Islands and that greater protection was necessary (Moffitt et al. 2006). This led to the implementation of additional bottom fishing restrictions including a 6 mo seasonal closure, reduced non-commercial bag limits, mandatory permits, vessel marking and a revised system of BRFAs that came into effect on June 1, 2007. The new system of BRFAs reduced the overall number of BRFAs to 12 but increased the area protected to include more bottomfish habitat. The present study presents fishery-independent data collected during the first year of implementation of the revised system of BRFAs as part of an ongoing monitoring program designed to assess the effectiveness of these BRFAs.

Evaluating the effectiveness of MPAs to protect targeted fish species requires careful choice of sampling technique and/or gear used to minimise potential biases (Willis & Babcock 2000, Watson et al. 2005). The chosen technique must be non-destructive, accurate and efficient (i.e. cost and time) but also appropriate for the species of interest and question being addressed (Thresher & Gunn 1986, Willis et al. 2000). Biases can include selectivity associated with the gear (e.g. gear attraction and avoidance, observer bias and size selectivity) or the biology or behaviour of the fish including size, detectability, habitat association and mobility (Willis et al. 2000, Watson et al. 2010). Most surveys of MPAs have used SCUBA-based underwater visual census (UVC) methods (Russ & Alcala 1996, Cappo et al. 2003). However, UVC is restricted by depths accessible to divers (<20 m), may be affected by observer bias and can be unreliable for behaviourally adaptable species (i.e. diver attraction and avoidance) (Cole 1994, Willis & Babcock 2000). Few non-destructive survey techniques are available to assess fish populations beyond the limits of diver-based UVCs. Some underwater visual surveys have been conducted using submersibles and remotely operated vehicles (Ralston et al. 1986, Adams et al. 1995, Trenkel et al. 2004). However, these techniques can be costly and time consuming and have produced variable and biased population density estimates as a result of avoidance or attraction to moving equipment and/or artificial lights (Ralston et al. 1986, Trenkel et al. 2004). Underwater 'video fishing' is an alternative non-destructive sampling technique developed to combine the advantages of underwater visual survey and extractive fishing techniques (e.g. selective trapping, hook and line fishing or trawling) (Willis & Babcock 2000, Willis et al. 2000, Cappo et al. 2003). While avoiding many of the biases and selectivity associated with alternative techniques, baited underwater video also has its limitations, including the reliance on good visibility and an unknown area sampled by the bait. Comparative research has demonstrated baited underwater video stations to provide a consistent and comparable method for assessing fish relative abundance (Willis et al. 2000, Cappo et al. 2004, Watson et al. 2005, Harvey et al. 2007). However, the distance over which fish may be attracted to the bait will remain unknown until the development of a robust method for modelling dispersal and distance of attraction to the bait (Priede & Merrett 1996, Harvey et al. 2007).

Much research has been undertaken to critically examine the biases and limitations of techniques assessing fish populations (e.g. Ralston et al. 1986, Thresher & Gunn 1986, Willis et al. 2000, Cappo et al. 2004, Watson et al. 2005). The major finding of this comparative research has been that different techniques can survey significantly different components of the fish fauna, and authors have cautioned researchers to choose a sampling technique that is appropriate for the species of interest (Willis et al. 2000, Cappo et al. 2004). For example, Cappo et al. (2004) compared baited remote underwater video stations (BRUVs) with prawn (shrimp) trawls and found that the trawls captured significantly more small, sedentary or cryptic species, while the BRUVs recorded a greater number of larger mobile species from a much wider size range of families. Watson et al. (2010) also established that BRUVs recorded a greater number of large-bodied targeted species in higher abundance when compared with diver-operated stereo video. Harvey et al. (2007) found the number and diversity of fish to be greater and more consistent from baited versus unbaited video stations and, as a result, suggested that this technique may enable stronger statistical testing of the relative abundance of large predatory species in particular. For our research, it was decided that baited underwater video was the most appropriate and efficient non-destructive survey method for assessing targeted Hawaiian bottomfish. We used a baited stereo-video camera system, the bottom camera bait station (BotCam), designed specifically as a non-destructive fishery-independent tool for monitoring deepwater Hawaiian bottomfish and their habitat (Merritt et al. 2011). The advantage of stereo video over a single camera is the ability to make precise and accurate length measurements of fish to provide information on size-frequency distributions (Harvey & Shortis 1996, Harvey et al. 2002).

The focus of this research was to provide the first comprehensive fishery-independent baseline assessment of commercially important bottomfish populations within the main Hawaiian Islands across the revised system of BRFAs. Data collected during the first year after their implementation were evaluated to establish baseline bottomfish relative abundance and size-frequency distributions both inside and outside 6 of the 12 redefined BRFAs. The aim of the research was to test whether there was a significant difference in mean relative abundance or size of harvested species inside versus outside the BRFAs.

MATERIALS AND METHODS

Study site

The research was conducted across 6 of the 12 redefined BRFAs within the main Hawaiian Islands (Fig. 1). In some instances, the original regions closed to bottom fishing in 1998 overlapped with the redefined BRFAs implemented in 2007 offering continued protection, while closed areas not encompassed within 2007 areas became open to fishing in 2007. Two BRFAs were chosen to represent areas of continued closure (BRFA B, Ni'ihau and BRFA L, Hawai'i), 2 BFRAs were chosen because they were newly closed areas (BRFA D, west O'ahu and BRFA H, Pailolo Channel) and 2 represented BRFAs that encompassed smaller pre-existing closed areas (BRFA E, east O'ahu and BRFA F, Penguin Bank). Paired sampling was conducted inside and outside each of the BRFAs from May 2007 to July 2008 between depths of 100 and 300 m.

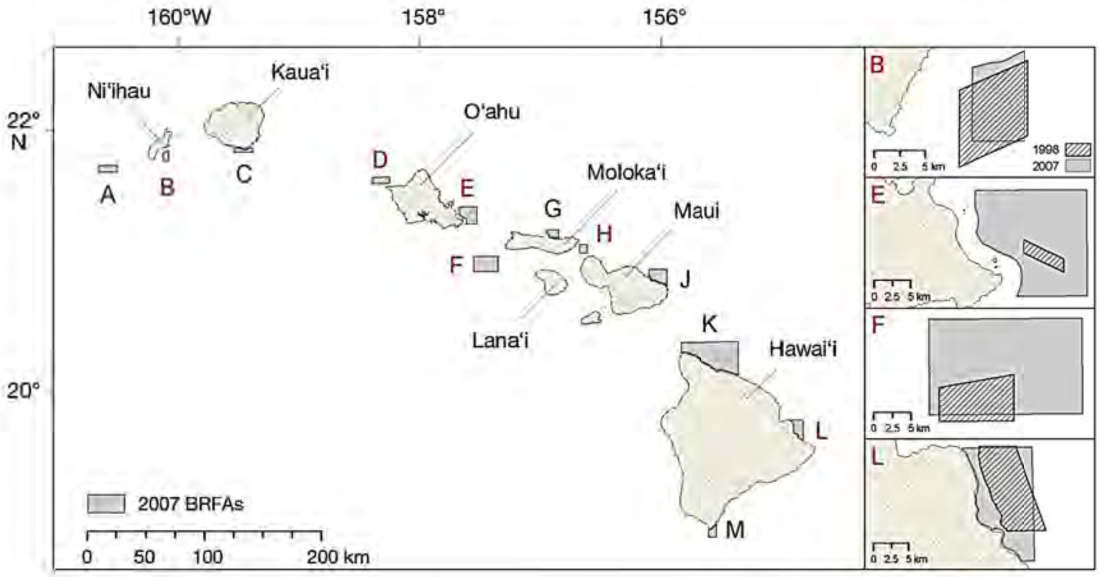

Fig. 1. Map of the main Hawaiian Islands showing location of revised Bottomfish Restricted Fishing Areas (BRFAs) implemented in 2007. Each BRFA has been coded by the State of Hawai'i with a letter, proceeding from west to east. BRFAs surveyed are indicated by red lettering. Inserted to the right are enlarged maps of the 2 ongoing BRFAs (B and L) and the 2 BRFAs encompassing smaller pre-existing closed areas (E and F). Diagonal hatching indicates location of old 1998 BRFAs

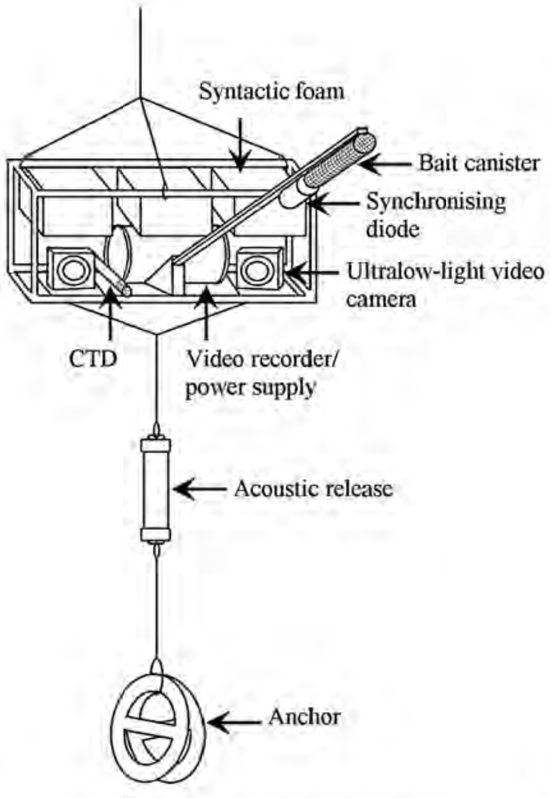

Fig. 2. Schematic of the bottom camera bait station (BotCam) used for assessing bottomfish relative abundance

Sampling technique

The baited stereo-video camera system used in the present study was the BotCam developed by the NOAA PIFSC in collaboration with the Hawai'i Undersea Research Laboratory (Fig. 2). The BotCam was designed specifically as a fishery-independent tool for monitoring Hawaiian deepwater fish and their habitat after comparative research by Ellis & DeMartini (1995) found that baited video stations provided an accurate tool for sampling juvenile *Pristipomoides filamentosus*. Detailed information on the design of the system can be found in Merritt et al. (2011), who found the redesigned baited video system to be a flexible and economic tool for assessing these deepwater bottomfish. The system consists of 2 ultralow-light monochrome video cameras used to record under ambient light conditions to a depth of 300 m. Ambient lighting is preferred, as artificial lights have been noted to repel bottomfish species (Ralston et al. 1986). Sampling was completed between 08:00 and 16:00 h to avoid crepuscular changes in fish behaviour and maximise available ambient light at depth. Attached in front of the video cameras is a light diode used to synchronise the stereo-video pair, enabling accurate fish length measurements. Also attached is a plastic mesh bait canister. Bait consisted of approximately 800 g of mixed chopped anchovy *Engraulis mordax* and market squid *Doryteuthis opalescens*, frozen to ensure it did not dissipate before reaching the seafloor. This bait was chosen to be consistent with that used by local bottom fishers. The system was moored to 2 anchor weights designed to rest on the seafloor, while syntactic foam blocks, attached to the camera frame, allowed the video cameras to float approximately 3 m off the bottom, optimising the field of view. The small footprint of the anchor allows the system to sit securely on high profile substrates, thus enabling it to record bottomfish in the steep and rugose terrain often favoured by these species. The system is retrieved via its surface-buoyed tether, the acoustic release used only if the anchor becomes lodged on the seafloor. Each system was deployed by boat and left to record on the seafloor for 45 min to optimise the number of fish observed with time (Harvey & Cappo 2001). Previous bait stations conducted from a submersible (Kelley & Ikehara 2006) found that 30 min was an adequate length of time to capture the peaking feeding activity stimulated by the bait (C. D. Kelley pers. comm.). To reduce the likelihood of an overlap in bait plumes and of sampling the same fish twice, concurrent deployed systems were placed a minimum of 400 m apart, with actual distances usually much greater (mean separation of 1938 ± 102 m). Ellis & DeMartini (1995) estimated the greatest distance of attraction to bait over a 10 min period was between 48 to 90 m based on maximum recorded bottom current speeds of between 0.1 to 0.2 m s^{-1} and a swimming speed for *P. filamentosus* of 0.6 m s^{-1}. Based on this calculation, the greatest distance of attraction over a 45 min period would be between 250 and 408 m.

Table 1. Number of samples collected within each substrate category for each of the 6 Bottomfish Restricted Fishing Areas (BRFAs). High: high profile (slope ≥20°); low: low profile (slope <20°)

BFRA	Location	Protection	Substrate				Total
			Reef		Sediment		
			High	Low	High	Low	
B	Ni'ihau	Yes	12	8	8	4	64
		No	12	8	8	4	
D	W. O'ahu	Yes	–	26	2	4	64
		No	12	8	8	4	
E	E. O'ahu	Yes	8	10	10	4	64
		No	12	8	8	4	
F	Penguin Bank	Yes	11	8	8	4	63
		No	12	8	8	4	
H	Pailolo Channel	Yes	–	27	–	4	62
		No	–	27	–	4	
L	Hawai'i	Yes	12	8	8	4	61
		No	9	8	8	4	

Sampling design

Fish were sampled both inside and outside each of the redefined BRFAs using a stratified random sampling protocol recommended by the PIFSC and the University of Miami. Each BRFA location was designated 64 replicates, with 32 replicates inside the BRFA and 32 outside. The outside, unprotected area adjacent to each BRFA was an area equivalent to the bottomfish habitat inside the BRFA (100 to 300 m) on either side of the BRFA, following the bathymetric contours. Multibeam bathymetry and backscatter data, collected at a resolution of 20 m, were used to classify habitat into 4 categories: high or low profile rocky substrates (hereafter called 'reef' for simplicity) and high or low profile sediment. The multibeam data were used to designate the substrate as either high profile (slope ≥20°) or low profile (slope <20°), while the backscatter data were used to designate the substrate as either reef (consolidated hard substrate) or sediment (unconsolidated soft substrate). Habitat was categorised at a resolution of 200 × 200 m by assigning the habitat category comprising the majority of each 200 × 200 m grid cell. BotCam deployments were randomly designated within each of these 4 habitat categories, with replicates weighted towards preferred bottomfish habitat to ensure greater replication where fish were expected to be found (detailed in Table 1). Where some habitat categories were not present, particularly at west O'ahu (D) and Pailolo Channel (H), additional replicates were allocated to the next most preferred bottomfish habitat as dictated by previous studies, which found that adult bottomfish often associate with rocky substrates and/or higher profile substrates (Polovina et al. 1985, WPRFMC 1998). The sampling protocol resulted in some differences in depths sampled, with the mean sample depth between each BRFA and its control site varying between 5 and 31 m.

Image analysis

Video data were analysed to measure bottomfish relative abundance and size-frequency distribution. Relative abundance was quantified as the maximum number (MaxN) of each species observed in a single frame during the entire 45 min video (Priede et al. 1994, Cappo et al. 2004). Individuals that could not be identified to species were identified at the highest taxonomic resolution possible. Schools of fish exceeding 50 individuals were rounded to the nearest increment of 5, with an exact number unrealistic with so many individuals moving in and around one another. Where individual fish were observed in both cameras, fork lengths were measured using 1 of 2 stereo-photogrammetric systems, initially Vision Measurement System (VMS) v.7.5 (Geometric Software) and then PhotoMeasure v.1.74 (SeaGIS), which replaced VMS. To avoid repeated measures of the same fish, measurements were made at a single time in the video where the relative abundance of the fish was maximised. Replicate measurements of each individual fish were taken where possible to increase precision and accuracy of the measurement. This was particularly important in this study, as the low light conditions affected the resolution of the video imagery and, in turn, the accuracy of length measurements. Any length measures with a residual parallax and a RMSE of >10 mm and a ratio of precision of measurement to length of fish of >10% were removed from the analysis.

Measurements were taken on average 1.9 ± 0. 98 m from the cameras, with a maximum distance of 8.2 m. The visual area sampled was noted to expand and contract due to the nature of working at these depths and changes in visibility. Based on these measurements and using minimum and maximum view distances 2 to 10 m from the cameras, the visual area sampled was estimated to be between 4 and 416 m^2.

Data analysis

Eight bottomfish species were examined: *Aphareus rutilans*, *Etelis carbunculus*, *E. coruscans*, *Hyporthodus quernus*, *Pristipomoides filamentosus*, *P. sieboldii*, *P. zonatus* and *Seriola dumerili*. The first 7 species, locally referred to as the 'Deep 7', were chosen because they are the most commercially and recreationally valuable species. For simplicity, we refer to these 7 species as the 'Deep 7' throughout this paper. The eighth species, *S. dumerili*, was also included, as it was once a valuable component of the fishery and is now the most important bycatch species (WPRFMC 1998). Two of the Deep 7, *A. rutilans* and *P. zonatus*, were detected infrequently in very low relative abundances and could not be included in all statistical tests (species-specific permutational multivariate ANOVA [PERMANOVA] or examination of length frequency, see below). A PERMANOVA (Anderson 2001) was chosen for the analyses, as the experimental design was unbalanced and the relative abundance of fish was highly skewed with many zero counts. The PERMANOVA was conducted for the Deep 7 assemblage and 6 of the 8 individual species using the PERMANOVA+ software in PRIMER 6 v.6.1.11, with the number of permutations set to 9999 (Anderson et al. 2008). The data were first fourth-root transformed to downweight the effect of large schools of fish recorded, and then the dissimilarity matrix was built using Bray-Curtis dissimilarities, which are appropriate for this type of ecological data (Bray & Curtis 1957). The sampling design consisted of 2 fixed and crossed factors: closure (2 levels) and substrate (4 levels). Paired sampling was conducted at the same time inside versus outside each BRFA to ensure that we controlled for time of year. However, the effect of BRFA location was not tested, as data were collected from each BRFA location at differing times during the year due to logistical constraints. Therefore, temporal changes in species distributions and relative abundances, such as species migratory patterns and spawning aggregations, could not be accounted for between BRFA locations.

Size-frequency distributions of individual species were tested using Kolmogorov-Smirnov tests to determine whether they differed inside versus outside the BRFAs. This test was only possible where >10 length measurements were available for a given species both inside and outside the BRFA. Significant differences in the size-frequency distributions inside versus outside the BRFAs are presented graphically. The Wilcoxon 2-sample test was used for smaller sample sizes ($4 \leq n < 10$). This test is based on rank order and is appropriate for small sample sizes with unknown distributions (Wilcoxon 1945).

RESULTS

Relative abundance

Analysis using PERMANOVA, examining the effect of closure and substrate on harvested species, found no significant difference in relative abundance between open and closed areas for the Deep 7 assemblage or the 6 individual species (Table 2). However, several significant responses to substrate were detected for the Deep 7 assemblage and for some individual species. The Deep 7 had a significant response to substrate at the BRFAs E, F, H and L,

Table 2. Differences in relative abundance for the Deep 7 assemblage and individual harvested bottomfish species for each of the 6 Bottomfish Restricted Fishing Areas in response to the factors closure (C), substrate (S) and their interaction (C × S). *E.*: *Etelis*; *H.*: *Hyporthodus*; *P.*: *Pristipomoides*; *S.*: *Seriola*. *0.05 > p > 0.01; **0.01 > p > 0.001; ***p < 0.001

	B - Ni‘ihau							**D - west O‘ahu**						
	C (df = 1)		S (df = 3)		C × S (df = 3)		Error	C (df = 1)		S (df = 3)		C × S (df = 2)[a]		Error
	MS	*F*	MS	*F*	MS	*F*	MS	MS	*F*	MS	*F*	MS	*F*	MS
E. carbunculus	36.1	0.77	52.6	1.13	52.6	1.13	46.7	37	0.5	59.7	0.8	72.5	0.97	74.7
E. coruscans	1.6	0.01	255.3	1.38	74.2	0.4	185.1	5.7	0.14	5.6	0.14	8.4	0.21	40.5
H. quernus	211.5	1.73	168.9	1.38	82.2	0.67	122.2	1.3	0.03	19.2	0.53	1.9	0.05	36.6
P. filamentosus	16.5	0.07	312.7	1.27	48.3	0.20	246.3	3.2	0.01	363.4	1.48	482.5	1.97	245.5
P. sieboldii	719.5	2.39	581.6	1.93	256.9	0.85	301.6	29.3	0.35	30.2	0.36	25.2	0.3	83.1
S. dumerili	0.46	0.00	359.0	1.14	124.0	0.39	315.7	4.4	0.02	205.1	0.88	364.4	1.56	233.7
Deep 7	1.57	1.12	2.46	1.74	0.96	0.68	1.41	0.15	0.23	0.52	0.80	0.71	0.10	0.64
	E - east O‘ahu							**F - Penguin Bank**						
	C (df = 1)		S (df = 3)		C × S (df = 3)		Error	C (df = 1)		S (df = 3)		C × S (df = 3)		Error
	MS	*F*	MS	*F*	MS	*F*	MS	MS	*F*	MS	*F*	MS	*F*	MS
E. carbunculus	299	2.29	22.4	0.17	34.9	0.27	130.7	5.6	0.02	571	1.85	435.3	1.41	308.8
E. coruscans	1.7	0.01	85.6	0.64	331.1	2.48	133.6	20.2	0.07	766.7	2.78*	115.3	0.42	276.1
H. quernus	67.7	1.33	87	1.71	87	1.71	51	6.6	0.12	55.9	1.04	9.4	0.17	53.9
P. filamentosus	64.4	0.18	1614	4.50**	262.3	0.73	359.1	1.2	0.00	276.2	0.67	264.2	0.64	410.1
P. sieboldii	218.4	1.49	117.9	0.8	20.9	0.14	146.6	36.3	0.12	1342	4.43**	1054	3.49*	302.2
S. dumerili	736.6	2.92	578.8	2.29	523.8	2.08	252.3	682.7	2.83	215.0	0.89	126.4	0.52	241.6
Deep 7	0.76	0.71	2.48	2.33*	0.85	0.80	1.06	0.17	0.08	4.26	2.08*	3.03	1.48	2.04
	H - Pailolo Channel							**L - Hawai‘i**						
	C (df = 1)		S (df = 3)		C × S (df = 3)		Error	C (df = 1)		S (df = 3)		C × S (df = 3)		Error
	MS	*F*	MS	*F*	MS	*F*	MS	MS	*F*	MS	*F*	MS	*F*	MS
E. carbunculus	140.3	0.43	5359	16.2***	140.3	0.43	329.5	0.4	0.002	472.9	2.19	436.2	2.02	216.3
E. coruscans	210.6	0.59	2029	5.72*	210.6	0.59	354.4	192.2	1.82	137.7	1.3	73.5	0.69	105.8
H. quernus	6.7	0.06	89.1	0.76	6.7	0.06	117.4	120.7	2.19	57.1	1.04	57.1	1.04	55.1
P. filamentosus	22.2	0.14	171.8	1.07	22.2	0.14	160.2	927.5	2.56	1412	3.89*	1154	3.18*	362.9
P. sieboldii	380.9	1.35	604.2	2.14	380.9	1.35	282.0	116.9	0.34	411.9	1.19	70.6	0.2	345.8
S. dumerili	0.36	0.00	178.7	0.77	583.7	2.51	232.5	435.2	2.88	49.5	0.33	9.0	0.06	151.1
Deep 7	1.06	0.65	9.89	6.07***	1.06	0.65	1.62	3.38	1.89	4.05	2.27*	2.32	1.30	1.78

[a]Term has one or more empty cells in the model

with pairwise comparisons establishing that there were significantly lower mean relative abundances detected over low profile sediments. Relative abundance over the remaining 3 substrate classes demonstrated an inconsistent preference for reef and/or high profile substrates (Fig. 3). Higher relative abundances of the Deep 7 assemblage were detected over reef and high profile substrates at BRFAs E and L, with significantly more over low profile reef than high profile sediment. In contrast, at BRFA F, while there were also higher relative abundances of the assemblage over reef and high profile substrates, there were significantly more over high profile sediment than low profile reef. Only 2 substrate categories were present and sampled at BRFA H. Here, significantly more Deep 7 species were found over low profile reef than low profile sediment. Deep 7 relative abundances were consistently low across all substrates at D, and relative abundances at BRFA B were highly variable, hence the difference between high profile reef and low profile sediment having a low but insignificant p-value of 0.066.

Examination of individual species responses to substrate (Table 2) revealed that *Pristipomoides filamentosus* had significantly higher relative abundance over low profile reef at BRFA E. *Etelis coruscans* had a significantly higher relative abundance over high profile reef at BRFA F and low profile reef at H (where high profile reef habitat was absent). All *E. carbunculus* at BRFA H were found over low profile reef while being absent over low profile sediment. Significant interactions between closure and substrate were detected for *P. sieboldii* at BRFA F and for *P. filamentosus* at L. These responses mirror those found for the Deep 7 assemblage and suggest that *P. filamentosus* was driving the significant response to substrate in the Deep 7 assemblage at BRFAs E and L, with a preference for high and low profile reef. The same could be true for *E. carbunculus* and *P. sieboldii*, with these species potentially

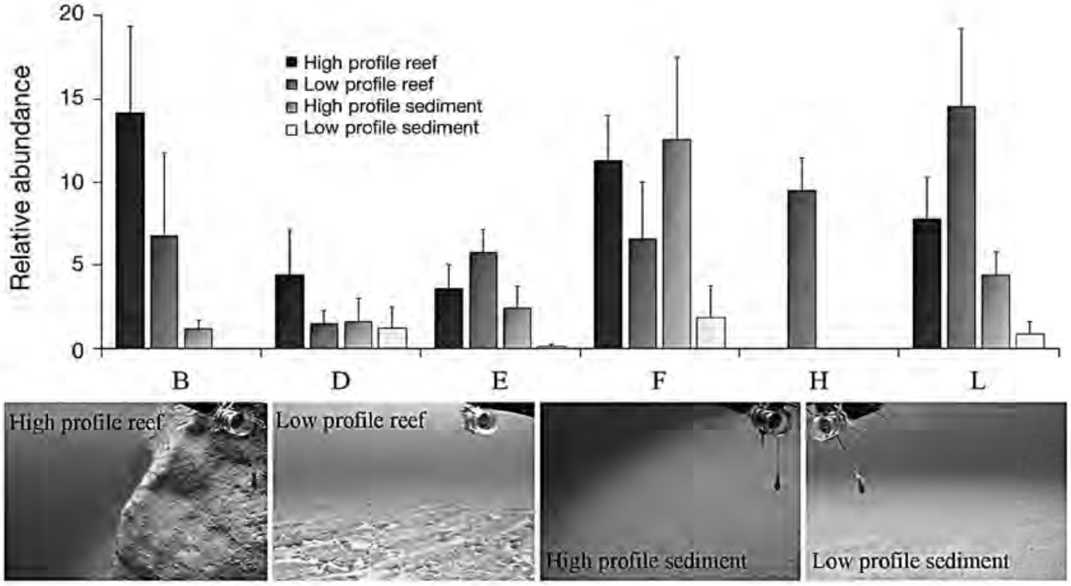

Fig. 3. Mean relative abundance (+1 SE) of deep 7 bottomfish species within each of the substrate categories from each of the 6 Bottomfish Restricted Fishing Areas (BRFAs): (see Fig. 1, Table 1). Note: Two substrate categories (high profile reef and high profile sediment) were absent from BRFA H. Example images, recorded by the bottom camera bait station (BotCam), are provided to illustrate the 4 substrate categories

driving the preference for high profile substrates, including both high profile reef and high profile sediment, at BRFA F. *E. carbunculus* and *E. coruscans* were possibly driving the preference for low profile reef at BRFA H.

Large variability in the relative abundance of the 8 species was observed across the 6 locations and between individual species (Fig. 4). This variability was related to the schooling behaviour of some of the species and their very patchy distribution. Large numbers (MaxN > 10) were recorded for 5 of the 8 harvested species; the exceptions were *Aphareus rutilans*, *Hyporthodus quernus* and *Pristipomoides zonatus*. The 2 largest individual schools were both recorded at B, with ~100 *P. sieboldii* recorded outside the BRFA and ~80 *Etelis coruscans* recorded inside the BRFA. When examining mean relative abundance, the largest MaxN and maximum prevalence (frequency of occurrence) across BRFA locations and between species, it was rare for these statistics to coincide (Fig. 4). Some general trends included some of the highest mean relative abundances and frequencies of occurrence inside the newly-closed BRFA H (i.e. *E. carbunculus*, *E. coruscans* and *P. filamentosus*) and some of the largest schools inside (*P. filamentosus* and *P. zonatus*) and outside (*A. rutilans* and *E. carbunculus*) the expanded BRFA F. In contrast, high mean relative abundance and high prevalence was recorded for several species outside the ongoing BRFA L (i.e. *A. rutilans*, *P. filamentosus* and *P. zonatus*). In general, low relative abundance and prevalence was recorded for all species off BRFAs D and E. The exception, *Seriola dumerili*, was found to have the highest mean relative abundance and largest MaxN inside the expanded BRFA E.

Size-frequency distributions

Statistical tests examining differences in the mean size (Wilcoxon test where $n < 10$) or the size-frequency distribution (Kolmogorov-Smirnov test where $n \geq 10$) of harvested species inside versus outside the BRFAs revealed some significant differences (Table 3; Fig. 5). *Pristipomoides filamentosus* and *Etelis coruscans* were both significantly larger inside BRFA B (Fig. 6). The mean fork lengths of *P. filamentosus* and *E. coruscans* inside the BRFA were 97.4 mm and 83.3 mm larger, respectively, than the mean fork length outside the BRFA. Individual *P. filamentosus* measured inside the BRFA were between 560 and 785 mm, while individuals outside the BRFA were between 464 and 679 mm. *E. coruscans* inside the BRFA were between 576 and 911 mm, while those outside the BRFA were between 574 and 825 mm. *P. sieboldii* was also found to have a significantly larger size-frequency distribution within BRFAs F and H. The mean fork

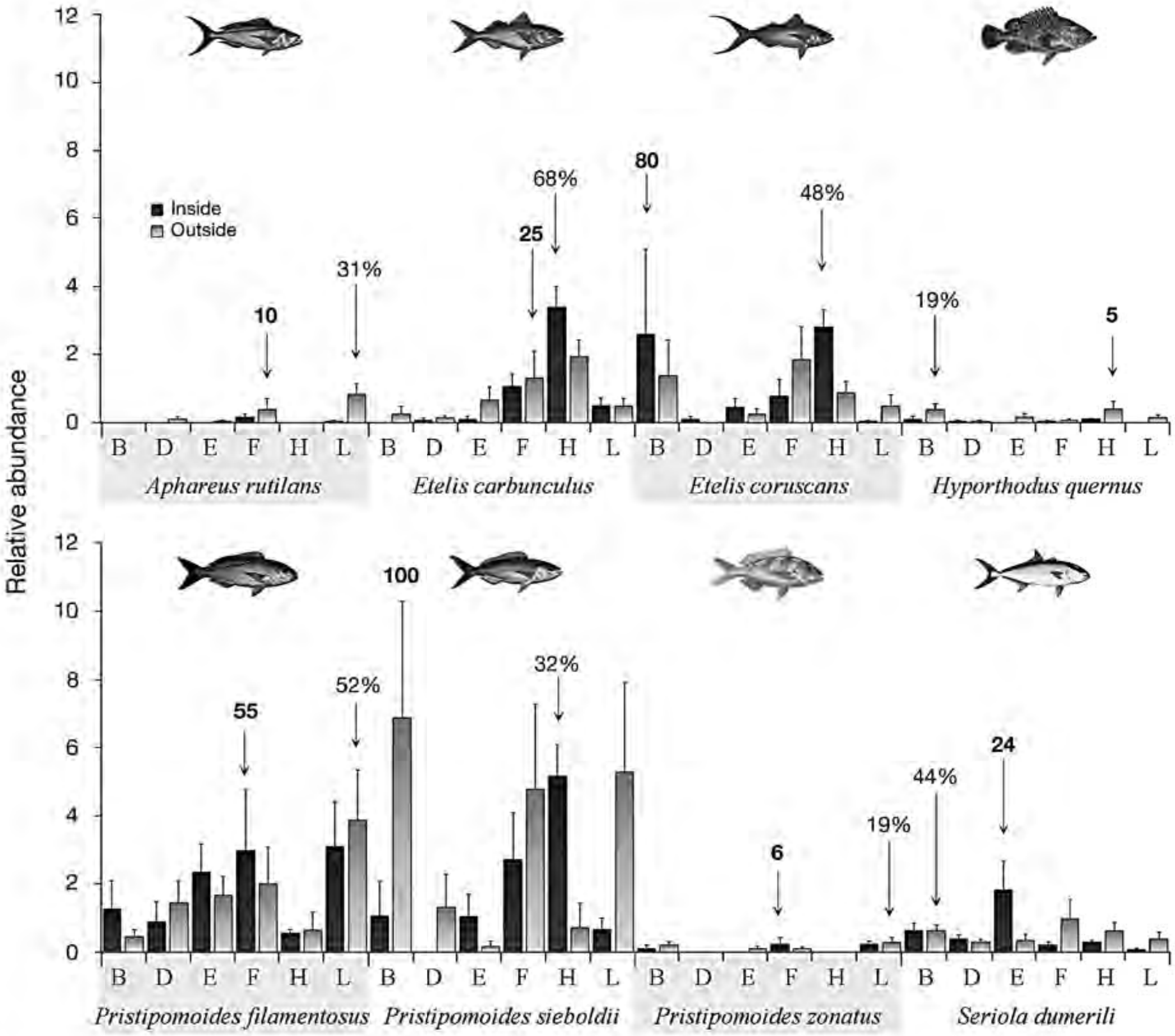

Fig. 4. Mean relative abundance (+1 SE) of harvested bottomfish species recorded inside and outside the 6 BRFAs (see Fig. 1, Table 1). Maximum relative abundance for each species is shown in **bold**, and percentages indicate maximum prevalence (percent of sites where species were recorded). Illustrations by Les Hata©, Hawaii Department of Land and Natural Resources

lengths of *P. sieboldii* inside the BRFAs were 111 mm larger in F and 79.7 mm larger in H than outside both BRFAs. *Seriola dumerili* were significantly larger within BRFA H, with a mean fork length 330.2 mm larger outside the BRFA. However, only 6 *S. dumerili* were measured within H.

A few species were found to have significantly larger individuals outside the BRFAs (Table 3; Fig. 5). The size-frequency distribution of *Pristipomoides filamentosus* at BRFA D revealed a distinctive split in the size distribution, with mostly juveniles within the BRFA and adults outside the BRFA (Fig. 6). *Etelis carbunculus* had a significantly larger size-frequency distribution outside BRFA H. Individuals outside H tended to be larger, in contrast a more even size-frequency distribution with equal representation of small, intermediate and larger individuals within H (Fig. 6). No species were found to have significantly larger size-frequency distributions within BRFA L, however, *P. filamentosus* and *P. sieboldii* were both found to be significantly larger (82.9 and 73.3 mm, respectively) outside L.

DISCUSSION

The present study found no increases in the relative abundance of Hawaiian bottomfish inside the 6 redefined BRFAs across the main Hawaiian Islands. This was as expected for the newly-established BRFAs and those BRFAs where a small fraction of the area had been previously protected. However, evalu-

Table 3. Mean fork length (mm) of harvested bottomfish species sampled within and outside the Bottomfish Restricted Fishing Areas (BRFAs). These were calculated using the Wilcoxon test (W) for small sample sizes or the Kolmogorov-Smirnov test (*D*) where 10 or more fish measures were available both inside and outside the BRFA. Values in **bold** are significant (<0.05); –: not tested as too few measurements

Species	Inside Mean ± SE (n)	Outside Mean ± SE (n)	Statistic W	Statistic *D*	p	Inside Mean ± SE (n)	Outside Mean ± SE (n)	Statistic W	Statistic *D*	p
	B - Ni'ihau					**D - west O'ahu**				
E. carbunculus	–	389.6 ± 25.5 (3)	–	–	–	349.0 ± 7.4 (2)	397.2 ± 23.8 (3)	–	–	–
E. coruscans	763.6 ± 24.2 (19)	680.3 ± 14.6 (23)	–	0.450	**<0.05**	546.9 ± 57.7 (3)	–	–	–	–
H. quernus	619.6 ± 34.6 (2)	752.5 ± 56.6 (6)	–	–	–	423.3 ± 0.0 (1)	481.6 ± 0.0 (1)	–	–	–
P. filamentosus	662.0 ± 12.1 (19)	564.6 ± 21.9 (12)	–	0.600	**<0.01**	459.7 ± 27.3 (17)	546.5 ± 17.4 (21)	0.570	**<0.01**	
P. sieboldii	355.2 ± 7.7 (14)	340.7 ± 5.1 (66)	–	0.240	0.45	–	236.4 ± 4.2 (29)	–	–	–
Seriola dumerili	797.2 ± 36.0 (15)	815.2 ± 37.6 (17)	–	0.320	0.317	710.2 ± 95.9 (8)	889.3 ± 94.6 (6)	52	–	0.414
	E - east O'ahu					**F - Penguin Bank**				
E. carbunculus	302.3 ± 25.1 (2)	404.1 ± 38.0 (7)	–	–	–	375.6 ± 15.6 (15)	386.8 ± 20.8 (19)	–	0.180	0.926
E. coruscans	599.6 ± 67.2 (6)	549.0 ± 0.0 (1)	–	–	–	634.2 ± 53.8 (4)	654.0 ± 25.0 (18)	42	–	0.770
H. quernus	–	472.0 ± 28.7 (2)	–	–	–	–	572.3 ± 138.1 (2)	–	–	–
P. filamentosus	403.2 ± 11.0 (43)	426.2 ± 15.7 (34)	0.153	0.728	404.8 ± 13.2 (54)	446.2 ± 24.2 (27)	–	0.190	0.524	
P. sieboldii	367.8 ± 14.3 (5)	–	–	–	–	331.8 ± 4.3 (58)	220.8 ± 6.8 (44)	–	0.900	**<0.001**
S. dumerili	623.4 ± 14.3 (46)	723.3 ± 42.0 (8)	311	–	**<0.05**	587.3 ± 90.5 (3)	690.4 ± 21.7 (26)	–	–	–
	H - Pailolo Channel					**L - Hawai'i**				
E. carbunculus	388.4 ± 10.1 (63)	446.6 ± 15.1 (24)	–	0.381	**<0.01**	380.9 ± 25.0 (11)	325.5 ± 19.4 (8)	61	–	0.127
E. coruscans	405.3 ± 16.2 (27)	332.8 ± 44.1 (11)	–	0.364	0.198	499.4 ± 0.0 (1)	705.4 ± 37.1 (9)	–	–	–
H. quernus	775.4 ± 210.7 (2)	613.0 ± 21.2 (5)	–	–	–	–	652.4 ± 31.4 (4)	–	–	–
P. filamentosus	489.1 ± 18.2 (7)	491.7 ± 28.6 (9)	64	–	0.681	249.5 ± 5.3 (86)	322.4 ± 13.3 (74)	–	0.420	**<0.001**
P. sieboldii	245.2 ± 9.5 (30)	165.5 ± 6.6 (7)	33	–	**<0.001**	218.8 ± 12.9 (12)	292.1 ± 5.5 (70)	–	0.730	**<0.001**
S. dumerili	952.7 ± 78.4 (6)	622.5 ± 15.7 (16)	110	–	**<0.01**	900.1 ± 0.0 (1)	949.0 ± 46.6 (11)	–	–	–

ation of size-frequency distributions revealed that the 2 most commercially valuable species, *Etelis coruscans* and *Pristipomoides filamentosus*, were significantly larger inside the ongoing BRFA B. When comparing the 97.4 mm increase in average size of *P. filamentosus* with age and growth information taken from Andrews et al. (2011), the larger fish inside BRFA B are estimated to be approximately 10 yr older than those in neighbouring unprotected areas. While we have no data from when before the BRFAs were declared and therefore cannot make any firm conclusions, this increase in size corresponds well with a 10 yr restriction on bottom fishing.

One key question managers of MPAs want answered is the time required for an MPA to be effective. For targeted species, managers need information on recovery rates i.e. how long will it take to see an increase in the target species' abundance and size? While some studies have detected increases in fish size and density in just 1 to 3 yr after protection (Roberts 1995, Halpern & Warner 2002), others have demonstrated that these results may take decades (Jennings 2000, Micheli et al. 2004, Russ & Alcala 2004). Differences, including size and age at sexual maturity, reproductive biology, diet, mobility and behavior, will all have a profound influence on how a species responds to protection and when these responses become apparent (Willis et al. 2003, Berkeley et al. 2004, Russ & Alcala 2004, Claudet et al. 2010). As the majority of the Hawaiian bottomfish species have a relatively high age at maturity, long life span and slow growth rate, it follows that it will be some time for changes to take effect (Ralston & Polovina 1982, Haight et al. 1993). For example, *Etelis coruscans*, *Hyporthodus quernus* and *Pristipomoides sieboldii* are particularly slow growing and slow to mature, taking an estimated 6 or more years to reach maturity (Everson et al. 1989, Williams

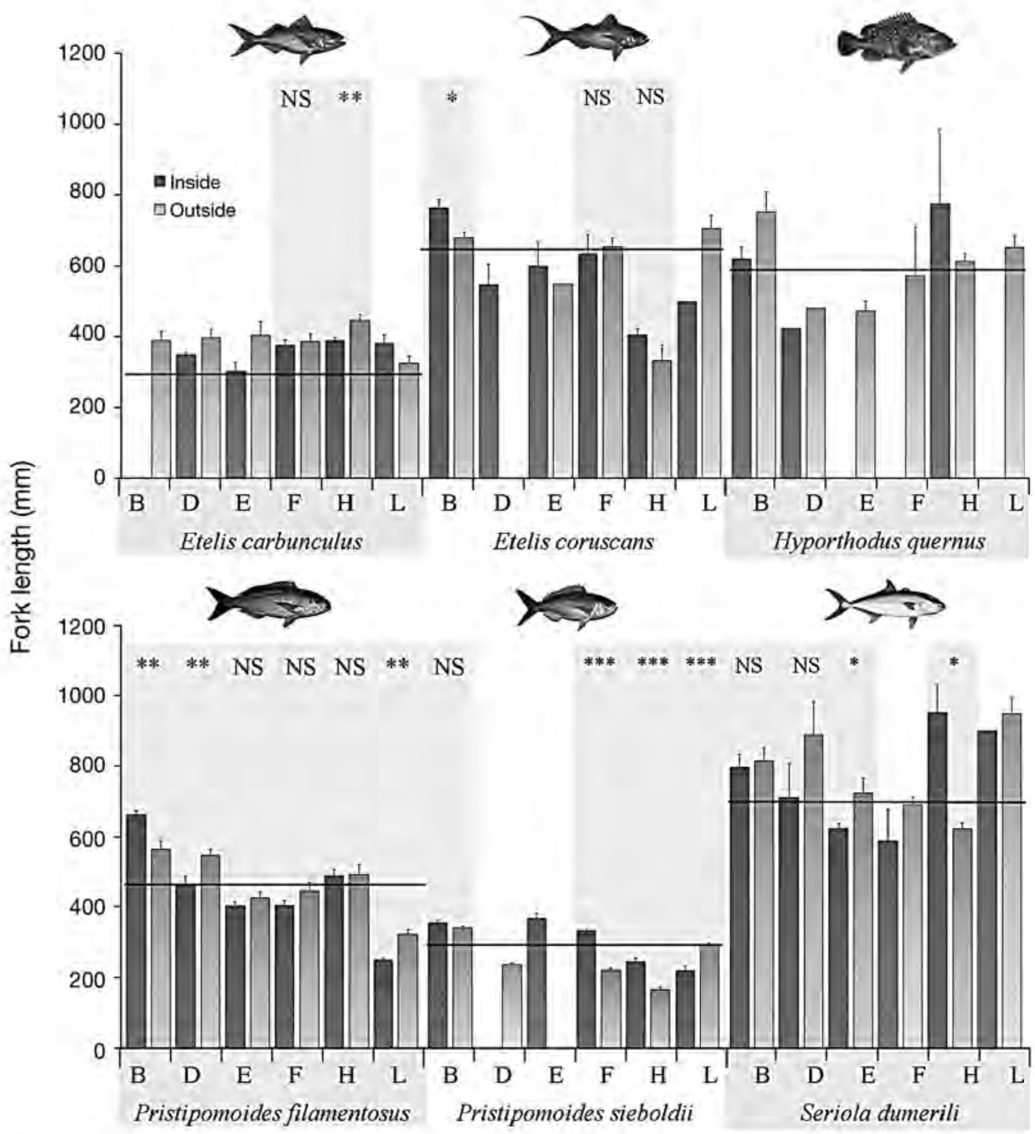

Fig. 5. Mean fork length (+1 SE) of harvested bottomfish species recorded inside and outside the 6 Bottomfish Restricted Fishing Areas (BRFAs) (see Fig. 1, Table 1). Test results are shown for significant differences in size-frequency distributions, calculated using the Wilcoxon test for small sample sizes or the Kolmogorov-Smirnov test where 10 or more fish measures were available both inside and outside the BRFA. Those species and sites that could be tested (n ≥ 4 both inside and outside the protected area) are indicated by grey shading. Horizontal lines indicate L_{50}, the reported size (length) at which 50 % of the population is sexually mature (Kikkawa & Everson 1984, Everson et al. 1989, DeMartini & Lau 1999, Harris et al. 2007, DeMartini et al. 2011). Illustrations by Les Hata©, Hawaii Department of Land and Natural Resources. NS: not significant; $^{*}0.05 > p > 0.01$; $^{**}0.01 > p > 0.001$; $^{***}p < 0.001$

& Lowe 1997, DeMartini et al. 2011). In addition, *H. quernus* is a protogynous hermaphrodite (DeMartini et al. 2011). Therefore, while females mature at 58 cm total length (TL) or 6 to 7 yr, changing sex from female to male occurs at 89 to 90 cm TL, which is broadly estimated to be at more than 20 yr of age (DeMartini et al. 2011). Overfishing of this species could result in changes in the sex ratio, sperm limitation or a reduced size or age at maturity (Alonzo & Mangel 2004, Molloy et al. 2007). For such a long-lived species with such a high age at maturity, it is not difficult to see how it may be decades before an effect of protection is detected. These life history traits combined with the results for *P. filamentosus* and *E. coruscans* inside the ongoing BRFA B suggest very slow recovery rates for Hawaiian bottomfish species. This is an important outcome for managers as it highlights the need to ensure the long-term protection and management for species demonstrating low rates of recovery.

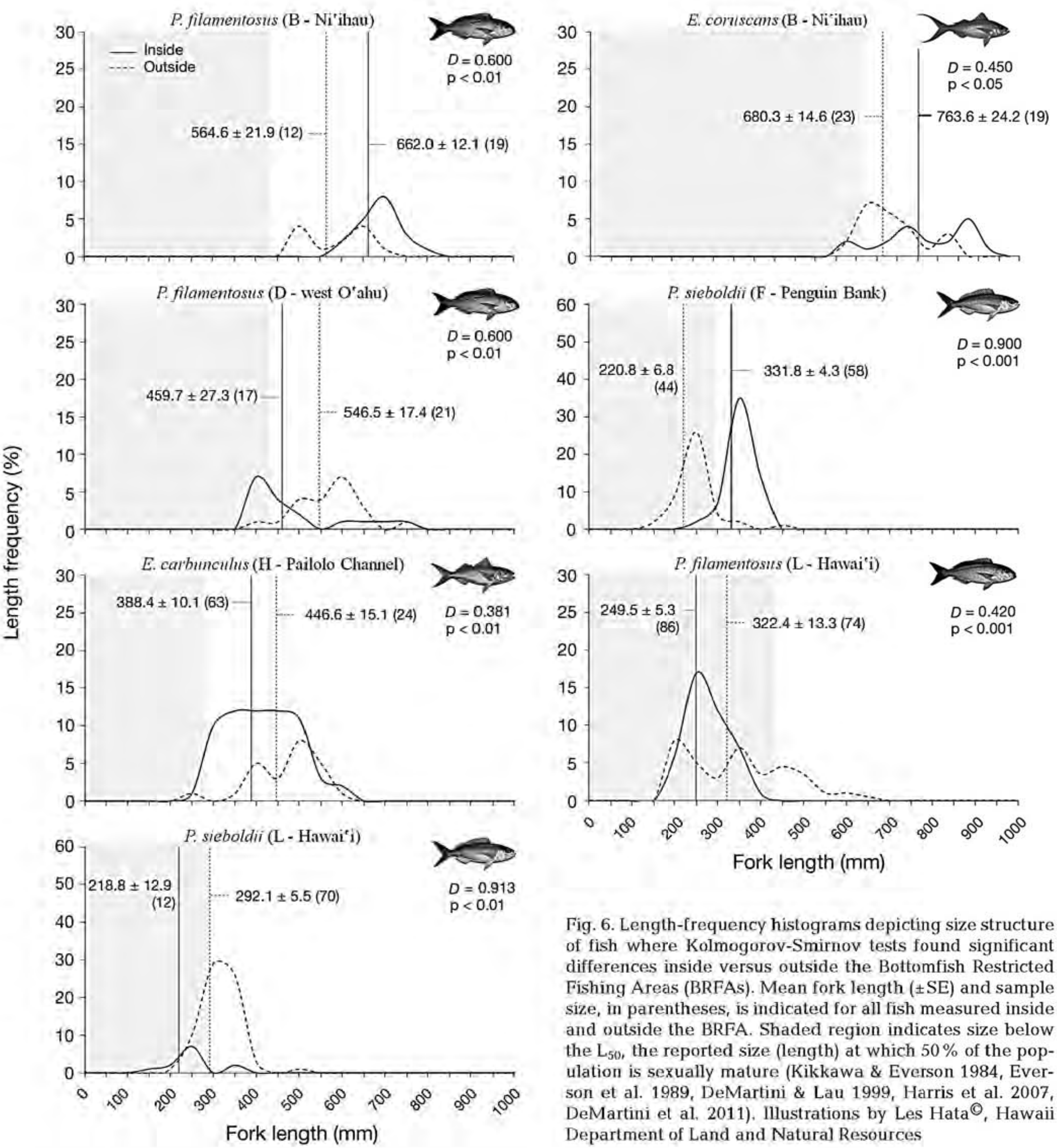

Fig. 6. Length-frequency histograms depicting size structure of fish where Kolmogorov-Smirnov tests found significant differences inside versus outside the Bottomfish Restricted Fishing Areas (BRFAs). Mean fork length (±SE) and sample size, in parentheses, is indicated for all fish measured inside and outside the BRFA. Shaded region indicates size below the L_{50}, the reported size (length) at which 50% of the population is sexually mature (Kikkawa & Everson 1984, Everson et al. 1989, DeMartini & Lau 1999, Harris et al. 2007, DeMartini et al. 2011). Illustrations by Les Hata©, Hawaii Department of Land and Natural Resources

While most of the harvested bottomfish species are long lived and slow to mature, there are some notable exceptions. For example, *Seriola dumerili* reaches sexual maturity in as little as 1.3 yr and at the largest size of any species, indicating an extremely fast growth rate (Uchida & Uchiyama 1986, Harris et al. 2007). It may follow that this species should be the first to be detected in higher abundance and greater size within the ongoing BRFAs. However, this was not the case, with few differences detected in abundance and size of this species. This result could be attributed both to its high mobility and to sociological influences (Uchida & Uchiyama 1986). Generally, MPAs have not been considered effective at enhancing densities of highly mobile species, as migratory species with home ranges larger than the size of the

MPA will not be fully protected. However, recent reviews by Palumbi (2004) and Claudet et al. (2010) found evidence that protection can benefit very vagile benthic, bentho-pelagic and pelagic commercially exploited species. A suggested mechanism for this response is that protection also increases habitat quality and densities of prey species, thereby attracting and retaining highly mobile species within MPAs (Rodwell et al. 2003). In addition, there is growing recognition that while these reserves may not be protecting highly mobile species 100 % of the time, they are still offering a potential reduction in fishing mortality (DeMartini 1993, Palumbi 2001). This reduction in fishing mortality is proportional to the time spent within MPAs, which is, in turn, proportional to the percent of the species distribution that is protected. It is therefore hypothesised that positive effects will be seen for a species where a comprehensive network of MPAs is protecting a significant portion of its distribution (DeMartini 1993, Palumbi 2001). A second consideration when interpreting the response of *S. dumerili* is its change in status in the commercial catch. In the past, *S. dumerili* was an important component of the bottomfish fishery in Hawai'i. However, it is currently the least valuable of the captured species, and for the past few decades landings have dropped significantly, with it now appearing as bycatch in catch reports. This change has been attributed to its association with ciguatera poisoning and a resulting ban on commercial sales (Kikkawa & Everson 1984, Uchida & Uchiyama 1986). Therefore, in addition to the species' biological characteristics, changes in fishing priorities will also influence the detected responses to the BRFAs and how we interpret those responses.

While some potentially promising responses to protection were detected for BRFA B, quite the opposite was recorded for BRFA L, the second ongoing BRFA. Here, 2 harvested species (*Pristipomoides filamentosus* and *P. sieboldii*) were significantly larger outside the BRFA. In addition, all of the 74 individual *P. filamentosus* measured inside the BRFA were too small to be sexually mature, as was the single *Etelis coruscans* recorded inside the BRFA. Again, no data were collected before this BRFA was declared, but these are not the results expected after 10 yr of protection. In this situation, it is particularly important to examine other potentially confounding factors. In contrast to BRFA B located off the most remote of the main Hawaiian Islands, BRFA L is located next to the second largest port in Hawai'i, Hilo, offering easier access, higher population pressure and more problematic enforcement with few resources available to ensure compliance. Sites assessed outside BRFA L to the south towards Kumakahe Point have limited accessibility. This rocky, cliff-lined coast has no ports and faces northeast, directly into the prevailing trade winds. Recent studies have now documented strong correlations between accessibility and proximity to population centres and reductions in top predator abundance and size and include studies from Hawaii (Friedlander & DeMartini 2002, Williams et al. 2008).

In addition to considering accessibility, when BRFA L was originally declared in 1998, a large portion of preferred bottomfish habitat was not included. The major habitat within this BRFA is a coastal terrace extending along its length. Originally, only the deepest sections of this terrace (approximately 200 to 400 m) were included in the BRFA, with the shallower sections (<200 m) open to fishing. This shallow terrace section has since been closed to fishing with the implementation of the revised system of BRFAs in 2007. Having the shallower section remain open to fishing is likely to have had a profound influence on bottomfish populations and on *Pristipomoides filamentosus* in particular. Research by Merritt et al. (2011), using the same BotCam as that used in this research, examined the distribution and relative abundance of *Etelis coruscans* and *P. filamentosus* between 100 to 300 m depth in BRFA F. They established that *E. coruscans* was primarily restricted to depths of >200 m, whereas *P. filamentosus* was restricted to depths of <250 m. Research by Ralston et al. (1986), using observations from submersible dives off Johnston Atoll (1250 km southwest of O'ahu), also reported the depth range of *P. filamentosus* to be shallower (120 to 260 m) than that of *E. coruscans* (250 to 355 m). Therefore, it follows that while the original BRFA should have been providing protection for *E. coruscans*, it is unlikely to have offered the same protection for *P. filamentosus*. While it is possible that other ecological and environmental factors such as competition, habitat quality, oceanographic processes and productivity may be influencing the distribution of *P. filamentosus*, overfishing is also likely to explain why no adult *P. filamentosus* and *E. coruscans* were recorded within BRFA L. Establishing exactly what is driving these differences will require ongoing monitoring and more detailed information on species-specific habitat and ecological requirements, in conjunction with a formal assessment of enforcement.

Hawaiian bottomfish are known to have strong habitat associations. Larger species are reported to form large aggregations near high relief features, and smaller species are found near hard substrates with high structural complexity (Ralston et al. 1986, Kelley et al. 2006, Parke 2007, Merritt et al. 2011). Some significant effects of habitat were detected for both the Deep 7 assemblage and for a few individual species. However, the effects were inconsistent between research locations and between species. While the Deep 7 were found in significantly lower mean relative abundances over low profile sediment, relative abundances over the remaining substrate classes demonstrated an inconsistent preference for reef and/or high profile substrates. This differs from our current understanding that Hawaiian bottomfish species are affiliated with hard, high profile substrates (Parke 2007). For example, *Pristipomoides filamentosus* was found to prefer reef including high and low profile reef at both BRFAs E and L, while *Etelis coruscans* showed a preference for high profile substrates including high profile reef and high profile sediment at BRFA F. The results suggest that the species are responding to their habitat in a more complex manner than previously thought and that there are species-specific differences in habitat preferences. The habitat classification used was quite broad, with substrate classified into 4 categories based on the dominant substrate within a 200 × 200 m grid. While more detailed and fine-scale information on species–habitat associations would be very useful, in the absence of a robust model of the area of attraction to the bait, the classification of habitat at this scale was more appropriate. Based on the calculations by Ellis & DeMartini (1995), if a video station is set in high currents, it is possible that individuals could be attracted from a maximum distance of between 250 and 408 m. This means that some stations set in high current may have also sampled an adjacent grid possibly dominated by a different substrate type. While not affecting our assessment of protection, as samples were placed a minimum of 400 m apart, this is something that needs to be addressed to enable us to obtain a clearer picture of bottomfish habitat associations. Having a solid understanding of species-specific habitat preferences would allow us to better understand the amount and quality of bottomfish habitat offered by each of the BRFAs. This understanding would, in turn, provide insight into the contribution that individual BRFAs provide for protecting Hawaiian bottomfish and the overall contribution that the network of BRFAs will make for protecting Hawaiian bottomfish populations.

A final consideration is the sampling technique. Many studies have demonstrated bias and selectivity among sampling methods used to assess fish populations (Thresher & Gunn 1986, Willis et al. 2000, Trenkel et al. 2004), with comparative research suggesting that no single technique will provide information on all fish species (Cappo et al. 2004, Watson et al. 2010). Instead, authors recommend careful choice of sampling method to suit the species of interest and question being addressed (Thresher & Gunn 1986, Willis et al. 2000). Stereo-baited underwater video was chosen for this research as it has been demonstrated to provide a more effective and efficient means for sampling larger mobile target species (Cappo et al. 2004, Harvey et al. 2007, Watson et al. 2010). Video-based techniques have been increasingly used to monitor MPAs as they are non-destructive, cost effective, not limited by depth and remove some major sources of bias, including observer bias and diver attraction or avoidance (see Cappo et al. 2003 for review, Willis & Babcock 2000, Willis et al. 2000). Most video systems are baited as data collected using unbaited systems can be very sparse, with much greater replication and field effort required to provide statistically testable results (Harvey et al. 2007). However, one of the key issues yet to be addressed is the effect of using bait on fish behaviour and on the distance over which fish are attracted. Some studies have attempted to model the area of attraction using current velocity, fish swimming speed and models of bait plume behaviour (e.g. Priede & Merrett 1996). However, these models currently require some unverifiable assumptions. The research by Harvey et al. (2007) comparing baited versus unbaited underwater video stations conceded that fish behaviour (e.g. swimming speed, schooling behaviour, levels of curiosity and aggression) and life history (e.g. reproduction, feeding ecology, mobility and home range) do presumably affect attraction to the bait. However, they still found baited underwater video stations able to collect robust and reproducible data. They found that variances of individual species sampled within habitats decreased when using bait, which was noted as particularly important for larger, more mobile species with relatively low densities and patchy distributions. We addressed the problem of an unknown area sampled by the bait by using a relatively short soak time (45 min) and adequate separation between samples to avoid overlap of the bait plume and ensure that samples were independent of one another (Ellis & DeMartini 1995, Cappo et al. 2004). The development of an accurate model of the area of attraction to the bait would enable us to take these results one step further to establish bottomfish abundance (number of fish per unit area) and more detailed habitat affiliations.

CONCLUSIONS

From this first comprehensive fishery-independent assessment, we have robust baseline data on commercially important bottomfish populations across the main Hawaiian Islands. Few significant differences were detected from this first assessment. However, differences detected in size-frequency distributions inside BRFA B (ongoing BRFA) provide evidence for very slow recovery rates for these deepwater species, which is consistent with their known life histories. Our results suggest that differences in size-frequency distributions can be detected after 10 yr but that it will take more years, possibly decades, before we detect large changes in species abundances. Having established the value of solid size-frequency distribution data, sampling intensity will be increased in future assessments to increase the number of length measurements for all harvested species. Another consideration is a better understanding of species-specific habitat requirements, both to ensure that high-quality bottomfish habitat is included within the BRFAs and to strengthen our sampling and statistical analyses. Individually, each BRFA is providing a unique set of habitat characteristics and environmental conditions affecting the distribution and relative abundance of bottomfish. Future research must examine the effect that the whole network of BRFAs is having on bottomfish populations across the main Hawaiian Islands. Differences in accessibility and proximity to population centres are likely to influence levels of enforcement and compliance. It is highly recommended that this be formally assessed to obtain a full picture of whether or not these BRFAs are achieving their purpose. While no data were collected on bottomfish populations before implementation of the system of BRFAs, this research has provided comprehensive baseline data on targeted bottomfish populations and some valuable information on expected recovery rates. This is just a first step, and only with ongoing monitoring and research will a full understanding of Hawaiian bottomfish populations and recovery rates be achieved.

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Establishing species–habitat associations for 4 eteline snappers with the use of a baited stereo-video camera system

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Abstract—With the use of a baited stereo-video camera system, this study semiquantitatively defined the habitat associations of 4 species of Lutjanidae: Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*). Fish abundance and length data from 6 locations in the main Hawaiian Islands were evaluated for species-specific and size-specific differences between regions and habitat types. Multibeam bathymetry and backscatter were used to classify habitats into 4 types on the basis of substrate (hard or soft) and slope (high or low). Depth was a major influence on bottomfish distributions. Opakapaka occurred at depths shallower than the depths at which other species were observed, and this species showed an ontogenetic shift to deeper water with increasing size. Opakapaka and Ehu had an overall preference for hard substrate with low slope (hard-low), and Onaga was found over both hard-low and hard-high habitats. No significant habitat preferences were recorded for Kalekale. Opakapaka, Kalekale, and Onaga exhibited size-related shifts with habitat type. A move into hard-high environments with increasing size was evident for Opakapaka and Kalekale. Onaga was seen predominantly in hard-low habitats at smaller sizes and in either hard-low or hard-high at larger sizes. These ontogenetic habitat shifts could be driven by reproductive triggers because they roughly coincided with the length at sexual maturity of each species. However, further studies are required to determine causality. No ontogenetic shifts were seen for Ehu, but only a limited number of juveniles were observed. Regional variations in abundance and length were also found and could be related to fishing pressure or large-scale habitat features.

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The catch of deepwater fisheries comprises a multitude of species that live on continental slopes and deep topographic oceanic structures, such as seamounts, ridges, and banks to depths below 2000 m. In the Indo-Pacific region, deepwater snappers (Lutjanidae), groupers (Serranidae), and jacks (Carangidae) that inhabit deep slopes and seamounts at depths of 100–400 m make up a major component of this fishery. The deepwater handline or "bottomfish" fishery of Hawaii also targets these groups of fishes (Haight et al., 1993a). Some of the commercially important bottomfish species can live in excess of 35 years (Andrews et al., 2011; Andrews et al., 2012)—a longevity that indicates low rates of natural mortality and susceptibility to overfishing (Haight et al., 1993a). Four of these key bottomfish species are the focus of this study: Crimson Jobfish (*Pristipomoides filamentosus*), Lavender Jobfish (*Pristipomoides sieboldii*), Flame Snapper (*Etelis coruscans*), and Ruby Snapper (*Etelis carbunculus*). In Hawaii, these species are known by a different set of common names, and these names will be used for simplicity throughout this article. *Pristipomoides filamentosus* is commonly called Opakapaka, *P. sieboldii* is called Kalekale, *E. coruscans* is called Onaga, and *E. carbunculus* is called Ehu. Opakapaka and Onaga rank first and second in total landed weight and value in the Hawaiian Archipelago, and the smaller species, Ehu and Kalekale, are abundant but lower in value and landings (WPRFMC[1]).

From the late 1980s to early 2000s, the Division of Aquatic Resources (DAR) of the Hawaii Department of Land and Natural Resources (DLNR) and the Western Pacific Re-

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[1] WPRFMC (Western Pacific Regional Fishery Management Council). 2006. Bottomfish and seamount groundfish fisheries of the western Pacific region, 2005 annual report, 113 p. [Available from Western Pacific Regional Fishery Management Council, 1164 Bishop Street, Suite 1400, Honolulu, HI 96813.]

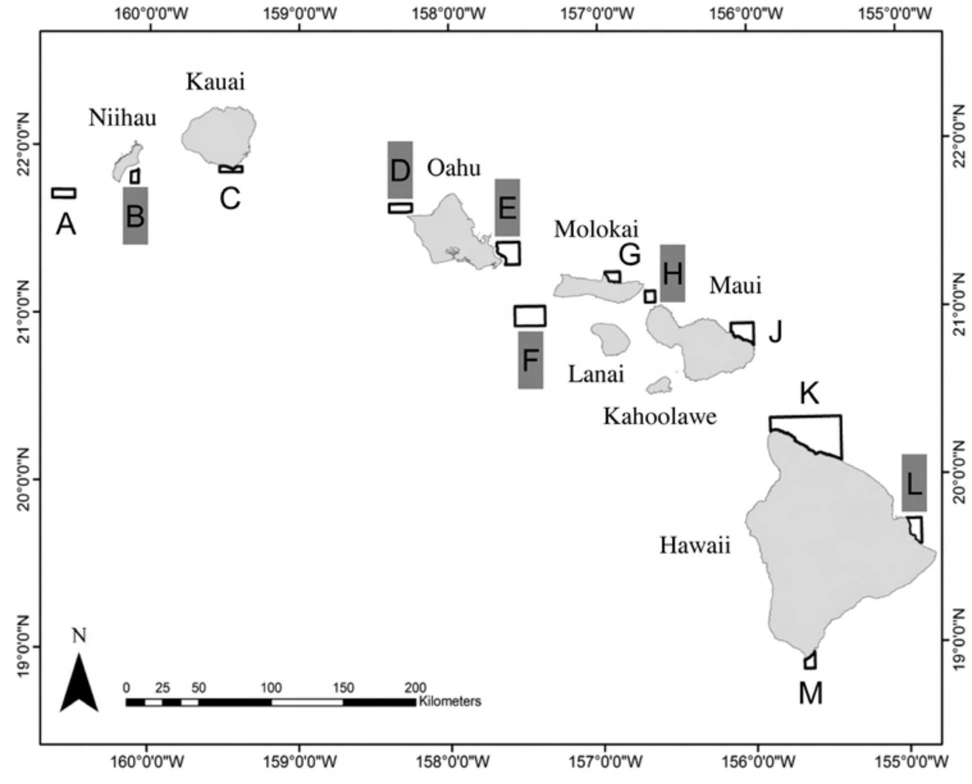

Figure 1

Map of the current bottomfish restricted fishing areas (BRFAs) in the main Hawaiian Islands. Highlighted letters indicate the 6 BRFAs—(B) Niihau, (D) Kaena, (E) Makapuu, (F) Penguin Bank, (H) Pailolo Channel, and (L) Hilo—that were sampled from May 2007 to June 2009 with the use of a baited stereo-video camera system for the study of the habitat associations of 4 snapper species.

gional Fishery Management Council (WPRFMC) assessed bottomfish stocks in the main Hawaiian Islands (MHI) by calculating their estimated spawning potential ratios (SPRs) from annual commercial catch data and established the critical threshold for designation of a stock in a state of recruitment overfishing at a SPR of 20%. Two bottomfish species, the Onaga and Ehu, had SPRs well below 20% for most of this period (DAR[2]) and were, therefore, considered to be in a state of recruitment overfishing.

In 1996, the Magnuson-Stevens Fishery Conservation and Management Act imposed a mandate on regional fishery councils to restore the stocks of overfished species to healthy levels (i.e., SPR >20%) within a 10-year time period. To address this problem, the WPRFMC turned to the DAR, which created 19 bottomfish restricted fishing areas (BRFAs) and prohibited bottomfishing in them (Div. Aquatic Resources, Department of Land and Natural Resources, Chapter 13-94, Bottomfish Management, Hawaii Administrative Rules). These BRFAs, which took effect on June 1, 1998, were designed to protect 20% of deepwater areas in the depth range of 100–400 m, where most Onaga and Ehu are found (Parke, 2007). However, identification of suitable geographic areas for closure was difficult at that time because of a lack of adequate habitat data—a common problem for most deepwater fisheries given the logistical challenges involved in sampling the deep sea.

In 2007, the DAR revised the BRFA system with data from surveys conducted with a multibeam sonar system, fishing surveys, and analysis of video collected during surveys with a submersible—all of which provided a great deal of new information on bottomfish habitats. The original BRFAs established in 1998 were retained, expanded, relocated, or opened to fishing, and the 12 BRFAs established in 2007 (Fig. 1) contained significantly more of the hard, steep habitat believed to be preferred by most bottomfish species (Parke, 2007). This belief was formed on the basis of results from submersible and fishing surveys that found some species in the water column adjacent to areas of high relief, such as underwater headlands, ledges, outcrops, and pinnacles

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[2] DAR (Division of Aquatic Resources). 2006. Hawaii's bottomfish fishery, Land Board briefing paper, 17 p. [Available from Division of Aquatic Resources, Hawaii Department of Land and Natural Resources, 1151 Punchbowl St., Rm. 330, Honolulu, HI 96813.]

(Ralston et al., 1986; Haight et al., 1993a). More recent submersible surveys have supported those studies and have indicated that substrate type may be an important factor that influences distributions of adult bottomfishes (Kelley et al., 2006). However, information on species-specific and age-specific habitat associations for bottomfishes remains limited. Although the preferred habitat of juvenile Opakapaka has been observed to be soft substrates with little to no relief (Moffitt and Parrish, 1996; Parrish et al., 1997), variations in habitats between adults and juveniles, if any, have yet to be identified for other species of deepwater bottomfishes.

Information that can identify fish–habitat associations is fundamental to fisheries science. In addition to the requirement to improve overfished stocks, the Magnuson-Stevens Act required federal fishery management plans to identify the essential fish habitat (EFH) for their managed species (Rosenberg et al., 2000). The EFH for the bottomfish fishery in Hawaii currently is designated as depths from 0 to 400 m without species-specific habitat requirements, despite the notion that habitat requirements probably differ between bottomfish species and ontogenetic stage of these species. To guide management decisions on the protection and sustainable use of bottomfish resources in Hawaii, this EFH designation should be as complete and as specific as possible (Kelley et al., 2006).

New data are needed to obtain a greater understanding of the habitat associations of bottomfish species. Common shallow-water sampling techniques, such as diver transects, however, are not logistically feasible at depths below 100 m, and fishing surveys can be destructive to local populations. The need for a different survey method has led to the emergence of baited camera systems as cost-effective, nonextractive tools for the estimation of relative abundances of fish species at depths >100 m (Merritt et al., 2011; Moore et al., 2013).

With the use of a baited stereo-video camera system, we aimed to improve our understanding of the habitat associations of 4 species of bottomfishes, within different size classes, in the MHI. Data specific to each species can be used to assess the amount of suitable habitat present in management areas and to relate catch per unit of effort (CPUE) to habitat type. Most important, through expansion of our understanding of the ecology of bottomfishes, more specific and refined EFH designations can be forged and ecosystem-based management strategies can be further developed.

Materials and methods

The Bottom Camera Bait Station (BotCam) developed by the Coral Reef Ecosystem Division of the NOAA Pacific Islands Fisheries Science Center is a remote, fully automated, baited system with stereo-video cameras; it was designed specifically for nonextractive, fishery-independent sampling of deepwater bottomfish species in their habitat and depth range (Merritt, 2005; Merritt et al., 2011). The method for sampling fish populations with a baited stereo-video camera system has been found to generate more consistent data than have comparable unbaited systems (Harvey et al., 2007), has the ability to detect mobile fish species (Harvey et al., 2007; Watson et al., 2010), and has been determined to be effective in sampling bottomfishes in Hawaii (Ellis and DeMartini, 1995; Merritt et al., 2011). The BotCam is a means by which bottomfish abundance estimates can be made within actual bottomfish habitats and fish lengths can be accurately measured.

Upon deployment, the BotCam sits about 3 m off the bottom of the seafloor, and, depending on the depth of deployment, amount of light, and water clarity, the field of view may expand or contract. Moore et al. (2013) estimated that the visual area sampled by the BotCam was between 4 and 400 m^2. The BotCam makes use of ambient light, which allows for an operating depth of up to 300 m and is operational on multiple bottom types, including steep slopes and high relief. In our study, the BotCam recorded 30 to 45 min of continuous video at each of the 6 deployment locations. Depth data were taken from a conductivity, temperature, and depth profiler attached to the system. The bait canister attached to the BotCam was filled with ~800 g of ground anchovy and squid, a mix that is similar to the bait used by bottomfish fishermen (Merritt et al., 2011).

Bottomfish habitat types in the MHI were characterized with multibeam bathymetry and backscatter data that originated from a variety of mapping surveys conducted with multibeam sonar systems in and around the MHI since the late 1990s. The U.S. Geological Survey in collaboration with the Monterey Bay Aquarium Research Institute carried out the first survey in the MHI in 1998 (U.S. Geological Survey Digital Data Series DDS-55, http://pubs.usgs.gov/dds/dds-55/index.html; MBARI Hawaii Multibeam Survey, http://www.mbari.org/data/mapping/hawaii/index.htm) with a 30-kHz Simrad[3] EM 300 multibeam sonar system (Kongsberg Maritime AS, Kongsberg, Norway). Both the bathymetry and backscatter data from this survey were processed at a grid resolution of 20 m. The majority of the remaining data came from subsequent surveys conducted from 2002 to 2006 by researchers at the Hawaii Undersea Research Laboratory, University of Hawaii at Manoa, with a 95-kHz Simrad EM 1002 multibeam sonar system. The editing and processing of raw data were carried out by the Hawaii Mapping Research Group of the University of Hawaii at Manoa using the SABER multibeam editing program (SAIC, Inc., McLean, VA) and other proprietary software. Bathymetry data were processed at a 20-m grid resolution, and backscatter data were processed at either a 10-m resolution or a 20-m resolution, depending on the survey. The processed

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[3] Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA

data from these cruises have not been made publicly available, with the exception of the bathymetry data that have been incorporated into a 50-m-resolution synthesis of the entire MHI that is available from the Hawaii Mapping Research Group (http://www.soest.hawaii.edu/hmrg/multibeam/index.php).

Multibeam backscatter data in grids with a 20-m resolution cannot be used effectively to identify specific substrate types, such as mud, sand, pebbles, cobbles, boulders, and bedrock, because more than one of these substrate types often can be found on the seafloor in an area of 20×20 m. Similarly, more than one type of slope can be found in areas of that size because of the presence of small carbonate ledges, large boulders and blocks, sand dunes, and other small-scale topographic features common to seafloors in the Hawaiian Archipelago. Multibeam data values for each grid cell (20×20 m) are typically derived through calculation of either the Gaussian weighted means (bathymetry) or the medians (backscatter) of the sonar footprints within each cell. For these reasons, only 4 general habitat types were derived from these multibeam data: hard substrate with high slope (hard-high), hard substrate with low slope (hard-low), soft substrate with high slope (soft-high), and soft substrate with low slope (soft-low). Bathymetry data from the different sonar systems generally were consistent.

After a number of slope analyses were conducted in ArcGIS 9.1 (Esri, Redlands, CA), a value of 20° was determined to be a reasonable boundary between the high and low slopes that appeared in the bathymetry images. Backscatter data, however, are often inconsistent between systems with different frequencies. Furthermore, the backscatter data used in this study were processed in different ways by different technicians. As a result, boundary values between hard and soft substrates had to be determined on a basis of per system and per cruise. A value of 187 was used as the boundary between hard and soft substrates for the EM 300 data and was validated through examination of video from submersible surveys. Boundary values for the EM 1002 data ranged from –41 to 150 and were established through comparison of areas of overlap with EM 300 data and analysis of video from submersible surveys.

Habitat was classified at a resolution of 200×200 m for areas in and around BRFAs. Polygons for high and low slopes and hard and soft substrates were generated with the Raster calculator in ArcGIS 9.1. Intersects of slope and hardness resulted in polygons for the 4 habitat types. A grid cell (200×200 m) was superimposed over these polygons, and the areas of the habitat types within each grid cell were calculated. Each grid cell was assigned a habitat type on the basis of which habitat type was observed in the greatest proportion in that area.

A stratified-random sampling approach was used to select locations for BotCam sampling. Although the purpose of our study was to evaluate species–habitat associations, another goal of this project was to evaluate population changes inside and outside of BRFAs. This objective affected our sampling design. We used data from 625 deployments of the BotCam conducted inside and outside of 6 of the 12 current BRFAs (Fig. 1) between May 2007 and June 2009. The 6 BRFAs that were sampled are located off Niihau (BRFA B), Kaena (BRFA D), Makapuu (BRFA E), and Penguin Bank (BRFA F), in Pailolo Channel (BRFA H), and outside of Hilo (BRFA L). The Niihau and Hilo BRFAs were areas of continued closure from the initial implementation of BRFAs in 1998. The Makapuu and Penguin Bank BRFAs were expanded versions of smaller preexisting BRFAs from 1998, and the BRFAs off Kaena and in Pailolo Channel were areas newly closed in 2007.

The BotCam was lowered to depths of 100–300 m. Although the EFH for deep bottomfishes in Hawaii extends to 400 m, the video cameras work under ambient light to only 300 m, thus limiting the depth range of our sampling. Sampling effort was weighted toward known preferred bottomfish habitats to ensure greater replication where fish densities were expected to be higher. Because previous studies have found bottomfishes associated with hard substrates, high slopes, or a combination of both (Polovina et al., 1985; Ralston et al., 1986; Haight et al., 1993a; Parke, 2007), for our study, hard-high habitats were considered the most suitable and soft-low habitats the least suitable. To sample a BRFA, 32 BotCam deployments inside and 32 outside but adjacent to a BRFA were completed over grids of each habitat type with the following replication: 12 hard-high, 8 hard-low, 8 soft-high, and 4 soft-low. BotCam deployments targeted centroids of randomly selected grid cells (200×200 m) and were kept a minimum of 400 m apart to reduce the likelihood of sampling overlap. In regions where a given habitat type was not present, sampling intensity was increased in the next most suitable habitat. This approach led to skewed sampling across habitat types in Pailolo Channel because only low-slope habitats were identified at a resolution of 200×200 m. When BotCam deployments did not yield usable video (e.g., no recordings or extremely dark imagery), the BotCam was redeployed at that location on another day. As often happens during sampling efforts in the field, not all targeted grids were sampled because of weather and equipment issues. In the 2-year sampling period covered by this study (2007–09), 4 of the 6 BRFAs (Niihau, Makapuu, Penguin Bank, and Pailolo Channel) were sampled twice and the Kaena and Hilo BRFAs were sampled only once.

BotCam video footage was reviewed in the laboratory to estimate the relative abundance, recorded as the maximum number of a particular species observed in a single frame of video (MaxNo), of Opakapaka, Kalekale, Onaga, and Ehu with VF Deep Portal (Deep Development Corp., Sumas, WA) and Adobe Premiere Pro CS4 (Adobe Systems, Inc., San Jose, CA) software programs. Fishes were identified to the most specific taxonomic classification possible with a species identification reference (Randall, 2007). MaxNo is a conservative abundance estimate that avoids the potential problem of counting the same fish multiple times as it re-enters a camera's field of view. Many studies have determined that MaxNo is positively correlated with fish density (Ellis and DeMartini, 1995; Priede and Merrett, 1996; Willis et al., 2000; Willis and Babcock, 2000; Yau et al., 2001; Cappo et al., 2003). This parameter also has been found to be highly correlated with the traditional parameter of CPUE used in fishing surveys (Ellis and DeMartini, 1995). MaxNo was recorded for all fishes present in the BotCam video footage, but only data for the 4 species of interest were analyzed.

Permutational analysis of variance (PERMANOVA) of the data was performed in Primer 6 (PRIMER-E Ltd., Ivybridge, UK) with PERMANOVA+ (Anderson et al., 2008). With PERMANOVA, the data are not assumed to be normally distributed; therefore, this technique was deemed appropriate for analysis of our data, which included a highly skewed (overdispersed) relative abundance distribution due to an unbalanced experimental design and frequent zero counts. The 4 species considered in our study do not all occupy the entire depth range sampled (Polovina et al., 1985; Haight, 1989; Everson et al., 1989; Merritt et al., 2011). To constrain the data to an appropriate range for each species, the depths at which each species had its greatest MaxNo had to be identified. For the initial analysis, depth was divided into 30-m bins from 90 to 300 m. Relative abundance values were square-root transformed to compensate for numerous zero counts and occasional large numbers. A Euclidean distance matrix was used in the statistical test with a type-III sum of squares. If a significant difference ($P<0.05$) was observed across depth bins, a subsequent pair-wise PERMANOVA was performed to determine the preferred depths of each species. Subsequent analyses (MaxNo and fork length [FL]) were then constrained to the depth preferences identified for each of the 4 species studied.

Through identification of habitat preferences, the influence of BRFA location (i.e., combined area inside and outside a BRFA) and protection (i.e., area inside versus outside a BRFA) could not be overlooked. PERMANOVA in a 3-way crossed design was used to determine how BRFA location (BR, 6 levels, fixed), protection (PR, 2 levels, fixed), habitat type (HA, 4 levels, fixed), and the interaction of these factors affected the relative distribution of each species. MaxNo values were square-root transformed, and the PERMANOVA was run on a Euclidean distance matrix with type-III sum of squares. Where significant results ($P<0.05$) occurred, pair-wise testing was performed to identify specific differences.

For individual fish visible in both BotCam cameras, FL was measured with stereo-photogrammetric measurement software: Visual Measurement System 7.5 (Geometric Software Pty. Ltd., Coburg, Victoria,

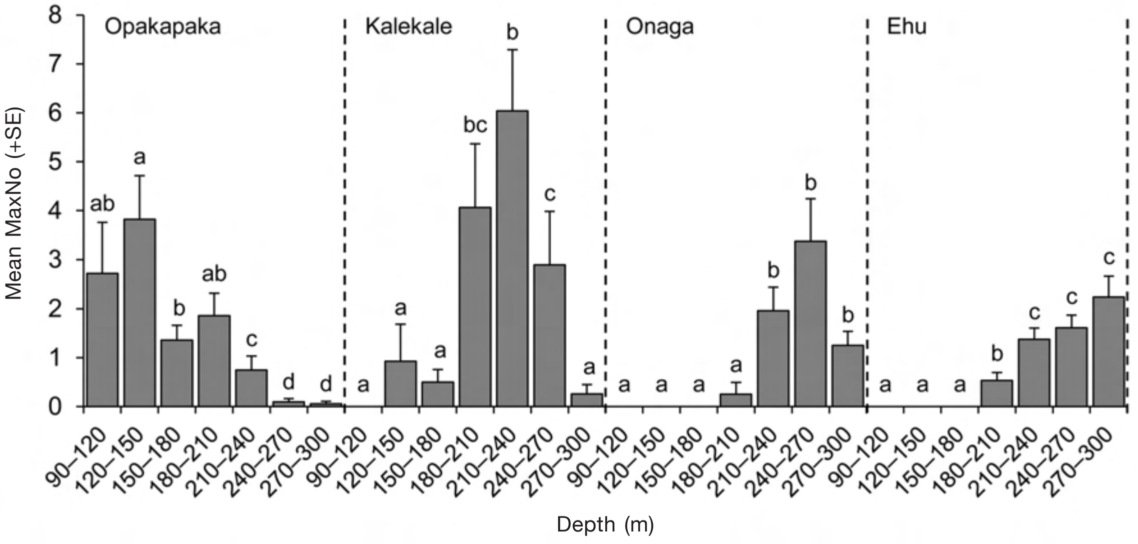

Figure 2

Mean relative abundance (MaxNo) with standard error (SE) across 7 depth bins for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) from surveys of these species conducted in the main Hawaiian Islands from May 2007 to June 2009 with the use of a baited stereo-video camera system. Columns with the same letter are not significantly different from each other ($P>0.05$, post hoc permutational analysis of variance [PERMANOVA] testing). Error bars indicate ±1 SE of the mean.

Table 1

Results of permutational analysis of variance (PERMANOVA) with relative abundance (MaxNo) data from our surveys of 4 species—Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*)—in the main Hawaiian Islands between May 2007 and June 2009. The following factors were tested within the preferred depths of each species: bottomfish restricted fishing area location (BR), protection (PR), and habitat type (HA). Preferred depths are noted in the column head for each species. df=degrees of freedom; *F*=PERMANOVA *F*-statistic; *P*=PERMANOVA *P*-value. Asterisks indicate statistical significance at *P*<0.05.

	Opakapaka (90–210 m)			Kalekale (180–270 m)			Onaga (210–300 m)			Ehu (210–300 m)		
Factor	df	*F*	*P*	df	*F*	*P*	df	*F*	*P*	df	*F*	*P*
BR	5	2.86	0.02*	5	2.07	0.09	5	1.54	0.17	5	4.78	0.00*
PR	1	0.00	1.00	1	0.07	0.79	1	0.07	0.78	1	0.31	0.58
HA	3	8.28	0.00*	3	1.68	0.18	3	3.87	0.02*	3	2.83	0.04*
BR×PR	5	0.63	0.66	5	0.55	0.72	5	0.56	0.70	5	0.81	0.54
BR×HA	13	0.64	0.80	12	1.89	0.06	13	0.69	0.71	13	2.33	0.01*
PR×HA	3	0.62	0.59	3	0.87	0.45	3	0.56	0.62	3	0.93	0.42
BR×PR×HA	12	1.02	0.42	10	0.44	0.91	9	0.59	0.76	9	0.58	0.79
Residual	247			282			295			295		

Australia) and PhotoMeasure 1.74 (SeaGIS Pty. Ltd., Bacchus Marsh, Victoria, Australia). Measurements of individual fish were taken at the point of MaxNo or at the point in the video where the most fish could be measured to ensure that individuals were not repeatedly measured at various times during video analysis. Replicate measurements were taken for individual fish to increase the accuracy of the measurement. An LED device was used to ensure synchronicity of the video footage from the left and right cameras. A root-mean-square error or residual parallax >10 mm and a precision-to-FL ratio >10% were indicative of inaccurate measurements. To ensure the quality of fish length data, these measurements were removed from the analyses in this study. The same 3-way crossed design from the PERMANOVA of relative abundance (BR, PR, HA) was used to test FLs for each species. Transformation of FLs, however, was not necessary because these data typically were normally distributed.

Because only variations in mean length were evaluated with the previously described approach, additional analyses were undertaken to investigate size-related changes in habitat association. A linear regression was used to evaluate the relationship between depth and FL for each of the 4 species studied to identify ontogenetic shifts with depth. As part of our examination of ontogenetic shifts across habitat types, a contingency table (tested with Pearson's chi-square test) was used to determine whether the size-class distribution of each species was independent of habitat type. Fork lengths were grouped into 10-cm bins. This size interval was chosen to maximize the number of observations in each size bin. Merritt et al. (2011) tested and found measurements from BotCam video to be accurate to within 0.3–0.9 cm, making such a grouping very robust.

Results

For all 4 species studied, significant differences in relative abundance were found across depth bins (PERMANOVA, *P*<0.05). Pair-wise comparisons of MaxNo from the 7 depth bins highlighted the depth preference of each species (Fig. 2). MaxNo was highest from 90 to 210 m for Opakapaka (post hoc PERMANOVA, *P*<0.05). The preferred depths of Kalekale were 180–270 m, and both Onaga and Ehu had the deepest range among species at 210–300 m (post hoc PERMANOVA, *P*<0.05).

Within the preferred depths of a species, either BRFA location, habitat type, or the interaction of these 2 factors had an effect on the relative abundance of 3 of the 4 species studied (Table 1). Protection and the interaction of all other factors with protection, however, did not have an effect (PERMANOVA, *P*>0.05). BRFA location and habitat type were each significant factors for Opakapaka. Hilo had the highest relative abundance of this species among sampled locations, and hard-low habitats yielded greater abundance estimates for Opakapaka than other habitat types (Fig. 3; post hoc PERMANOVA, *P*<0.05). Although no significant location or habitat effects were observed for Kalekale, the interaction of BRFA location and habitat type was marginal (*P*=0.06; Table 1); 2 of the largest counts of this species (100 and 85 individuals) occurred on hard-high habitats at Niihau and led to a high mean MaxNo (Fig. 3).

Habitat type was the only factor that affected the relative abundance of Onaga. Hard substrate habitats, with either high or low slope, had greater mean MaxNo for Onaga than soft substrate habitats (Fig. 3; post hoc PERMANOVA, *P*<0.05). BRFA location, habitat type, and the interaction of these 2 factors were significant

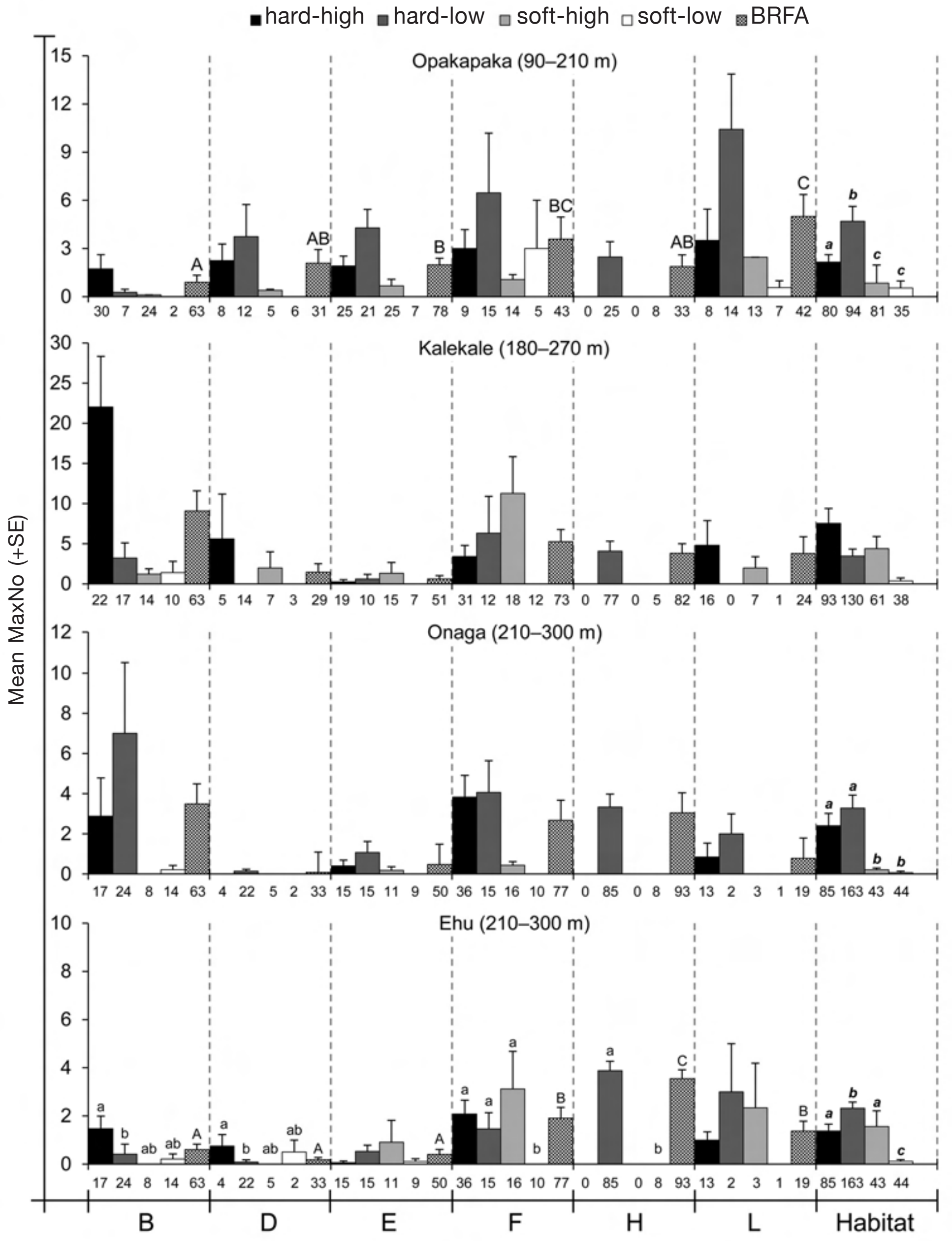

Figure 3

Mean relative abundance (MaxNo) with standard error (SE) by location of bottomfish restricted fishing area (BRFA; i.e., combined area inside and outside a BRFA), by habitat type combined for all BRFA locations (habitat), and by habitat type in each BRFA location (BRFA×habitat) for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) within the preferred depths of each species in the main Hawaiian Islands. A baited stereo-video camera system (BotCam) was used to collect data from May 2007 to June 2009. Columns with the same letter (uppercase type for BRFA; lowercase, bold, italic type for habitat; lowercase type for BFRA×habitat) are not significantly different from each other ($P>0.05$, post hoc permutational analysis of variance [PERMANOVA]). The number below each column is the number of BotCam deployments. The 4 habitat classifications used in our study were derived from data collected with multibeam sonar systems: hard substrate with high slope (hard-high), hard substrate with low slope (hard-low), soft substrate with high slope (soft-high), and soft substrate with low slope (soft-low). The 6 sampled BRFAs were (B) Niihau, (D) Kaena, (E) Makapuu, (F) Penguin Bank, (H) Pailolo Channel, and (L) Hilo. Error bars indicate ±1 SE of the mean.

Table 2

Results of permutational analysis of variance (PERMANOVA) with fork length data from our surveys of 4 species—Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*)—in the main Hawaiian Islands between May 2007 and June 2009. The following factors were tested within the preferred depths of each species: bottomfish restricted fishing area location (BR), protection (PR), and habitat type (HA). Preferred depths are noted in the column head for each species. df=degrees of freedom; *F*=PERMANOVA *F*-statistic; *P*=PERMANOVA *P*-value. Asterisks indicate statistical significance at *P*<0.05.

	Opakapaka (90–210 m)			Kalekale (180–270 m)			Onaga (210–300 m)			Ehu (210–300 m)		
Factor	df	*F*	*P*	df	*F*	*P*	df	*F*	*P*	df	*F*	*P*
BR	5	36.04	0.00*	5	28.20	0.00*	4	11.05	0.00*	4	4.90	0.00*
PR	1	14.24	0.00*	1	1.43	0.23	0	No test		0	No test	
HA	3	11.39	0.00*	3	18.38	0.00*	1	0.48	0.49	2	1.77	0.17
BR×PR	5	2.02	0.08	3	16.57	0.00*	3	4.82	0.00*	4	0.84	0.52
BR×HA	9	7.66	0.00*	4	1.16	0.33	1	23.69	0.00*	5	1.62	0.16
PR×HA	2	0.45	0.64	2	0.21	0.82	2	0.48	0.61	2	1.31	0.27
BR×PR×HA	5	3.42	0.01*	2	0.21	0.81	1	13.26	0.00*	0	No test	
Residual	419			446			242			274		

for Ehu. The highest relative abundance for this species was in Pailolo Channel, and the lowest levels were seen at Niihau, Kaena, and Makapuu (Fig. 3; post hoc PERMANOVA, *P*<0.05). Overall, hard-low habitats had significantly greater numbers of Ehu than did other habitat types. By BRFA location and habitat type, the mean MaxNo of Ehu in Pailolo Channel was higher for hard-low than for soft-low habitats, and similar abundance estimates were found for hard-high, hard-low, and soft-high habitats on Penguin Bank. Niihau and Kaena differed from the other sampled locations in that hard-high habitats had a greater relative abundance of Ehu than did hard-low habitats.

In our evaluation of mean lengths, BRFA location, protection, and habitat type were all important factors, and the interactions between them were sometimes significant (Table 2). BRFA location, protection, habitat type, the interaction of BRFA location and habitat type, and the interaction of all 3 factors were significant for Opakapaka. Niihau had the largest Opakapaka on average (65.29 cm FL) among sampled locations, and the smallest Opakapaka (28.35 cm FL; Fig. 4; post hoc PERMANOVA, *P*<0.05) were seen at Hilo. The smallest individual at Hilo measured ~16 cm FL, and the largest individual at Niihau was ~79 cm FL. Opakapaka from outside protected areas had a mean length of 42.89 cm FL and were larger than those fish observed inside the sampled BRFAs (40.53 cm FL; PERMANOVA, *P*<0.05). The smallest mean lengths of this species were found over hard-low habitats compared with other habitat types overall, other habitats at each BRFA location, and other habitats either inside or outside a particular BRFA (Fig. 4; Table 3; post hoc PERMANOVA, *P*<0.05).

BRFA location, habitat type, and the interaction of BRFA location and protection were significant for Kalekale. Pair-wise comparisons showed that this species had its smallest mean length (23.64 cm FL) at Kaena, was largest in hard-high habitats (31.46 cm FL) and smallest in soft-low habitats (8.64 cm FL, *n*=2), and was larger inside the Penguin Bank and Pailolo Channel BRFAs and outside the Hilo BRFA than in other sampled areas (Fig. 4; Table 3; post hoc PERMANOVA, *P*<0.05). The smallest individual Kalekale, however, measured 7.63 cm FL at Niihau. BRFA location, the interaction of BRFA location with protection, the interaction of BRFA location with habitat type, and the interaction of all 3 of these factors were significant for Onaga. Mean length for Onaga was smallest in Pailolo Channel (42.80 cm FL) than at other locations (Fig. 4) but larger inside the Pailolo Channel BRFA than outside this protected area (Table 3; post hoc PERMANOVA, *P*<0.05). The smallest individual Onaga measured 15.05 cm FL. Although the interaction of BRFA location and habitat type and the interaction of BRFA location, protection, and habitat type had significant results for Onaga, no clear trends were seen. BRFA location was the only factor that had an influence on mean length for Ehu (Table 2; PERMANOVA, *P*<0.05). Overall, mean sizes were very similar for this species but were smallest at Makapuu and Hilo (Fig. 4).

For all sampled locations combined, size-related shifts in species–habitat associations were evident. The linear regressions of FL against depth for each species showed that size increased with depth for Opakapaka (coefficient of determination [r^2] =0.438, *P*<0.01) but did not for the other 3 species (Fig. 5). In our evaluation of the proportion of fish measured in each habitat type by size class, habitat associations clearly varied by size for Opakapaka, Kalekale, and Onaga (Fig. 6). Ehu had very similar habitat associa-

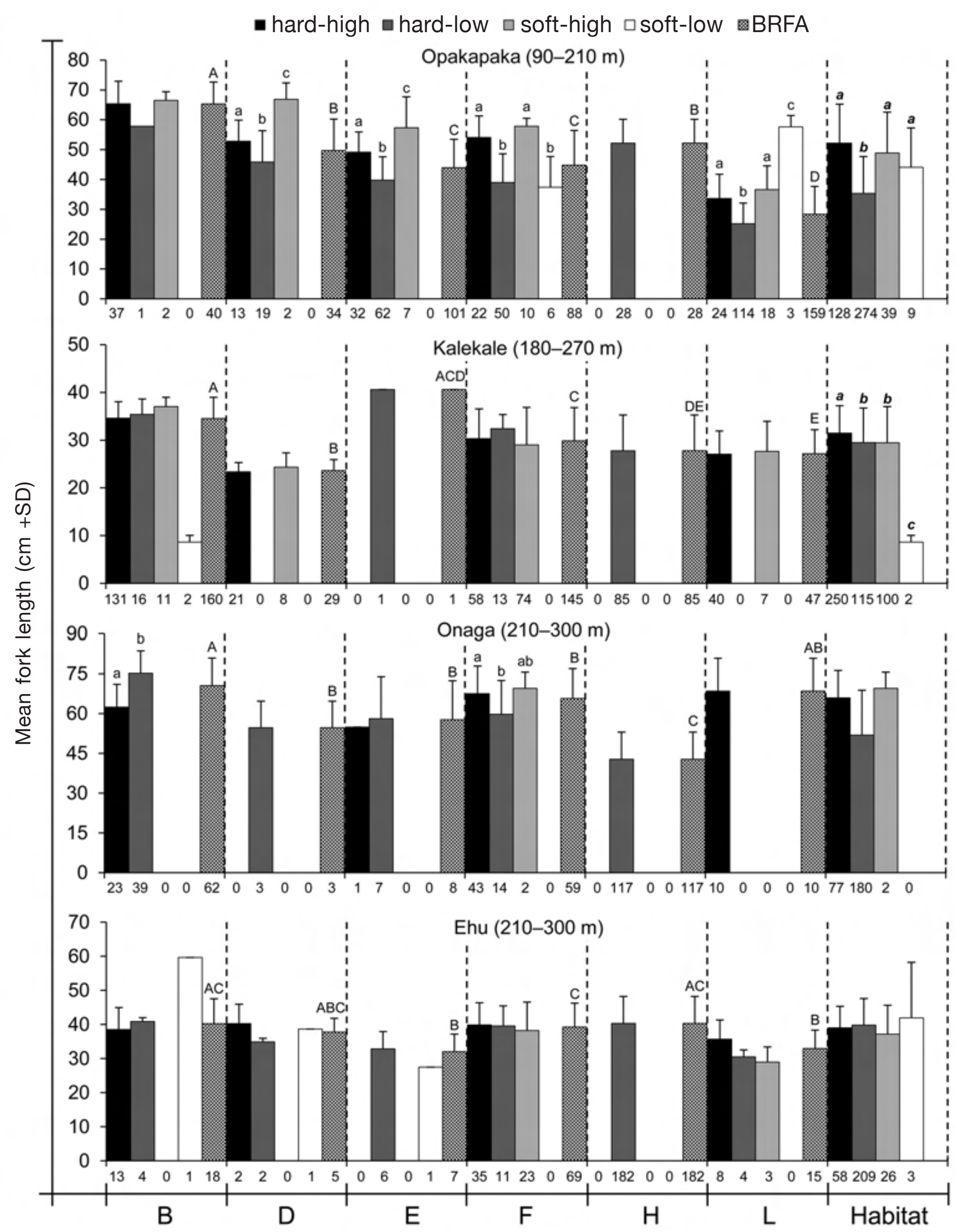

Figure 4

Mean fork length with standard deviation (SD) by location of bottomfish restricted fishing area (BRFA; i.e., combined area inside and outside a BRFA), by habitat type combined for all BRFA locations (habitat), and by habitat type in each BRFA location (BRFA×habitat) for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) within the preferred depths of each species in the main Hawaiian Islands. A baited stereo-video camera system (BotCam) was used to collect data from May 2007 to June 2009. Columns that have the same letter (uppercase type for BRFA; lowercase, bold, italic type for habitat; lowercase type for BRFA×habitat) are not significantly different from each other ($P>0.05$, post hoc permutational analysis of variance [PERMANOVA] testing). Number below each column is the number of fish measured. For protection effects, refer to Table 3. The 4 habitat classifications used in our study were hard substrate with high slope (hard-high), hard substrate with low slope (hard-low), soft substrate with high slope (soft-high), and soft substrate with low slope (soft-low). The 6 sampled BRFAs were (B) Niihau, (D) Kaena, (E) Makapuu, (F) Penguin Bank, (H) Pailolo Channel, and (L) Hilo. Error bars indicate ±1 SD of the mean.

Table 3

Summary of significant comparisons from post hoc permutational analysis of variance (PERMANOVA) of fork lengths for bottomfish restricted fishing area location (BR), protection (PR), habitat type (HA), and the interaction of these factors for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) within the preferred depths of each species from our study of these species in the main Hawaiian Islands between May 2007 and June 2009. Locations of the 6 BRFAs where sampling was conducted are the following: Niihau (B), Kaena (D), Makapuu (E), Penguin Bank (F), Pailolo Channel (H), and Hilo (L). Protection is designated as inside (in) or outside (out) a BRFA. Habitat types are hard-high (HH), hard-low (HL), soft-high (SH), soft-low (SL). NS=nonsignificant comparisons. Preferred depths are noted under the species name in the first column.

	BR	PR	HA	BR×PR	BR×HA	PR×HA	BR×PR×HA
Opakapaka (90–210 m)	Largest in B Smallest in L	Larger outside	Smallest in HL	NS	(D) largest in SH, smallest in HL (E) largest in SH, smallest in HL (F) largest in high slope, smallest in low slope (L) largest in SL, smallest in HL	NS	(D in) SH>HL (E in) HH,SH>HL (E out) SH>HH>HL (F in) HH,SH>HL (F out) HH,SH>SL (L in) HH>HL (L out) SL>HH>SH>HL
Kalekale (180–270 m)	Smallest in D	NS	Largest in HH	(F) larger inside (H) larger inside (L) larger outside	NS	NS	NS
Onaga (210–300 m)	Smallest in H	No test	NS	(H) larger inside	(B) larger in HL than HH (F) similar mean size	NS	(B in) HL>HH (F in) HH>HL
Ehu (210–300 m)	Similar mean size	No test	NS	NS	NS	NS	No test

tions in all size classes and did not show any habitat shifts with size (Pearson's chi-square, $P>0.05$). Opakapaka had a shift from hard-low habitats to hard-high habitats with an increase in size. There was a greater proportion of sexually mature individuals (≥43 cm FL; Kikkawa, 1984) for this species over hard-high habitats, and individuals <43 cm FL were seen mostly in hard-low habitats. Although less evident than the habitat shift by Opakapaka, a habitat shift by Kalekale to hard-high from other habitat types was observed within the size class of 25–35 cm. Onaga and Ehu were recorded mostly in hard-low habitats in all size classes. For Onaga, however, the smallest individuals (<55 cm FL) were found only in hard-low habitats, and, as size increased, hard-high habitats were equally dominant for this species.

Discussion

Depth has a significant influence on the distribution of bottomfishes in Hawaii. Two distinct depth groupings were seen within the sampling range of this study. Opakapaka was dominant in the shallower end of the sampling depths (<200 m), and Kalekale, Onaga, and Ehu were observed more frequently toward the deeper end (>200 m). This finding is consistent with that of previous studies in Hawaii (Haight, 1989; Everson et al., 1989; Merritt et al., 2011) and in the Mariana Archipelago (Polovina et al., 1985). When establishing species-specific differences in distribution, depth must be the first factor evaluated.

Although the limitations of our sampling methods have been discussed in previous studies (e.g., Merritt et al., 2011; Moore et al., 2013), it is important to review them here before further discussion of our results. The absence of a quantifiable sampling area, variability in the field of view of the BotCam, and the scale at which habitats were classified are confounding factors that limit the interpretation of the results of this study to a semiquantitative nature. Because the BotCam makes use of ambient light and because environmental conditions, such as water clarity can differ from site to site, variability in the visual area sampled was unavoidable. However, unlike other visual survey methods, where quadrats or transect lines are used, this approach reduces, but does not eliminate, the effect of visual area because it relies on attracting fishes close to the cameras. What may be more important is the effect of the attracting bait-odor plume.

It was our working assumption that any fish seen on BotCam video was from the targeted grid area

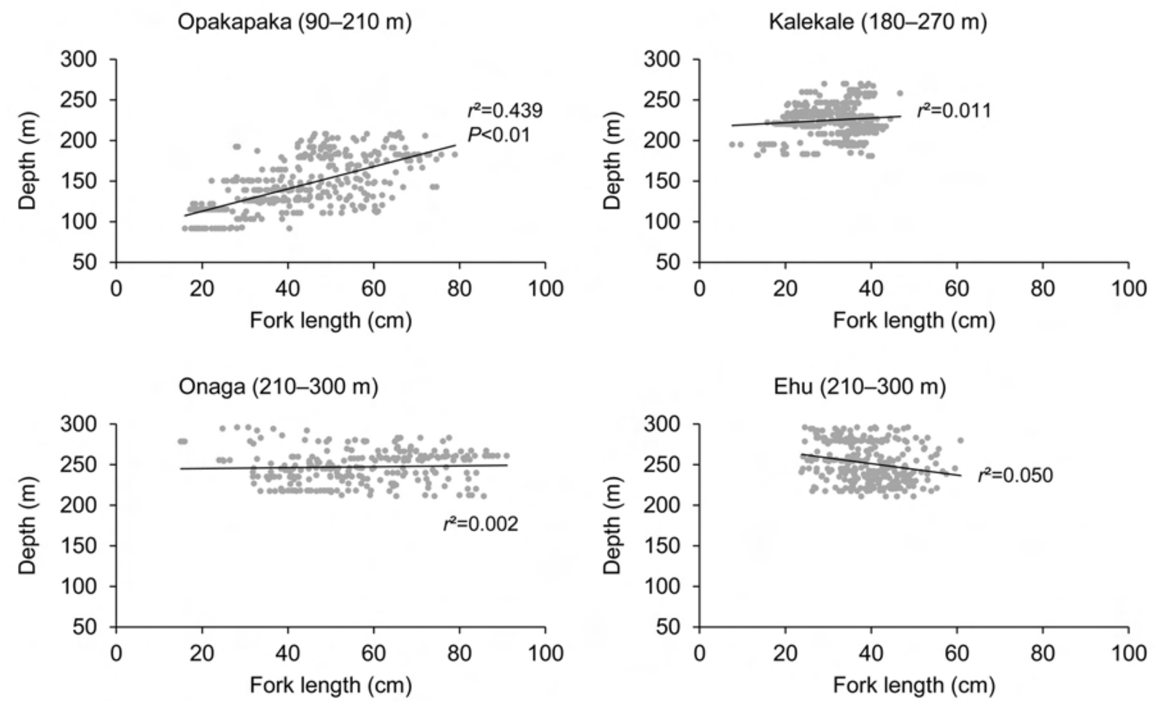

Figure 5

Comparison of fork length with depth for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) within the preferred depths of each species for our study of habitat associations of these species in the main Hawaiian Islands from May 2007 to June 2009. The coefficient of determination (r^2) is given for each species.

(200×200 m) regardless of the visual area observed in the video. This assumption was made on the basis of the limited information available on the distance of bottomfish attraction to bait stations. Ellis and DeMartini (1995) estimated that the greatest distance of attraction for juvenile Opakapaka to their baited cameras was between 48 and 90 m. Merritt et al. (2011), in their baited camera survey of Penguin Bank, used a 200-m distance between deployment locations to avoid a cross influence of bait.

The area of fish attraction (sampling area) has been quantified at abyssal depths by Priede and Merrett (1996) through the use of current velocity, fish swimming speed, and a bait dispersal model. Their determination of the area of attraction, however, relied on assumptions (i.e., fish are evenly dispersed) that do not apply to the fish species and shallower depth ranges in this study. Furthermore, bottom current variability, habitat variability, and small-scale bathymetric features at mesophotic depths around Hawaii make the quantification of the area of attraction to bait extremely challenging. In a comparison of baited and unbaited underwater video stations, Harvey et al. (2007) acknowledged that fish behavior and life history also may affect attraction to bait. All the species in this study are regularly attracted to bait and are taken on baited hooks, but other behavioral traits (e.g., mobility, schooling, and reproductive cycles) could affect species-specific responses to a bait-odor plume. Given the difficulty involved in the determination of the actual area of bait influence, the appropriateness of the habitat-classification scale chosen for use in this study cannot be evaluated. Until an effective scale of attraction can be verified for deepwater snappers and other bottomfishes, a fully quantitative assessment of species–habitat associations is not yet possible.

Although previous studies have indicated that habitats with hard substrates and high slopes, such as headlands and promontories, are preferred by many bottomfish species (Ralston and Polovina, 1982; Ralston et al., 1986; Parrish, 1987; Kelley et al., 2006; Parke, 2007), we determined that other habitat types, such as hard-low habitats, are important to eteline snappers and that species-specific differences in habitat preference exist. On the basis of relative abundance, we found that the overall habitat preference of Opakapaka was for low-sloping hard substrates. Onaga was associated with hard-high and hard-low habitats, and Ehu was seen mostly on hard-low habitats. The observed association of juvenile Opakapaka and Onaga with hard-low habitats may be driving their preference for this habitat type. In contrast, the finding for Ehu could

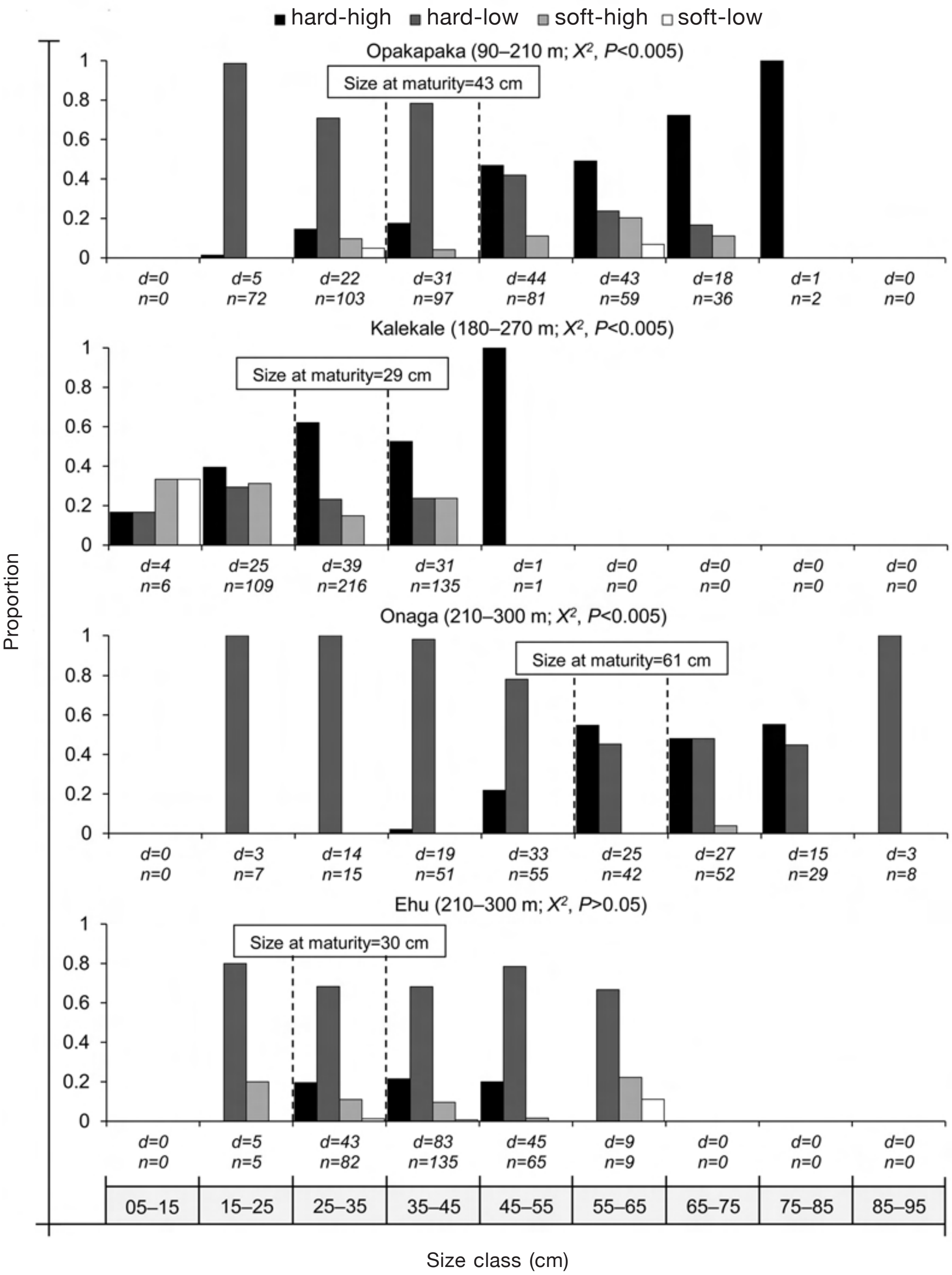

Figure 6

Proportion of fish found in each habitat type by size class tested with Pearson's chi-square (χ^2) test for Opakapaka (*Pristipomoides filamentosus*), Kalekale (*P. sieboldii*), Onaga (*Etelis coruscans*), and Ehu (*E. carbunculus*) within the preferred depths of each species in the main Hawaiian Islands. A baited stereo-video camera system (BotCam) was used to collect data from May 2007 to June 2009. The 4 habitat classifications used in our study were hard substrate with high slope (hard-high), hard substrate with low slope (hard-low), soft substrate with high slope (soft-high), and soft substrate with low slope (soft-low). *d*=number of BotCam deployments; *n*=number of fish measured. References for size at maturity: Kikkawa, 1984 (Opakapaka); DeMartini and Lau, 1999 (Kalekale); Everson et al., 1989 (Onaga); Everson, 1986 (Ehu).

have been the result of a sampling artifact caused by the lack of habitat types other than the hard-low environments in Pailolo Channel, where many observations of this species were made. Regardless, the results of this study clearly show the importance of this habitat type for Ehu. Kalekale were observed often in large schools in our video footage. For defense against predators, this species may rely on its schooling behavior instead of associating with the bottom habitat. The lack of a significant habitat preference for Kalekale could, consequently, be driven by this defense mechanism. Assessment of species–habitat associations, therefore, requires an understanding of species behaviors and the changes in habitat use by life stage.

Clear ontogenetic shifts in habitat associations were evident for 3 of the 4 species studied. For Opakapaka, there was a distinct ontogenetic progression in habitat association that expands what is known for this species. The known habitat for juveniles of this species at 7–25 cm FL is shallow, low-sloping, soft substrates (Moffitt and Parrish, 1996). Juvenile Opakapaka have been observed at depths of 65–100 m offshore of Kaneohe Bay (Parrish, 1989; Moffitt and Parrish, 1996) and more recently off Waikiki, Oahu, at depths of 37–42 m (J. Drazen, unpubl. data). These juveniles move out of their nursery grounds and presumably merge with the adult schools in deeper waters after about 1 year (Parrish et al., 1997). Within the preferred depth range identified in our study for Opakapaka (90–210 m), the smallest mean lengths were found over hard-low habitats at 4 of the 6 sampled locations. We recorded Opakapaka as small as 16 cm FL within our sampling depths over hard-low habitats. On the basis of growth curves from DeMartini et al. (1994), the juvenile Opakapaka in our study were just under 1 year old and could be recent migrants from a surrounding nursery area. The results of this study show that these fish continue to stay in hard-low habitats until they reach 45 cm FL or about 5 years of age and, thereafter, increasingly use hard-high habitats. It is possible that this species uses hard substrates with low slopes as a transitional habitat before a move into hard-high habitats. Opakapaka reaches sexual maturity at ~43 cm FL (Kikkawa, 1984). The shift in habitat from hard-low to hard-high could be a response to reproductive maturity, which is discussed later.

Size-related habitat shifts also were evident for Kalekale and Onaga but were observed without a change in their depth of occurrence. Previous studies also showed a lack of depth change with size for these species (Kelley et al[4]; Ikehara, 2006). The move into hard-high habitats with increasing size coincided roughly with the onset of sexual maturity in both species. The size (25–35 cm FL) at which Kalekale shifted to hard-high habitats from other types includes the size (29 cm FL) at which this species reaches sexual maturity (DeMartini and Lau, 1999). The onset of sexual maturity for Onaga occurs at 61 cm FL (Everson et al., 1989)—a size larger than the size (55 cm FL) at which a shift in habitat use was observed in our study. On the basis of size-at-age curves, the onset of sexual maturity occurs between the ages of 3 and 6 years for Kalekale (Williams and Lowe, 1997) and 5 to 6 years for Onaga (Everson et al., 1989).

In contrast to the other 3 species, no size-related habitat shifts were observed for Ehu, but very few juveniles of this species were measured (Fig. 6; n=37). Juvenile Ehu, along with other smaller bottomfishes, are highly vulnerable to predation by demersal carnivores, such as the Greater Amberjack (*Seriola dumerili*) (Humphreys and Kramer, 1984). A few instances where Greater Amberjack seemed to scare away Kalekale and Ehu were observed in the BotCam video collected during our study. No aggressive behavior toward the target species by other predators was seen, but it is possible that carnivorous species could have affected our ability to sample certain size ranges of bottomfishes, particularly Ehu. Smaller snappers may have moved out of the BotCam's field of view before predators entered. Even if they were possibly in the vicinity of the BotCam, juveniles may have remained close to the bottom of the seafloor for protection and out of the unit's field of view. Until very small Ehu (i.e., 5–15 cm FL) can be observed regularly, a complete ontogenetic assessment of habitat for this species will not be possible. However, it is important to note that the size range of Ehu harvested by the fishery is represented in this study.

The ontogenetic habitat shifts observed for Opakapaka, Kalekale, and Onaga could be related to shifts in diet, increases in reproductive output, and predator avoidance at smaller sizes. Szedlmayer and Lee (2004) reported a shift in the diet of the shallow-water juvenile Red Snapper (*Lutjanus campechanus*) from crustaceans to fishes and cephalopods with increasing size. This change in diet was associated with the migration from nursery habitats to coral reefs. For deepwater snappers, diet shifts have yet to be documented. DeMartini et al. (1996) examined the diet of juvenile Opakapaka from the nursery in Kaneohe Bay and discovered that it was composed of crustaceans (shrimps and stomatopods), gelatinous organisms (salps and heteropods), nekton (fishes and squids), and benthic organisms (demersal octopods, echinoids, and microgastropods). With the exception of benthic prey, a similar diet was found for Opakapaka caught at depths of 100–300 m in Penguin Bank by Haight et al. (1993b). It is possible that smaller individuals (<43 cm FL) of this species associate with low-sloping, hardbottom habitats to feed on the benthos and then shift to a pelagic diet when they move into hard-high habitats where the postulated increase in water flow increases prey availability (Ralston et al., 1986; Haight et al., 1993a; Kelley et al., 2006).

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[4] Kelley, C. D., B. C. Mundy, and E. G. Grau. 1997. The use of the Pisces V submersible to locate nursery grounds of commercially important deepwater snappers, family Lutjanidae, in Hawaii, 62 p. Paper presented at the 5th Indo-Pacific Fish Conference; Nouméa, New Caledonia, 10–16 November.

With the hypothesis that the levels of bottomfish prey and current speed are greater over hard-high habitats than over other environments (Ralston et al., 1986; Haight et al., 1993a; Kelley et al., 2006), it could be inferred that Opakapaka, Kalekale, and Onaga move into this habitat type upon reaching sexual maturity to increase their foraging rates and maximize reproductive output and gamete dispersal. On coral reefs in Hawaii, the Yellow Tang (*Zebrasoma flavescens*) has been found to shift into habitats with increased food resources when it reaches reproductive size to possibly improve its reproductive ability (Claisse et al., 2009). No actual bottomfish spawning events were recorded during our study. Opakapaka and Onaga are known to spawn at night (C. Kelley, unpubl. data), and camera deployments were restricted to daytime hours. Other than seasonality, habitat and environmental parameters of bottomfish spawning have yet to be determined. It remains possible, however, that the observed ontogenetic habitat shifts occurred as a result of a reproductive cue—given that the change in habitat roughly coincided with sexual maturity.

Another factor that may influence ontogenetic habitat shifts is habitat complexity. Laidig et al. (2009) found that juvenile rockfishes on the continental shelf off central California were associated with boulder and cobble habitats before they moved into the slope habitats used by adults. It is plausible that juveniles and smaller species of bottomfishes use more complex habitats in a similar manner for protection and predator avoidance. However, because habitats were classified at a 200-m scale, our study did not take into account habitat heterogeneity within grid cells and smaller-scale habitat characteristics, such as complexity or rugosity. Structural complexity and the combination of habitat types in a given area are likely to influence fish distributions at their respective scales. Future work is needed to investigate the role of habitat complexity and heterogeneity on size distributions of bottomfishes and to look more closely into how specific habitat types are used. Such an approach could provide more information about the cause of the ontogenetic habitat shifts observed in this study.

The regional variations in relative abundance and mean length could be related to differential fishing pressure or large-scale habitat features. It can be expected that remote locations, such as Niihau, would have less fishing pressure than locations closer to major ports and, thereby, would have greater relative abundances and lengths of target species. Contrary to this expectation, the highest levels of relative abundance were found at Hilo for Opakapaka and in Pailolo Channel for Ehu. Both areas are easily accessible to fishing; therefore, other factors may have driven the observed distributions. Protection did not have an influence on the relative abundance of any of the 4 species studied, a finding that is consistent with the results of Moore et al. (2013). In terms of mean length, the largest Opakapaka may have been found at Niihau because of the remote location and longevity of the protection of this small island. The Niihau BRFA has been closed to fishing since 1998. The opposite may be true for Hilo, where the smallest Opakapaka were observed. Before the implementation of the revised system of BRFAs, fishing in the depth range of Opakapaka was permitted because the BRFA boundary began at 200 m. How protection and fishing pressure affect abundance and size distributions of bottomfishes should be investigated further because these factors may confound any trends attributed to habitat or other environmental variables.

Mega-scale habitat features (scale from Greene et al., 1999: macro=1–10 m; meso=10–1000 m; mega=1–10 km), such as pinnacles, banks, terraces, and even featureless carbonate flats, also could be influencing bottomfish distributions. In this study, juvenile Opakapaka and Onaga were found to associate with hard-low habitats. There is a large terrace at Hilo, where most juvenile Opakapaka were observed, and flat, hardbottom habitats predominate in Pailolo Channel, where most Onaga juveniles were present. These large-scale features predominantly have low slopes and hard bottoms and match the observed habitat preference of these species at the meso-scale. However, because of the difference between the habitat classification scale (200×200 m) used in our study and the size of mega-scale features, further investigation is required to establish a conclusive connection between the bottomfish distributions observed in this study and mega-scale features. In the case of Pailolo Channel, for example, with its large, flat areas of hardbottom habitat, our results agreed with a finding of another survey effort. Previous fishing surveys have indicated that this area possibly was a nursery ground for Onaga (C. Kelley, unpubl. data). Because the smallest mean length (42.80 cm FL) and about 75% of all juveniles of this species measured (<61 cm FL) in this study came from Pailolo Channel, it is highly likely that a nursery ground for Onaga exists in this area.

Conclusions

This study has improved our understanding of the species-specific ecology of 4 bottomfish species in the MHI. Analyses of habitat preferences on the basis of relative abundance and length-frequency distributions showed that habitat types other than hard-high environments are important to each of the species studied, often as a result of ontogenetic shifts in habitat use. Given that these bottomfishes are found throughout the Indo-Pacific region, these findings may provide the framework for the prediction of species distributions outside of Hawaii. Because juveniles of Opakapaka and Onaga were associated mostly with hard-low habitats, it is imperative that future definitions of the bottomfish EFH take into account habitat associations by life stage. Although some species share similar preferences, it also is clear that bottomfish distributions are species-specific and cannot be generalized for all members of the bottomfish fishery in Hawaii. Because it has increased our knowledge of the ecology of individual species, the results of this study can aid in the improvement of ecosystem-based management strategies and definitions of species-specific EFHs. Moving forward, to further improve our understanding of the habitat requirements of bottomfish species in Hawaii, research on bottomfish habitat should focus on development of models to determine the dispersal range of bait-odor plumes, identification of the effective scale of attraction to bait stations, standardization of sampling areas, and inclusion of habitat heterogeneity and macroscale habitat characteristics in future analyses of bottomfish distributions.

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BotCam: a baited camera system for nonextractive monitoring of bottomfish species

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Abstract—A stereo-video baited camera system (BotCam) has been developed as a fishery-independent tool to monitor and study deepwater fish species and their habitat. During testing, BotCam was deployed primarily in water depths between 100 and 300 m for an assessment of its use in monitoring and studying Hawaiian bottomfish species. Details of the video analyses and data from the pilot study with BotCam in Hawai`i are presented. Multibeam bathymetry and backscatter data were used to delineate bottomfish habitat strata, and a stratified random sampling design was used for BotCam deployment locations. Video data were analyzed to assess relative fish abundance and to measure fish size composition. Results corroborate published depth ranges and zones of the target species, as well as their habitat preferences. The results indicate that BotCam is a promising tool for monitoring and studying demersal fish populations associated with deepwater habitats to a depth of 300 m, at mesohabitat scales. BotCam is a flexible, nonextractive, and economical means to better understand deepwater ecosystems and improve science-based ecosystem approaches to management.

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The ability to monitor stocks targeted by a fishery in order to understand the effects of regulatory measures, such as spatial or temporal fishing closures, is important to stakeholders. An understanding of species composition, age- and size-class distributions, habitat use, and other population parameters is critical for developing resource management programs and for monitoring their effectiveness (Jennings, 2001). However, acquisition of data for stock assessments within, and adjacent to, marine protected areas (MPAs) may be compromised by restrictions on extractive sampling or fishery-dependent data. Further, monitoring deepwater species is challenging because of limitations (both logistical and regulatory) on diving in deep water; catch-and-release, or other nonlethal techniques typically are used in shallow water. Because deepwater fisheries have developed rapidly over the last few years, it is important to develop reliable, nonextractive, and fisheries-independent methods for stock assessment and monitoring that will enable managers to assess fishery impacts, evaluate MPAs, and implement ecosystem-based management (Roberts, 2002).

Camera systems provide a fisheries-independent and nonextractive tool for monitoring fish stocks, associated communities, and habitat preferences. Baited camera systems have been used in a number of fisheries habitat studies (Ellis and DeMartini, 1995; Gledhill et al., 1996; Priede and Merrett, 1996; Francour et al., 1999; Willis et al., 2000; Cappo et al., 2003). Most of these studies involved deepwater deployments (>1500 m) for the study of deep-sea scavengers or they involved deployments in relatively shallow waters (<100 m) as a supplement to scuba surveys (Willis et al., 2000; Watson et al., 2007). Currently, there is a need to develop systems for use at intermediate depths.

In Hawai'i, the bottomfish fishery targets snappers, groupers, and jacks that inhabit waters down to 400 m around the archipelago. The most important commercial species live below 100 m and are often referred to as the “deep 7” (WPRFMC, 2007). Six of these are snappers that include *Etelis coruscans* (flame snapper, onaga), *Etelis carbunculus* (ruby snapper, ehu), *Pristipomoides zonatus* (oblique-banded snapper, gindai), *Pristipomoides sieboldii* (lavender snapper, kalekale), *Pristipomoides filamentosus* (pink snapper, opakapaka), and *Aphareus rutilans* (silvermouth snapper, lehi). The seventh species is an endemic grouper called *Epinephelus quernus* (Hawaiian grouper, hapu`upu`u)

(Randall, 2007). Most of these species are long-lived, slow-growing, and are assumed to have a low annual natural mortality rate and limited reproductive capacity (Haight et al., 1993a). These characteristics make these bottomfish stocks especially susceptible to overfishing and habitat destruction (Ralston et al.[1]).

The Hawaiian bottomfish fishing is primarily conducted by jigging hooks and lines on motorized reels. All of the deep 7 species eat a variety of fish and invertebrate species opportunistically. For example, *E. coruscans* are known to feed on species within the water column near the bottom, whereas *E. carbunculus* targets species on the bottom. All target species are caught by using both fish, such as mackerel (*Decapterus* spp.) and invertebrates (such as squid) as bait. Fishing vessels that anchor will often use a *palu* bag containing a mixture of baits.

Although the entire range of depths used by the Hawai`i deepwater bottomfish assemblage has not been determined, the Western Pacific Regional Fishery Management Council (WPRFMC) has defined the deepwater bottomfish essential fish habitat as all depths between 100 and 400 m, and adult habitat areas of particular concern as slopes and escarpments between 40 and 280 m depth (WPRFMC, 1998). Low light levels at these depths complicate the use of cameras. However, surveys with submersibles and remotely operated vehicles (ROVs) indicate that ambient lighting is preferable to artificial area lights or strobes because the artificial lights may repel or attract target species (Ralston et al., 1986; Ryer et al., 2009).

To address the need for a nonextractive, fishery-independent method for monitoring Hawaiian bottomfish stocks, a baited stereo-video camera system (BotCam) has been developed by the National Oceanic and Atmospheric Administration's Pacific Islands Fisheries Science Center (PIFSC) in collaboration with the Hawai`i Undersea Research Laboratory. BotCam is designed to survey the distribution, relative abundance, and size composition of bottomfish, and associated biological and physical characteristics of their habitat.

A pilot study was designed to test BotCam as a tool in making stock assessments. The main purpose of the study was to determine whether, from an operational perspective, BotCam can consistently and reliably collect the same types of data collected by other baited stereo-video camera systems, as reported in the literature, on the commercially important Hawaiian bottomfishes. More specifically, we asked if the system could obtain a metric of relative abundance, accurate information on habitat associations, and a length-frequency distribution for fish of a given fishery.

Materials and methods

Baited stereo-video camera system

BotCam was designed as a fully autonomous baited stereo-video camera system (Merritt, 2005). Most of the components are housed in an aluminum frame (1.2 m wide×0.5 m deep×0.45 m tall) designed to protect the cameras and maintain fixed camera positions to one another for accurate length measurements (Fig. 1). The system consists of two ultralow-light video cameras (Monochrome Navigator, Remote Ocean Systems, San Diego, CA), the video capture electronics and system controller (Viperfish Deep, Deep Development Corporation, Sumas, WA), a temperature and pressure recorder (SBE 39TP, Seabird Electronics Inc., Bellvue, WA), a custom-built battery pack and relay used to trigger a delayed bait release-system (BWR, Sexton Photographics LLC, Salem, OR), and syntactic foam blocks for positive buoyancy (Flotation Technologies, Biddeford, ME). The frame also allows for the attachment of oceanographic instruments such as current meters, temperature and depth recorders, and hydrophones. The system is moored to the bottom by anchor weights attached to an anchor line and is designed to float above the bottom and to record video by pointing horizontally down-current with a nominal downward angle of 15°. This orientation improves the view of the benthic habitat without sacrificing the field of view. Each camera provides an 80° diagonal field of view in water. Because of the depth of targeted deployments, motions of the floating system are not affected by surface waves and the platform moves only by means of the currents, which are generally driven by tides, and are therefore stable on the order of several minutes. BotCam does often rotate and change the field of view relative to the substratum over the duration of a deployment. This floating design was chosen to address a couple of concerns. First, the target species are known to school in the water column several meters above the bottom. Second, the habitat of these target species is found on extremely steep and rocky slopes and setting a system directly on the bottom would be problematic for both the deployment and recovery of the system. An extension arm attached to the frame can carry both a stereo-video synchronizing (SVS) device and a bait canister or bag in view of the cameras (Fig. 1). The SVS, a grid of lights that flash in rapid succession, was custom made by Sexton Photographics LLC (similar to a system used by Harvey and Shortis (1996)) and allows two video streams to be synchronized by time for accurate stereo-video measurements. The lights flash at 30 Hz for 1 second every minute and no reaction to the lights has been observed by any of the target species. The first of two baiting modes involves simply attaching a bait bag or trap feeder to the extension arm. The second method involves the use of a 1.7-L Niskin bottle to hold bait sealed inside; at a predetermined time the bottle opens, exposing the bait.

An acoustic release (AR701, Ixsea, Boston, MA) was placed between the bottom of the frame and a set of two

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[1] Ralston, S., S. Cox, M. Labelle, and C. Mees. 2004. Western Pacific Regional Fishery Management Council bottomfish stock assessment workshop final panel report; January 13–16, 20 p. [Available from Western Pacific Fishery Management Council, 11643 Bishop Street, Suite 1400, Honolulu, HI 96813.]

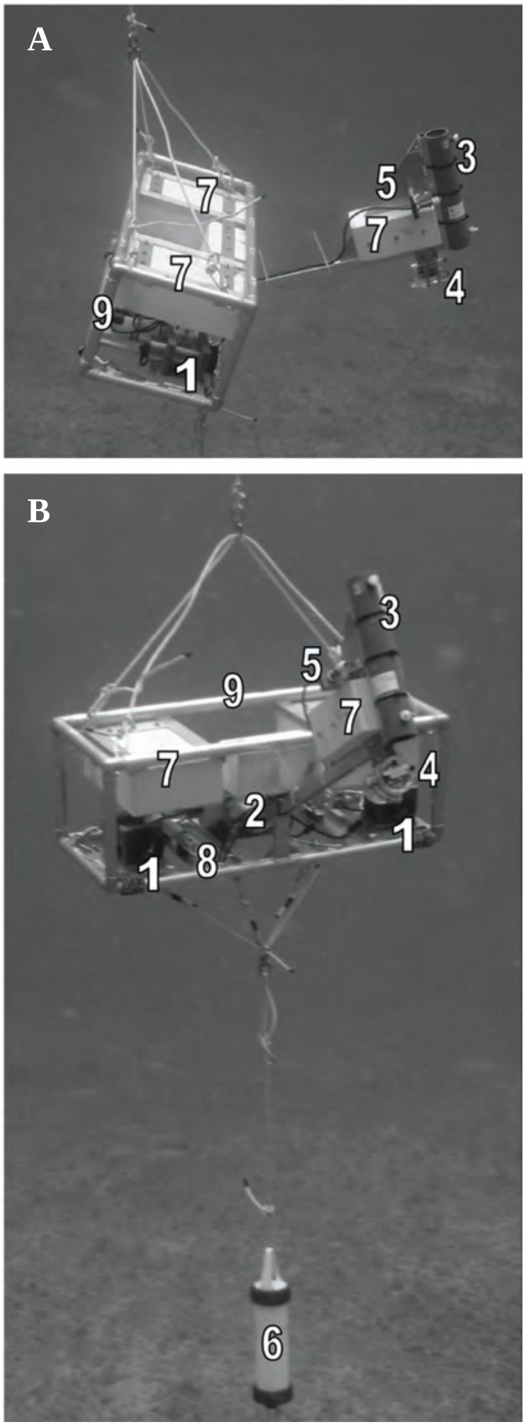

Figure 1

(**A**) side view and (**B**) front view of stereo-video baited camera system (BotCam). Components include (1) ultralow-light video camera, (2) controller-power supply-video capture device, (3) bait container, (4) stereo-video synchronization device, (5) bait release system, (6) acoustic release, (7) syntactic foam flotation, (8) pressure and temperature sensor, (9) aluminum frame. Not shown below the acoustic release is the anchor (concrete blocks).

or three concrete blocks that served as the sacrificial anchor. Concrete was used because it is environmentally benign, inexpensive, and readily available. BotCam was set to float 3 m above the seafloor, thus allowing deployments along steep, rocky slopes without risking entanglement of the instrument on the bottom. It was recovered when it floated to the surface after the acoustic release was triggered to separate the sacrificial anchor from the buoyant instrument frame. The instrument can also be tethered to a surface buoy to allow recovery by a line haul.

The complete system, as used during the pilot study, cost approximately $40,000; however, the systems being used presently with very similar capabilities are about $25,000 per unit. The largest single expense is the pair of ultra-lowlight cameras. In addition, charter time for an appropriate survey vessel in Hawaii runs about $1000 per day.

Study design

During its development, BotCam was tested in approximately 50 deployments around Hawai`i, Wake Atoll, Guam, and the Commonwealth of the Northern Mariana Islands at depths down to 400 m. It was determined that 300 m was the maximum reliable deployment depth under ambient light conditions that would allow accurate species identification and sizing. Further, it was determined that by using a 30- to 60-minute recording time, a single BotCam unit could be deployed, recovered, and ready for redeployment in 90 minutes (Merritt, 2005). Ten- to 60-minute deployments are also consistent with other shallow baited camera studies (Ellis and DeMartini, 1995; Willis et al., 2003).

Given these constraints and a limited number of available charter vessel days, a study site was selected relatively close to Honolulu, home port for the charter vessel and the Pacific Islands Fisheries Science Center. The site was centered on bottomfish habitat located along the west side of Penguin Bank, between the Hawaiian Islands of Oahu and Molokai. Penguin Bank has historically been a productive bottomfish area and its proximity to the highly populated island of Oahu has resulted in high fishing pressure on both the east and west sides of the bank (Haight et al., 1993b).

Previous studies with submersibles and anecdotal evidence from bottomfish fishermen have indicated that the deep 7 bottomfish species generally prefer high-slope, hard-bottom habitats (Kelley et al., 2006; Parke, 2007), which are present at Penguin Bank. Twenty-meter resolution bathymetry and backscatter data derived from multibeam sonar were available for the entire study area and were incorporated into a geographic information system in order to derive intersections of depth, slope, and substratum hardness (i.e., backscatter). The upper and lower depth boundaries for BotCam deployments were 100 and 300 m, respectively, set by the biological and logistical constraints given above, with a resulting sampling area of 24.9 km^2. Within this depth range, four habitat types were defined on the basis of intersecting substratum (bottom) hardness and slope: 1) hard bottom–high slope (HB–HS); 2) hard bottom–low slope (HB–LS); 3) soft bottom–high slope (SB–HS); and 4) soft bottom–low slope (SB–LS). High slope values were considered to be 20 degrees or greater and hard substrata had backscatter values equal to or greater than 41 on a scale of 0–100 (actual maximum measurement was 92). The sampling locations were randomly selected within these four habitat types and weighted towards the preferred bottomfish habitat. A total of 38 sites were sampled on HB–HS, 14 on HB–LS, 17 on SB–HS, and 13 on SB–LS. In this way greater replication was performed where fish densities were expected to be higher and replication was lower where few or no fish were expected to be found. Adjacent sampling locations were no closer than 200 m and to avoid cross influence of the bait, no two adjacent sites were sampled on the same day.

The BotCam system was set to begin recording after its release from the boat but before its arrival on the bottom. For each deployment, the recording period was between 45 and 60 minutes. The bait consisted of equal parts of ground squid and mackerel, and the volume of bait used for each deployment was standardized to approximately 1 liter. This mixture was designed 1) to be similar to what bottomfish fishermen typically use on their rigs; 2) to provide multiple types of scent; and 3) to provide food similar to the natural diets of the "deep 7" which include both fish and cephalopods (Haight et al., 1993b).

The bait was placed in a simple plastic mesh container that allowed the bait scent to disperse as soon as the system was placed in the water. The bait station was considered to have started when BotCam arrived at the seafloor, as determined from the video recording. From that point, the cameras were allowed to record for a minimum of 30 minutes before BotCam was recovered.

Data analysis

Each video stream from the two cameras was viewed independently. Each video was viewed in 3-minute intervals to allow for flexibility in analyzing the data. The data from the 10 intervals per 30-minute station could be combined into larger intervals or a subset could be randomly selected for statistical comparison with data from other bait stations. The maximum number (MaxNo) of each species seen in any one frame within the time interval (Ellis and DeMartini, 1995) and the exact time from the start of the deployment to the time of first arrival (TFA) of each species seen over the entire 30 minutes were recorded. Further, the largest MaxNo from all the increments was noted as the MaxNo for the deployment for each species observed.

For the purposes of this study, enumeration and measurements were performed only for the two primary bottomfish species of interest, *P. filamentosus* and *E. coruscans*, which were also the two most frequently observed of the "deep 7" species and represent the majority of the bottomfish catch in the Hawaiian Islands (Haight et al., 1993a; Parke, 2007).

Bottomfish fork-length measurements were made from the video recordings by using a software package called Visual Measurement System (SVS) (Geomsoft, Victoria, Australia). With this software, the video streams were synchronized by time by using the SVS device, and then viewed simultaneously frame by frame. Measurements of lengths for *E. coruscans* and *P. filamentosus* were conducted by using the MaxNo video frame and adjacent frames to avoid repeat measurement of individual fish congregating around the bait. Each individual fish was measured six times from different video frames to evaluate the consistency of the measurement technique. This method of only measuring at MaxNo may bias the data by possibly selecting for smaller schooling fish (Willis et al., 2003).

To specifically test the precision and accuracy of the stereo-photogrammetric method of fish measurement, a separate experiment was performed in shallow water. BotCam video was used to measure four different fish models (foam cutouts shaped like fish) of varying length (469.9 mm, 581.0 mm, 628.7 mm, and 997.0 mm) and body depth. The models were filmed at various locations in the field of view at distances of 3 m and 6 m from the cameras. The BotCam was rotated by a diver so that the fish traversed the field of view to simulate swimming. The models were moved vertically to obtain coverage of the models throughout the fields of view of the cameras and the models were measured at haphazard angles. Length measurements on each fish were made by three scientists using stereophotometric software.

The relative distributions of each species across substratum and slope categories described above were evaluated within the framework of a generalized linear model based on a Poisson distribution and log-link function. The model development for predictor variables was based on likelihood ratio tests with a comparison of the full and reduced models. A Pearson chi-square goodness-of-fit test was used to evaluate the appropriateness of the model fits (Kutner et al., 2005). Model fitting included habitat and depth categories and their two-way interaction.

Results

Thirty-three sampling trips were conducted between June 2006 and February 2007, on which a total of 102 BotCam deployments were completed. The fabrication of a second BotCam system toward the end of the study increased the average number of deployments per boat trip to 5.5. Six to eight drops could easily be conducted per day depending on travel time from port to the deployment sites. Of the 102 BotCam deployments, 82 were successful and were distributed amongst habitat and depth categories as outlined above (Table 1). Of the 20 that failed, four landed below 300 m so their recording was too dark; four landed above 100 m outside the

Table 1

Number of baited stereo-video camera (BotCam) deployments that fell within the 100-m to 300-m depth contours and recorded video at Penguin Banks, Hawai'i, between June 2006 and February 2007. Deployments are separated by habitat classification (substratum and slope), depth by 50-m bin, and time period, and the average maximum number (AveMaxNo) and standard error (SE) of counts of *Etelis coruscans* and *Pristipomoides filamentosus* by habitat type and depth. na=not available.

Multibeam habitat classification	Depth (m)	Sample size Total	Jun 6	Jul 6	Aug 6	Dec 6	Feb 7	*Etelis coruscans* AveMaxNo	SE	*Pristipomoides filamentosus* AveMaxNo	SE
Hard bottom–high slope	100–150	3	0	0	3	0	0	0.0	na	1.3	1.3
	150–200	9	1	6	2	0	0	0.0	na	3.0	1.5
	200–250	16	1	10	5	0	0	1.9	1.0	1.5	0.6
	250–300	10	1	4	5	0	0	6.1	3.0	0.0	na
Soft bottom–high slope	100–150	1	0	0	0	0	1	0.0	na	1.0	na
	150–200	2	1	1	0	0	0	1.0	1.0	4.5	0.5
	200–250	5	0	1	0	2	2	0.0	na	1.8	1.2
	250–300	6	1	1	1	1	2	0.2	0.2	0.0	na
Hard bottom–low slope	100–150	1	0	0	0	0	1	0.0	na	4.0	na
	150–200	6	0	4	1	0	1	0.3	0.2	5.5	4.3
	200–250	6	0	2	1	1	2	1.7	1.6	1.4	0.5
	250–300	4	2	1	1	0	0	4.3	4.3	0.0	na
Soft bottom–low slope	100–150	2	0	2	0	0	0	0.0	na	0.0	na
	150–200	1	0	0	0	1	0	0.0	na	5.0	na
	200–250	6	0	0	0	6	0	0.7	0.7	1.0	0.8
	250–300	4	0	2	0	1	1	0.8	0.8	0.0	na

Hawaiian bottomfish essential fish habitat; nine did not record because of technical failures; and three failed as a result of human errors. No equipment was lost during the study.

All of Hawaii's "deep 7" bottomfish species were recorded on videotape (Fig. 2). Other species of note observed included goldflag snapper (*Pristipomoides auricilla),* greater amberjack (*Seriola dumerili*), large-head scorpionfish (*Pontinus macrocephalus*), dawn boarfish (*Antigonia eos*) (Randall, 2007), shortspine spurdog (*Squalus mitsukurii*), and numerous carcharhinid sharks. The appearances of each species under ambient light conditions were noted, and a photo library of BotCam videotapes was developed for species identification.

MaxNo values for *E. coruscans* and *P. filamentosus* recorded by BotCam varied between 0 and 29. MaxNo distributions for the two species across the study area are shown in Figure 3, A and B, respectively. *Etelis coruscans* was recorded at 21 locations and *P. filamentosus* at 30 locations and both species were present throughout the study area. No linear relationship between MaxNo and TFA was detected, although the apparent pattern for both species was similar (Fig. 4). For both species, most TFAs were less than 200 seconds (3.3 minutes) and all MaxNos higher than five were reached within the first 200 seconds.

Depth and the interaction of depth and habitat significantly affected *E. coruscans* MaxNo ($P<0.05$). The greatest MaxNo of *E. coruscans* was reached at depths between 250 and 300 m ($P<0.01$, Fig. 5A). Within this depth category, greater mean MaxNo for *E. coruscans* were found in habitats with a slope greater than 20 degrees with either hard or soft bottom substratums ($P<0.05$, Fig. 5A). *Pristipomoides filamentosus* was more widely distributed than *E. coruscans* across the sampled depth range and substratum types. Habitat, depth, and their interaction significantly affected the MaxNo for *P. filamentosus* ($P<0.05$). The interaction of depth and slope significantly affected the MaxNo for *P. filamentosus* with the highest MaxNo observed between 150 and 200 m regardless of habitat type ($P<0.01$, Fig. 5B). No significant relationships were found between temperature and the MaxNo for either species ($r^2<0.10$, $P>0.05$).

In the experiment where model fish were measured, the average residual measurement error (the difference between the actual measurement and the measurement estimated from the photos) of the stereo-photogrammetric analysis was –3.1 mm (percent error of 0.5%) when the models were a distance of 3 m from the camera, and –8.8 mm (percent error of –1.3%) when models were 6 m from the camera. However, the percent error does not appear to be a function of fish size within the range of models measured; therefore, the residual error appears to be a more relevant statistic to use when assessing variance (Table 2).

In the video analysis from the actual survey, it was possible to measure 56 individual *E. coruscans* out of

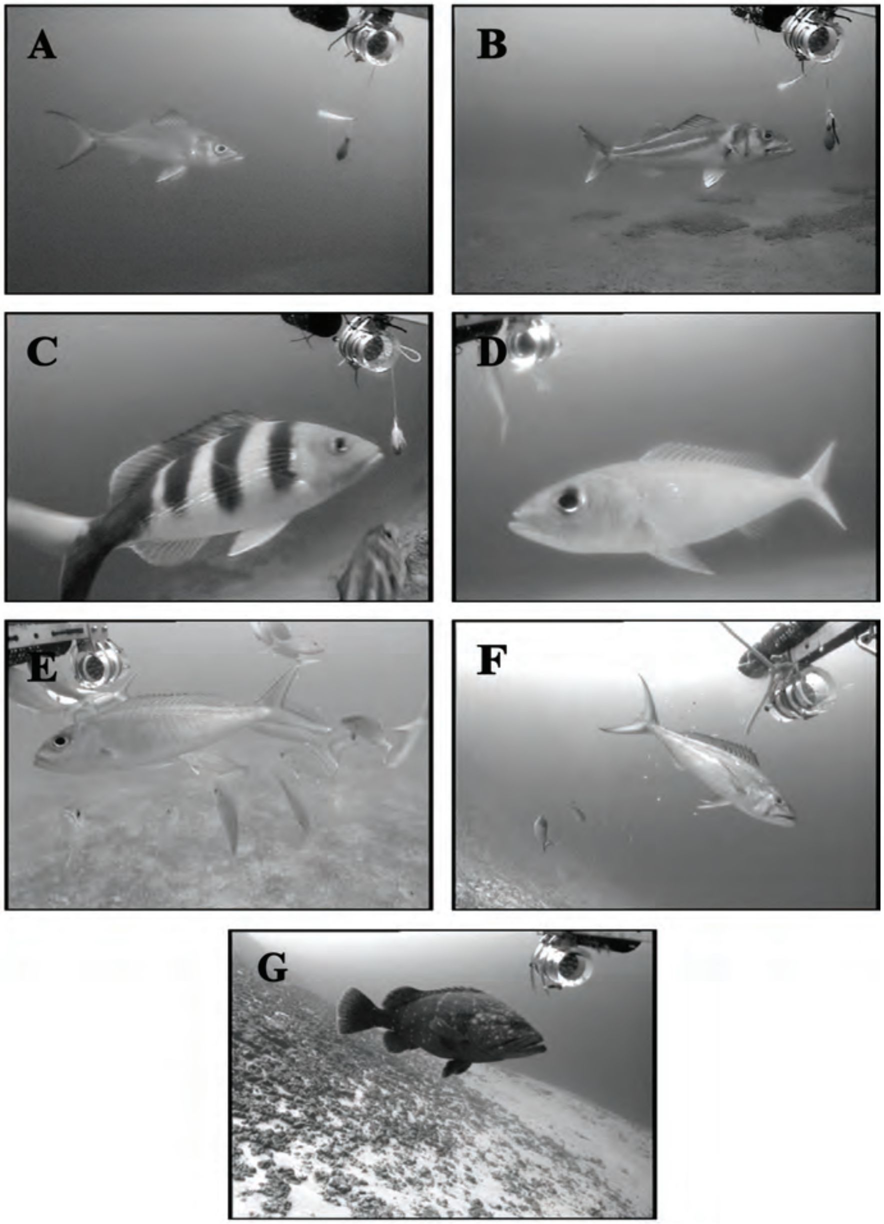

Figure 2

Hawaiian deepwater bottomfish fishery target species referred to as the "deep 7" as recorded by BotCam in Hawaiian waters from depths between 100 m and 300 m. (**A**) *Etelis coruscans* (longtailed red snapper or onaga), (**B**) *Etelis carbunculus* (red snapper or ehu), (**C**) *Pristipomoides zonatus* (Brigham's snapper or gindai), (**D**) *Pristipomoides sieboldii* (von Siebold's snapper or kalekale), (**E**) *Pristipomoides filamentosus* (pink snapper or opakapaka), (**F**) *Aphareus rutilans* (ironjaw snapper or lehi), and (**G**) *Epinephelus quernus* (Hawaiian grouper or hapu`upu`u).

129 counted at the time of MaxNo (43%), and to measure 78 *P. filamentosus* out of the 134 counted (58%). The ability to measure a fish was constrained by the angle of orientation of the fish to the camera, distance from the camera, amount of overlap with other fish, and video clarity. *Etelis coruscans* fork lengths ranged between 432 and 833 mm (mean ±standard deviation [SD] = 605.7 ±26.8 mm, Fig. 6A), and *P. filamentosus* fork lengths ranged between 344 and 660 mm (mean ±SD = 518.0 ±10.9 mm, Fig. 6B).

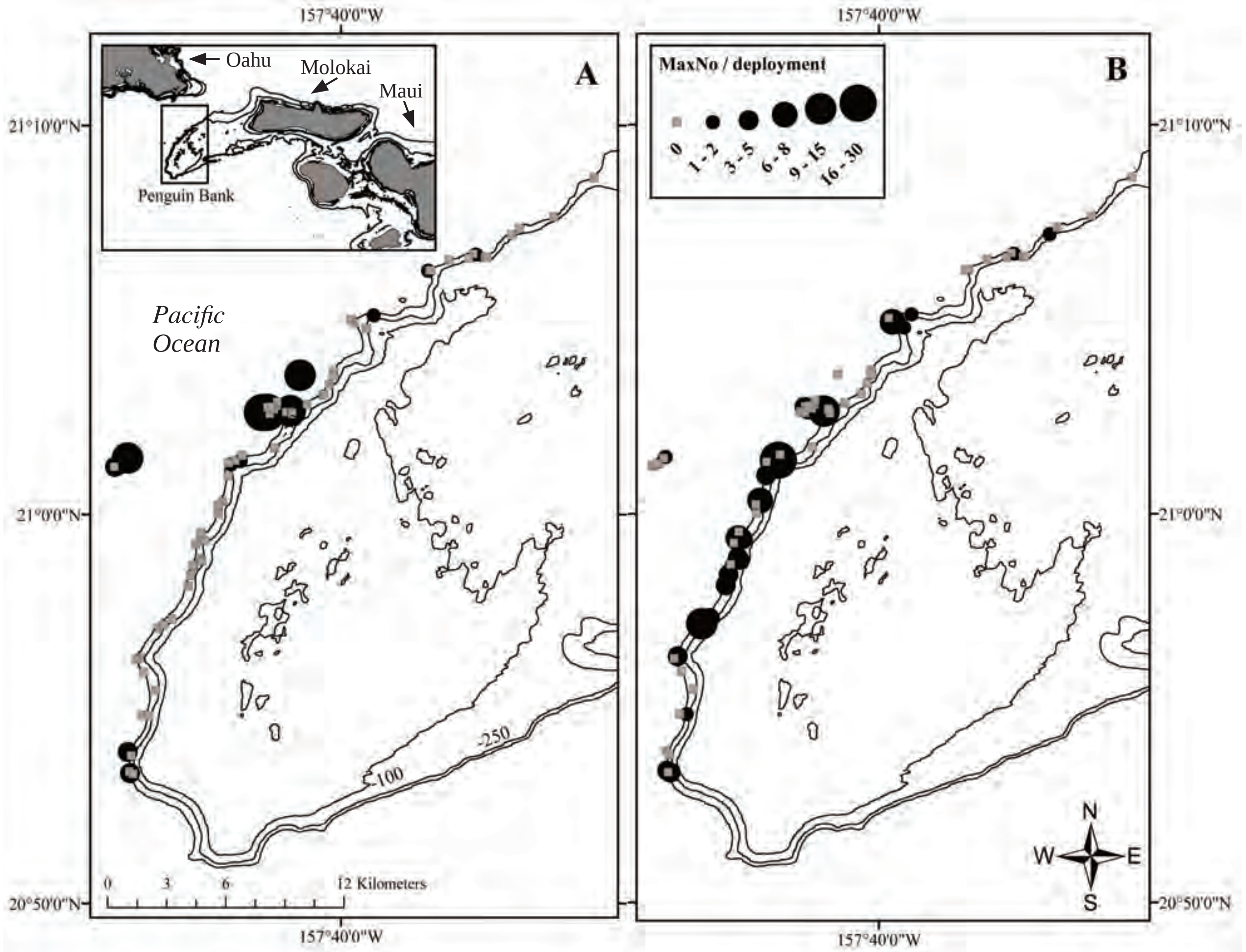

Figure 3

Distribution of (**A**) *Etelis coruscans* and (**B**) *Pristipomoides filamentosus* seen on the BotCam video at Penguin Banks, Hawai`i, between June 2006 and February 2007. Shown is the MaxNo (maximum number in a single frame) of each species seen at each camera deployment site, and the location of all 82 successful deployments.

Table 2

Measurement statistics for testing the precision and accuracy of the stereo-video camera system. A BotCam video camera was used to measure four different models of fish of varying length (469.9 mm, 581.0 mm, 628.7 mm, and 997.0 mm) and body depth. The fish models were filmed in approximately 10 m of water off the South Shore of Oahu, Hawai'i, at distances of 3 m and 6 m from the cameras. The BotCam was rotated by a diver so that the fish traversed the field of view to simulate swimming. The models were moved vertically to obtain coverage throughout the fields of view of the two cameras and were measured at haphazard angles. Length measurements on each fish were made by three scientists (user 1, 2, and 3) using Vision Measurement Software (Geomsoft, Victoria, Australia). Error is defined by the following: Error = actual fork length–fork length measured by stereo-video (also called residual).

	User 1		User 2		User 3		Total	
	3 m	6 m	3 m	6 m	3 m	6 m	3 m	6 m
Number of measurements	193	113	192	134	249	0	634	247
Average error (mm)	–2.3	–6.2	–0.8	–17.0	–7.5	na	–3.1	–8.8
Standard deviation of average error (mm)	22.2	50.8	25.6	42.0	30.8	na	27.7	51.6
Percent error (%)	–0.3	–1.2	0.0	–2.2	–1.1	na	–0.5	–1.3

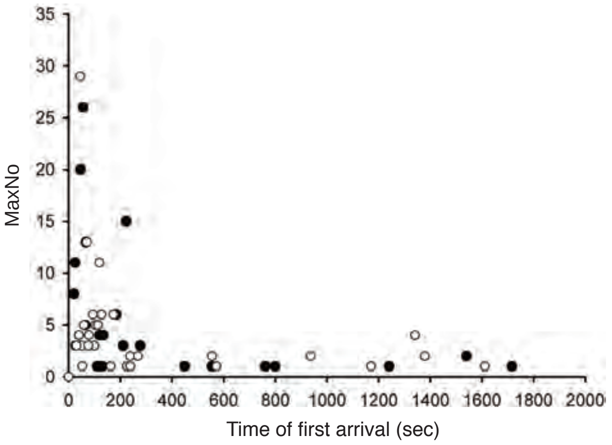

Figure 4

MaxNo (maximum number in a single frame) as a function of TFA (time of first arrival) for *Etelis coruscans* (●) and *Pristipomoides filamentosus* (○) recorded from BotCam deployments at Penguin Banks, Hawai'i, between June 2006 and February 2007.

Discussion

The primary objective of this research was to investigate whether, from an operational perspective, BotCam can provide reliable fishery-independent data on Hawaiian deepwater bottomfish populations that are of similar quality to data obtained from camera systems placed in shallower waters. The results indicate that BotCam can be a useful tool and furthermore illustrate the different types of data it is capable of collecting. Of particular importance, 80% of the deployments were successful in hitting their target sites and recording for the planned time interval. All of the "deep 7" species were attracted to BotCam and were recorded on videotape during the study. Thus from an operational standpoint, BotCam has the potential to collect data useful for assessment of bottomfish populations. Studies are underway to compare results of the pilot study with those from subsequent deployments to determine whether the method can lead to a greater understanding of the temporal and spatial dynamics of bottomfish populations.

As with data collected with other methods, fish count data collected with underwater video systems are confounded by a number of factors, especially when a baited design is used. One factor that affects variance is the inconsistent size of the sampling area due to an unknown size of the bait-plume. One of the outstanding questions about baited camera stations is how extensive is the area of influence of the bait (Priede and Merrett, 1996; Willis et al., 2000). Initial attempts to measure bait dispersal with the stereo-video system proved inadequate; however, measurements of current speeds were promising (Merritt, 2005). Watson et al. (2005) compared baited and unbaited stereo-video surveys with underwater visual surveys in a shallow-water environment and found that the baited stereo-video system was the best technique for obtaining consistent fish counts with the least sampling effort, and that unbaited techniques would require a high level of replication to yield similar results (see Harvey et al., 2007). Heagney et al. (2007), working in the open-water column, found

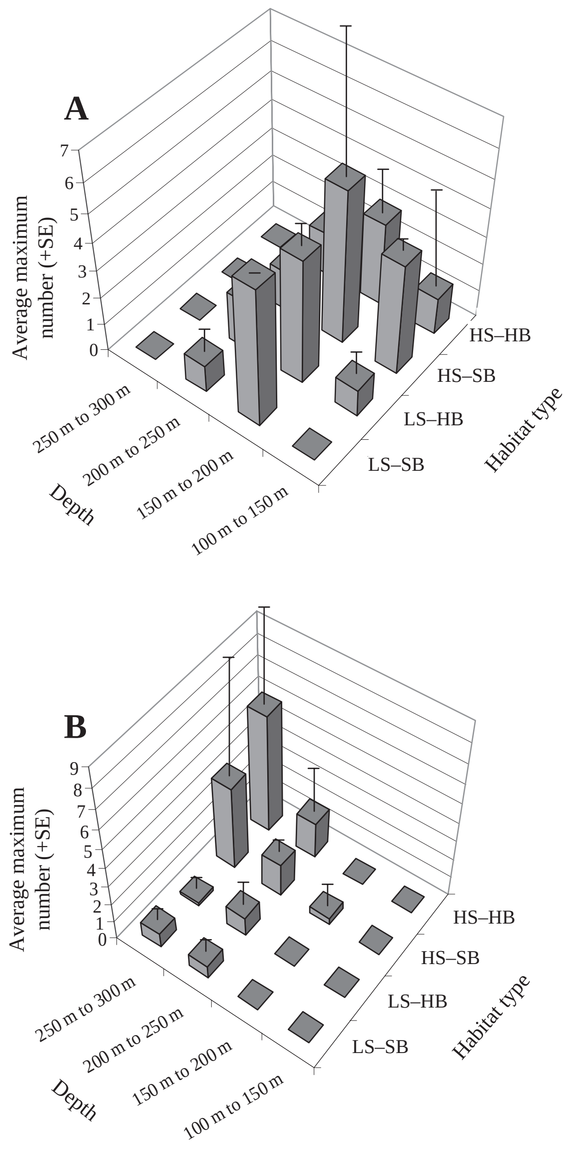

Figure 5

Average MaxNo (maximum number of individuals in a single frame) and standard error (SE) of (**A**) *Etelis coruscans* and (**B**) *Pristipomoides filamentosus* at Penguin Banks, Hawai'i, between June 2006 and February 2007. Depth bins and bottom or substratum types (defined by bottom slope and hardness) were derived from multibeam data during deployment planning. Error bars indicate standard error. HS: high slope, LS: low slope, HB: hard bottom, SB: soft bottom.

that an area-based bait plume model worked well to explain variation in their count data but were unable to determine if the correlation between counts and current was a result of the bait plume size or an indication of the preferred habitat of the fishes. Further work with BotCam is necessary to evaluate the area of influence of the bait, but the skewed relationship between MaxNo and TFA (Fig. 4) indicates that attraction to the bait is rapid and, therefore, local in its effect.

Another confounding factor is the visual attraction of fish to the camera system itself. Watson (2005) refer to this as the "curiosity" effect and although it is a difficult value to quantify, it is clear from the video recordings that fish do react to the camera system. Unbaited deployments need to be carried out to better understand the magnitude of this effect.

Baited camera systems have historically been used to determine either TFA or MaxNo to estimate relative density of the attracted fishes (Bailey et al., 2007). In many studies, TFA has been used in an inverse-square model as a metric of abundance (Priede et al., 1994). It is assumed with the use of TFA that individuals are uniformly distributed in space, act independently of each other (i.e., there is no schooling behavior), all fishes that contact the odor plume swim up current to the camera, and the effect of the bait plume on fish counts is linear and dependent on local current speed. Thus, short TFAs imply greater densities than long TFAs. In more recent statistical models, the arrival rate instead of the TFA has been used, which allows an estimate of a confidence interval (Farnsworth et al., 2007), but both measures are based on the same basic assumptions. These metrics have been applied primarily to deep sea fishes (>1000 m) inhabiting low-energy, bathymetrically monotonous environments (Priede and Merrett, 1996). They are also hypersensitive at rapid TFAs (<~5 min) and insensitive at long TFAs (>~120 min; King et al., 2006; Yeh and Drazen, 2009). Shallower water environments, such as those surveyed in the current study, are more dynamic ecologically and physically than in the deep sea and therefore fishes tend to be less evenly distributed in space.

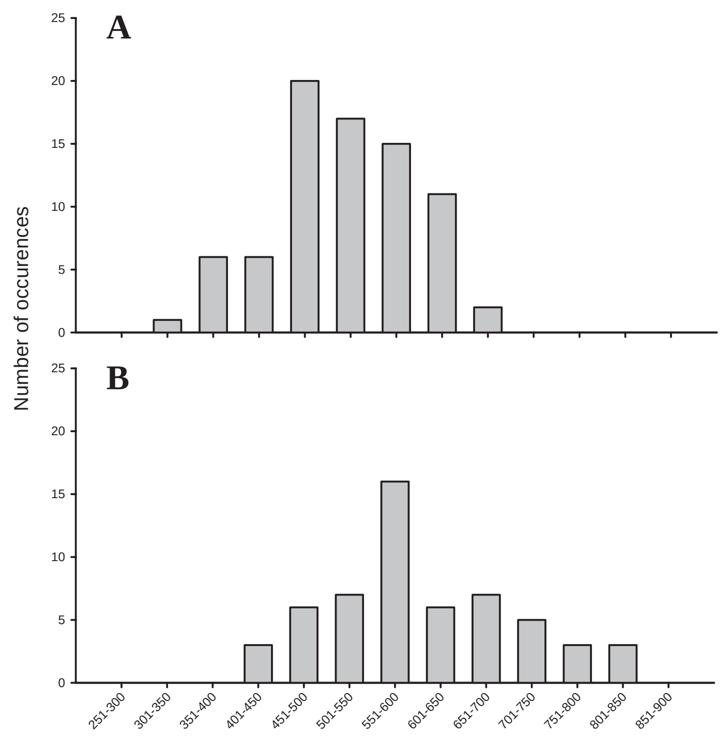

Figure 6

Length-frequency distribution of (**A**) *Etelis coruscans* and (**B**) *Pristipomoides filamentosus* from BotCam deployments at Penguin Banks, Hawai'i, between June 2006 and February 2007 as measured by stereo-video software Vision Measurement System (Geomsoft, Victoria, Australia). Only fish identified at the time of MaxNo (maximum number of individuals in a single frame) were measured. Each fish seen around the time of MaxNo was measured six times (from six different frames of the video) in order to tease out errors due to fish motions and human error. The average fork lengths are binned in 50-mm intervals.

The assumptions about the uniform distribution of the target fishes or linearity of responses to the odor plume required by TFA models often cannot be met. As a result, studies examining shallow-water fishes (Ellis and DeMartini, 1995; Willis et al., 2000; Watson et al., 2005; Kelley and Ikehara, 2006; Stoner et al., 2008) have used MaxNo as an index of relative density which avoids the potential for recounts of the same fish as they exit and reenter the field of view during the survey period. Ellis and DeMartini (1995) found that MaxNo is positively correlated to catch per unit of effort (CPUE) and concluded that it is a useful index of abundance. Likewise, Stoner et al. (2008) concluded that MaxNo was the optimal measure because it is correlated with seine hauls and is consistent across habitat types. Willis et al. (2000) compared a baited camera system with visual surveys and angling surveys and also concluded that video survey techniques with MaxNo provided reliable estimates of relative density. In the present study, TFAs were very short (Fig. 4) and could produce highly variable and spuriously high estimates of abundance (King et al., 2006). This is associated with the lack of sensitivity of TFA to small densities where arrival time is dependent on the position and response to bait of the closest fish. We assumed that the bait plume was not uniform because of the variability in conditions (i.e., currents) and rugged bathymetry. Furthermore, it is well known that some species of bottomfish school, whereas others associate only with hard substrate; therefore in any sampling there will be an aggregated distribution rather than a random or uniform one (Haight et al., 1993a; Kelley and Ikehara, 2006). Indeed, the present results show that MaxNo, similar to many other types of count data, were not normally distributed; many camera deployments resulted in zero fish and others with up to 29 fish (Fig. 4). MaxNo appears to be a more appropriate metric than TFA for estimating relative abundance in this case, but will likely require analysis with statistical models that are designed for nonuniform dispersion patterns.

Knowledge of the distribution of fishes among habitats is of importance to fisheries management, and such information can readily be obtained with the BotCam system. The distributions of *E. coruscans* and *P. filamentosus* among depth bins and habitat substrata types in our study (Fig. 5) indicate that *E. coruscans* on Penguin Bank prefer high slopes and deeper water, whereas *P. filamentosus* do not have a strong preference for a particular bottom type but are found in the shallowest three quarters of the depth range sampled. Modeling the distribution of both species across depth, slope, and substrate type indicated that these factors were important in understanding the association of these species with their habitat. Currently, the essential fish habitat for these species is simply defined as all waters between 100 and 400 m deep. Although beyond the scope of this study, the results show that additional work with BotCam would enable fisheries scientists to more accurately define essential fish habitats and habitat areas of particular concern on a species-by-species basis. Combined with direct observation of habitat, BotCam is also a tool that will allow for a much finer resolution of habitat classification (i.e., bedrock versus boulders versus cobbles) and enable species preferences to be discerned (see Stoner et al., 2008). Parrish et al. (1997) applied this technique to investigate habitat affinity of juvenile *P. filamentosus* and identified premium habitat by using direct observations from video cameras.

One objective of this study was to evaluate the precision and accuracy of the stereo-photogrammetric technique for obtaining accurate size measurements of bottomfishes. After analyzing repeated measurements of *E. coruscans* and *P. filamentosus*, a discrepancy was apparent between the species. The smaller number of *E. coruscans* measured and the larger standard deviation of the measurements relative to *P. filamentosus* were likely the result of *E. coruscans* being found in deeper water, where visibility and image quality decrease, making video measurement more difficult. Nonetheless, valuable information about the size distribution of these fishes was collected (Fig. 6), indicating that BotCam could be useful as a nonextractive tool for sampling size distributions for stock assessment. Additional experience in both calibrating the camera system and in using the stereo-video software will improve the precision and accuracy of size measurements as evidenced by previous studies where a similar system and software were used (Harvey et al., 2003). Harvey et al. (2002) compared fish length estimates from stereo-video and scuba divers and found video to provide consistently more accurate and precise data. Additionally, Harvey et al. (2010) conducted a similar study on the accuracy and precision of stereo video camera system and found that the length of the object measured was a major factor in reducing variance during measuring. In contrast to this finding, we suggest that size was not a factor, although our study supports the finding that precision degrades with distance away from the camera.

The size distributions of *P. filamentosus* and *E. coruscans* estimated in our study were consistent with published data for both species. Haight et al. (1993a) estimated the length at maturity of *P. filamentosus* to be 430 mm, and maximum length to be 780 mm. Our estimates for *P. filamentosus* ranged from 344 mm to 660 mm, normally distributed throughout the reported size range (Fig. 6). Everson et al. (1989) estimated the length at maturity of *E. coruscans* to be 663 mm, and maximum length to be 925 mm. Our estimates for *E. coruscans* ranged from 432 mm to 832 mm, again normally distributed across the reported size range (Fig. 6). These results indicate that BotCam can estimate relative size frequencies, both pre- and post-sexual maturity and therefore could be used for monitoring recruitment and changes in spawning potential ratios. In neither species was a fish measured near its reported maximum size. The reasons for this could be low sampling effort, size-related differences in behavior or habitat use, bias caused by measuring only at MaxNo, or simply that individuals of such large size were absent from the sampled area. Juveniles of these species were also absent from the video recordings, possibly because they remained close to the bottom near cavities because of their vulnerability to predation, as typical of other bottom associated fishes. Juveniles could have been in the vicinity of BotCam, but because of the presence of larger fish, such as *S. dumerili*, were possibly unwilling to come up to the cameras.

Monitoring deepwater fishes and their habitat is a difficult and costly undertaking. We tested the effectiveness of a new baited stereo-video camera system (BotCam) and found it an efficient tool in places where diver surveys are impossible and ROV or submersible surveys are cost prohibitive or provide data of uncertain quality (Kelley et al., 2006; Stoner et al., 2008). The success rate of data collected per deployment in this study supports the use of BotCam for studying biologic assemblages at depths ranging from 0 to 300 meters. As a nonextractive method, BotCam could prove particularly valuable in marine protected areas, where restrictions on fish removal may limit the usefulness of traditional sampling methods (Willis et al., 2003; Denny et al., 2004; Willis and Millar, 2005). Future work must include careful calibration of BotCam data with traditional population assessment data, including measures of relative abundance based on fisheries-dependent data such as CPUE. In addition, calibration with other nonextractive methods, such as acoustic surveys, is needed.

In future studies with the BotCam system, current meters should be used to model bait dispersal and its effects on fish counts and other measurements. The development of a diverse suite of methods for assessing fish stocks, including baited camera systems such as BotCam, strengthens the scientist's toolkit and allows for more reliable stock assessments and cross-validation of these assessments.

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RESEARCH REPORT

Assessing Kaho‘olawe Island Reserve’s bottomfish populations: a potential benchmark for main Hawaiian Island restricted fishing areas

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INTRODUCTION

The deep water bottomfish fishery is an important component of Hawaiʻi's industry. Four species of eteline snappers, the onaga, *Etelis coruscans*, the ehu, *Etelis carbunculus*, the ʻopakapaka, *Pristipomoides filamentosus*, the uku, *Aprion virescens*, and one endemic species of grouper, the hāpuʻupuʻu, *Epinephelus quernus* compose most of the catch (Haight et al. 1993a). Their essential fish habitat (EFH) has been loosely defined as the 100-400 m depth range around each island and bank in the Hawaiian archipelago. These designations are currently under review and will be more refined in 2010 (C. Kelley, unpub. report to Western Pacific Fisheries Management Council). Some species like ʻopakapaka and uku are typically found shallower than ehu and onaga. These species have varying diets from piscivory to predation on macroplankton (Haight et al. 1993b) but all take bait and are readily attracted to baited cameras. They along with two species of jacks or kahala (*Seriola dumerili* and *Seriola rivoliana*) comprise the major predators in this ecosystem, below elasmobranches and marine mammals in the food-web.

Unfortunately, these stocks have been overfished in the main Hawaiian Islands (MHI) for at least a decade. In 1997 the Department of Aquatic Resources (DAR) created a new bottomfish management plan and funded research on bottomfish to provide additional information on these species. A key element in the plan was the creation of nineteen bottomfish restricted fishing areas (BRFAs) where bottomfishing was prohibited. In 2005 a review of the system suggested that it did not protect an adequate amount of preferred bottomfish habitat and baseline data on populations within these reserves were not collected so their performance could not be determined. A new BRFA system was created and took effect on July 1, 2007. This time there was a much greater understanding of the distribution of MHI bottomfish habitat as a result of multibeam sonar mapping. The number of BRFAs was reduced from 19 to 12 and their boundaries were designed to protect selected habitats but also to facilitate spillover and thereby sustain adjacent habitats open to fishing. Both fisheries biologists at DAR and NOAA's Pacific Islands Fisheries Science Center (PIFSC) have stressed the importance of obtaining baseline data as well as population monitoring, so that this time there is a way to assess their performance.

Currently our lab group is studying the efficacy of these reserves to build back the exploited stocks. We began studying a representative subsample of the BRFAs (6 of 12) in 2007 and will continue for 5 years. The data gathered will allow us to assess the performance of the reserves by monitoring changes in abundance and average size of fishes. Indeed, marine reserves are becoming an important management tool for fisheries and conservation (Palumbi 2001; Pisco 2002). In addition to simply reducing fishing mortality, marine reserves can protect habitat, preserve a spawning stock that can enhance recruitment to nearby fished areas, generate large populations of adults which then spillover to adjacent fished areas, and allow populations to occur without the artificial selection of fishing and its evolutionary effects.

The Kahoʻolawe Island Reserve (KIR), with protection from fishing since 1994 (~15 years) may provide many of these functions for deep-water bottomfishes. Aside from several small coastal reserves protecting reef populations such as Hanuama Bay, KIR is the oldest marine reserve in Hawaiʻi. With its large area and long term protection Kahoolawe Island Reserve it is already well known that its coral reef resources are more abundant than other locations in the main

Hawaiian Islands. Its reefs have greater proportions of top predators present, greater overall fish abundance, and greater fish sizes (Friedlander and Demartini 2002).

The ocean resources plan for KIR states a mission of determining the status of the ocean resources surrounding Kahoʻolawe and improving the health of offshore areas. KIRC appears to be meeting this goal in reef areas but it is unclear if this is the case in deeper waters. This document reports our activities and findings to study the deep bottomfish within KIR in order to meet this mission goal and to generate data on what are some of the least exploited bottomfish populations in the main Hawaiian Islands. It was our goal to gather data on what should be a benchmark goal for the BRFAs in the rest of the main Hawaiian Islands. As a first step we determined if stocks within KIR are more abundant, more diverse, and/or whether the fishes are of a larger size than other areas in the main Hawaiian Islands. Hawaii's bottomfish resources are below conventional SCUBA limits making non-extractive monitoring of their populations difficult. Thus to accomplish our goals, we utilized a baited underwater video system as a non-extractive tool to measure bottomfish relative abundance and size frequency distributions.

METHODS

To assess KIR's deep bottomfish resources we utilized a cutting edge deep-water stereo baited camera system called the Bottom camera bait station (BotCam; Fig. 1). It was initially developed by the PIFSC Coral Reef Ecosystem Division (Merritt 2005; Merritt et al. in revision) and it has been modified since. It has an operating depth of up to 350 meters and provides a non-extractive, and hence very attractive, method to monitor fish populations within restricted fishing areas or marine protected areas. The system consists of two video cameras which are programmed to take images simultaneously. Numbers of fishes and their identity are determined and used in modeling relative abundance of each species. The stereo camera arrangement facilitates size and distance determinations of fish and habitat features in the field of view.

Sixty-four sites were randomly selected within the 100-300m depth range using GIS software. Typically this is done where multibeam bathymetric and acoustic backscatter coverage allows for classification of the substrate into either hard or soft bottoms and either steep (>20º) or shallow slopes. Some multibeam data was available for KIR waters but almost no acoustic backscatter. Using NOAA charts and the existing multibeam bathymetry, we gridded the 100-300m region into 200x200m grids as we would do for the BRFAs. It was possible to assign steep or shallow slopes to these grids even without multibeam from the distance between the 50 and 100 fathom isobaths on the NOAA charts. Thus we randomly selected 24 low slope 40 high slope grids. BotCams were deployed at each site for a recording time of ~45 minutes. These deployments were made during two separate trips one March 17-20th, 2009 and the other February 5-8th, 2010.

For each BotCam deployment, both video streams were analyzed to determine habitat type. Habitat information extracted from the video included substrate type (hard or soft), steepness of the slope (high or low), primary and secondary substrate type (bedrock, boulders, cobbles, sediment), relief of the terrain, and the presence of cavities, which are often used by some bottomfish or their prey as shelter. A temperature depth recorder recorded the depth and the temperature throughout the deployment.

All fish in the videos were identified to the lowest taxonomic level possible which was most often species. For commercially important bottomfishes often kalekale (*Pristipomoides seiboldii*) and ‘opakapaka (*P. filamentosus*) were indistinguishable when far from the camera so the category *Pristipomoides* was also used for analysis and incorporates both species whether they were identifiable to species or not. A similar situation occurs for kahala. Two species, *Seriola dumerili* and *S. rivoliana* can be difficult to distinguish. Thus all data are simply treated *Seriola* for analysis. Fish data included the maximum number of fish observed at any one time (n_{max}) for each species or genus. This metric relates directly to other more conventional fish abundance techniques such as diver transects and CPUE from fishing surveys (Ellis and Demartini 1995; Willis et al. 2000). In addition we recorded the time that each species or genus first arrived in the field of view or the time of first arrival (t_0). This metric has been related to relative abundance in some instances (Priede et al. 1994) but is not often a good indicator of abundance for schooling fishes such as deep Hawaiian bottomfish (Merritt et al. in revision). Results using this metric are not reported here.

Data were collected on all species observed not just commercially important ones, so that species diversity could be assessed. The data for this analysis included counts for fish that were identified to species and in a few cases counts for genera if no species level identifications were available. In a few instances pelagic species such as thresher shark and a manta ray were observed and these strictly pelagic taxa were not included as they were only incidentally observed during deployment. Several diversity indices were calculated using the area aggregated n_{max} data for each area using PRIMER v6 software. The indices were total number of species (S), total number of fish (N), Margalef’s species richness (d), Pielou’s evenness index (J′), and Shannon’s diversity index (H′) and the equations for each follow:

$$d = \frac{(S-1)}{\log(N)} \qquad J' = \frac{H'}{\log(S)} \qquad H' = -\sum_{i=1}^{S} (P_i * \ln(P_i))$$

In addition rarefaction curves were used. These curves estimate the average number of species sampled at a given number of individuals sampled. For instance, the ES100 represents the estimated species (ES) at 100 individual fish sampled. This method accounts for variations in sampling intensity between areas so it has been used in with a variety of ecological samples from diet diversity (Drazen et al. 2001) to sediment infauna diversity (Glover et al. 2002).

The fork lengths of fishes commercially important species including the snappers, groupers and large jacks (*Seriola dumerili* and *S. rivoliana*) were measured at the time the greatest number were visible on both cameras using PhotoMeasure software. For length measurements species were not pooled into either *Pristipomoides* or *Seriola* because each species reach different sizes. Instead measurements were restricted to those fish for which identifications were possible. Each BotCam frame was calibrated in a pool prior to each sampling trip. A light-sync device consisting of a series of LEDs illuminated in series in the field of view of both cameras was used to ensure synchronicity of both video files. Measurements using this type of stereo-video technique are accurate to within 0.1-0.5cm (Harvey et al. 2002; Merritt et al. in revision) however, lower precision is achieved in low light conditions such as is found below 250m.

To place the results from Kahoʻolawe into context and to address the question of how protection in KIR may have resulted in larger and more abundant bottomfish, we compared data to three other sites in the main Hawaiian Islands. These sites are now protected but the data we employ come from just before or within 11 months of the reserves being implemented in 2007. Penguin Bank, including the bottomfish restricted fishing area (BRFA) F, is a very large submerged bank extending off Molokaʻi's southwestern corner. Its summit is less than 100m deep and the flanks consist of walls and "fingers" that extend out into waters between 500-1000m. It is the location of most of the commercial bottomfish catch in the main Hawaiian Islands. Makapuʻu Pt, including BRFA E, encompasses a few pinnacle features and banks off the southeast corner of Oahu. It is heavily fished both commercially and recreationally due to its proximity to Honolulu. Pailolo channel is an area between Maui and Molokaʻi and now includes a small BRFA (H). This area is mostly at depths of 175-275 m and consists of relatively low slope rocky environments with a few small pinnacles. All of these areas will be refered to throughout the text by their BRFA letter and Kahoʻolawe Island Reserve will be abbreviated as KIR.

Data on relative abundance (n_{max}) and size (fork lengths) were compared between these areas and between the four principal habitat types using permutational ANOVA or PERMANOVA using PERMANOVA+ for PRIMER (Anderson et al. 2008). This test is permutational and thus does not assume that the data is normally distributed which is ideal for the count data used in this study. Data for n_{max} were square root transformed to deal with the large number of zeroes and few very large numbers and the PERMANOVA was run on a Euclidean distance matrix using type III sums of squares. The data was analyzed in the same way for the lengths except that these data often approached a normal distribution and, of course, there are no zeroes, so the data did not require any transformation.

RESULTS

Data

The raw data which includes all of the video and temperature-depth recorder files have already been given to KIRC on hard drives. Copies of all of these are also kept at UH. In this way either set serves as a backup for the other. If for any reason our drives are compromised or KIRC drives are, the other party should be informed immediately.

The data will be summarized in the subsequent sections of this report however, the data from which it is derived is also being provided as an MS Access database. It includes all the architecture of the full main Hawaiian Islands database designed and assembled by the bottomfish project at the University of Hawaiʻi. All the data for the deployments in KIR are available for and a variety of queries and a few reports could assist in future data exploration.

Summary of data from KIR

The bottomfish habitat (depths between 100 and 400m) are principally located along the south and east shores of Kahoʻolawe (Fig. 2). Previous fishing studies have found the best bottomfish habitats occur in these locations (C. Kelley, unpub. data). A total of 74 BotCam deployments were conducted of which 59 were acceptable for analysis (Fig. 2) spanning 114 to 319m depth. The most common problem with drops that were not acceptable was that they were dark with grainy video. This was largely the result from having poor bathymetric data in planning drop locations and placing BotCam units in water too deep. Only four of the drops were unacceptable due to electronics malfunction with caused either no recording of the cameras or a premature shutoff of recording. Attached to this report is an extract of the database which gives all detailed deployment information for each BotCam drop, such as latitude, longitude, time, date, associated temperature and depth, etc.

Without multibeam bathymetric coverage of the area sampled we had to rely on visual observations to determine the substrate type for each deployment. A good representation of all four basic habitat types was made (Table 1; Fig 3) except for hard substrate low slope (hard-low) which reflects a general lack of this type of habitat within the sampling area. Of the 28 sites with hard substrate classification 26 were principally bedrock and 2 were cobbles and/or pebbles. In 27 of these sites sediment was also present and in the other site no secondary type of substrate was determined. Of the 31 soft-substrate sites the secondary substrate type was bedrock (n=3), boulders (n=3), cobbles/pebbles (n=17), indeterminate (n=2), or none (n=6) suggesting that even where sediments were predominant some type of hard substrate was also present in much smaller amounts.

Over the depth range sampled the temperature declined linearly with depth from ~22° C at 120m to 10° C at 315 m (Fig. 4). There were no evident differences between the two field efforts. Although only a small hydrographic section is represented, it is clear from the rapid change in temperature with depth that the habitat is below the permanent thermocline (~100m).

All the eteline snappers and other species of commercial bottomfish known to occur in Hawaiian waters were observed in KIR (see Fig. 5 for examples of fish images from the BotCam). Their depth ranges varied (Table 2). Species with shallower distributions include ‘opakapaka, lehi, yellowtail kalekale and hapu‘upu‘u. Species that have deeper distributions include onaga and ehu. Kahala (*S. dumerili* and *S. rivoliana*) were observed at all depths. The most frequently observed bottomfish in KIR were ‘opakapaka and kahala (*Seriola* spp.; Table 3). ‘Opakapaka, onaga, lehi, hapu‘upu‘u, and *Seriola* spp. were observed on all substrate types but most of these species were most frequently observed on hard-high substrate. ‘Opakapaka were observed several times to form large midwater schools just shoreward of a steep, sedimented shelf-break (soft-high habitat) that then followed the BotCam to the seafloor (Fig. 5). Kalekale, Randall’s snapper and gindai were only observed on hard-high substrate. The yellowtail kalekale, which is rare in Hawai‘i, was observed once ($n_{max} = 3$) in a hard-high site. Ehu were observed only in high slope habitats regardless of whether the substrate was hard or soft.

Sixteen other taxa (species, genera, or family) were observed (Table 4). None was commonly observed but the most frequent were sharks (unidentified *Carcharhinus* spp. and sandbar sharks). One BotCam deployment on hard-high substrate at 124m in Kamohio Bay (Fig. 2) was very diverse and the only location at which 6 of the non-target species were observed. Most of these non-target species are deep-reef species or reef species near the lower boundary of their depth distributions so most non-target species are observed shallower than 200m. No ta‘ape (*Lutjanus* *kasmiri*) were observed although this species has been observed in other areas of the main Hawaiian Islands down to 155m. 15 BotCam deployments occurred from 119-155 meters depth but most of these (10) were on soft substrates on which taape are not normally observed.

Relative abundance and diversity in KIR and in comparison to other areas

The frequency of occurrence of each bottomfish species varied with area and habitat type (Table 3). The distribution of deployments amongst the four habitat types was not even between areas so comparisons are most sound within a habitat and between areas. Pailolo channel stands out as having very high frequency of occurrence of ehu and onaga, particularly in hard low habitats at 50 and 75 percent of the deployments respectively. However, in KIR 59% of hard-high deployments had ehu, the next largest occurrence of ehu in any habitat type in the data set. KIR had notably higher frequency of occurrence of gindai (22.7% of hard-high) and both lehi and hapu‘upu‘u were found, in all habitat types and in relatively high proportions compared to the other areas. *Seriola* were very common everywhere but more so in the Makapu‘u and Penguin Bank areas than at Pailolo channel and KIR. Some of these trends were paralleled in the relative abundance data.

Relative abundance (n_{max}) of commercially important bottomfish species was highly variable (Fig. 6). The species with the greatest relative abundance in KIR were ‘opakapaka and ehu followed by onaga, kalekale, and *Seriola* spp. Less abundant were hapu‘upu‘u, gindai, lehi, and Randall’s snapper. The data were not normally distributed. Many of bottomfish species school forming aggregations (Fig. 5) so that the data more closely approximate a negative binomial distribution. This means that many samples contain zeros, less have low n_{max} (1-5) and a few have much larger values (5-50). Representative distributions for ‘opakapaka and onaga, the two most commercially important species, are given in Fig. 7. Schooling is also evident in kalekale which were seen at 5 sites in KIR with n_{max} of 2, 2, 3, 18, and 38 fish.

Comparisons between the four habitat types and between KIR and the three other Hawaiian areas indicated significant ($p<0.05$) species-specific differences (Fig 6). Significant differences between areas were found only for ehu and ‘opakapaka. For ehu there was a significantly greater n_{max} in Pailolo channel (area H) compared to all of the other areas. Conversely, there were significantly fewer opakapaka in this area. Hapu‘upu‘u were also more abundant in area H and KIR compared to E and F but not significantly so. Lehi were never very abundant and showed the greatest mean n_{max} in KIR although this was not significant. Finally *Seriola* spp. were more abundant in E and F but not significantly so. For some species such as kalekale and gindai no differences between areas were evident, probably because these species are less frequently seen than other species (Table 3) so variability in the data is very high

Most of the significant differences observed were between habitat types rather than between areas and in no case did the two-way PERMANOVA indicate significant area x substrate interactions effects. Generally soft-low habitats exhibited the lowest n_{max} and hard-high and often hard-low the highest (Fig. 6). Specifically, ‘opakapaka were significantly more abundant in hard-low habitats compared to soft substrates and soft-low slope substrates had the least fish of all. Kalekale and gindai were significantly more abundant in hard-high habitats compared to the other habitat types. At the generic level, *Pristipomoides* spp., is driven strongly by the common and abundant ‘opakapaka but also by infrequent but often abundant kalekale. In this case relative abundance was significantly higher in hard-high habitats compared to soft substrates and lowest in soft-low habitats. Onaga were more abundant in hard-low habitats compared to soft-high and soft-low. It was also more abundant in hard-high compared to soft-low habitats. The data for onaga were particularly variable as is evidenced by large error bars. Ehu were more abundant on hard-high than soft-low. Abundance in hard-low habitats was marginally higher than that in soft-low ($p=0.053$). Ehu abundance in KIR was quite high in soft-high habitats compared to the other areas although there was no significant interaction between area and substrate for this species ($p>0.05$). Hapu‘upu‘u were more abundant in hard-high habitats compared to all others but the variance in abundance in soft-low substrates gave a mean value nearly as high as in hard-high substrates. *Seriola* were significantly less abundant on soft-low habitats compared to the others except hard-high ($p=0.08$).

Diversity of all fishes observed, both those of commercial importance and those that are not, were evaluated for each site. Depth is a very important determinant of diversity particularly in the depth range examined in this study as it occurs in a zone of great faunal change associated with the permanent thermocline and rapid decreases in light levels. Thus for the comparisons to KIR only samples from depths greater than 114 m were used to match the depth ranges. This resulted in the omission of 4, 3 and 1 samples from E, F, and H respectively. Also water clarity, which greatly affected the ability to identify fishes varied between the habitats. It was lowest in Pailolo channel. KIR had many drops which were murky but had better overall water clarity than Pailolo channel. Makapu‘u and Penguin Bank were often very clear. To ensure that water clarity did not result in the trends observed we took two approaches. First, small fishes which are difficult to identify even in clear water, such as slopefishes, threespot chromis and a few others were omitted from the analysis. Also in some cases the data were condensed to genera or even family. For instance, there are several unicornfishes in the genus *Naso*. It is easy to identify them to *Naso* but not to a particular species within that genus. Thus counts at the generic level were used. Second, we examined only the diversity of the commercial bottomfish species including the deep snappers, jacks and hapu‘upu‘u. All of these species are strongly attracted to the bait and can be identified with consistency even in poor water clarity.

Using the first approach, KIR had the greatest species richness both in terms of total number of species and Margalef’s richness value which standardizes for differing sample sizes (Table 5). Pailolo channel (H) had the lowest richness of all the sites. Evenness was also low for H reflecting the dominance of ehu and onaga in this area. Evenness was roughly similar for Penguin Bank and KIR and slightly higher for Makapu‘u. Shannon’s diversity index which takes into account both richness and evenness was lowest for H and similar for the other three areas, with the highest value being found for Makapu‘u. The rarefaction curves show that the diversity in KIR is the highest but only above about 450 fish (Fig. 8). The absence of clear asymptotes in the lines for all but Makapu‘u suggest that additional sampling will reveal greater diversity.

Using the second approach of including only the 11 commercial bottomfish species revealed that only in KIR and Penguin Bank were all species observed and with only 8 of the 11 in Pailolo channel (Table 5). Margelef’s index suggested the greatest species richness of commercial bottomfish in Makapu'u and KIR. Importantly eveness was greatest in KIR and this likely resulted in the greatest estimate of Shannon diversity. These results are supported by the rarefaction curves (Fig. 8). In short, our observations in KIR often found many more species of bottomfish co-occuring such that there wasn't as much dominance by one or a few species compared the other areas.

Bottomfish length data

A total of 1391 commercially important bottomfish were measured for the current analysis, of which 297 were from KIR. Of these data a total of 1034 and 257 from KIR were found acceptable for analysis. Some measurements were excluded because either only a single measurement could be made (4-5 replicates are the goal), replicate measurements that were highly variable (coefficient of variation of measurements > 10%) indicating poor video quality and little confidence in the measurement, or the fish were at a great angle to the plane of the cameras (angle >30º or angle > 25º at a distance of 5+m) which results in very inaccurate measurements.

In most cases bottomfish mean fork length was larger in KIR compared to one or more other areas (Fig. 9). ‘Opakapaka were bigger in KIR (493 ± 99 mm) than they were in Makapu‘u (432 ± 87) or Penguin Bank (420 ± 107; p<0.05; Fig. 8). Mean size of ‘opakapaka in Pailolo channel were similar but the sample size was very small (n=18). Kalekale were principally seen in hard-high habitat and only were observed outside this habitat type in Pailolo channel. Thus, only a one-way PERMANOVA was possible between the areas. Makapu‘u and KIR had similar sized fish but only 5 individuals were measured in Makapu‘u making a robust test impossible. However, KIR kalekale were significantly (p<0.05) larger (345 ± 41) than those at Penguin Bank (283 ± 68) and H (221 ± 44). Penguin Bank also had significantly larger fish than Pailolo channel. Ehu were significantly larger in KIR as compared to those at Penguin bank (376 ± 70). Mean fork length of ehu was the largest in KIR (427 ± 65) but not significantly greater than the mean lengths of fish at Makapu‘u and Pailolo. The largest onaga (629 ± 89) were found at Penguin Bank, significantly larger than those in Pailolo channel and KIR. Six fish from Makapu‘u had a similar size to those at Penguin Bank. KIR fish were ~7 cm larger than those in Pailolo (p<0.05). Onaga was the only species that was significantly smaller in KIR compared to another area. Greater amberjack were the smallest at Penguin Bank but only by ~2 cm compared to those from Makapu'u. The grouper, hapu‘upu‘u, were significantly smaller in the Makapu‘u area compared to Pailolo and KIR. The fish in KIR were the largest on average (662 ± 86) but statistically indistinguishable from those from Penguin Bank and Pailolo. Makapu‘u had much smaller almaco jack than the other areas (486 ± 100; p<0.05). Mean fork length in KIR (623 ± 76) was similar to that in Penguin Bank and smaller than the fish from Pailolo (668 ± 97) although not significantly so. Only 9 gindai were measured preventing any statistical tests, but the two fish in KIR were the two largest (404 and 414mm). Few lehi measurements were available (n=27) and no significant differences in size were found between areas or habitats.

Differences between habitats and interactions between area and habitat were also evident among some of the bottomfishes (Fig. 9). For ‘opakapaka habitat had a significant effect and interactions between area and substrate were evident. Fish from high slope habitats were significantly larger than those from low slope habitats. The interaction between area and habitat was significant ($p<0.05$) so tests were performed within each habitat type. These results showed that within hard-high and hard-low habitats KIR had larger fish than Makapuʻu and/or Penguin Bank ($p<0.05$). The largest ʻopakapaka in soft-high habitats occurred at Penguin Bank and in soft-low habitats the fish were smaller in Penguin Bank compared to Makapuʻu but, few measurements were available reflecting the low abundance in this type of habitat. For onaga, mean length in hard-low habitats were larger than those in soft-low but with only 5 fish from Pailolo channel measured in the later habitat ($p<0.05$). Greater amberjack fish from hard-high habitats had larger fish than those from the soft substrate areas. Within hard-high habitat KIR had the largest greater amberjack (837 ± 73) compared to Makapuʻu and Penguin Bank but with very small sample size.

DISCUSSION

The best design to evaluate the ecological effects of creating a marine reserve is to sample it before and after protection and in relation to a control site – the so called before-after-control-impact or BACI design (Palumbi 2001; Gell and Roberts 2003). This was not possible for our work in the Kahoʻolawe Island Reserve (KIR). In this report we evaluate the reserves effectiveness in protecting deep water bottomfish species by comparing the results to other areas which at the time of sampling were not protected. A major complaint by critics of MPAs is that any differences observed in such a study are the result of differences in habitat between the reserve and control areas. However, we have carefully evaluated the habitat type for each sample utilized in the study control for such differences.

Given this caveat, KIRs deep water resources are more robust than those examined at other sites in the main Hawaiian islands in many respects. First, some species occurred more frequently in KIR BotCam samples than in other areas and some species such as hapu'upu'u were found more frequently in what is considered less suitable habitat such as soft substrates (Table 3). However, kahala were observed more frequently in more heavily fished areas such as Makapuʻu and Penguin Bank. Kahala are not fished commercially and very little recreationally because of ciguatera and parasite concerns. The Honolulu fish auction ceased allowing their sale in the 1990's as a result. Kahala, *Seriola rivoliana* and *S. dumerili,* do not have as well developed gas bladders as the snappers and often survive capture from depth and subsequent release. It is also important to note that kahala are predators of the smaller bottomfish species. Ehu and kalekale have been observed to rapidly leave bait before kahala arrive (Drazen and Kelley, pers. obs.) and both have been found in the stomachs of kahala (Humphreys and Kramer 1984).

Our estimates of relative abundance (n_{max}) derived from baited cameras did not suggest that there were more bottomfish within KIR compared to the other areas. A few species had greater mean n_{max} in KIR such as hapuʻupuʻu and lehi but with few fish observed and great variability in the data the differences were not significant. This was not expected because marine reserves often show elevated abundances of species that are fished outside of the reserve (Palumbi 2001; Friedlander and Demartini 2002; Pisco 2002; Gell and Roberts 2003; Willis et al. 2003). Also an earlier submersible survey found greater abundances of ehu, gindai, and ‘opakapaka inside KIR compared to Penguin Bank and Raita bank in the NWHI (Kelley et al. 2002).

There are several possible explanations of these findings. The reserve may be too small in relationship to the movement patterns of Hawaiian bottomfish. Very little is known about their movements but a study along the southwestern boundary of KIR did show that ‘opakapaka moved across the border regularly (Ziemann and Kelley 2007). Conventional floy tag studies are also suggesting that ‘opakapaka and onaga may move between the islands and across deep channels at least occasionally (Okimoto and Clay Tam (DAR), unpub data). However, observations of species such as gindai, ehu, and hapu‘upu‘u from submersibles suggest that they reside directly on the bottom and may not exhibit such great movements. It is some of these species which show the greatest reserve effect in terms of fish size and other metrics. The duration of protection may not be adequate to see a robust abundance response. Some studies have found that increases in abundance may take very long times to observe in long lived species, up to decades (Russ and Alcala 1996; Pisco 2002; Gell and Roberts 2003; Unsworth et al. 2007). Species such as ehu and onaga are relatively long lived, reaching sexual maturity in ~10 years and ‘opakapaka only reach maturity in ~6 years but these ages may be increased based on ongoing age and growth data (Andrews and Humphreys, pers. comm.). Also enforcement of the fishing ban is paramount to seeing any benefit of reserve creation. Previous submersible surveys show that fishing debris (i.e. anchors, weights, and lines), which is generally indicative of fishing intensity, is much lower in KIR compared to Penguin Bank but higher than some areas of the NWHI (Kelley and Ikehara 2006). The age of such debris is very difficult to determine. The south shore of Kaho’olawe is exposed to the ‘Alenuihaha channel winds making it a difficult fishing location but it is also obscured from view by high cliffs which might make detection of illegal fishing by anything other than direct patrols a challenge. Some studies have found that recreational fishing alone is enough to eliminate a reserve effect on the targeted species (Denny and Babcock 2004).

Finally, the lack of any evident enhancement in bottomfish abundance might be the result of poorer quality habitat inside KIR or inadequately characterized habitat. As described in this report, Hawaiian bottomfish species showed strong habitat preferences with most species present in greater abundances on hard substrates and often in hard substrate high slope environments (Fig. 6). This is well known from fishing surveys as well (Kelley et al. 2000). We made visual habitat classifications and these reflect the immediate area sampled by BotCam but not the entire area from which fish are drawn to bait. So we were able to compare similar habitats at least on a very small spatial scale between KIR and the other sites. In this way we have minimized any biases of habitat effects. However, without complete multibeam and backscatter coverage in the reserve it is difficult to characterize the habitat of the reserve overall or to expand the spatial scales of our habitat analysis. Shortly multibeam bathymetry and backscatter data will be gathered in KIR (funding to HURL via NOAA/PIRO, C. Kelley, pers. comm.). Given time and funding this data will greatly inform our habitat classifications and could result in different results than those presented in this report. For instance, some sites appeared to have soft sediments but in fact could be bedrock. Perhaps more importantly with the multibeam data we can evaluate the habitat surrounding each deployment location at various scales (i.e. within 10, 50, 100, 200m) which will help us to compare apples to apples at a bottomfish relevant scale.

We found a greater diversity inside KIR compared to elsewhere. This result, while common in marine reserves (Gell and Roberts 2003; Friedlander et al. 2007), should be approached with caution. The BotCam is designed to image commercial bottomfish species which are large, attracted to bait, and commonly swim in the water column above the seafloor by many meters. Thus many smaller species of fishes, particularly in poor visibility situations, may be missed and diversity may be underrepresented. During our sampling the waters of KIR had only moderate water clarity likely due to the islands erosion and runoff situation. Thus, the diversity is likely higher than that reported here. To evaluate total demersal fish diversity in KIR and elsewhere other tools such as submarine or ROV transects would be needed. Indeed two submersible transects in KIR found greater densities of deep-water corals and other cnidarians but diversity was not evaluated (Kelley et al. 2002; Kelley and Ikehara 2006).

A more robust and interesting result was the finding that within the 11 large bottomfish species (the snappers, hapuʻupuʻu, and two kahala species) the diversity in KIR was higher. All species were observed but there was also a very high evenness. This means that these species were often seen together and in more even numbers relative to one another than at the other locations studied. In more heavily fished areas, not only can overall abundance decline but diversity declines as a result of selective harvest of the most desirable species (Pisco 2002; Westera et al. 2003; Unsworth et al. 2007). Protection at KIR may have eliminated or reduced such selective harvest.

The length data showed a clear affect of protection in KIR (Fig. 9). Many species were significantly larger within the reserve compared to at least one if not all three of the other areas studied. ʻOpakapaka were 6-7 cm larger in mean size or 14-17% larger than the fish in Makapuʻu or Penguin Bank. Kalekale were 22 and 56% larger than the fish at Penguin bank and Pailolo respectively. Hapuʻupu'u and ehu mean length was the largest in KIR but not significantly so in all cases. Only onaga were significantly smaller in KIR compared to another area and this species is the most sought after and arguably the most mobile of those examined.

It is important to evaluate the differences in size in relationship to the life history of these animals. In particular, the proportion of fish which have reached maturity may be more important than their specific size per se. For instance, many of the bottomfish fishing restrictions enacted in the last 5 years have been out of concern that the spawning potential ratio of onaga has dipped below 20% and under the Magnuson-Stevens act must be elevated above this level (WPFRMC 1998). This species reaches maturity at very large size and analysis of the size of fish sold at auction suggests there are very few large fish left in the wild. Table 6 takes data for size at 50% maturity from the literature and then estimates the proportions of fish measured in each BRFA that were greater than or equal to this size. KIR had greater proportions of mature ehu, greater amberjack, hapuʻupu ʻu, and ʻopakapaka. The same is true for kalekale except in comparison to 5 fish which were all mature off Makapuʻu. Onaga were notably not mature in KIR (only 6%) compared to 17 and 24% off Makapuʻu and Penguin Bank respectively. These results strongly suggest that KIR is playing an important role in protecting populations of larger sexually mature and reproducing fish. This type of effect has been documented in other marine reserves. For instance lingcod in Puget Sound marine reserves were thought to produce 3 times more eggs than fished populations outside the reserve (cited in Pisco 2002). In an important study of scallops living in closed fishing areas of Georges Bank in the North Atlantic, the protected areas had many times more scallops but the number of scallops in neighboring areas also went up four fold as a result of the export of larvae, settlement and growth (Murawski et al. 2000). Due to the lack of information on the early life history of Hawaiian bottomfish species it is not clear how production of eggs and larvae in the reserve might affect the entire population or whether the larvae are more likely to recruit back to the KIR or to other areas in the islands. However, Ana Vaz, an Oceanography PhD candidate has almost completed an oceanographic/biological model which will couple physical and early life history information to produce theoretical larval trajectories. This model could be very useful in understanding the importance of KIR to Hawaiian bottomfish conservation.

For fisheries management, a major goal of marine reserves is the creation of populations that spillover into fished habitats and augment catch (Gell and Roberts 2003; Amargós et al. 2010; Goni et al. 2010). Ideally this spillover should exceed the biomass or abundance of fishes lost due to closure of an area. Such effects are a great challenge to observe but one recent study has shown that for lobsters in the Mediterranean one reserve similar in size to the Hawaiian BRFAs has resulted in a 10% overall gain in the biomass of catch 5 years after its creation (Goni et al. 2010). With the present study design we can only hypothesize as to whether the spillover of adults from KIR into fished areas might occur. Certainly there is contiguous habitat across the reserve boundaries with the major conduits being to the west towards lanai and a small band of habitat between KIRs northeast boundary and the island of Maui (Fig 10). Available tagging studies do show migration of ‘opakapaka across the southwest boundary of the reserve (Ziemann and Kelley 2007). Therefore, it is likely that spillover is occurring.

In summary, our findings suggest that KIR is protecting Hawaiian bottomfish populations and that these populations are more diverse, many of the fishes are larger, and a greater proportion of the fishes are of reproductive size. This implies that KIR may provide important reproducing populations whose offspring may be important to the rebuilding of stocks in newly created reserves and augmenting recruitment in fished areas. Spillover of adults is also likely but additional study is required to evaluate abundance trends and movement patterns. While the results here are the first to evaluate the importance of KIR to bottomfish protection in the Hawaiian Islands the results could become even more clear with additional habitat data – specifically multibeam and backscatter data to provide continuous habitat characterization in KIR waters. Time and funding permitting, the present analysis could be extended with this information and it could include additional statistical treatments to address the highly non-normal abundance data. As a result, we urge you not to make management decisions based on the present analysis of the data without consulting us first.

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Table 1. Substrate classifications of 59 BotCam deployments in KIR based on either bathymetry (determined prior to deployment) or visually from the BotCam video.

Bathymetry based		**Visually classified**		
			Hard substrate	Soft substrate
High slope	35	High slope	22	14
Low slope	24	Low slope	6	17

Table 2. Depth ranges of bottomfish species in the main Hawaiian Islands as observed with BotCam sampling.

Scientific name	Common name	Depth range (m)
Aphareus rutilans	Lehi	123-278
Etelis carbunculus	Ehu	195-319
E. coruscans	Onaga	210-319
Pristipomoides spp.		108-278
Pristipomoides auricilla	Yellowtail kalekale	124-197
P. filamentosus	‘Opakapaka	108-278
P. seiboldii	Kalekale	158-276
P. zonatus	Gindai	168-247
Randallichthys filamentosus	Randall’s snapper	131-282
Epinephelus quernus	Hapu‘upu‘u	119-229
Seriola spp.	Kahala	96-319

Table 3. Proportion of drops in which bottomfish were present by BRFA and basic habitat type. Yellowtail kalekale were observed only once in KIR and once at Penguin Bank, both in hard-high substrate. Area E – Makapu‘u, F – Penguin Bank, H – Pailolo channel, KIR – Kaho‘olawe Island Reserve.

area	n	Pristipomoides	‘opakapaka	Kalekale	Gindai	Onaga	Ehu	Lehi	Randall’s	Hapu‘upu‘u	Seriola
E	64	42.2%	35.9%	7.8%	3.1%	9.4%	7.8%	1.6%	3.1%	4.7%	65.6%
hard-high	10	90.0%	80.0%	30.0%	10.0%	10.0%	10.0%	0.0%	10.0%	30.0%	90.0%
hard-low	20	60.0%	55.0%	5.0%	0.0%	20.0%	10.0%	5.0%	0.0%	0.0%	85.0%
soft-high	18	27.8%	16.7%	5.6%	5.6%	5.6%	5.6%	0.0%	5.6%	0.0%	61.1%
soft-low	16	6.3%	6.3%	0.0%	0.0%	0.0%	6.3%	0.0%	0.0%	0.0%	31.3%
F	63	46.0%	36.5%	20.6%	7.9%	23.8%	27.0%	9.5%	11.1%	4.8%	66.7%
hard-high	32	53.1%	34.4%	40.6%	15.6%	34.4%	40.6%	6.3%	18.8%	6.3%	75.0%
hard-low	13	46.2%	46.2%	0.0%	0.0%	23.1%	15.4%	7.7%	0.0%	0.0%	53.8%
soft-high	8	62.5%	62.5%	0.0%	0.0%	12.5%	12.5%	37.5%	12.5%	12.5%	100.0%
soft-low	10	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
H	62	27.4%	11.3%	14.5%	0.0%	37.1%	59.7%	0.0%	3.2%	8.1%	50.0%
hard-high	1	0.0%	0.0%	0.0%	0.0%	100.0%	100.0%	0.0%	0.0%	0.0%	0.0%
hard-low	40	37.5%	15.0%	20.0%	0.0%	50.0%	75.0%	0.0%	5.0%	7.5%	55.0%
soft-low	21	9.5%	4.8%	4.8%	0.0%	9.5%	28.6%	0.0%	0.0%	9.5%	42.9%
KIR	59	44.1%	40.7%	8.5%	8.5%	22.0%	27.1%	13.6%	3.4%	15.3%	55.9%
hard-high	22	59.1%	50.0%	22.7%	22.7%	36.4%	59.1%	13.6%	9.1%	27.3%	63.6%
hard-low	6	50.0%	50.0%	0.0%	0.0%	16.7%	0.0%	16.7%	0.0%	16.7%	66.7%
soft-high	14	50.0%	50.0%	0.0%	0.0%	21.4%	21.4%	21.4%	0.0%	7.1%	64.3%
soft-low	17	17.6%	17.6%	0.0%	0.0%	5.9%	0.0%	5.9%	0.0%	5.9%	35.3%

Table 4. Other species observed in KIR, the number of observations in each habitat type and depth range. When a genus is given it includes all observations where identification to species was not possible.

		# of observations				
Scientific name	Common name	Hard-high	Hard-low	Soft-high	Soft-low	Depth range (m)
Carcharhinus spp.		2		1	1	114-234
Carcharhinus galapagensis	Galapagos Shark				1	136
Carcharhinus plumbeus	Sandbar Shark		3		1	114-149
Plesiobatis daviesi	Giant Stingray				1	164
Dasyatis lata	Brown Stingray	1	1		1	117-204
Decapturus moroadsi	Amberstripe Scad		1			211
Aprion virescens	Uku	1		1	1	119-124
Caranx melampygus	Bluefin Trevally	1				124
Carangoides spp.		1				124
Erythrocles scintillans	Golden Rover	1		1		211-299
Chaetodontidae		1				124
Apolemichthys arcuatus	Bandit Angelfish	1				124
Luzonichthys earlei	Earle's Anthias			1		123
Bodianus albotaeniatus	Hawaiian Hogfish	1				124
Sufflamen fraenatum	Bridled Triggerfish				1	121
Naso spp.	Unicornfish	1				124

Table 5. Diversity indices of all fishes for each of the four areas sampled and for only the commercial bottomfish species. Total number of species (S), total number of fish (N), Margalef's species richness (d), Pielou's evenness index (J′), and Shannon's diversity index (H′) are given. Area abbreviations are as for Table 3.

all fishes	Area	S	N	d	J′	H′
	E	24	473	3.73	0.745	2.37
	F	24	749	3.48	0.701	2.23
	H	16	508	2.41	0.664	1.84
	KIR	27	671	4.00	0.692	2.28

commercial bottomfishes	Area	S	N	d	J′	H′
	E	10	264	1.61	0.743	1.71
	F	11	657	1.54	0.773	1.85
	H	8	479	1.13	0.778	1.62
	KIR	11	479	1.62	0.809	1.94

Table 6. The proportion of fish measured which were at or above the size of 50% maturity published for the species (Kikkawa 1984; Kikkawa and Everson 1984; Everson et al. 1989; Everson 1992; Demartini and Lau 1999; Humphreys, unpub. data). Only those species for which the size at 50% maturity is known are given below. For total numbers of measurements in a given area see Fig. 9. nd = no data.

	Ehu	Greater Amberjack	Hapu‘upu‘u	Kalekale	Lehi	Onaga	‘Opakapaka
size at maturity (mm)	279	780	579	290	727	700	520
E	78%	15%	0%	100%	nd	17%	17%
F	92%	19%	67%	53%	50%	24%	21%
H	91%	32%	69%	3%	nd	0%	28%
KIR	97%	46%	82%	91%	41%	6%	43%

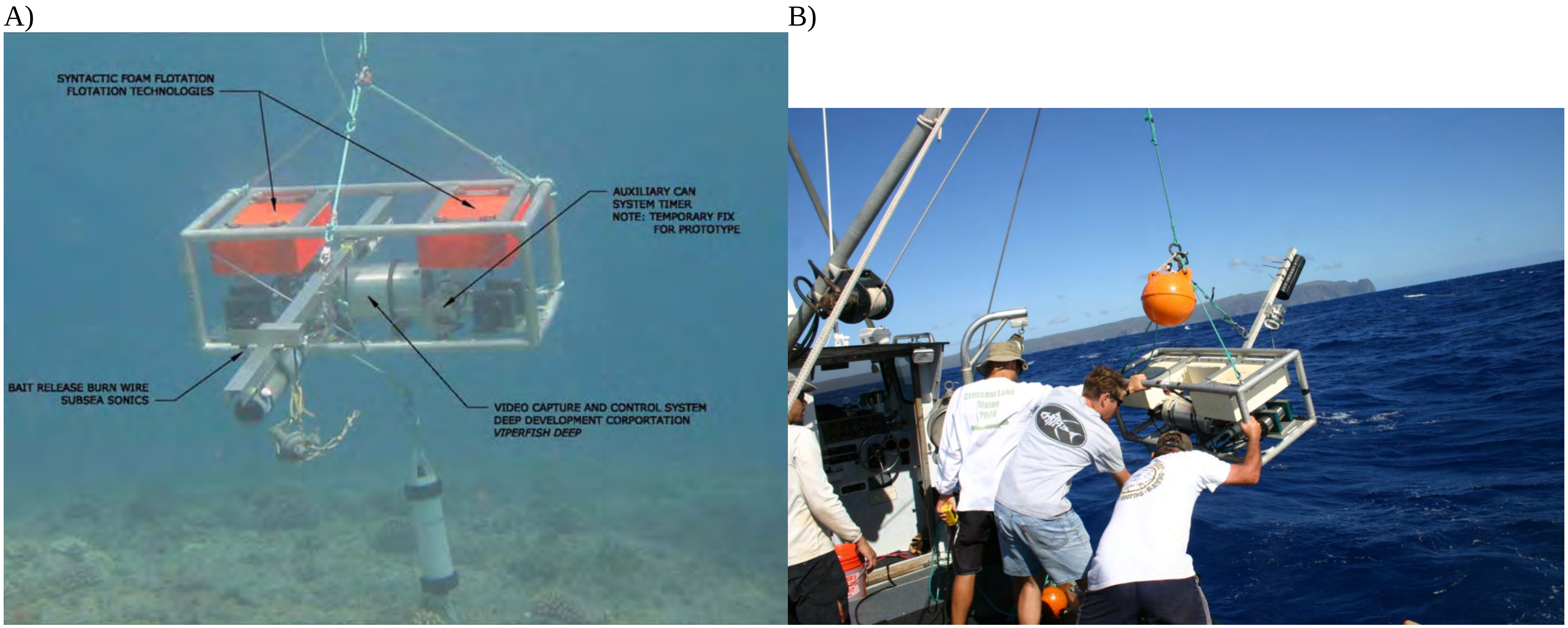

Figure 1. The bottom-camera bait station (BotCam). A) Shown on the seafloor in an earlier configuration which illustrates the suspension of the instrument off the seafloor using an anchor, acoustic release (white cylinder under the instrument) and surface floats. B) The BotCam in the configuration used for the present study in KIR being deployed at sea.

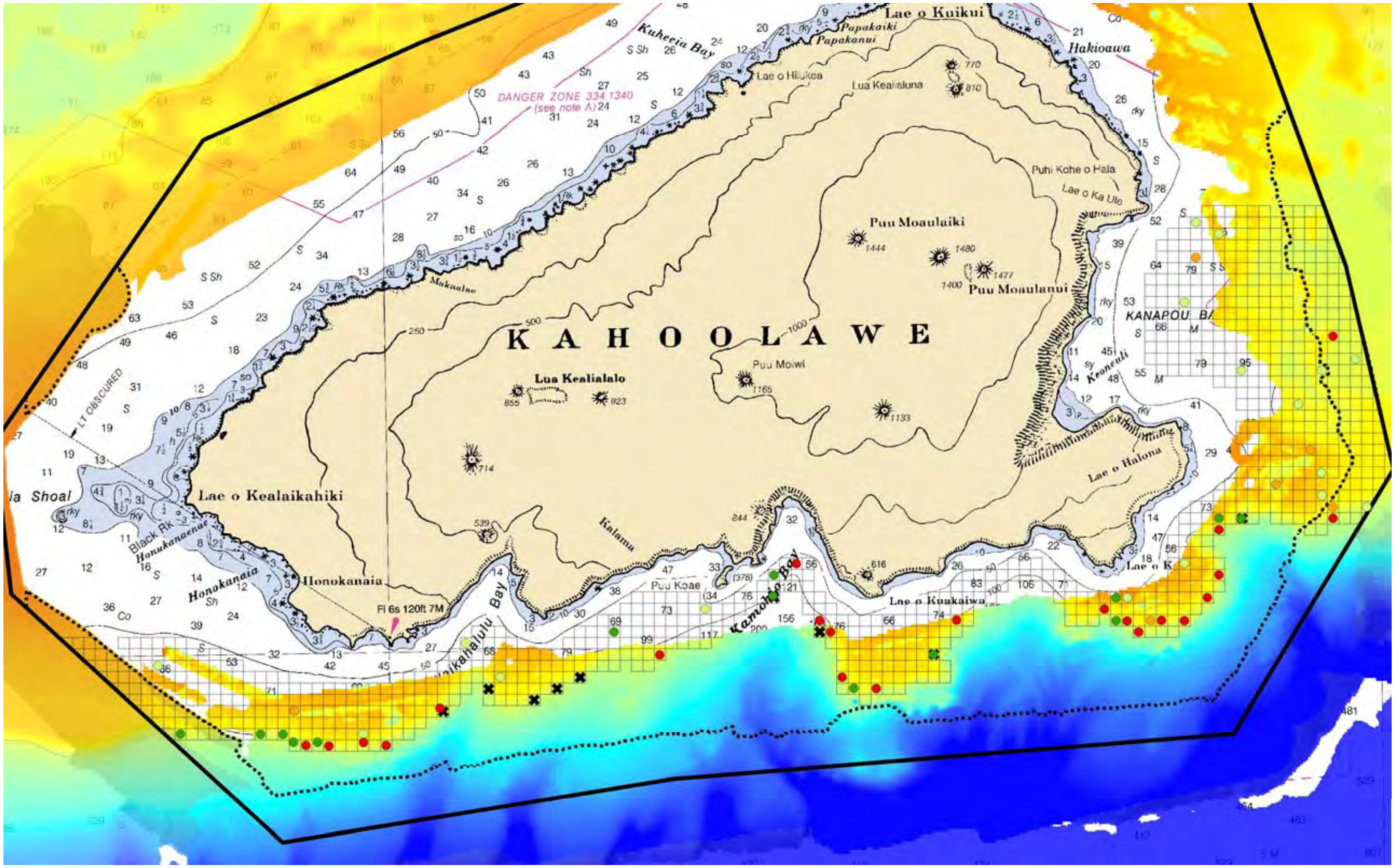

Figure 2. Bathymetric map of Kaho'olawe Island Reserve and BotCam deployment sites. All existing multibeam bathymetry is given in colors while the most current NOAA nautical chart fills in the rest of the region showing the 50 and 100 fathom contours. The solid bold line delineates the boundary of the reserve and the dotted line denotes the farthest shoreward extend of acoustic backscatter data. Bottomfish habitat between 100 and 300 m along the south and east shores is gridded (200x200m). Grids with symbols are those that were sampled with a BotCam unit (total of 74 deployments): x – rejected deployment, red – hard-high habitat, orange – hard-low, dark green – soft-high, and light green – soft-low.

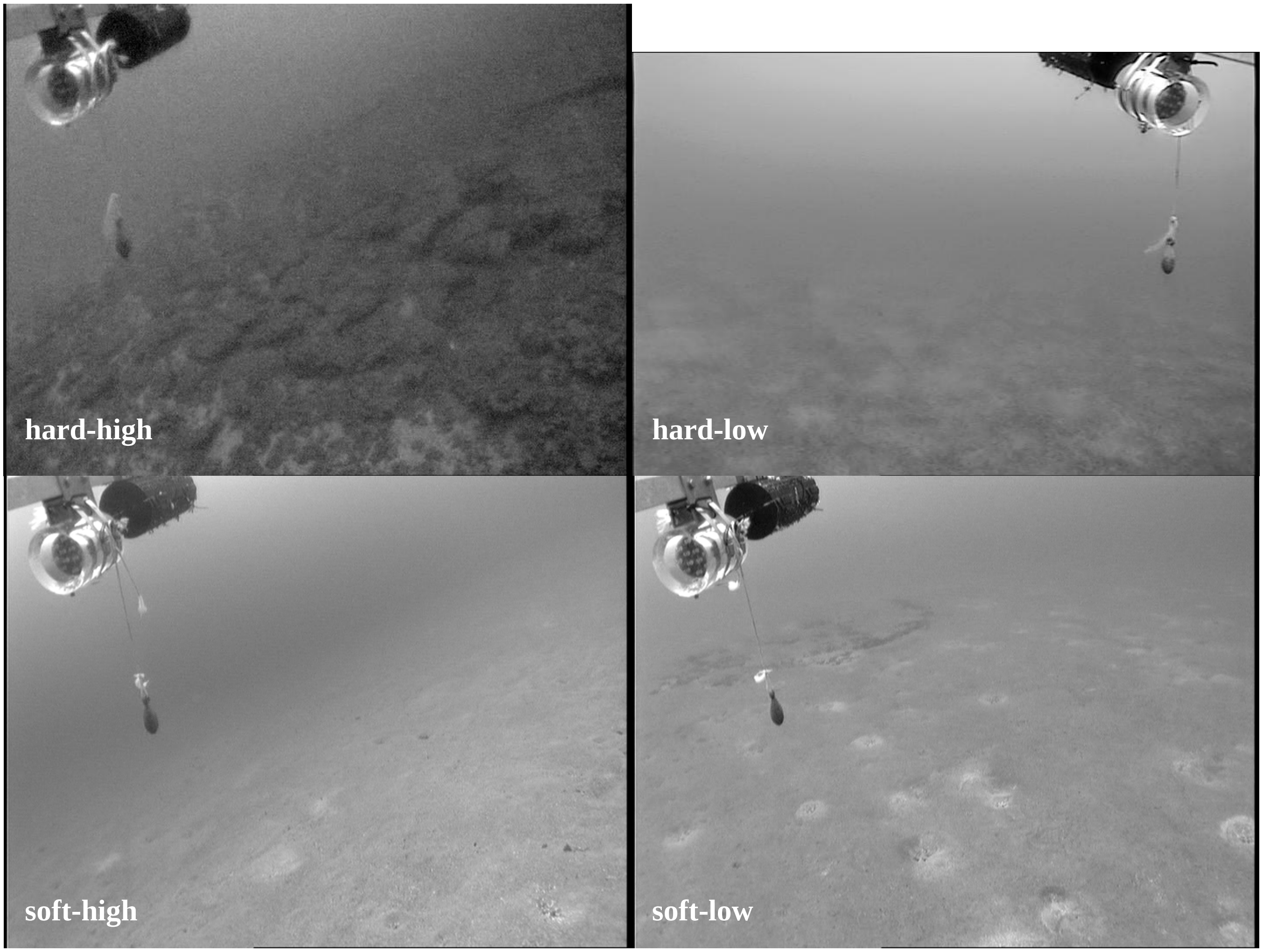

Figure 3. Examples of the four main substrate types as observed in the Kahoʻolawe Island Reserve. The light sync is visible in each frame with a plumb weight hanging below it. Behind the light sync is the bait canister.

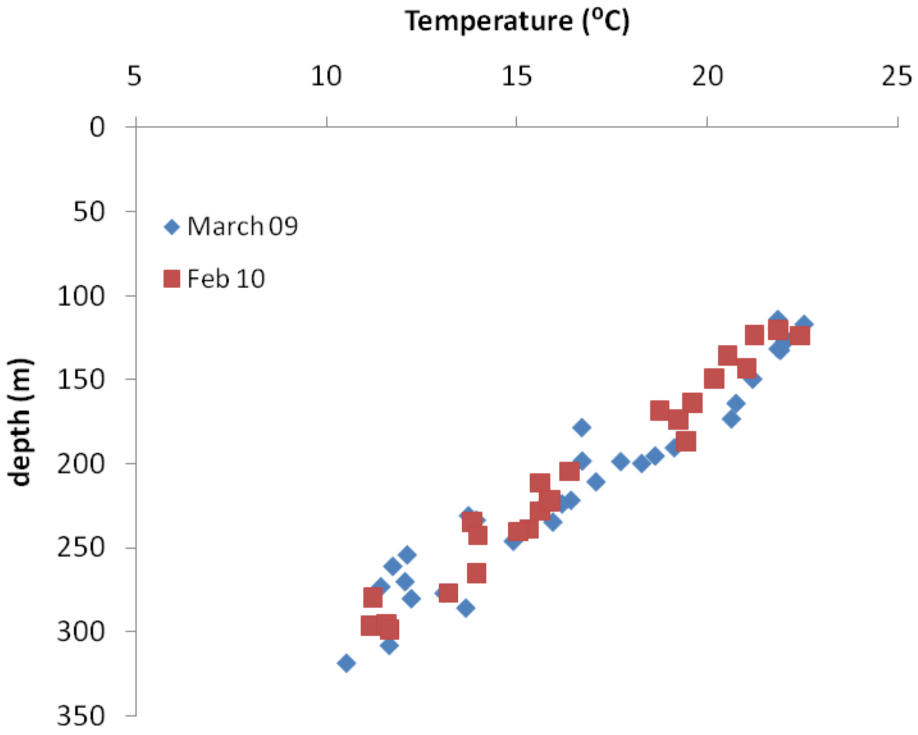

Figure 4. Temperature as a function of depth for the BotCam deployments in KIR.

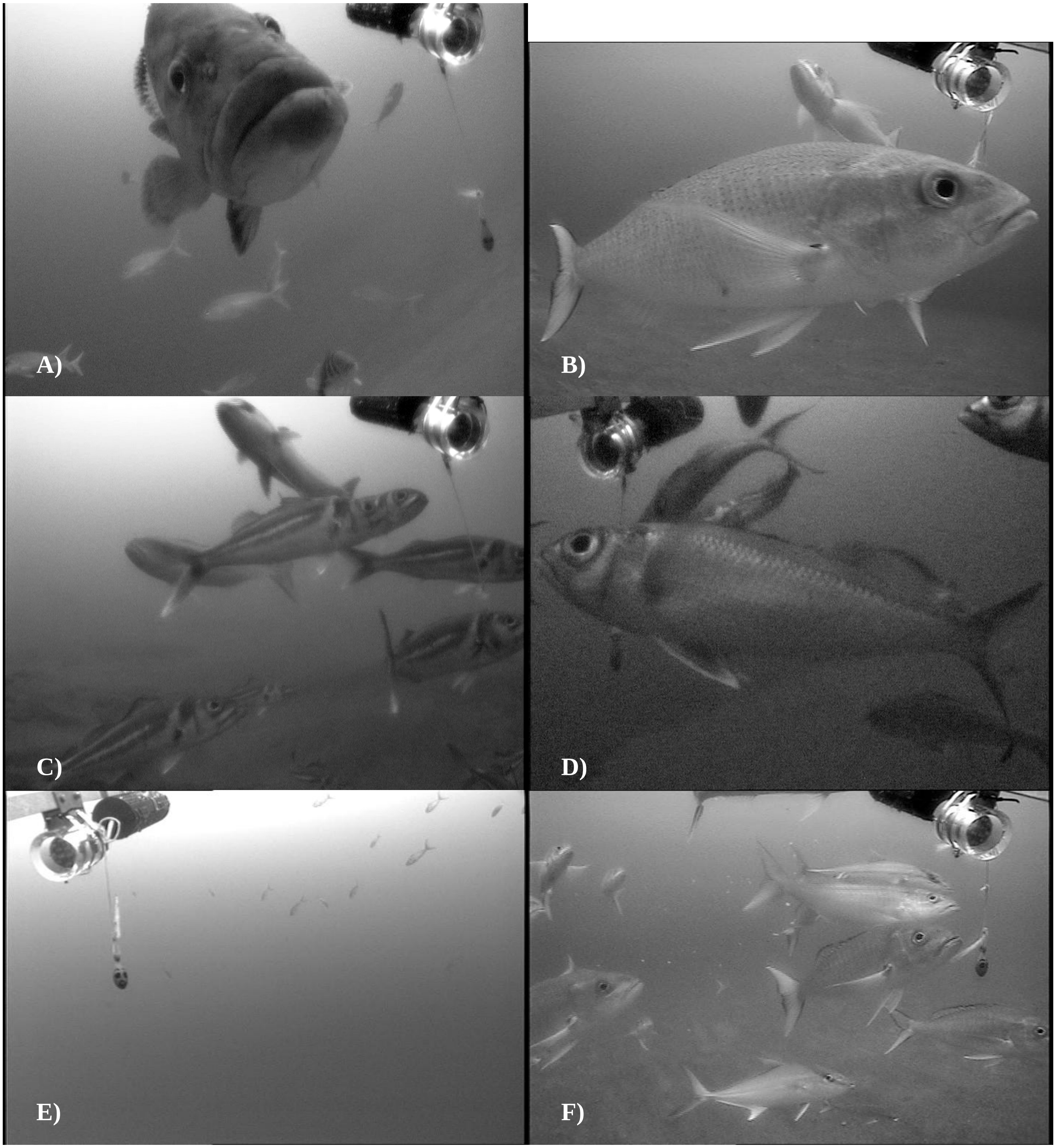

Figure 5. Images of bottomfish in KIR. A) Hapu‘upu‘u, kalekale, and one gindai B) ‘Opakapaka, C) Ehu, one ‘opakapaka and one kahala, D) Onaga, E) school of ‘opakapaka and lehi in midwater during BotCam descent and F) school of ‘opakapaka and kahala on the seafloor.

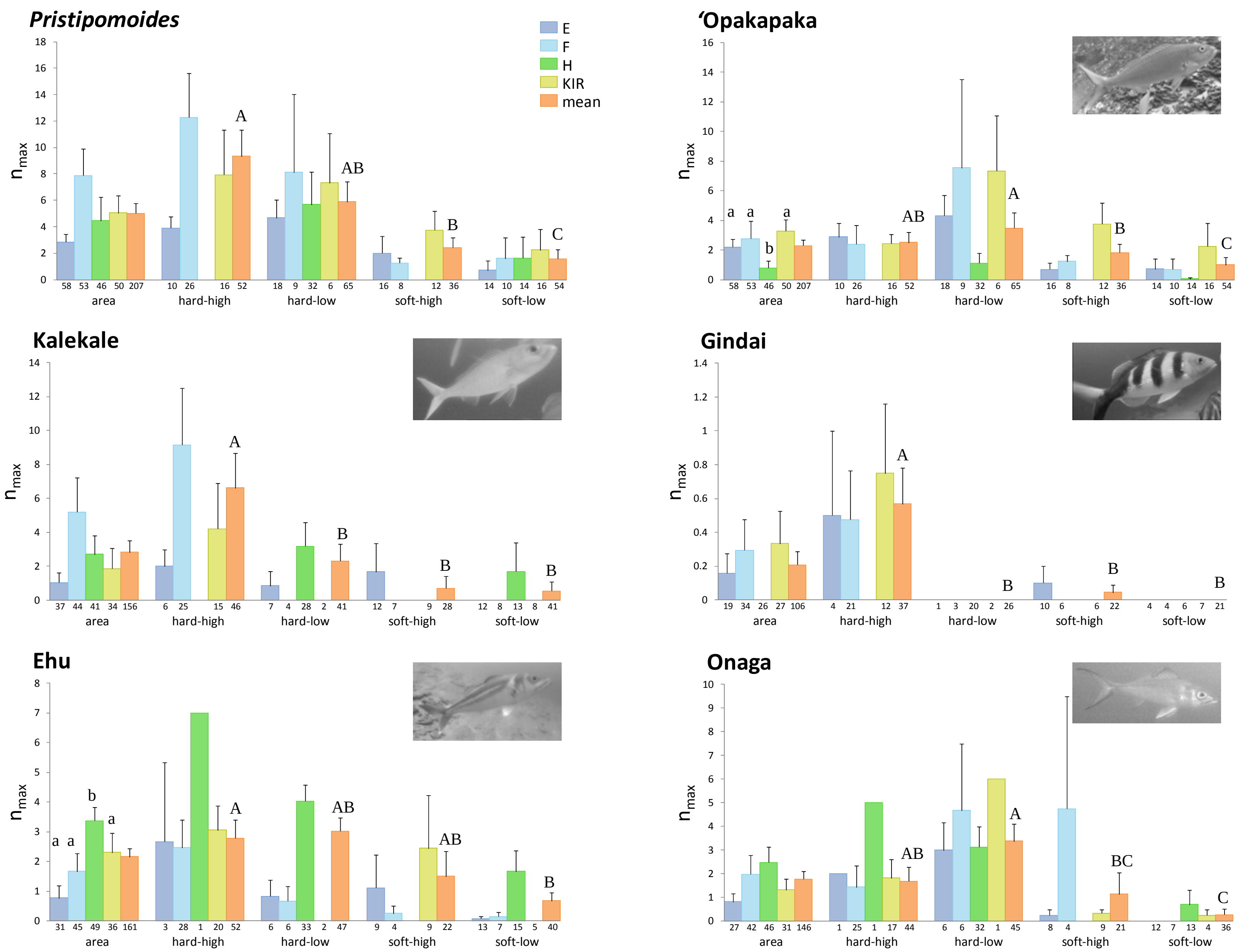

Figure 6. see caption on next page

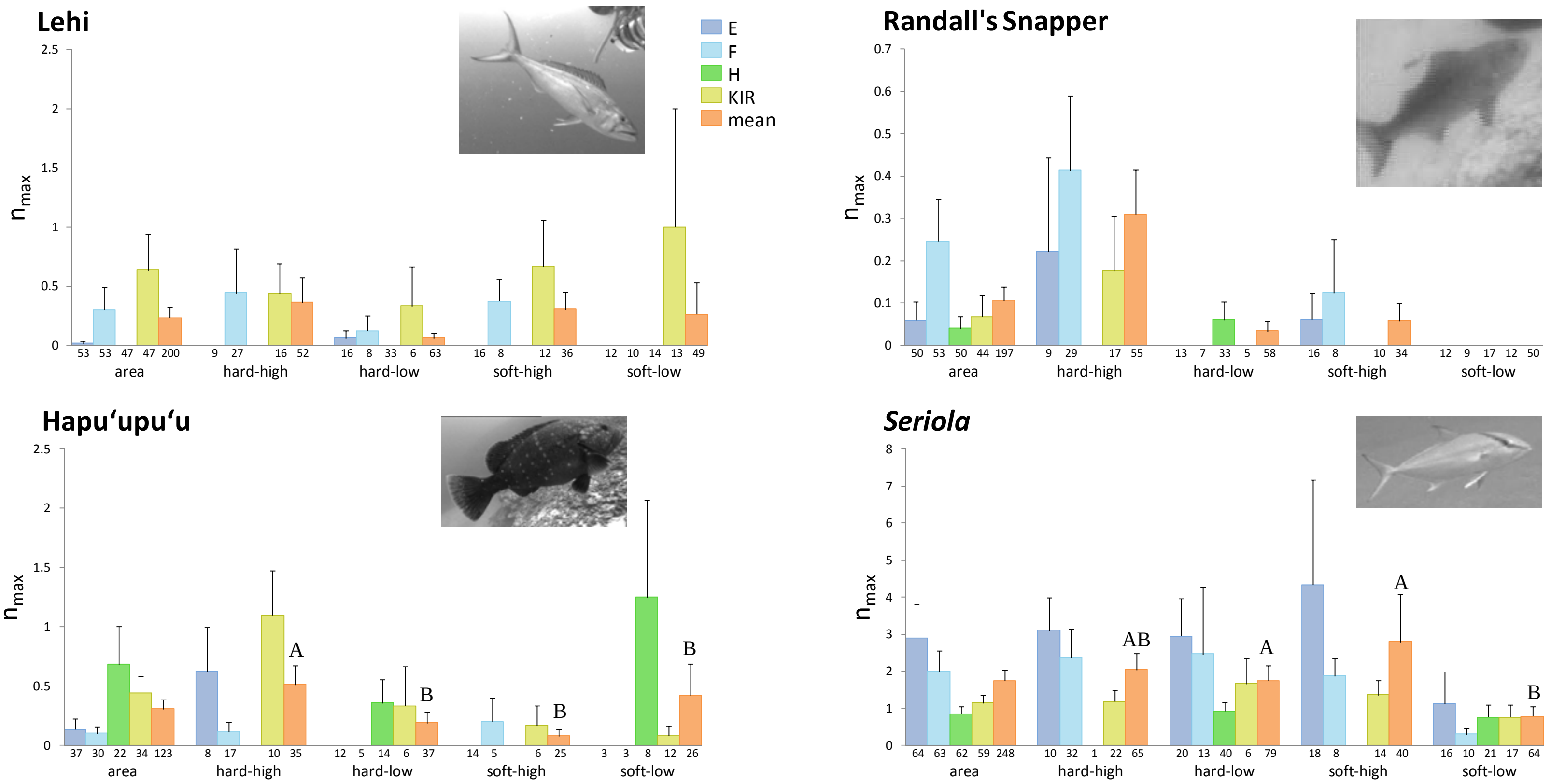

Figure 6. Relative abundance (mean ± std. error of n_{max}) of bottomfish in four areas of the main Hawaiian Islands (E – Makapuu, F – Penguin Bank, H- Pailolo channel, KIR – Kaho‘olawe Island Reserve). The data are derived from those drops within each species depth range (Table 2) and the sample sizes are given below each bar. Note the very different scale of the y-axis between panels. Bars within each stanza have the same lowercase letter if there are not significantly different from each other. Between stanzas, upper case letters indicate a lack of significant differences between habitat types overall.

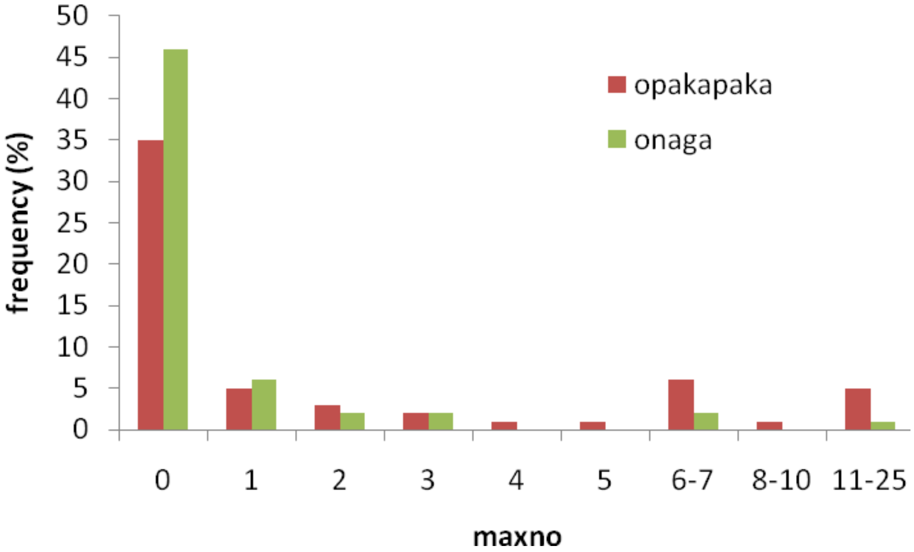

Figure 7. Frequency distribution of the n_{max} for ‘opakapaka and onaga in 59 KIR samples showing the non-normal, aggregated, distribution indicative of the aggregated distribution in these fish resulting from schooling and the patchy distribution of preferred habitat.

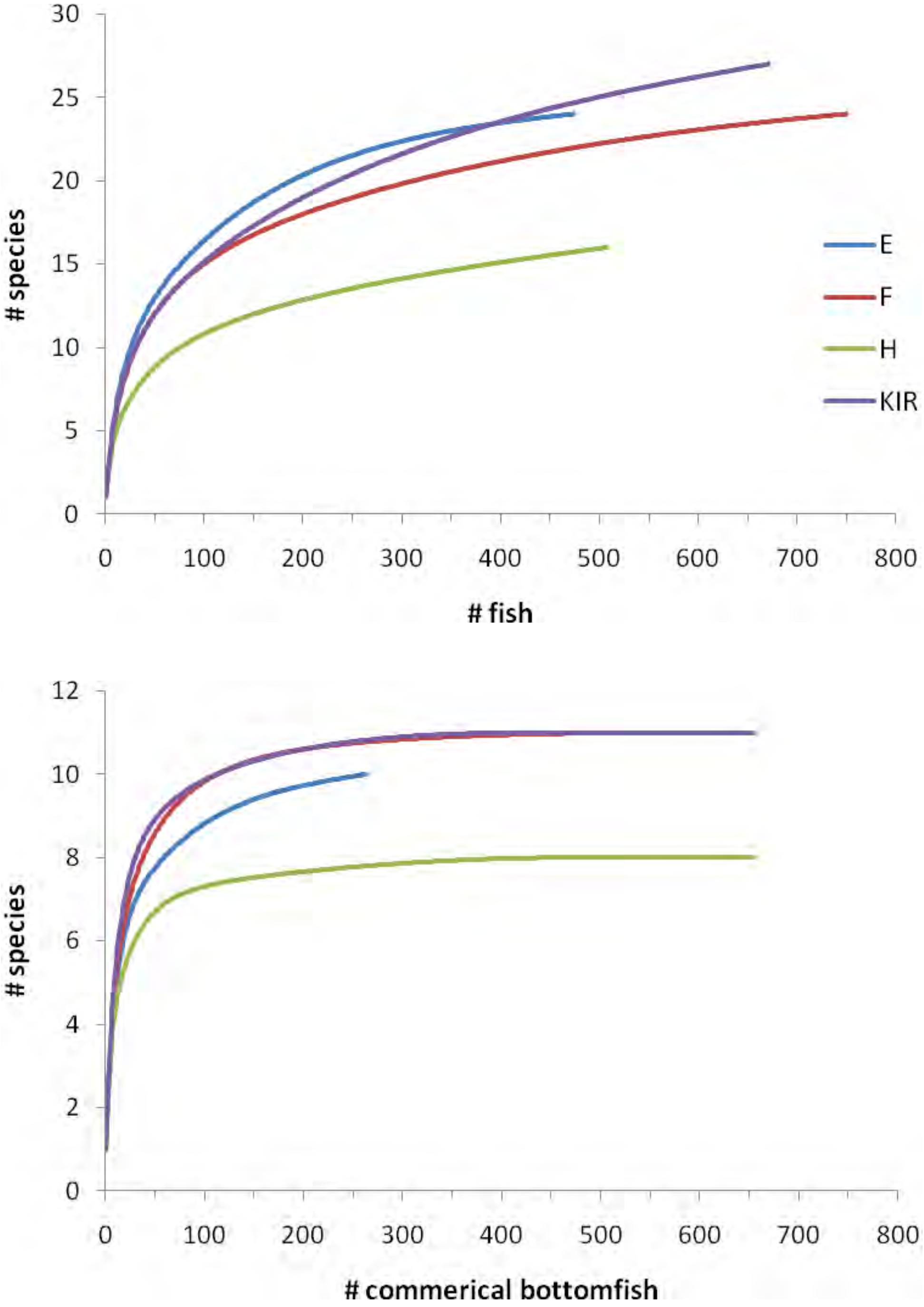

Figure 8. Rarefaction curves showing the mean number of species in each sampled areas as a function of the number of fish observed.

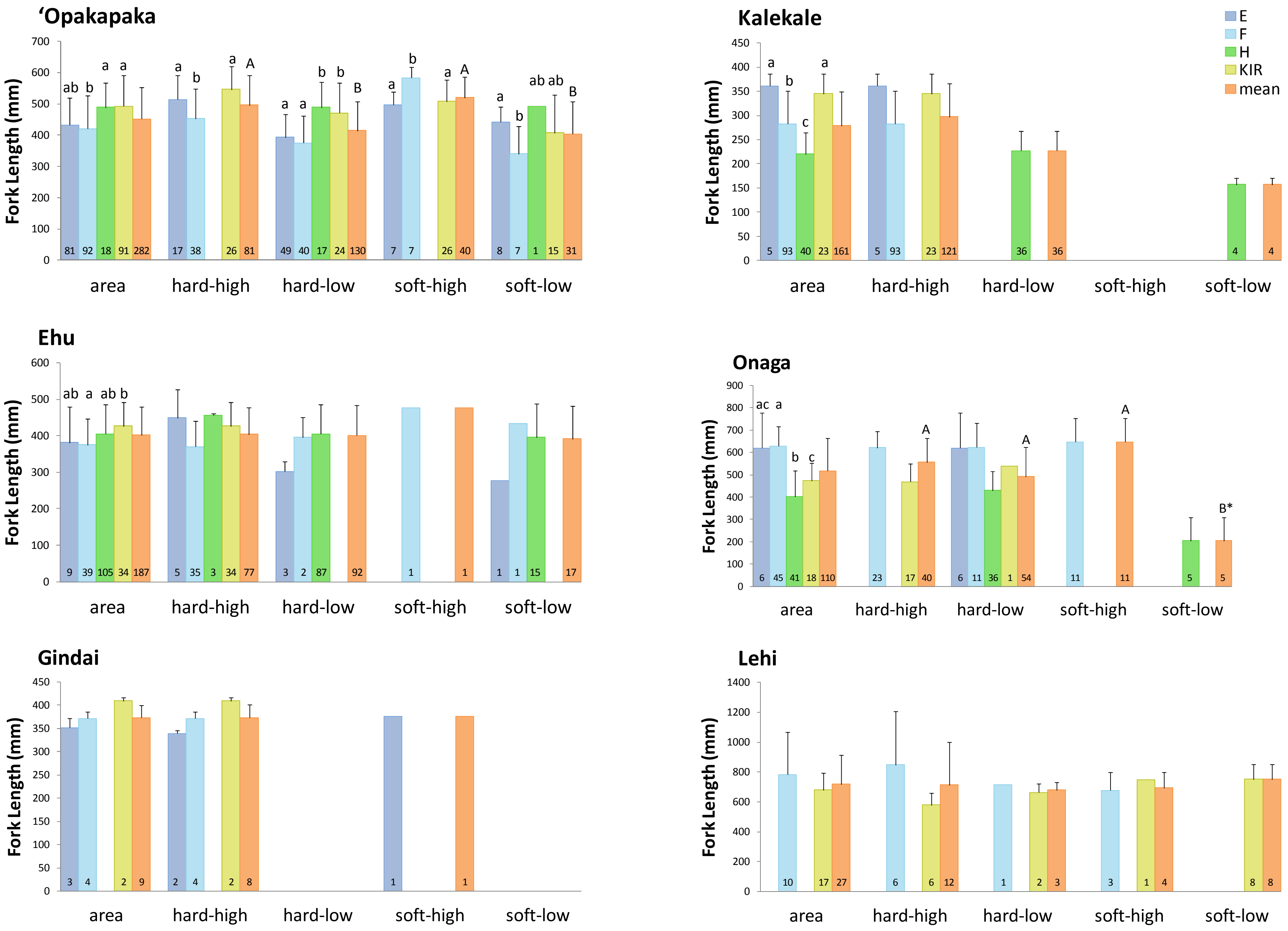

Figure 9. See next page for caption.

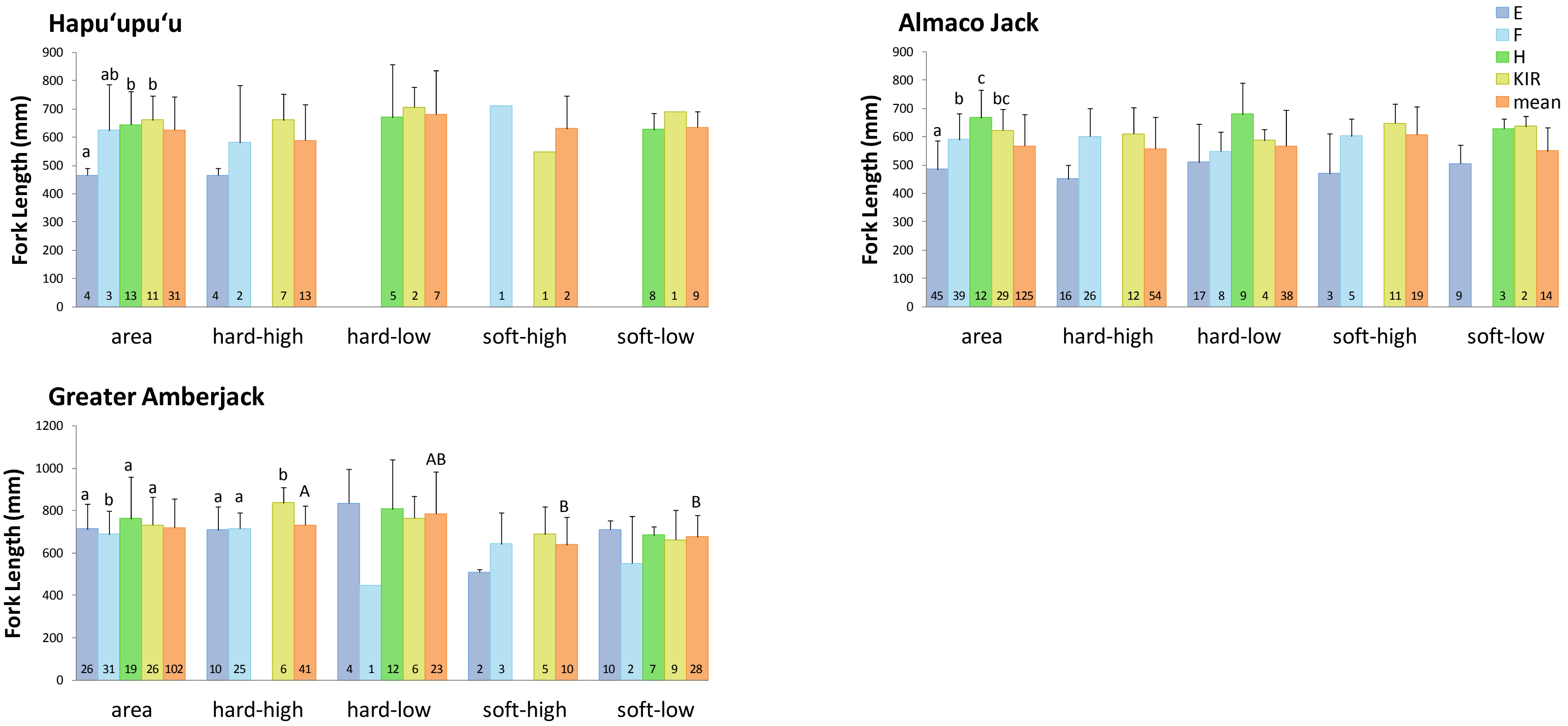

Figure 9. Fork lengths (mean ± std. dev) of bottomfish in four areas of the main Hawaiian Islands (E – Makapu‘u, F – Penguin Bank, H-Pailolo channel, KIR – Kaho‘olawe Island Reserve). The number of fish measured is given at the bottom of each bar. Bars within each stanza have the same lowercase letter if there are not significantly different from each other. Between stanzas, upper case letters indicate a lack of significant differences between habitat types overall. In the panel for onaga B* over the soft low habitat bar indicates that a test could only be performed between this habitat and hard-low because data was present for Pailolo channel in these two habitat categories only.

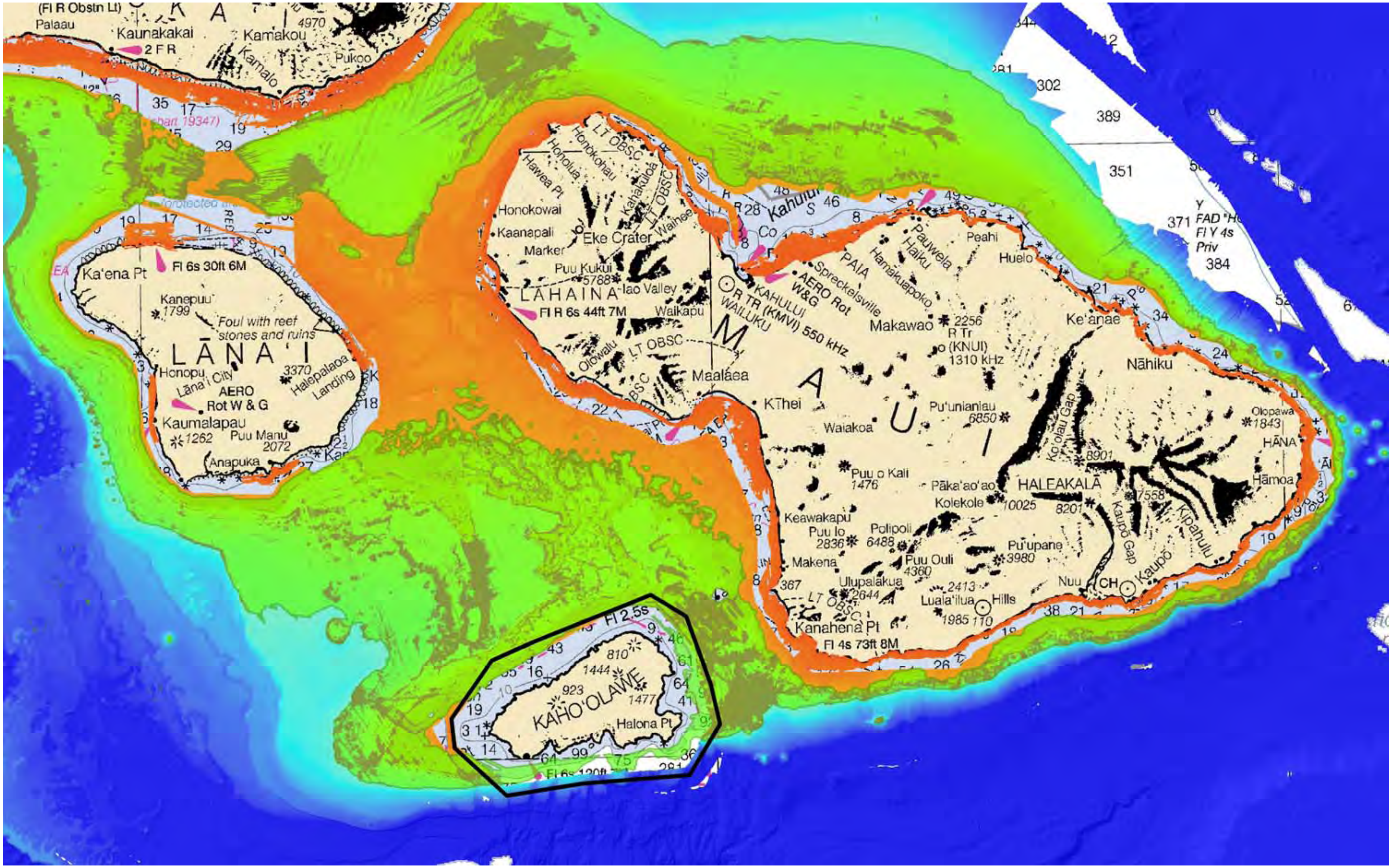

Figure 10. Map showing potential adult bottomfish spillover conduits from KIR to adjacent habitats. The green areas are those between 100 and 400m designated as Hawaiian bottomfish adult habitat. Darker shades represent hard substrates. There is suitable bottomfish habitat to the west and to the northeast of KIR. Movements of bottomfish might occur across the ‘Alenuihaha channel to Hawaii but the frequency of such transits is not known.

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Dear Commercial Bottomfish fishers and dealers-Please let us know what kind of information is useful to you. We welcome your feedback! Any feedback about the newsletter, positive or negative, is greatly appreciated! PLEASE CALL statistical staff member, Jessica Miller, (808) 587-0594 or e-mail dlnr.ar.bf@hawaii.gov. Mahalo! - DAR Statistical Unit

Thank you to everyone for your ideas and assistance in editing the Bottomfish Newsletter Volume 14! A special thanks to: Caitlin Burgess, Francis Oishi, Reginald Kokubun, Eric Yokomori, Jeffrey Drazen, Dana Sackett, Ana Vaz, and Cordelia Moore .

Editor: Jessica Miller

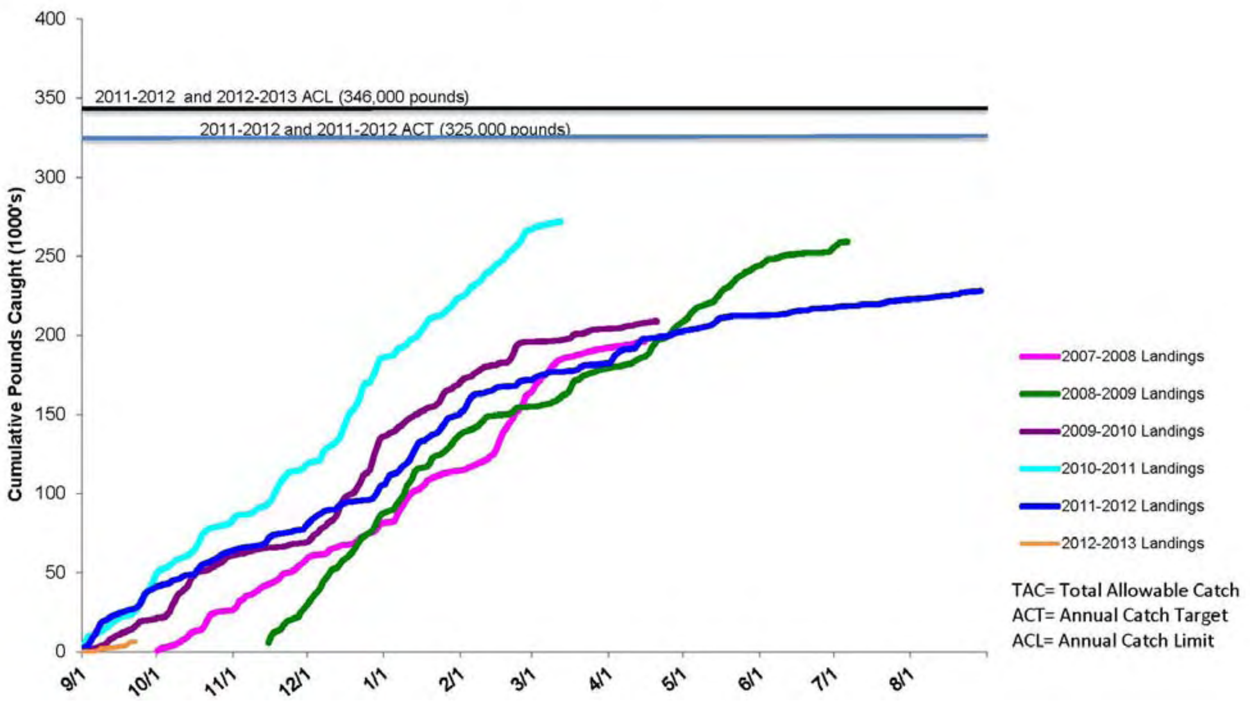
BOTTOMFISH NEWS

Volume 14
October 2012

Summary of the 2011-2012 MHI Deep 7 Bottomfish Fishing Year

The 2011-2012 bottomfish fishery closed on August 31, 2012. 228,388 pounds of Deep 7 bottomfish were reported landed (70.3% of the 325,000 pound Annual Catch Target (ACT)). A total of 468 fishers reported 3,075 Deep 7 bottomfish trips. This is the first year since catch limits began in 2007 for the Deep 7 Bottomfish fishery that the fishery remained open the full year. Many fishers credit weather as the major factor that prevented the ACT from being reached in 2011-2012 fishing year. The 2012-2013 fishery opened on September 1, 2012. Sixty commercial fishers have reported making 105 Deep 7 bottomfish trips and landing 6,411 pounds of Deep 7. (Data as of September 25, 2012.)

Comparison of MHI Deep 7 Bottomfish Landings from 2007 to Present with current catch limits (as of 09/25/2012)



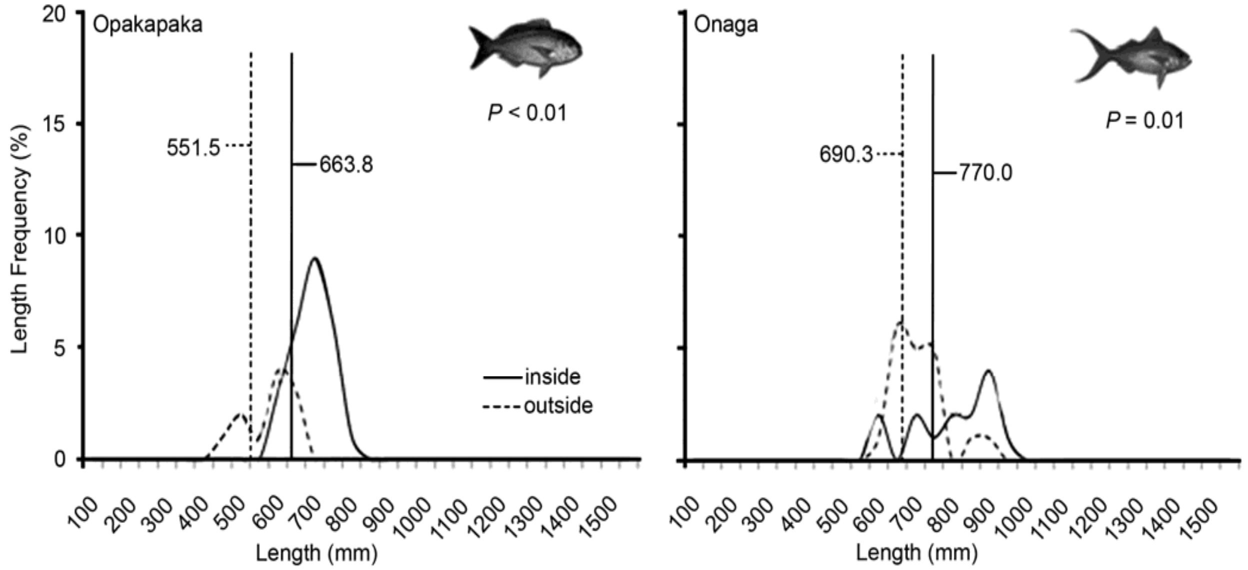
7th Annual Hawaii Fishing and Seafood Festival

Don't miss the Hawaii Fishing and Seafood Festival on Sunday October 7, 2012 from 9am-4pm. It will be at the Honolulu's Fishing Village at Pier 38. For general information please visit their website, http://www.hawaiifishingfestival.com/. The festival has over 100 vendors and fun for the whole family. This year the Deep 7 Hawaii Bottomfish booth will have information about current Deep 7 Bottomfish research along with scientists on hand to answer questions regarding their research. Deep 7 monitoring staff will also be on hand to answer questions regarding reporting.

Bottomfish Restricted Fishing Areas and the deep 7: a report of current monitoring results.

In 1998, the State of Hawaii, Department of Land and Natural Resources implemented 19 BRFAs throughout the Main Hawaiian Islands (MHI). Because of new information on bottomfish and their habitat the system was substantially revised on June 1, 2007. The new system of BRFAs reduced the overall number to 12, but increased the area protected to include more Essential Fish Habitat (those areas necessary for fish feeding, growth to maturity or spawning) for bottomfish. The state's goals were to increase the size and abundance of bottomfish inside the reserves with the long term goal of realizing spillover to neighboring fished habitats and higher production of eggs and juveniles from larger fishes. Monitoring of the new BRFAs was mandated by the state to determine their effectiveness. This monitoring was tasked to Dr. Jeff Drazen's lab at the University of Hawaii. While other successful marine reserves have measured increases in fish abundance and fish size as a result of protection, bottomfish in the MHI, can live up to 40 years and can take nearly a decade to reach maturity; so benefits from these BRFAs may take a while to detect. However, after examining nearly four years of data we are starting to see some positive benefits as a result of the BRFAs.

Relative abundance and fish size have often been monitored using data collected from fishing (i.e. catch totals or CPUE). However, fishing was not a viable method inside of the BRFAs. Instead we used a baited camera system (BotCam), designed specifically for monitoring Hawaiian bottomfish and their habitat. Fish can be identified, counted and the system allows precise and accurate length measurements. Measuring relative abundance is important because it can tell us whether bottomfish populations are changing over time as a result of protection. Also because large females contribute most of the eggs for the next generation of fish, increases in fish size as a result of protection can have a big impact on recruitment and the size of bottomfish populations in the future.



Graphs showing a significant increase in the average length of opakapaka and onaga inside versus outside the Ni ʻihau BRFA.

Results have shown significantly larger opakapaka and onaga inside the Ni ʻihau BRFA compared to outside. This particular BRFA had been protected for ten years; since the inception of the original BRFAs in 1998. Interestingly, the increase in fish size within the BRFA was equivalent to 10 years of growth. In contrast, the Hilo BRFA had significantly smaller opakapaka and kalekale within the reserve compared to out. These results are likely because (1) the shallow depth range (< 100 fathoms) of this BRFA was not protected prior to 2007; this is particularly important for opakapaka, which most often reside within this shallower range, and (2) the areas outside the BRFA to the south have limited accessibility possibly creating a natural reserve. Indicative of the potential success of the BRFAs, monitoring results from the Kaho'olawe Reserve (KIR), which was established in 1994, has shown significant improvements in bottomfish diversity, fish size, and an increase in the proportion of mature fish within the reserve.

Bottomfish Restricted Fishing Areas and the deep 7: a report of current monitoring results (cont.)

An image from BotCam of a school of opakapaka.

More recently, we have been examining fish size and abundance over time. This type of analysis tells us whether fish size and abundance are changing over time as a result of protection or whether our results have always been there (for example if there have always been large fish in the BRFAs). So far, our results show an increase in the size of ehu and onaga inside the BRFAs as a result of protection. We also found that the size of opakapaka and kalekale has increased inside and outside the BRFAs. Although, we found higher fish abundance inside many of the reserves compared to out, this has not changed over time.

One of the many benefits of areas with populations of large fish is that older and mature individuals tend to produce higher quality eggs and more of them when compared to younger fish of the same species. Fish eggs and larvae are small in size and have limited swimming capabilities, so ocean currents can help transport these small organisms. Contribution to the rebuilding of fishing populations is contingent on the offspring of large fish staying or being transported to good habitat. A recent study at the University of Hawaiʻi (Ana Vaz) simulated the dispersal of eggs and larvae from three of the Deep-7 fish species (opakapaka, ehu and onaga). Results indicate that eggs released inside BRFAs in the MHI are very likely to be transported to areas open to fisheries. Study results also indicate that eggs released in the area between Hawaiʻi and Oʻahu stay in this region, while eggs released around Kaua'i, Niʻihau, Ka'ula and Middle Bank stay around the MHI and are exported to the Northwestern Hawaiian Islands. However, eggs spawned in the Northwest Hawaiian Islands do not contribute to the MHI populations of bottomfish.

These results suggest that the BRFA system is protecting larger fish, which could lead to increased recruitment and ultimately increases in bottomfish abundance. However, help is needed from our local communities to protect our important bottomfish resources. We all have the same goal, to continue the harvest of bottomfish for generations to come; if we work together in protecting these small refuges so that bottomfish can eat, grow and contribute to larger future populations of bottomfish, the fishery as a whole will benefit.

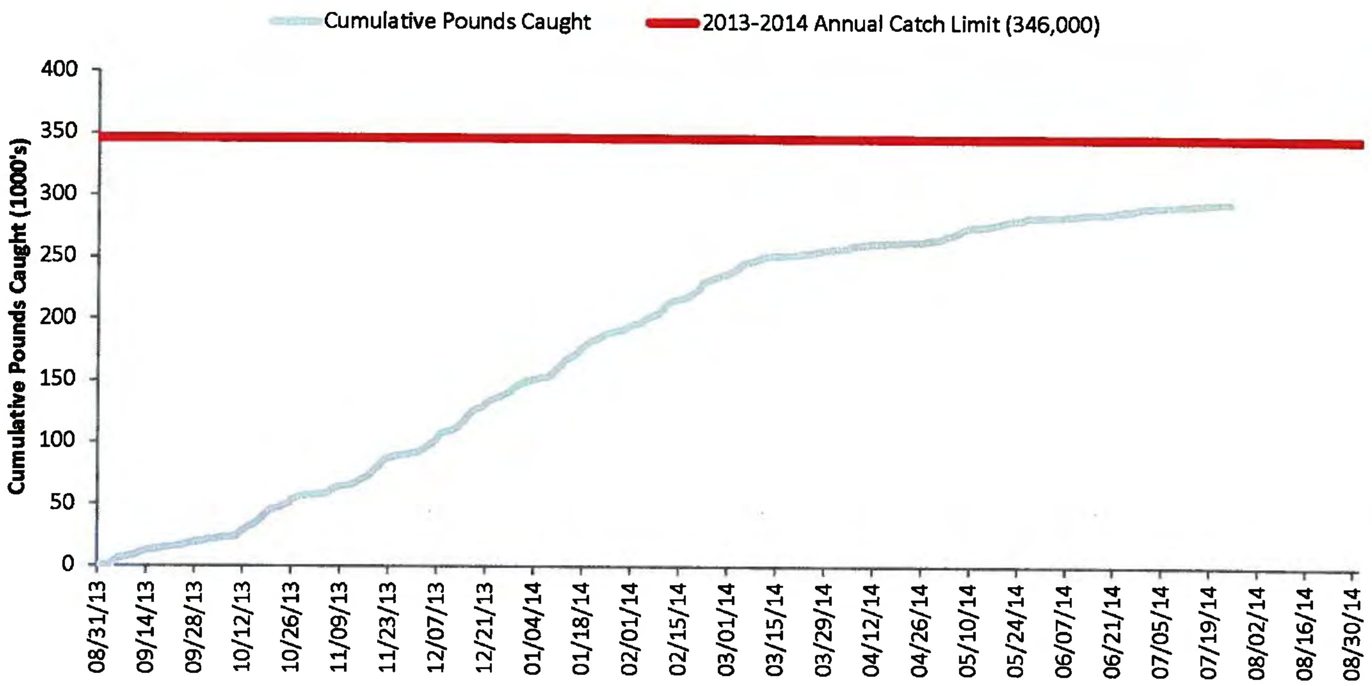
Jeffrey Drazen, Dana Sackett, Ana Vaz, and Cordelia Moore - University of Hawaii

BOTTOMFISH NEWS

Volume 17

August 2014

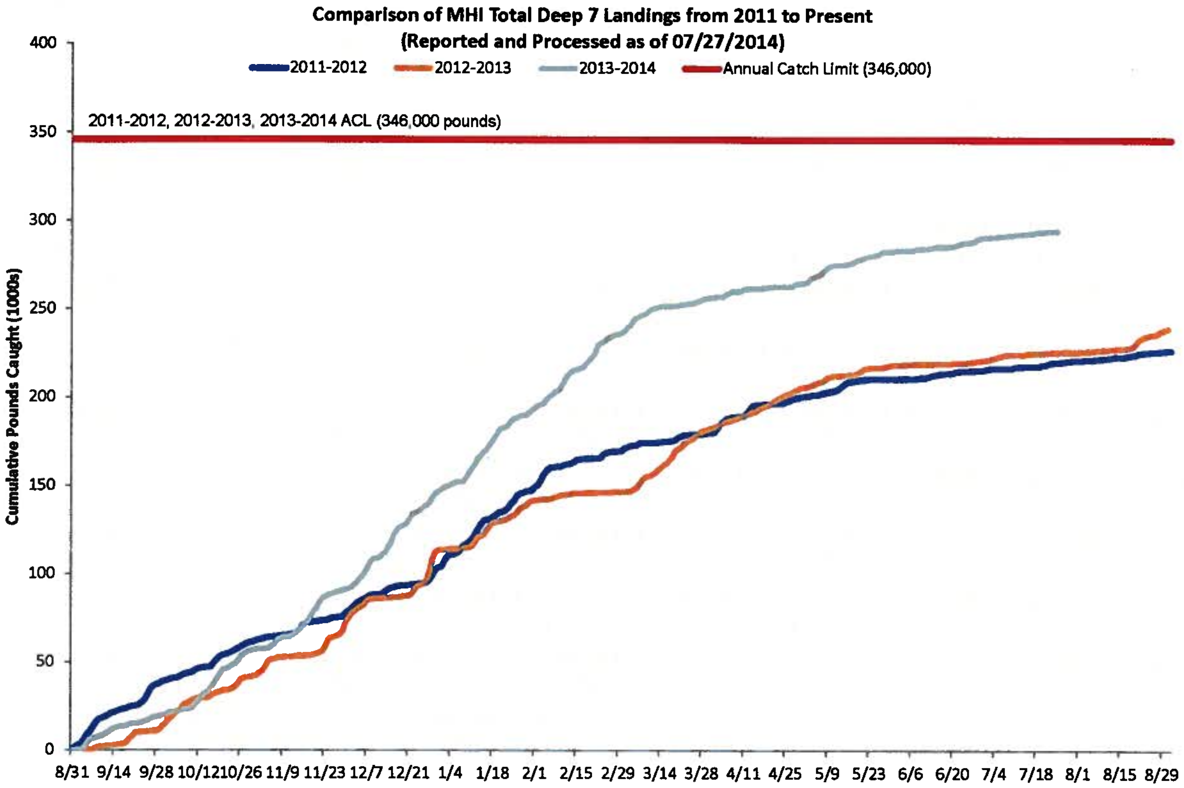
2013-2014 MHI Deep 7 Bottomfish Landings



Main Hawaiian Islands Monthly Catch of Deep 7 Bottomfish (Includes ONLY data Recieved and Processed as of 07/27/2014)

As of 7/27/2014, 407 commercial bottomfish fishers have reported landing 294,405 pounds of Deep 7 bottomfish, or 85% of the 346,000 annual catch limit (ACL). This is the largest landings accumulated in a fishing year since the 346,000 pound ACL was implemented on September 1, 2011. In March 2014, due to the high landings, fishery managers began closely monitoring the fishery. The trip report requirement and high compliance rate of 75% allowed fishery managers to make well informed decisions regarding the fishery. They found that the catch rate had decreased significantly lowering the probability that the fishery would reach the ACL. Before the 5 day trip report requirement was implemented in September 2011 there was a month lag time in the data making it difficult to accurately project when the catch limit would be met. Mahalo to all the commercial fishermen who are reporting accurately and on time. Your cooperation permitted the scientists to watch the landing trend more closely and determine that the catch rate was slowing allowing the fishery to continue to stay open. At this point the fishery is projected to remain open through August 31, 2014 and reopen the next day, September 1, 2014 with the same ACL of the last three years, 346,000 pounds. In the event that the landings significantly increase in the remaining month of the 2013-2014 fishing year, all commercial and non-commercial fishers who registered their vessel for bottomfishing will receive a letter. Mahalo for your cooperation!

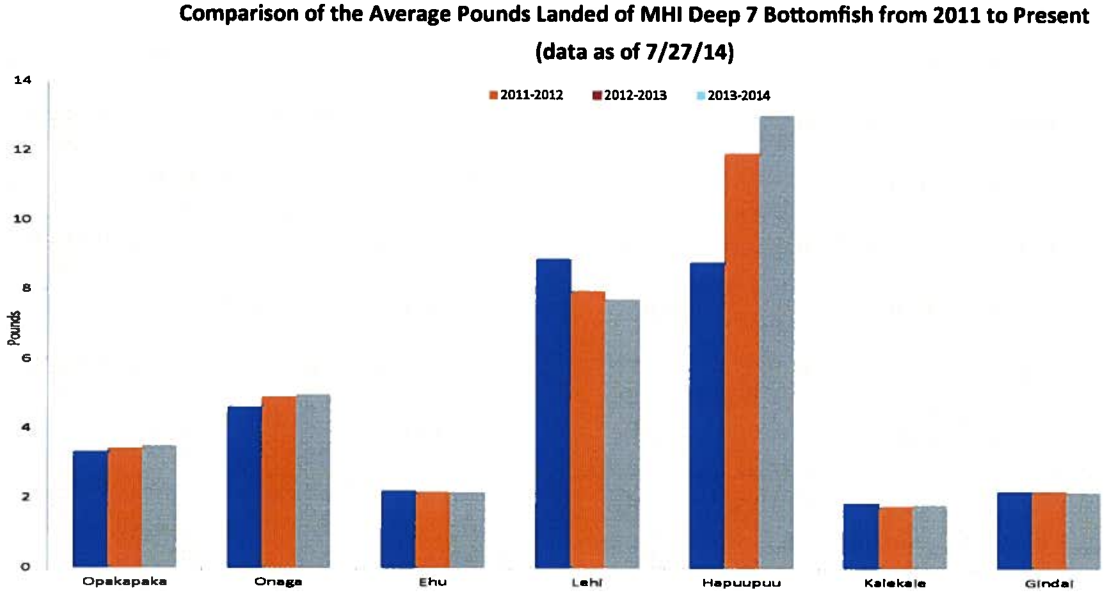
A Comparison of MHI Deep 7 Bottomfish Landings over the last 3 years with an ACL of 346,000 pounds



Comparison of MHI Total Deep 7 Landings from 2011 to Present
(Reported and Processed as of 07/27/2014)

Fishing Year	Number of Fishers	Sum of Landings	Number of Trips	Percent of ACL
2011-2012	479	226,702	3,106	65.5
2012-2013	458	239,034	2,987	69.1
2013-2014*	407	294,405	2,979	85.1

*Fishing year is still open until August 31, 2014



Comparison of the Average Pounds Landed of MHI Deep 7 Bottomfish from 2011 to Present
(data as of 7/27/14)

Larger average size of Deep 7 bottomfish during the 2013-2014 fishing year contributed to the higher landings, especially since there were less active commercial fishers and less trips compared with the two previous years.

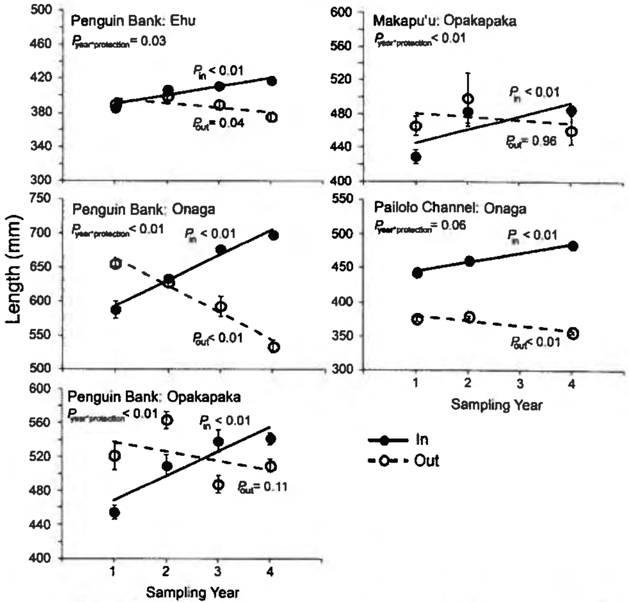
Bottomfish Restricted Fishing Areas and the Deep 7: an update on monitoring results.

Dana Sackett, Jeffrey Drazen, Virginia Moriwake –University of Hawaii

An image from BotCam of a school of opakapaka.

The bottomfish restricted fishing areas, more commonly referred to as BRFAs were initially established in 1998 and revised in 2007 by the State of Hawai'i, Department of Land and Natural Resources (DLNR) in response to declining catch rates and spawning potential ratios of the Deep 7. Monitoring began in 2007 when the system of BRFAs were revised, using a baited camera system (BotCam) to determine if these protected areas, like others, could cause increases in the size and abundance of bottomfish inside the reserves with the long term goal of a higher production of eggs and juveniles and spillover of larger adult fish into neighboring fished areas. Results so far are positive and suggest that both of these goals are beginning to be met.

In the first four years of monitoring data (2007-2011) from four BRFAs showed that fish length, and in some cases abundance, increased for one or more of the most economically important Deep 7 species (ehu, onaga and opakapaka) inside, while outside fish sizes and relative abundance declined or stayed the same. These results demonstrate that fish size and abundance are changing over time as a result of protection rather than having always been there (for example if there had always been large fish in the BRFAs). This study was peer-reviewed by other scientists and published in the journal *Marine Biology* this year. These results were also presented to the Western Pacific Regional Fisheries Management Council and at a Fishers' Talk Story session with DLNR Director William Aila.



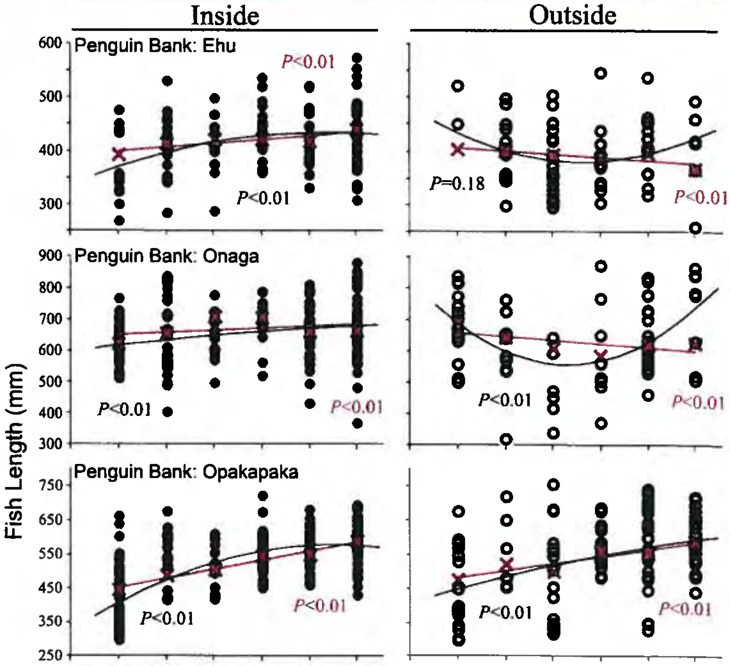
Predicted mean fish length over time inside and outside BRFAs.

Bottomfish Restricted Fishing Areas and the Deep 7: an update on monitoring results. (Continued)

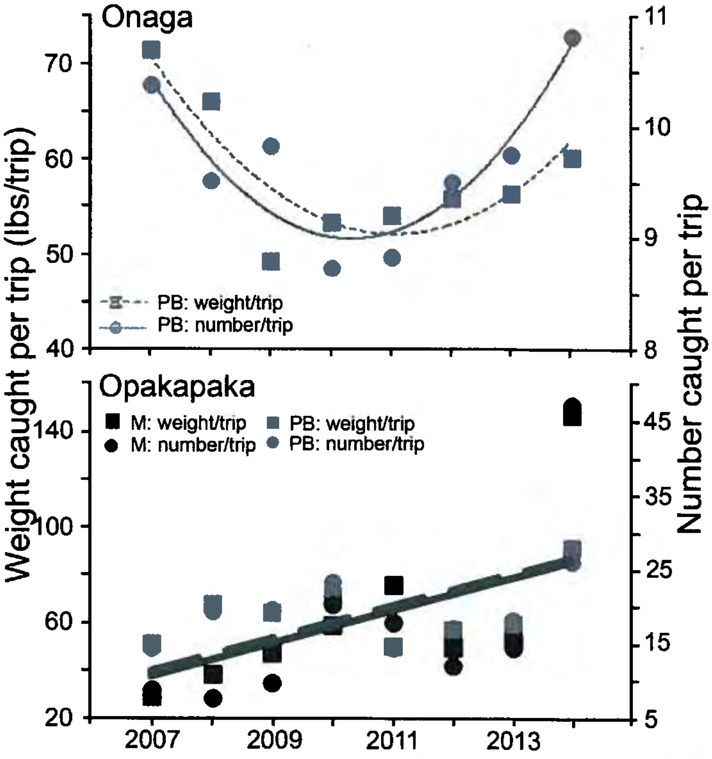
In addition to seeing clear benefits of the BRFAs to Deep 7 populations inside of them we wanted to determine if there was a benefit to the fishery through the spillover of fish into neighboring fished areas. We are using the last two years of monitoring data (2007-2013) that took place in Makapu'u and Penguin Bank BRFAs to achieve this goal. Spillover can be seen as a decline in abundance, size or diversity of fish with distance from the protected area; the result of the protected area acting as a source of more and larger fish to fished areas. Data collected in and around the BRFAs showed this pattern. The abundance, fish size and the number of Deep 7 species seen in a single sample all significantly declined with distance from both Makapu'u and Penguin Bank BRFAs. Furthermore, the declines with distance from the BRFAs only developed in the most recent years. Further supporting these conclusions, the size of ehu, onaga and opakapaka inside Penguin Bank initially increased but this upward trend is leveling-off inside the BRFA in the most recent years while outside fish sizes have begun to increase.

Fisher's catch data recorded in the fishing areas surrounding the BRFAs were also examined and demonstrated the same trends seen from data collected with BotCam outside of BRFAs. These data also showed that the size and number of fish caught per trip were increasing in recent years near the BRFAs. These results suggest that these BRFAs have begun to benefit the Deep 7 fishery.

MAHALO Dana Sackett, Jeffrey Drazen, and Virginia Moriwake for contributing to the Bottomfish News!

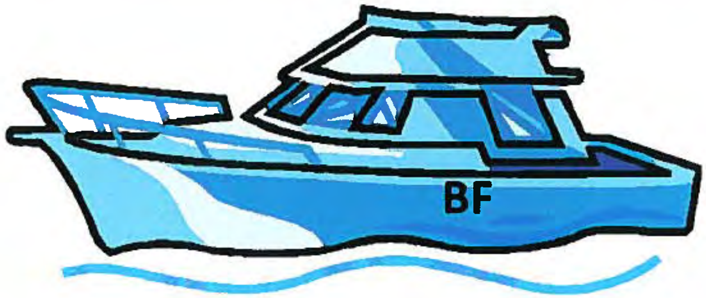


Changes in fish length over time inside and outside Penguin Bank and Makapu'u BRFA. Black = raw data. Red = linear model results.



The weight and number of onaga and opakapaka caught per fishing trip over time around Penguin Bank and Makapu'u BRFAs.

Deep 7 Bottomfish Vessel Registration Reminder

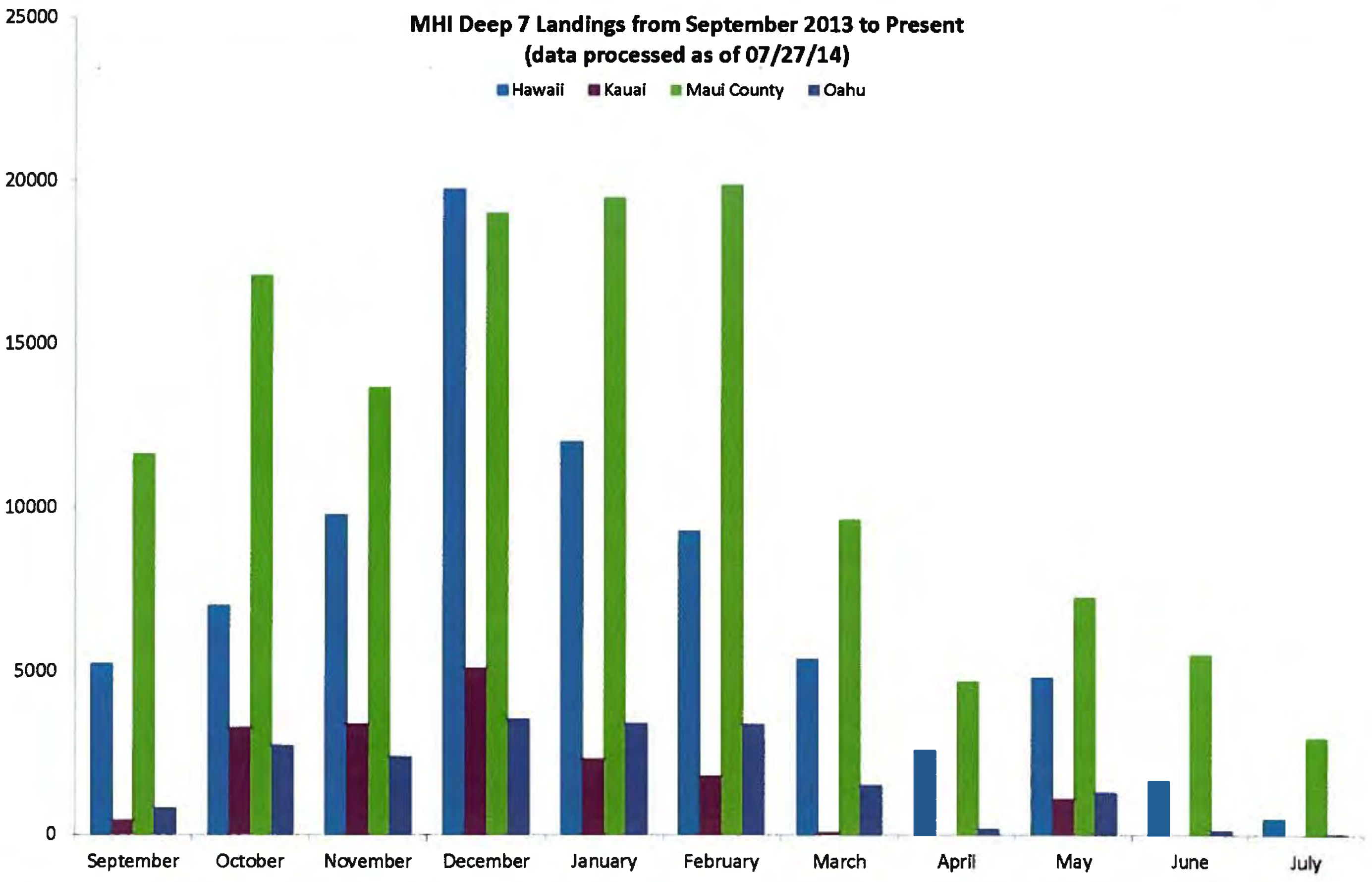


If you plan to catch Deep 7 bottomfish during the 2014-2015 fishing year, please renew your bottomfish vessel registration. There is no fee for this registration. This is the fourth year that fishers have been required to register their vessels annually if they plan to catch any of the Deep 7 species regardless if they are commercial or non-commercial. Fishing years begin September 1 and end August 31 or when the fishery reaches the Annual Catch Limit (ACL) each year. You can renew online: dlnr.ehawaii.gov/cmls or in a DAR office.

Reporting Reminder: Report everything you catch

Stock assessment scientists need to see everything caught on your catch reports in order to make an accurate stock assessment. Stock assessments are used to make management decisions such as setting Annual Catch Limits (ACLs). Please be sure to report every fish that you caught whether it is sold or kept for home consumption. In addition, please report every fish that is released or lost to a predator. Stock assessment scientists need to see the whole picture of your fishing trip in order to make accurate stock assessments allowing fishery managers to set an ACL.

Comparison of the 2013-2014 Fishing year by Island



Division of Aquatic Resources
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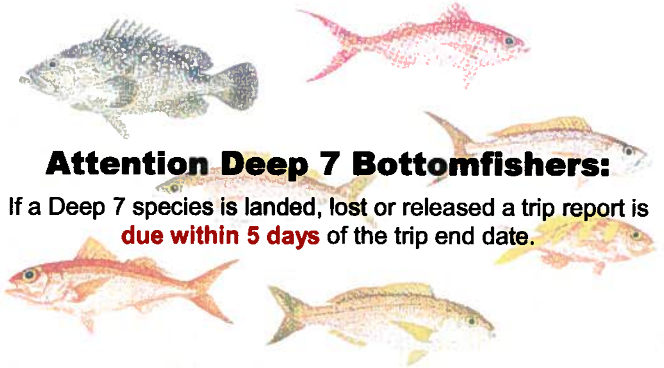
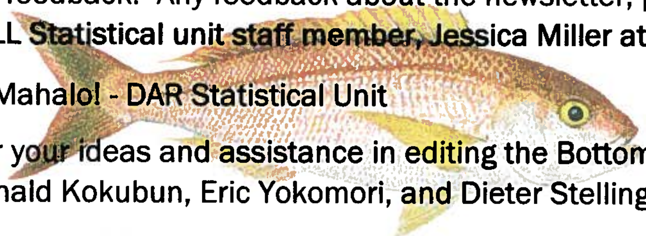


Attention Deep 7 Bottomfishers:

If a Deep 7 species is landed, lost or released a trip report is **due within 5 days** of the trip end date.

- Bottomfish trip reports are due within 5 days of trip end.
- Late trip reports are subject to a fine.
- Reports can be entered online: dlnr.ehawaii.gov/cmls or using a paper form, please call (808) 587-0109 for a trip report book or visit your local DAR office.

Division of Aquatic Resources
Hawaii Department of Land and Natural Resources



Dear Commercial Bottomfish fishers and dealers-Please let us know what kind of information is useful to you. We welcome your feedback! Any feedback about the newsletter, positive or negative, is greatly appreciated! PLEASE CALL Statistical unit staff member, Jessica Miller at (808) 587-0594 or e-mail

bottomfish@hawaii.gov. Mahalo! - DAR Statistical Unit

Thank you to everyone for your ideas and assistance in editing the Bottomfish Newsletter Volume 17!
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Editor: Jessica Miller

Authors: Dana K. Sackett[1], Virginia N. Moriwake[1], Christopher D. Kelley[1], Jeffrey C. Drazen[1]

Title: Spilling over deepwater boundaries; evidence of spillover from two deepwater protected areas in Hawaii

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Abstract. The net export of adults (spillover) is an important though contentious benefit of marine protected areas (MPAs). Controversy over spillover often exists because though intuitive arguments and theoretical modeling are prevalent, it is difficult to discern empirically. In addition, of those studies that have provided empirical evidence of spillover, few have demonstrated tangible benefits to fishery yield and nearly all of those are from shallow reef ecosystems. Here we examined two deepwater MPAs called bottomfish restricted fishing areas (BRFAs) in the Main Hawaiian Islands, established to benefit a complex of species collectively called the Deep 7. To study these fish we used a non-extractive baited camera system and fishery landings data. These data provided evidence of spillover for at least three of these species and benefits to fishery yield. Relative abundance, fish size, and species richness declined with distance from BRFAs, signifying that the recovering Deep 7 community inside these reserves had begun to spillover the boundary of the BRFAs and that BRFAs were a source of more and larger fish to fished areas. Further, a temporal analysis of these spatial trends indicated that declining fish length, MaxN, and species richness did not always exist but only developed in the most recent years. In addition, changes in fish size over time suggested both density independent and dependent processes contributed to spillover. Displaced fishing effort also likely caused initial declines in *Etelis coruscans* size and catch data that increased in later years. Identifying the ability and time span for a MPA to begin to benefit a fishery is crucial to resolving debates regarding the use of MPAs in fisheries management. Further, this is the first study to provide empirical evidence of spillover from deepwater protected areas and one of the first to temporally examine spatial trends around MPAs.

Key words: spillover; marine protected area; bottomfish; deepwater; density dependent; density independent; displaced fishing effort.

Introduction

The use of marine protected areas (MPAs) as spatial tools to maintain or enhance local fisheries has been extensively debated over the last several decades (Roberts & Polunin 1991; Bohnsack 1996; Agardy et al. 2003; Halpern 2003; Hilborn et al. 2004; Kaiser 2005; White and Kendall 2007; More and Sale 2011). Much of this debate focuses on the theory that exploited populations inside an MPA will grow and ultimately export adults (spillover) and recruits (larvae and/or juveniles) into surrounding fished areas, benefiting local fisheries (McClanahan and Mangi 2000; Sales et al. 2005; Kerwath et al. 2013). Controversy over spillover often exists because though intuitive arguments and theoretical modeling are prevalent, it is difficult to discern empirically (Roberts and Polunin 1991; Kellner et al. 2007; White and Kendall 2007; Halpern 2014). One of the reasons spillover is difficult to distinguish is that there are numerous other factors that can influence fish populations in open ocean environments and distinguishing population changes related to protection alone can be difficult. Further, it often takes several years to decades, depending on the life history of the target species, for spillover to begin and be statistically recognized in monitoring data; a much longer time span than many monitoring programs (Molloy et al. 2009; Babcock et al. 2010; Russ and Alcala 2010; Russ and Alcala 2011). In addition, protected areas require a form of data collection that is non-extractive, often relying on diver surveys and baited camera systems to measure relative abundance, which is often highly variable and statistically difficult to analyze (Pennington 1983; Stefánsson 1996; Martinez et al. 2011; Smith et al. 2012).

Despite these limitations, studies have shown how relative abundance, size, and biodiversity of exploited species have increased just outside of well designed and managed MPAs (Russ and Alcala 1996; McClanahan and Mangi 2000; Russ et al. 2004). Indeed, a recent study by Russ and Alcala (2011) demonstrated that after 25 years of protection the high level of biodiversity and community complexity inside the MPA had extended beyond the boundaries into adjacent fished areas due to the spillover of multiple species. While some have provided evidence of spillover few have demonstrated tangible benefits to fishery yield (Abesamis et al. 2006; Kerwath et al. 2013) and of those that have, nearly all are from shallow reef ecosystems (McClanahan and Mangi 2000; Roberts et al. 2001; Russ et al. 2004; Abesamis and Russ 2005). Thus, controversy still remains on whether spillover from MPAs is a reliable benefit to fisheries management, particularly in environments other than shallow reef ecosystems.

Among the Main Hawaiian Islands (MHI) there is a system of deepwater MPAs called bottomfish restricted fishing areas (hereafter referred to as BRFAs) that were initially established in 1998 and revised in 2007 by the State of Hawai‘i, Department of Land and Natural Resources. These BRFAs were created in response to declining catch rates and spawning potential ratios of an exploited group of fish called the Deep 7 (Ralston et al. 1986; Haight et al. 1993; Kelley et al. 2006; Parke 2007). The Deep 7 refers to a complex of deepwater bottomfish species that are relatively site attached and includes six snappers (Lutjanidae) in the subfamily Etelinae: deepwater red snapper *Etelis carbunculus,* deepwater longtail red snapper *E. coruscans,* crimson jobfish *Pristipomoides filamentosus,* lavender jobfish *P. sieboldii,* oblique-banded snapper *P. zonatus,* rusty jobfish *Aphareus rutilans*, and one grouper (Serranidae): Hawaiian grouper *Hyporthodus quernus* (Haight et al. 1993; Kelley et al. 2006). Of these the most economically important are *E. carbunculus, E. coruscans,* and *P. filamentosus*. These BRFAs restrict bottomfish harvest and protect the deepwater environment. Surface waters, however, are open to fishing for pelagic species such as tuna.

Monitoring of the BRFAs began after their revision in 2007, following detailed mapping of the seafloor to include more essential fish habitat (EFH; Rosenberg et al. 2000; Moffit 2006; Parke 2007; Kelley and Moriwake 2012; Sackett et al. 2014). Our previous work analyzing the first four years of monitoring data (2007-2011) from a subset of these BRFAs found that mean fish length, and in some cases abundance, increased for one or more of the most economically important Deep 7 species inside, while outside fish sizes and relative abundance declined or stayed the same (Sackett et al. 2014).

Monitoring continued in two of the BRFAs following that study, both of which had their boundaries expanded in 2007 (Makapu‘u from 10.2 km^2 to 220.2 km^2; Penguin Bank from 54.7 km^2 to 268.6 km^2). The six years of data (2007-2013) obtained from these two BRFAs were the focus of this study. Here we provide evidence that spillover began to occur in the fifth and sixth year of monitoring using fish size, relative abundance and species richness data collected from 2007-2013 with a baited camera system and fishery landings data collected by the Hawaii Division of Aquatic Resources (DAR) from 2007-2014.

Methods

Data collection and synthesis

Data collected in this study are part of an ongoing monitoring program examining deepwater bottomfish populations in the MHI using a fishery-independent approach, baited stereo-video camera system (BotCam). Consequently, detailed data collection methods, video analysis methods, and sampling strategies were previously described in Moore et al. (2011), Misa et al (2013) and Sackett et al. (2014). Briefly, the camera system used two paired ultra low-light video cameras that enabled accurate fish identification and length measurements under ambient light conditions to a depth of 310m (Shortis et al. 2008; Watson et al. 2010). In addition, $0.04km^2$ (200m x 200m) sample grid cells created in and around the two BRFAs (Penguin Bank and Makapu‘u; Figure 1) were randomly chosen, though stratified by protection (fished/unprotected or unfished/protected) and habitat to ensure comparisons inside and outside the protected areas were equivalent. The camera system was left to record 3m above the sea floor for approximately 40mins at each sample site. These specifications allowed our system to target the Deep 7, which are closely associated with the benthos and/or school in the water column above it (Sackett et al. 2014). Habitat designations were classified by slope (high ≥ 20 degrees, low < 20 degrees) and substrate type (hard = consolidated hard rocky substrate, soft = unconsolidated soft substrate) for every $0.04km^2$ area based on multibeam bathymetry and backscatter data (Misa et al. 2013). Relative abundance data for each sample was recorded using a metric call MaxN or the maximum number of fish observed in a single frame of the video (Parrish 1989; Priede et al. 1994; Moore et al. 2011). Species not seen were given a value of zero. In addition, because each species does not occupy the entire depth range sampled (90-310m), data collected outside of a species preferred depth range were excluded from the MaxN database (Misa et al. 2013; Sackett et al. 2014). Fork lengths for a single species were also measured only once during a video; when the most measurable fish were visible on both cameras. Species richness was measured by a count of the number of Deep 7 species that were present in a single video.

Statistical approach

We focused much of our analyses on the three most economically important and abundant of the complex of deep-dwelling bottomfish species called the Deep 7 (*E. carbunculus*, *E. coruscans*, and *P. filamentosus*; Haight et al. 1993; Kelley et al. 2006; Sackett et al. 2014). Where data were sufficient, results for other Deep 7 species are briefly presented as well (Table

1). To determine whether spillover was occurring for these species around Penguin Bank and Makapuʻu BRFAs, we first examined whether fish length, relative abundance (MaxN), and species richness declined with distance away from each BRFA using simple linear regression (length data), a generalized linear model with a negative binomial distribution (MaxN data; Martinez et al. 2011; Smith et al. 2012) and a generalized linear model with a Poisson distribution (species richness data). A decline in these metrics with distance from the BRFAs would indicate that the protected areas were a source of more and larger fish to the fished areas (i.e. spillover; McClanahan and Mangi 2000; Russ et al. 2004; Forcada et al. 2009; Russ and Alcala 2011). Distances were measured as the shortest straight line distance within the 100-400m depth range as this is the depth at which these species reside (Kelley et al. 2006; Parke 2007; Kelley and Moriwake 2012; Weng 2013). To reduce error we measured the shortest distance from the BRFA to the sample site within the 100-400m depth range three times in ESRI ArcMap 10 (ESRI Inc. Redlands, CA, USA) and took a mean from those measurements. In addition, those sites greater than 50km away from Penguin Bank BRFA were excluded from these analyses because of their close proximity to Makapu´u BRFA (~14-20km; see Figure 1). These analyses also used only those data collected from hard bottom habitat types to ensure results were not an artifact of differences in habitat type among sample sites and because hard habitat types are often preferred by Deep 7 species (Kelley et al. 2006; Parke 2007; Misa 2013; Sackett et al. 2014). Those sites inside the BRFA were given a distance of zero meters from the BRFA. We also examined the first two, second two and last two sampling years separately (e.g. three separate databases one with sampling years 1 and 2, one with sampling years 3 and 4, and one with sampling years 5 and 6) to determine whether spatial trends changed over time. We also examined differences in fish length over time, again using only data collected in hard habitat types, using non-linear (degree 2 polynomial fit) regression both inside and outside of protected areas. A nonlinear approach was chosen to ensure asymptotes and changes in data direction over time would be accounted for in our analyses. These analyses were presented using raw values instead of means to determine if groups of small fish were evident in later sampling years, presumably due to recruitment. Further, as these tests were parametric, MaxN and species richness data were not appropriate for these tests.

We also compared linear model results from Sackett et al. (2014) to model results in this study, which included the fifth and sixth years of monitoring data. The same models and techniques were used in these analyses as were used for Sackett et al. (2014) for consistency in comparisons. Briefly, standard least squares multiple regression models were used to analyze length data while generalized linear mixed models using a negative binomial distribution were used to analyze MaxN and species richness data (Martinez et al. 2011; Smith et al. 2012). Models accounted for differences in habitat type and depth. Further, Akaike's Information Criterion (AIC_c; Burnham and Anderson 1998) was used to rank candidate models to determine the model that best explained the variation in fish length, MaxN and species richness data.

Fisheries data collected by DAR from 2007 to 2014 were examined to determine whether spillover effects were evident in catch data. These data included effort measured as the number of trips (representing a day of fishing) anglers took in each fishing area (Figure 1) around the MHI. The total annual weight (lbs) and number of fish caught for each Deep 7 species from 2007 to 2014 were also recorded in those same fishing areas. To determine whether spillover from Penguin Bank and Makapuʻu BRFAs were having an impact on catch data, we summed data collected from those fishing areas that intersected each BRFA; thus creating a database of catch data that surrounded each BRFA. Catch data (total weight and number of fish caught) were then divided by the total number of trips in these areas from each year to standardize catch by effort and create two catch per unit effort (CPUE) metrics (lbs of fish caught per trip and number of fish caught per trip). The CPUE metrics were $\log_{10}$ transformed to meet assumptions of normality and equal variance then analyzed over time using linear and non-linear (degree 2 polynomial fit) regression to determine which fit the data best (based on the regression coefficient and a lack of fit test $P>0.05$). These analyses were run for pooled data from both BRFAs, because BRFAs shared a fishing area (see Figure 1), and for each BRFA separately.

Results

Distance from BRFA

Our spatial analyses of sampling years one and two, years three and four, and the most recent years five and six demonstrated that declines in fish length, MaxN and species richness with distance from the BRFAs only developed in the most recent years (Table 2). For instance, around Penguin Bank BRFA *E. carbunculus* ($P=0.06$, $r^2=0.08$), *E. coruscans* ($P<0.05$, $r^2=0.04$), and *P. filamentosus* ($P=0.07$, $r^2=0.04$) lengths increased or remained the same with distance from the BRFA in the first two years of monitoring. The same was true for Makapuʻu BRFA (*E. carbunculus*: $P=0.03$, $r^2=0.45$; *E. coruscans*: $P=0.85$; *P. filamentosus*: $P=0.07$, $r^2=0.04$). In sampling years three and four, *E. carbunculus* still showed an increase in length with distance from Makapuʻu BRFA ($P=0.04$, $r^2=0.37$) but had begun to show a decline with distance from Penguin Bank BRFA ($P<0.01$, $r^2=-0.20$). Similarly, *E. coruscans* length also began to decline with distance from Penguin Bank BRFA in sampling years three and four ($P<0.01$, $r^2=-0.24$). *P. filamentosus* demonstrated either no change (Penguin Bank, $P=0.41$) or a slight increase (Makapuʻu, $P=0.06$, $r^2=0.03$) in length with distance from the BRFAs in sampling years three and four. MaxN and species richness data showed the same general trends. In Makapuʻu and

Penguin Bank BRFAs, relative abundance of *E. carbunculus* and *E. coruscans* collected in the first two years of monitoring did not vary with distance from these BRFAs (Makapu‘u: *E. carbunculus* P=0.12, *E. coruscans* P=0.20; Penguin Bank: *E. carbunculus* P=0.22, *E. coruscans* P=0.48). In the second two years of monitoring *E. carbunculus* began to demonstrate a marginally significant decrease in MaxN with distance from Makapu‘u BRFA (P<0.10) though not yet from Penguin Bank BRFA (P=0.12). *E. coruscans* also began to demonstrate a decline in MaxN with distance from Penguin Bank BRFA in monitoring years three and four (P<0.01). In contrast, the relative abundance of *P. filamentosus* showed a significant or marginally significant decline with distance from both Penguin Bank and Makapu‘u BRFAs since sampling began (Makapu‘u$_{yr1,2}$: P<0.01; Makapu‘u$_{yr3,4}$: P=0.01; Penguin Bank$_{yr1,2}$: P=0.08; Penguin Bank$_{yr3,4}$: P=0.03; Table 2). Although the sample sizes for other Deep 7 species were limited (Table 1), other species did show similar trends. For example, *P. sieboldii* showed the same trend as *P. filamentosus* in Penguin Bank BRFA ($P_{1,2}$<0.01; $P_{3,4}$<0.01), while *H. quernus* showed no significant spatial trend in the first four years of sampling ($P_{1,2}$=0.38, $P_{3,4}$=0.48). *A. rutilans* also showed no spatial trends with distance from Penguin Bank BRFA in the first two years of monitoring, while in the second two years showed a significant (P<0.01) decline in MaxN with distance from this BRFA. Species richness of the Deep 7, or the number of Deep 7 species seen in a single sample, did not show any spatial trends over the first four years of monitoring in Makapu‘u BRFA (Makapu‘u$_{yr1,2}$: P=0.98; Makapu‘u$_{yr3,4}$: P=0.43). Similarly in Penguin Bank BRFA there were no spatial trends in the first two years of monitoring (Penguin Bank$_{yr1,2}$: P=0.56) while in the second two years there was a marginally significant decline in species richness with distance from the BRFA (Penguin Bank$_{yr3,4}$: P=0.05).

In the last two years of sampling, however, a decline in MaxN, fish length and species richness with distance from both BRFAs were evident for nearly every species with sufficient data to test (Table 2; Figure 2). All three of our target species (*E. carbunculus*, *E. coruscans* and *P. filamentosus*) showed significant ($P<0.05$) or marginally significant ($0.10<P<0.05$) declines in relative abundance (MaxN) with distance from both Penguin Bank and Makapu‘u BRFAs in the last two years of sampling (Figure 2c, 2d). *P. filamentosus* also showed significant ($P<0.01$) declines in fish length with distance from both the Penguin Bank (r^2=-0.18) and Makapu‘u (r^2=-0.09) BRFAs while, *E. coruscans* lengths significantly ($P<0.01$) declined with distance from the Penguin Bank BRFA (r^2=-0.08) in sampling years five and six (Figure 2a, 2b). *H. quernus* (P=0.04), *A. rutilans* (P=0.09) and *P. sieboldii* ($P<0.01$) also showed significant or marginally significant declines in MaxN data and significant declines in *P. sieboldii* length data ($P<0.01$; r^2=-0.19) with distance from the Penguin Bank BRFA in the fifth and sixth years of sampling. Species richness also declined significantly with distance from Penguin Bank ($P<0.01$) and Makapu‘u (P=0.03) BRFAs in the most recent years (Figure 2e, 2f). Further, the only area sampled where six of the Deep 7 were seen in a single sample was inside Penguin Bank BRFA in the fifth and sixth years of sampling (Figure 2e).

Changes over time

Non-linear regression analyses of *E. carbunculus*, *E. coruscans* and *P. filamentosus* showed an increase in fish length over time that began to reach an asymptote in the most recent years inside Penguin Bank BRFA ($P<0.01$; Figure 3). Outside of Penguin Bank BRFA *P. filamentosus* lengths increased while outside *E. coruscans* lengths initially declined over time then began increasing at approximately the same time the asymptote in fish length was reached inside the BRFA ($P<0.01$; Figure 3b, 3c). *E. carbunculus* showed a similar trend to *E. coruscans* outside Penguin Bank BRFA over time, however this relationship was insignificant (P=0.18; Figure 3a). *P. filamentosus* lengths also increased inside Makapu‘u BRFA over time (P<0.01) and had a distinct group of smaller immature fish (<450mm; Ralston and Miyamoto 1983) in sampling years four and six. Outside of Makapu‘u BRFA a similar distinct group of small immature *P. filamentosus* were recorded in the sixth year of sampling that were not seen in previous years and potentially represented recruitment. These smaller fish caused a decline in length trends over time outside of Makapu‘u BRFA in later sampling years (Figure 3d). *P. sieboldii* also showed a slight but significant increase inside (P=0.01, r^2=0.03) and a drastic increase outside (P<0.01, r^2=0.10) Penguin Banks BRFA over time both reaching an asymptote in the most recent years around a mean of approximately 350mm fork length.

Model results, which took habitat and depth into account but only expressed linear relationships, reflected non-linear fish length regression results in most cases (Table 3; Figure 3). Modeled *E. carbunculus* lengths continued to increase as expected inside Penguin Bank BRFA (P<0.01) over time, while outside lengths showed a slight but significant decline over time (P<0.01; Figure 3; Sackett et al. 2014). Predicted *E. coruscans* lengths increased inside Penguin Bank BRFA (P<0.01), while outside there was a significant linear decline. However, predicted values for each year mirrored non-linear results with increasing mean predicted values over the last three years of monitoring. Modeled *P. filamentosus* lengths also increased inside both Penguin Bank and Makapu‘u BRFAs over time (P<0.05; Figure 3). Outside Penguin Bank BRFA modeled *P. filamentosus* lengths also increased over time, so much so that the best model ranked by AIC_c no longer included year*protection, protection and habitat (Sackett et al. 2014) but just the factor year, demonstrating that length data were best described by a similar increase in fish length over time both inside and outside the BRFA (Table 3). Model results for *P.*

filamentosus inside Makapuʻu BRFA showed a slight but significant increase in fish length while outside linear results suggested that no changes were occurring over time. Model results for *E. carbunculus*, *E. coruscans*, and *P. sieboldii* also showed that there were significantly ($P<0.05$) larger fish inside Penguin Bank BRFA (means for *E. carbunculus*=448.0mm, *E. coruscans* =667.1mm, *P. sieboldii* =346.3mm) compared to outside (means for *E. carbunculus*=414.5mm, *E. coruscans* =639.1mm, *P. sieboldii* =313.8mm); while at Makapuʻu where data were only sufficient for *P. filamentosus*, model results also indicted that there were larger fish inside (mean=462.4mm) than outside (418.4mm) the BRFA, a result not seen in our previous analysis (Sackett et al. 2014). It is also important to note that in our previous study there were significantly larger *P. filamentosus* inside compared to outside Penguin Bank BRFA that were not seen here, suggesting that the addition of *P. filamentosus* size data from monitoring years five and six increased the overall mean of fish size outside the BRFA.

The best ranked MaxN model (year*protection, protection, habitat) for *P. filamentosus* in Makapuʻu BRFA showed no significant difference for the factor protection or the interaction between year and protection (Table 3). Results from Sackett et al. (2014) showed a significantly different trend inside and outside Makapuʻu BRFA over time. In addition, there were no significant models that explained MaxN data from Penguin Bank BRFA. The best significant model based on AIC_c for species richness data only included the factor protection (comparing pooled data collected inside to outside). These model results demonstrated that species richness was significantly ($P=0.01$) higher inside Penguin Bank BRFA (mean = 1.42 ± 0.016 SE) compared to outside (mean = 1.14 ± 0.012 SE), a result not previously seen in this BRFA (Sackett et al. 2014).

Fishery data

The overall number (P=0.01; r^2=0.83) and weight (P=0.03; r^2=0.74) of *E. coruscans* caught per fishing trip reflected data collected with BotCam outside Penguin Bank (compare Fig 3b to Fig 4a), demonstrating an initial decline in CPUE after the BRFA was enlarged in 2007 followed by an increase in CPUE in the last several years (P<0.05). The number of *E. coruscans* caught per fishing trip also increased linearly outside Makapuʻu BRFA; however, this result was only marginally significant (P=0.09). The number (P=0.02; r^2=0.34) and weight (P=0.01; r^2=0.39) of *P. filamentosus* caught per fishing trip around both Penguin Bank and Makapuʻu BRFAs together also increased linearly (results were pooled because BRFAs shared a fishing grid, see Figure1; Figure 4b). Separately, Makapuʻu showed a significant linear increase in CPUE data over time (number per trip: P=0.03, r^2=0.59; lbs per trip: P=0.02, r^2=0.65); however, Penguin Bank did not (P>0.05). For *P. sieboldii*, number (P<0.01; r^2=0.62) and weight (P=0.03; r^2=0.41) CPUE around both BRFAs together followed a polynomial increase over time. Results for the number of *P. sieboldii* caught per trip over time were significant (P<0.05) when analyzed separately for Penguin Bank and Makapuʻu BRFAs as well. Weight CPUE for *P. sieboldii* significantly increased along a polynomial regression around Penguin Bank (P=0.02; r^2=0.78) and Makapuʻu BRFAs separately as well, however this result was only marginally significant around Makapuʻu BRFA (P=0.07; r^2=0.66). Lastly, *P. zonatus* CPUE measures initially declined and then leveled off in recent years in a polynomial regression over time when data around both BRFAs were pooled (P_{noCPUE}=0.02, r^2=0.46; P_{wtCPUE}=0.02, r^2=0.43).

Discussion

The primary evidence for spillover often provided by others is a decline in relative abundance, fish size or biodiversity with distance from a MPA at a single point in time (McClanahan and Mangi 2000; Russ et al. 2004; Abesamis et al. 2006; Harmelin-Vivien et al.

2008; Forcada et al. 2009; Russ and Alcala 2011; Stamoulis and Friedlander 2013). These results suggest that the protected area is a source of more and larger fish to fished areas and that the increasing community complexity inside the protected areas is spilling over the boundary of the MPA (Kellner et al. 2007). For instance, McClanahan and Mangi (2000) measured biodiversity, fish sizes, and CPUE data with distance from the Mombasa Marine Park in Kenya to identify spillover and noted that all three declined with distance from the reserve within 5km. In addition, Russ and Alcala (2011) noted that species richness and community complexity declined with distance from a reserve protected in the Philippines for 25 years. These results were a direct consequence of the spillover of multiple species from this MPA. Here we temporally examined the spatial relationship between relative abundance, fish size and species richness with distance from two deepwater MPAs (Makapuʻu and Penguin Bank BRFAs) that were first established in 1998 and later expanded in 2007, and demonstrated that declines with distance only developed in the most recent years. Although, the fit of many of the significant spatial relationships in this study were relatively low, signifying that there were likely numerous other factors in the open ocean that contributed to the spatial distribution of these species (e.g. prey movements, currents). Also, as we were only able to broadly classify habitat it is likely that more specific habitat classification would better explain some of this variability (Misa et al. 2013; Sackett et al. 2014). Regardless, the relationships described here were consistent among species, independent factors (species richness, relative abundance, fish length) and areas and did significantly explain, at least in part, the spatial distribution of these species. These results signify that the recovering Deep 7 community inside these reserves has begun to spillover the boundary of the BRFAs and that BRFAs are a source of more and larger fish to fished areas. We also demonstrated that spillover from these BRFAs, while localized, occurred over a relatively large spatial scale compared to other studies (a scale of kilometers; e.g. McClanahan and Mangi 2000; Harmelin-Vivien et al. 2008). This is also one of the first studies to temporally examine spatial trends in species richness, relative abundance and fish length in relation to protected areas.

Two of the most economically important and abundant of the Deep 7 bottomfish complex (*E. coruscans* and *P. filamentosus*; Haight et al. 1993; Kelley et al. 2006) showed the strongest evidence of spillover from Penguin Bank BRFA. Fishing is size selective, often targeting larger individuals leading to decreases in the average size of fishes within a fishery (Bianchi et al. 2000, Berkeley et al. 2004, Tetreault and Ambrose 2007, DeMartini et al. 2008). Consequently, the significantly larger *E. coruscans* and *P. filamentosus* inside the BRFA, the temporal increases in fish lengths outside the BRFA, and that more and larger fish were seen closer to the BRFA, strongly suggested that more and larger fish from inside this protected area were being exported to fished areas. However, the spatial distribution of *P. filamentosus* relative abundance from both BRFAs did not change over time suggesting that this distribution may not be a result of spillover. Although, the spatial distribution in *P. filamentosus* sizes suggests that only recently larger fish were spatially distributed closer to the protected areas. Thus, larger fish likely spilled over the boundary of the BRFA, displacing the naturally more abundant fish there. Further, although suggested to be not as economically important (Haight et al. 1993; Kelley et al. 2006), *P. sieboldii* showed the same evidence of spillover, and a substantial increase in CPUE in recent years. Further, *E. carbunculus*, *H. quernus* and *A. rutilans*, also showed evidence of spillover beginning to occur from Penguin Bank BRFA in the most recent years.

The size selective nature of fishing pressure was demonstrated for *E. coruscans* outside of Penguin Bank BRFA in the first few years after the BRFA was revised in 2007 by declines in fish length. These results suggested that displaced fishing pressure caused initial declines in mean *E. coruscans* lengths (Halpern et al. 2004). However, it appears that the net export of large adults in later years was enough to sustain fishing pressure and cause an increase in fish size in recent years. Fishery data further supported our interpretation, demonstrating a parallel trend in CPUE data; an initial decline followed by an increase over time in the number and weight of *E. coruscans* caught per trip. This BRFA was, therefore, able to offset the initial decline in CPUE just 6 years after expansion . Identifying the ability and time span for a MPA to benefit a fishery is crucial to resolving debates regarding the use of MPAs in fisheries management (Hilborn et al. 2004; Hilborn et al. 2010).

Spillover is often suggested to be a result of both density independent and dependent processes. Density independent spillover occurs when fish movements unrelated to fish density inside the reserve causes fish to leave a MPA. A few types of density independent movements that could cause spillover include adult migration, ontogenetic migrations, or when the fish’s lifetime home range extends beyond the boundaries of the reserve (Gruss et al. 2011). Although few, studies on the mobility of deepwater bottomfish in Hawaii have suggested that many move beyond the boundaries of protected areas over their lifetime, though to varying degrees (Williamson 2005; Weng 2013). For instance, *P. filamentosus* have been seen to move between Penguin Bank and Makapuʻu BRFAs and tracking studies have indicated that they move from inside to outside of the Kahoʻolawe Island Reserve, a no-take protected area in Hawaii (Williamson 2005; PIFG 2013). *E. coruscans* and *E. carbunculus* have also been found to move from inside to outside of Niʻihau BRFA in Hawaii (Weng 2013) where, *E. coruscans* moved more frequently and over greater distances than *E. carbunculus*. Thus, the lifetime home range of these species likely extends beyond the boundary of the BRFAs, causing some degree of density independent spillover into fished areas (Gruss et al. 2011). Many studies have examined fish home range and movement on the ability of MPAs to cause spillover and found that species with intermediate levels of mobility were predicted to provide the greatest spillover benefits to nearby fisheries (Holland et al. 1996, Kramer & Chapman 1999). Our results support these finding as *P. filamentosus* and *E. coruscans* both have intermediate levels of movement (e.g. both spend enough time inside the reserve for protection to have an effect but also move outside of the boundaries of the BRFAs providing benefits to the fishery). Fishery data supported these conclusions of spillover for *P. filamentosus* and *E. coruscans* with CPUE increasing overtime around Makapuʻu and Penguin Bank BRFAs. The higher relative abundance of *E. carbunculus* closer to Penguin Bank BRFA suggested spillover as well. Changes in *E. carbunculus* size over time showed increases inside the BRFA that had begun to reach an asymptote, while outside and similar to *E. coruscans*, there was an initial decline that changed direction and began to increase in recent years. However, this relationship was not significant. As such, the cross boundary movement of *E. carbunculus*, which is lower than other species (Weng 2013), may not export enough fish to show a significant increase in the mean size of fish in fished areas (Tetreault and Ambrose 2007, DeMartini et al. 2008).

The asymptote in *E. carbunculus*, *E. coruscans*, *P. filamentosus*, and *P. sieboldii* lengths over time inside the reserve in the fifth and sixth years of monitoring may also indicate some degree of density dependent spillover. Density dependent spillover occurs when resources such as food or space are in short supply inside the reserve, causing fish to move outside of the protected area in search of these resources (Sanchez Lizaso et al. 2000). With the increase in larger individuals inside Penguin Bank BRFA, competition between larger fish could cause a net emigration from the BRFA, resulting in the increase in fish sizes seen outside the BRFA over time and that more and larger fish outside the BRFA were found closer to the boundary of the reserve. Also, because Deep 7 species often prefer specific depth ranges (Misa et al. 2013; Sackett et al. 2014) and are relatively site attached to high relief structures such as pinnacles (many of which the BRFA system were designed to protect; Ralston et al. 1986; Haight et al. 1993; Kelley et al. 2006; Parke 2007; Merritt et al. 2011), it is easy to conceive that space on these preferred habitat structures may be limited, particularly for a higher number of larger individuals inside the BRFA.

Spillover was also seen from Makapuʻu BRFA, where *E. carbunculus*, *E. coruscans* and *P. filamentosus* relative abundance and *P. filamentosus* sizes declined with distance from the boundary of the reserve in the last two years of sampling (McClanahan and Mangi 2000; Kellner et al. 2007). In addition, Deep 7 species richness declined with distance from Makapuʻu BRFA only in the last two year of sampling. Similar to Sackett et al. (2014), in Makapuʻu BRFA sample sizes for species other than *P. filamentosus* were limited and likely influenced our ability to detect protection effects or the lack thereof for other Deep 7 species. Additionally, changes in *P. filamentosus* size over time differed from Penguin Bank BRFA showing that while fish size increased inside and remained the same outside in model results, a distinct group of small immature (<450mm; Ralston and Miyamoto 198) *P. filamentosus* in the fourth and sixth year of monitoring reduced the rate of increase in mean length over time inside the reserve. This group of small *P. filamentosus* were estimated using age at size curves to be less than two years old (Andrews et al 2012), demonstrating recruitment to the area and gear (Ralston and Miyamoto 1983). Outside the BRFA the group of small individuals caused a decline in the overall length trend overtime. Although difficult to distinguish empirically, enhanced recruitment is one of the most important and contentious benefits to using MPAs as fishery management tools (Roberts and Polunin 1991; Bohnsack 2011). Evidence of enhanced recruitment from MPAs are often based on theoretical concepts, models and increases in adult fish sizes (Sackett et al. 2014; Vaz et al. 2014). Here we provide some empirical evidence of recruitment in and around Makapuʻu BRFA and demonstrate the influence it has on mean fish size over time; a result often used to prove the effectiveness of protection. Thus, empirically demonstrating that recruitment can cause a decline in mean fish sizes over time is a necessity in understanding, interpreting and managing MPAs.

Model results were generally consistent with our previous work, showing similar trends in fish length overtime (Sackett et al. 2014). However, comparisons between fish length model and nonlinear results indicated that linear models often miss when population trends change direction over time at this temporal scale. Relative abundance results were less consistent. For instance, in our previous work *E. coruscans* relative abundance increased inside Penguin Bank BRFA with no changes occurring over time outside, while here there were no significant differences in *E. coruscans* relative abundance over time inside compared to outside the BRFA. Additionally, a similar divergent trend inside versus outside Makapuʻu BRFA was seen for *P. filamentosus* relative abundance (Sackett et al. 2014). In this study model results differed, showing no significant differences in relative abundance over time. These results suggest the addition of data from years five and six changed linear trends in relative abundance, so that they were no longer significantly different. Another possible explanation may be that MaxN data are too variable to be reliable in complex models at this temporal scale (Pennington 1983; Stefánsson 1996; Martinez et al. 2011; Smith et al. 2012). The MaxN method is a conservative estimate of the number of fish in a given area (Parrish 1989; Priede et al. 1994). Recent research has indicated that MaxN increasingly underestimates abundance at higher levels of true abundance (Conn 2011; Schobernd et al. 2014). As such, when fish abundance inside a reserve increases, the level of recovery would taper off as detection using MaxN is saturated. The majority of our MaxN data were less than 10, well below the saturation level seen by Schobernd et al. (2014). However, it is those larger numbers and increases in MaxN that demonstrate recovery over time and this limitation in MaxN data could have affected our results. Further, the level and rate of saturation for MaxN data compared to true abundance is likely to vary as a consequence of species behavior, information lacking for Hawaiian deepwater bottomfishes.

Overfishing and the worldwide depletion of predatory fishes have caused global declines in marine biodiversity, population abundance and size structure, altered genetic diversity, and caused landscape level impacts on entire ecosystems (Friedlander and DeMartini 2002; Harmelin-Vivien et al. 2008; Estes et al. 2011; Mora and Sale 2011). Protecting areas from fishing has been one proposed solution to reverse declining marine biodiversity and other negative consequences of overharvest (Lester et al. 2009; Molloy et al. 2009; Russ and Alcala 2011). Besides conservation, well designed and managed MPAs can export adults (spillover) into fished areas as well as stabilize spawning stock and subsidize recruitment, ultimately improving fishery yield over time (McClanahan and Mangi 2000; Russ et al. 2004; Abesamis et al. 2006; Kerwath et al. 2013; Halpern 2010). Here we provide evidence that deepwater MPAs can and have begun to spread increased species richness seen inside to areas outside these reserves and have begun to benefit the local Deep 7 fishery through spillover of at least three deepwater snappers in the MHI.

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Tables

Table 1. Sample sizes for (a) fish length data, and (b and c) MaxN data collected within the preferred depth range of each species in hard habitat types for the first two, second two and last two years of monitoring two protected areas (Penguin Bank and Makapu‘u). Data are from 2007-2013 (years 1-6) from inside and outside each BRFA and for each of the Deep 7 species. Genera in the table are *Etelis, Pristipomoides, Hyporthodus,* and *Aphareus*.

a. Length	Years 1-2 Data				Years 3-4 Data				Years 5-6 Data			
	Penguin Bank		Makapu'u		Penguin Bank		Makapu'u		Penguin Bank		Makapu'u	
	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
E. carbunculus	30	17	5	6	30	34	4	8	58	20	10	8
E. coruscans	66	29	7	1	29	14	2	0	78	35	15	4
P. filamentosus	73	23	54	39	48	37	93	10	99	51	63	29
P. sieboldii	55	23	5	0	99	68	0	0	113	95	3	0
P. zonatus	6	1	0	1	5	9	0	2	11	4	1	2
H. quernus	1	1	0	4	3	7	0	0	10	1	0	0
A. rutilans	12	3	0	1	7	0	0	2	11	6	0	0

b. MaxN	Years 1-2 Data				Years 3-4 Data				Years 5-6 Data			
Makapu'u	In		Out		In		Out		In		Out	
Species	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence
E. carbunculus	15	0.20	17	0.18	3	0.67	8	0.13	14	0.29	18	0.17
E. coruscans	15	0.27	17	0.12	3	0.67	8	0.00	14	0.43	18	0.17
P. filamentosus	21	0.62	25	0.60	15	0.73	12	0.50	18	0.72	22	0.50
P. sieboldii	10	0.10	20	0.05	5	0.00	11	0.00	9	0.00	18	0.00
P. zonatus	21	0.00	24	0.17	11	0.09	15	0.27	13	0.08	26	0.12
H. quernus	24	0.00	24	0.21	13	0.00	17	0.00	16	0.00	27	0.04
A. rutilans	27	0.22	29	0.03	16	0.00	17	0.06	23	0.00	28	0.04

c. MaxN	Years 1-2 Data				Years 3-4 Data				Years 5-6 Data			
Penguin Bank	In		Out		In		Out		In		Out	
Species	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence	*N*	Presence
E. carbunculus	22	0.64	29	0.34	17	0.71	26	0.54	39	0.62	34	0.35
E. coruscans	22	0.68	29	0.48	17	0.59	26	0.46	39	0.59	34	0.47
P. filamentosus	15	0.73	9	0.44	22	0.50	13	0.62	39	0.56	42	0.50
P. sieboldii	19	0.37	24	0.21	17	0.59	30	0.53	38	0.45	36	0.25
P. zonatus	23	0.17	26	0.04	24	0.33	35	0.23	54	0.19	47	0.15
H. quernus	19	0.05	22	0.00	25	0.16	24	0.21	43	0.26	47	0.02
A. rutilans	23	0.35	25	0.16	28	0.21	24	0.04	50	0.14	53	0.15

Table 2. Summary of regression relationships between fish length, relative abundance (MaxN) and species richness with distance from two protected areas (Makapu‘u and Penguin Bank) in the Main Hawaiian Islands. Spatial trends were temporally delineated by the first two, second two and last two years of monitoring. Arrows represent the direction of the relationship between the independent variable and distance from the protected area. Slopes of each relationship are in parenthesis. Marginally significant relationships (0.10>*P*>0.05) are shaded grey. The full scientific name for each species is *Etelis carbunculus, E. coruscans, Pristipomoides filamentosus*.

			Sampling Years		
Data	**BRFA**	**Species**	**1 and 2**	**3 and 4**	**5 and 6**
Length	Makapu'u	*E. carbunculus*	**↑ (12.4)**	**↑ (10.9)**	**---**
		E. coruscans	**---**		**---**
		P. filamentosus	↑ (4.3)	↑ (4.2)	**↓ (-11.4)**
	Penguin Bank	*E. carbunculus*	↑ (2.0)	**↓ (-2.3)**	**---**
		E. coruscans	**↑ (1.9)**	**↓ (-5.1)**	**↓ (-4.9)**
		P. filamentosus	↑ (2.0)	**---**	**↓ (-3.0)**
MaxN	Makapu'u	*E. carbunculus*	**---**	↓ (-0.10)	↓ (-0.06)
		E. coruscans	**---**		**↓ (-0.20)**
		P. filamentosus	**↓ (-0.10)**	**↓ (-0.09)**	**↓ (-0.17)**
	Penguin Bank	*E. carbunculus*	**---**	**---**	**↓ (-0.07)**
		E. coruscans	**---**	**↓ (-0.04)**	**↓ (-0.04)**
		P. filamentosus	↓ (-0.05)	**↓ (-0.05)**	**↓ (-0.04)**
Species Richness	Makapu'u	Deep 7	**---**	**---**	**↓ (-0.05)**
	Penguin Bank	Deep 7	**---**	↓ (-0.02)	**↓ (-0.03)**

Table 3. Top ranked and significant models explaining MaxN, fish length, and species richness of the Deep 7 for each BRFA using Akaike's Information Criterion (AIC$_c$; Burnham and Anderson 1998). Model probability weights (W_i) indicated the probability that a model is the best of the set of models tested. The full scientific name for each species is *Etelis carbunculus, E. coruscans, Pristipomoides filamentosus*. $^{\#\#}$ indicates a significant model effect (P<0.05).

BRFA	Dependent Variable	Species	Top Model	P	R^2	N	W_i
Makapu'u	MaxN	*P. filamentosus*	year*protection, protection, substrate$^{\#\#}$	0.00	---	123	0.79
Makapu'u	Length	*P. filamentosus*	year*protection$^{\#\#}$, protection$^{\#\#}$, substrate$^{\#\#}$	0.00	0.41	335	1.00
Penguin Bank	Length	*E. carbunculus*	year*protection$^{\#\#}$, protection$^{\#\#}$, substrate	0.02	0.12	259	1.00
Penguin Bank	Length	*E. coruscans*	year*protection$^{\#\#}$, protection$^{\#\#}$, substrate	0.04	0.05	286	0.98
Penguin Bank	Length	*P. filamentosus*	year$^{\#\#}$	0.00	0.31	494	1.00
Penguin Bank	Species richness	Deep 7	protection$^{\#\#}$	0.01	---	491	0.98

Figures

Figure 1. Two out of a system of 12 deepwater marine protected areas called bottomfish restricted fishing areas (BRFAs) in the Main Hawaiian Islands and sample sites from hard habitat types inside and outside each BRFA. Depth contours from 100 to 400m are delineated on the map. Target species examined in this project inhabit depths within this range. Zones delineated for the bottomfish fishery by Hawaii Division of Aquatic Resources (DAR) are also indicated on the map. Data excluded from our spatial analyses using distance from Penguin Bank BRFA are also indicated on the map.

Figure 2. Relationships for (a and b) fish length, (c and d) relative abundance and (e and f) species richness (all Deep 7 species in a single sample) with distance from each bottomfish restricted fishing area (BRFA; Penguin Bank and Makapuʻu) from the fifth and sixth years of monitoring. A value of zero distance was assigned to data collected inside the BRFAs and are indicated by filled symbols while open symbols represent data collected outside the BRFAs. *P*-values refer to the regression lines indicated in each figure and color coded for each species. Simple linear regression (fish length), and generalized linear models with a negative binomial distribution (MaxN) and a Poisson distribution (species richness) were used for these analyses. In addition, while regressions were conducted using individual data points, data displayed here were averaged in 5km distance bins (10km=6-10km; 20km=16-20km; 30km=26-30km; 40km=36-40km; 50km=46-50km) with standard error bars to clarify figures.

Figure 3. Fish length data collected with a baited camera system in hard habitat types inside (filled in circles) and outside (open circles) Penguin Bank (a, b, and c) and Makapuʻu (d)

BRFAs. Significant polynomial regressions for raw data are in black while predicted mean lengths from the best ranked standard least squares multiple regression models (X) are depicted in red (see Table 3).

Figure 4. Catch per unit effort (CPUE) metrics associated with protected areas (fishing areas that intersected with bottomfish restricted fishing areas (BRFAs; see Figure 1)) over time, beginning when protected areas were revised in 2007 and monitoring began.

Figure 1

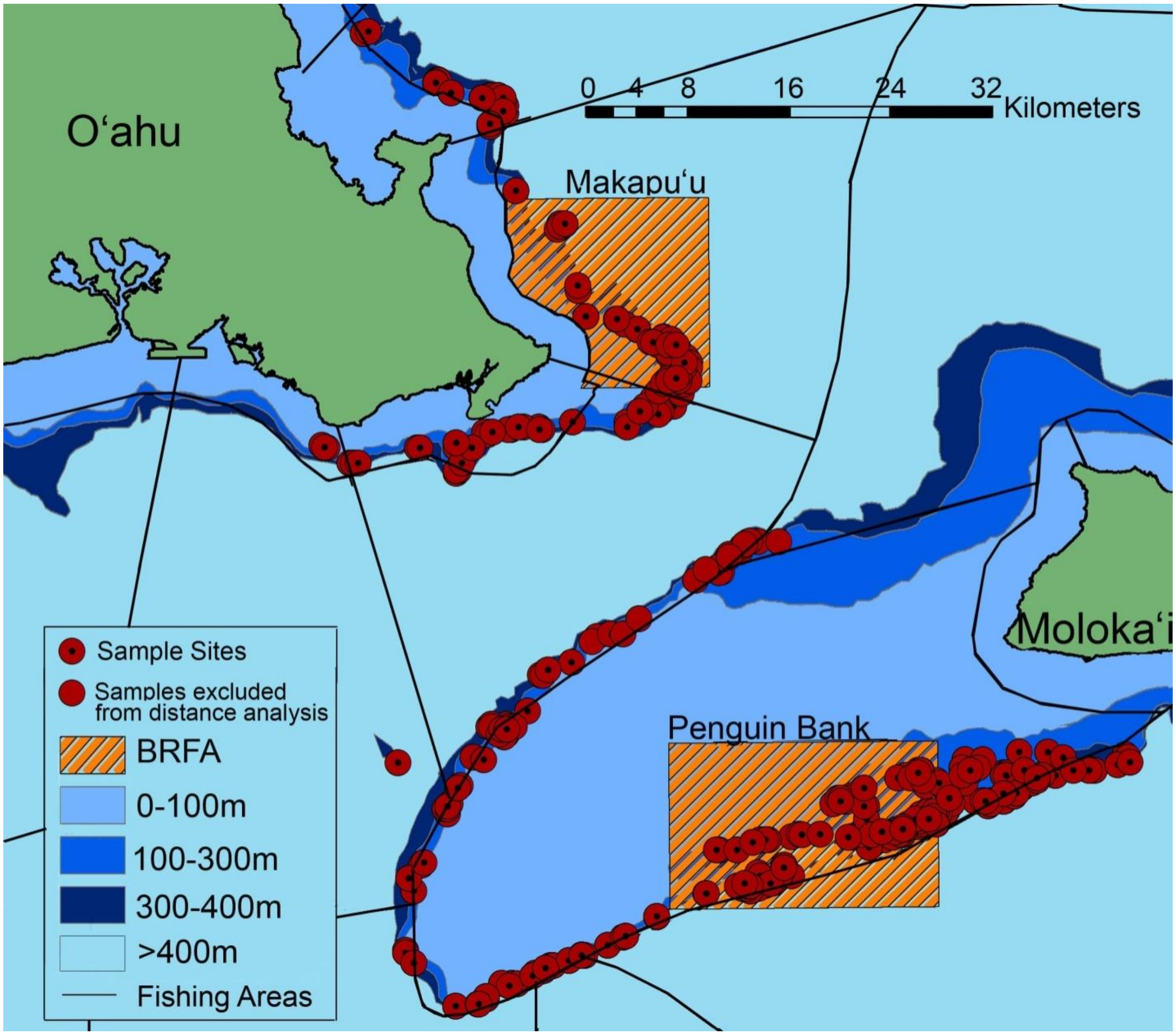
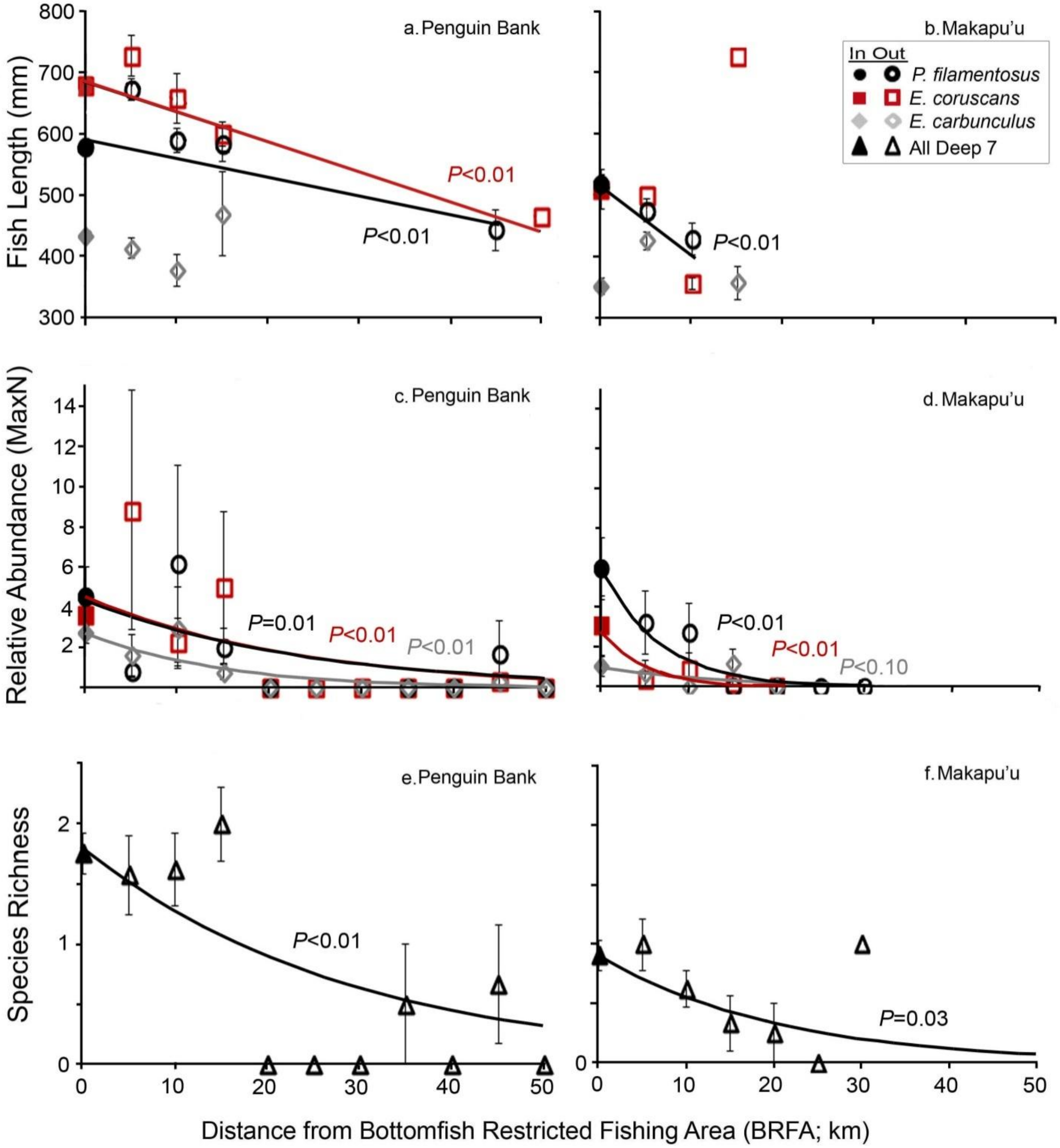
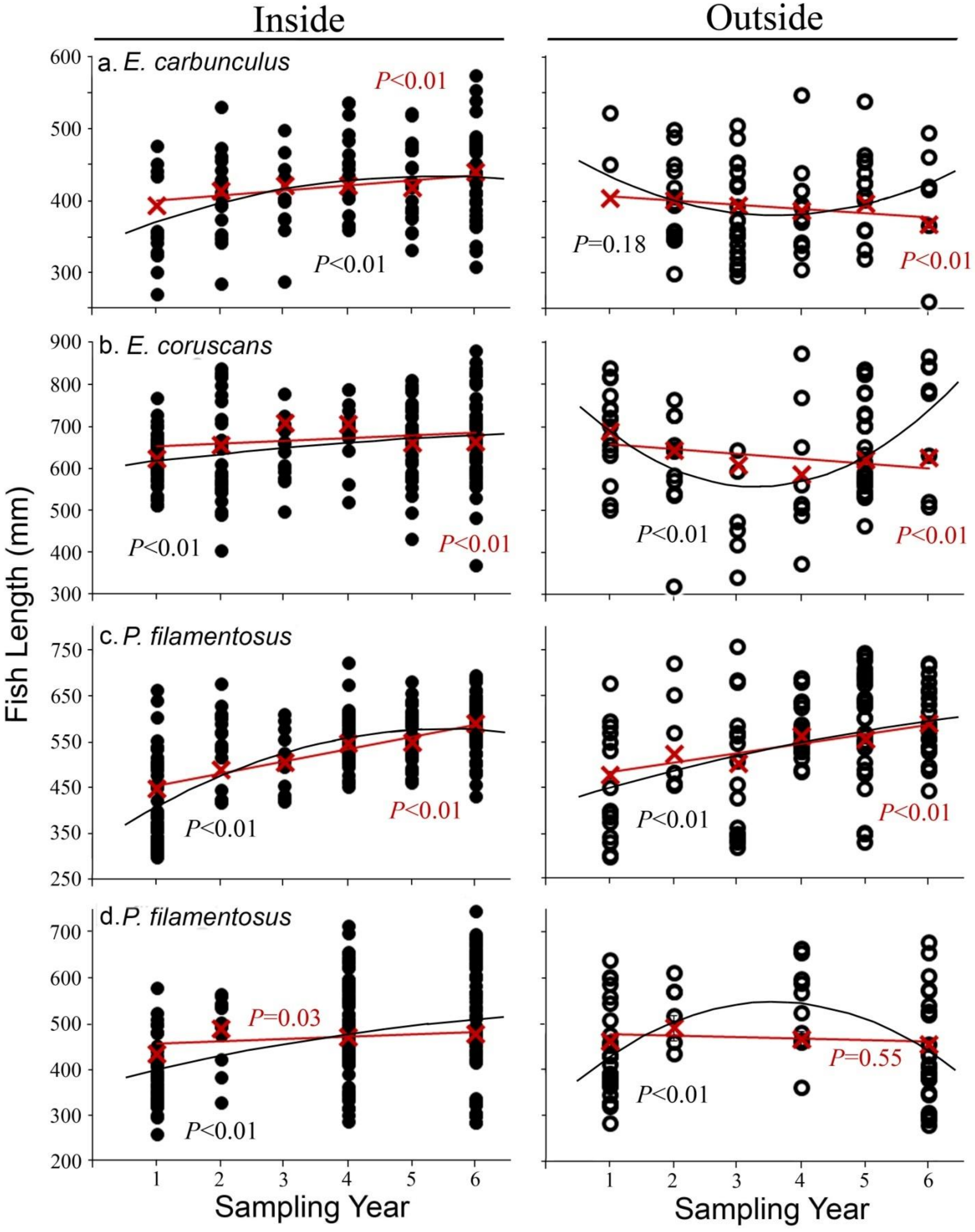

O‘ahu
Makapu‘u
Penguin Bank
Moloka‘i
0
4
8
16
24
32
Kilometers
● Sample Sites
● Samples excluded from distance analysis
▨ BRFA
0-100m
100-300m
300-400m
>400m
Fishing Areas

Figure 2.



Fish Length (mm)
800
700
600
500
400
300
a. Penguin Bank
P<0.01
P<0.01
b. Makapu'u
In Out
P. filamentosus
E. coruscans
E. carbunculus
All Deep 7
P<0.01
Relative Abundance (MaxN)
14
12
10
8
6
4
2
c. Penguin Bank
P=0.01
P<0.01
P<0.01
d. Makapu'u
P<0.01
P<0.01
P<0.10
Species Richness
2
1
0
e. Penguin Bank
P<0.01
f. Makapu'u
P=0.03
0
10
20
30
40
50
Distance from Bottomfish Restricted Fishing Area (BRFA; km)

787 Figure 3.



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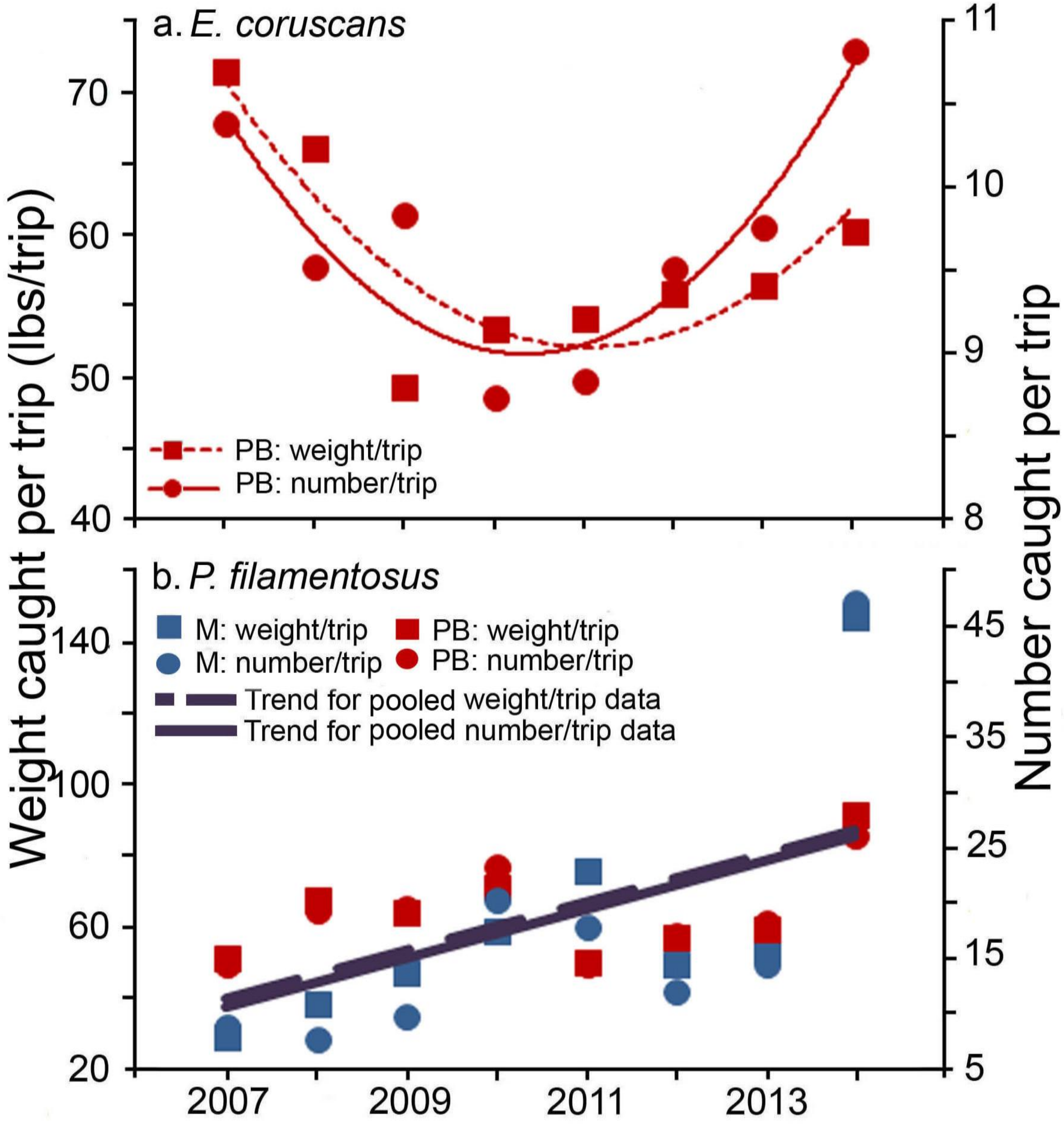
789

790 Figure 4.



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