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Mark Hixon

The Great Barrier Reef Marine Park is widely known as one of the best-managed marine protected areas in the world. The park is divided into different management zones, the most prevalent of which are marine national park zones, which are designated as no-take, and conservation zones in which some regulated fishing is permitted. I investigated the relationship between zoning designation and fish population metrics, specifically fish abundance by trophic guild in adjacent park and conservation zones in Opal Reef, Port Douglas, Australia. These areas have been designated as no-take and fishery regulated zones since 2004. From May 2011 to July 2011, underwater visual transect surveys were conducted by seven different school groups (n=7 samples) to obtain counts for several different fish families from multiple trophic levels, classified by feeding guild, at one site in the reserve and one site in the MPA. There were significantly higher counts of grouper and significantly lower numbers of butterflyfish in the no-take zones relative to the limited-fishing zones. These findings are consistent with the effects of a top-down trophic cascade in which an increase in the abundance of top predators (grouper), in this case caused by a release from fishing mortality in the no-take zone, results in a decrease in the abundance of their potential prey (butterflyfishes). These findings suggest that managers should continue to implement management strategies like marine reserves, because they have beneficial effects on entire ecosystems rather than just a single species. Involving student groups in the research process helped to raise awareness about reef degradation and management.

Key Words: Great Barrier Reef, Marine Reserves

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Effects of Marine Reserves on Populations of Herbivorous and Piscivorous Fishes on the Great Barrier Reef

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I understand that my project will become part of the permanent collection of Oregon State University, University Honors College. My signature below authorizes release of my project to any reader upon request.

Hilary J. Polis, Author

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Introduction

No-take marine reserves are progressively becoming an integral way of protecting marine ecosystems and biodiversity (Russ et al. 2008, McCook et al. 2010). Currently, the Great Barrier Reef is the largest network of marine reserves and marine protected areas (MPAs) in the world and spans an area of 344,400 square kilometers (Russ et al. 2008). According to NOAA, a marine reserve is an area where some or all fishing is prohibited and an MPA is defined as an area that has been set aside for the purpose of conservation of marine resources. The complex management system of the Great Barrier Reef was established in 1979 and is overseen by the Great Barrier Reef Marine Park Authority (GBRMPA), which is a cooperative authority between federal and state governments (Day 2002). The GBRMPA formulated a management plan in 2003, which divided the Marine Park into six different types of zones with corresponding colors: the light blue general use zone, the dark blue habitat protection zone, the olive green buffer zone, the green marine national park zone, the orange scientific research zone, the yellow conservation zone, and the pink preservation zone (Day 2002). Details of the specific activities allowed in selected zones are listed in Appendix E.

In 2004, the GBRMPA passed legislation to increase the area of the green national park or “no-take” zone from 4.5% to 33% due to the belief that the previous no-take zone area was inadequate to protect the tremendous biodiversity present in the park from anthropogenic threats (Fernandes 2005). The new system also included a Representative Areas Program (RAP) for the purpose of ensuring that the no-take zones included all different habitat types. The re-zoning process began after scientists identified 70 different "bioregions," which were considered to have unique biological and biophysical characteristics (GBRMPA 2009). These bioregions were then used to guide the re-zoning and mapping process to achieve a management goal of at least 20% of each bioregion conserved within no-take protected areas (Sweatman 2008). The initial zoning plan was revised several times following input from stakeholder groups and community members.

Since the re-zoning plan was developed in 2004, scientists have been active in developing monitoring programs at 100 different sites across the Marine Park. They are currently monitoring sites that have been no-take zones since 1983 and sites that were converted to no-take zones in 2004 (GBRMPA 2009). Scientists can determine the status of reef sites by using SCUBA divers to conduct underwater visual censuses (UVCs) on over 150 different species of reef fish, with a particular focus on species of importance to the fishing community, such as the coral trout (*Plectopomus leopardus*) (GBRMPA 2009). The divers collect data on fish numbers, biomass, and the biological condition of the survey area. Scientists then compare the data from new no-take zone sites to initial baseline data gathered from before the re-zoning began to get an idea of how the ecosystem structure at each site has changed with since re-zoning (GBRMPA 2009).

Although the monitoring is still in an early stage and data from many of the sites have yet to be analyzed, the initial results are showing indications of increases in densities of harvested fish species among areas converted to no-take zones (Russ et al. 2008, Lester et al. 2009, McCook et al. 2009). For example, researchers from James Cook University conducted surveys at 18 control sites open to fishing and 18 sites that had been converted to no-take zones on three inshore regions and five offshore regions of the reef (Russ et al. 2008). The researchers surveyed pairs of sites before the re-zoning and after 1.5 to 2 years of protection. The results showed that densities of coral trout increased in no-take zones for all three inshore regions of the reef and significantly so in two inshore regions. Coral trout densities also increased in no-take zones for all five offshore regions, significantly so in four. Sites in the corresponding locations that stayed opened to fishing showed non-significant increases in coral trout densities. Impressively, the trend of increased coral trout densities held for reefs spanning distances of over 1000 km from north to south. The trends also appeared with both inshore and offshore reefs, even though the environmental conditions for these different reefs differ significantly (Russ et al. 2008).

Lester et al. 2009 conducted an evaluation of no-take marine reserves utilizing 149 peer-reviewed scientific papers published between 1977 and 2006, on 124 marine reserves at 29 different locations across the world. The evaluation suggested that patterns of increases in exploited species densities within protected areas were so prevalent, that they cannot simply be explained by the hypothesis that reserves are simply placed in areas with more ideal fish habitat, and thus have higher fish densities at the outset (Lester et al. 2009). They also cannot be explained by the hypothesis that reserve implementation causes displaced fishing intensity to increase in areas around the reserves, thereby making reserves appear to work when they do not (Lester et al. 2009, McCook et al. 2009).

Marine reserves in the Great Barrier Reef have also been linked to increased larval dispersal rates for target species. Harrison et al. (2012) used DNA parentage analysis to determine the larval dispersal patterns for coral trout and stripey snapper (*Lutjanus carponotatus*) at a series of marine reserves within the Great Barrier Reef. The results revealed that, although marine reserves accounted for only 28% of the total reef area at the study site, they were responsible for producing 50% of all juvenile recruitment to fished and no-take areas within 30 km of the study site (Harrison et al. 2012). Christie et al. (2010) also found patterns of larval connectivity among populations of the exploited coral reef fish, yellow tang (*Zebrasoma flavescens*), in Hawaii. In this case, four parent-offspring pairs were identified through DNA parentage analysis. This identification revealed that larval dispersal distances ranged 15 to 184 km and in two instances, larvae dispersed from an MPA to an unprotected site. The finding of parent-offspring pairs among large adult populations of yellow tang indicates that there is significant larval connectivity between reefs. Therefore, if marine reserves are implemented correctly, they have the potential to provide an important source of recruitment to areas beyond reserve boundaries.

Changes in zoning and management tend to directly affect target species initially, and then may later affect overall community structure via indirect effects (Sweatman et al. 2008). Most of the analyses published on the results of the 2004 Great Barrier Reef re-zoning focus on numbers of coral trout and stripey snapper, which are target species for local fisheries (Russ *et al*. 2008, Harrison et al. 2012). Several studies have also been completed on the crown-of-thorns starfish (*Acanthaster planci*), a voracious predator of coral and can be an indicator of poor reef status, because it is usually prevalent in areas where its predators have been removed (Sweatman et al. 2008). In this case, predators of crown-of-thorns starfish are not necessarily target species. However, because fish populations are linked through predator-prey interactions and competition, shifts in the abundance of exploited species (which are often top predators, such as grouper) can have effects on many other species through trophic dynamics (Sandin et al. 2010). For example, the 1930's the decimation of cod stocks along the coast of Maine coincided with increases in lobster populations (Sandin et al. 2010). My study extends beyond simply exploring changes in fishery target species. I focus on analyzing numbers of fish at multiple trophic levels in order to identify changes that may be a result of both the direct and indirect effects of the 2004 re-zoning on the community structure of coral-reef fishes.

Overview of feeding guilds

This study examined fish from three different feeding guilds of coral-reef fish: herbivores or fish that feed on macroalgae; piscivores or fish that feed other fish; and corallivores or fish that feed on corals (Gerking 1994). The families from the feeding guild of herbivorous fishes included in this study were surgeonfish (Acanthuridae) and parrotfish (Scaridae). These fish feed by biting pieces of seaweed from the substrate (Gerking 1994). Parrotfish have a pharyngeal jaw apparatus that allows them to scrape and grind benthic algae and limestone from dead coral rock. They then digest the algae and release the byproduct as sand (Randall et al. 1990). Surgeonfish also feed on benthic algae, facilitated in several species by a thick-walled, gizzard-like stomach (Randall et al. 1990). Overexploitation of piscivorous reef fishes may result in increased quantities of herbivorous prey fish (Dulvy 2004).

The greatest number of families in this study were from the piscivore feeding guild. The piscivores featured in this study were from the families of jacks and trevallies

(Carangidae), grouper (Serranidae), and snapper (Lutjanidae), as well as a large wrasse (Labridae): the humphead or maori wrasse (*Cheilinus undulates*). The Carangidae commonly appear in large schools along the reef edges during the day, then disperse and feed at night. Jacks are very popular as seafood and are highly sought by both commercial and recreational fishermen (Randall et al. 1990). The Serranidae feed mainly on small fish and crustaceans, and are also caught both commercially and recreationally (White 2011). The primary target species of GBR fisheries is the piscivorous coral trout, a serranid, which was featured in the Reef Check Australia data (Graham et al. 2003). Coral trout generally inhabit one specific reef, but subpopulations are connected through planktonic larval dispersal, as is true for many reef fishes (Little et al. 2005, Harrison et al. 2012). The coral trout is a popular subject species for marine reserve studies, which can be attributed to the fact that its sedentary nature allows scientists to observe and model its population dynamics (Little et al. 2005). In addition, the fact that coral trout are a primary target species for reef fishermen drives a demand for information on how this grouper is affected by marine reserves.

Snappers (Lutjanidae) have similar desirable seafood characteristics to grouper, such as tasty flesh that makes them common targets for anglers and spearfishermen (White 2011, Randall et al. 1990). The humphead or maori wrasse is one of the largest piscivores on the Great Barrier Reef. This wrasse reproduces at a very slow rate and it is seen as a delicacy in East Asian cuisine. As a result, humpheads have become scarce, and there have been petitions to list them under the Endangered Species Act (Sadovy et al. 2003). The fact that the humphead wrasse has a very large home range and the transect area for this study was relatively small made it difficult to spot this species (Sadovy et al.

2003). The low numbers of humphead wrasse seen in this study made it difficult to run a statistical analysis on the data, so it was included as anecdotal evidence instead.

Only one family, the butterflyfishes (Chaetdontidae), was studied from the corallivore feeding guild. Butterflyfish commonly peck at coral with their elongated jaws and feed on coral polyps and small invertebrates (Randall et al. 1990). The capture of butterflyfish for the aquarium industry is very common, but chaetodontid species are not listed as endangered or vulnerable (Randall et al. 1990).

Coral reefs have a unique ecosystem structure in that piscivores generally comprise a greater percentage (up to 50%) of the entire of the species present, and as a result, the piscivores play prominent role in structuring of coral reef communities (Hixon 1991, Gerking 1994). Increased piscivore biomass and numbers may cause top-down trophic effects (McQueen et al. 1986). There is much concern that the overexploitation of predatory fishes with market value may indirectly change the structure and function of coral reef ecosystems (Dulvy *et al*. 2004). Yet at the same time, fishing is important to the coastal regions surrounding the Great Barrier Reef for both economic and cultural reasons. All three of the reef fishery sectors on the Great Barrier Reef -- commercial, recreational, and charter -- generally use hook-and-line as their primary gear (Little et al. 2005). It is important to note that the Great Barrier Reef fishery is multispecies, so this study focuses on several piscivore species that are targeted by the GBR line fishery, rather than just the primary target of coral trout.

Trophic Cascades

The research on top-down trophic cascades associated with marine reserves has shown that the removal of predatory fishes at the top of the food chain can significantly alter marine food webs (Madin et al. 2010). These trophic cascades generally occur in areas with high fishing intensity, because fishermen typically desire the large piscivorous fish from the highest trophic levels (Lamb and Johnson 2010). Fishermen may fish down the food web and target smaller fish from lower trophic levels as the large ones are depleted (Pauly et. al 1998). Still, overall, fishing intensity on species at upper trophic levels is disproportionately high on coral reefs (Lamb and Johnson 2010).

The trophic dynamics of coral reefs are very efficient, meaning that there is intense grazing and a high percentage of biomass is captured and recycled. As a result of the high turnover of lower trophic levels, un-fished reef ecosystems have the potential to support a biomass composition with up to 50% top-level piscivorous predators, yet this structure is rarely seen due to human influence (Sandin et al. 2010). High numbers of predatory species in the ecosystem result in higher rates of predation and as a result, juvenile mortality due to predation can reach 90% in many cases (Shulman and Ogden 1987, Almany and Webster 2006). Importantly, when large piscivorous fish are removed from the ecosystem, their prey species usually increase in biomass and numbers due to release from predation pressures (Frank et al. 2005). Conversely, when fishing declines, the biomass of piscivorous species is expected to increase. This increase in abundance can have a variety of direct and indirect effects down the food web (O'Sullivan et al. 2011).

Indirect effects occur when one species directly interacts with another species, and this interaction indirectly affects a third species (Menge 1995). Categories of indirect effects include trophic linkages (predation, competition), and environmental, chemical, and behavioral effects (Menge 1995). Trophic linkage indirect effects occur in response to changes in species abundance, environmental effects occur in response to changes in the abiotic environment, and chemical indirect effects occur through chemical pathways (Menge 1995). Behaviorally mediated indirect interactions (BMII) occur when an indicator (I) species causes the behavior of another transmitter (T) species to change, which in turn affects a third species, also known as the receiver (R). An example of a BMII occurs when schooling fish (T) form dense aggregations at the sea surface in order escape from larger predatory fish, which attack from below (I). When these schools are forced to the surface they become easier targets for flying predators (R), such as seabirds (Dill et al. 2003). Although this particular BMII, known as "synergistic predation," is not a trophic cascade, it occurs in coral-reef fishes (Hixon and Carr 1997). This type of BMII occurs only when juvenile mortality is density dependent, so data on mortality rates relative to catch sizes can be used to help detect synergistic predation.

Previous studies have shown several incidents of trophic cascades associated with prey release on coral reefs. A study by Madin et al. (2010) in conjunction with previous studies demonstrated that fishing for top predators on coral reefs indirectly affects prey feeding behavior, and can ultimately lead to behaviorally-mediated trophic cascades. Lamb and Johnson (2010) examined differences in fish biomass inside vs. outside the Exuma Cays Land and Sea Park marine reserve in the Bahamas and found that the fish within the reserve at the highest trophic levels increased in biomass, those at the intermediate levels decreased in biomass, and those at the lowest trophic levels increased in biomass, a clear trophic cascade.

Many previous studies have documented the significant effects of changes in herbivore abundance on species of primary producers in nearshore tropical ocean ecosystems (Hixon 1997). However, it is more difficult to find studies that demonstrate clear effects of changes in abundance of predatory species on species of lower trophic levels for these ecosystems (Sandin et al. 2010). One explanation for this lack of evidence is that top-down control is less likely to occur in coral reef ecosystems because they have complex food webs (Frank et al. 2007). This study seeks to determine whether the effects of the removal of fishing in the no take zones of Opal Reef is strong enough to create a trophic cascade. This would mean that fishing regulations for humans would result in higher abundances of target species, which would result in an increase in the rates of predation of target species on species from lower trophic levels. This increased predation would thus result in lower abundances of prey species.

Methods

Study Area

Fish surveys were first conducted in the North Queensland part of the Great Barrier Reef in the Cairns zoning region off of the coast of Port Douglas (Figure 1). The surveys were taken specifically at Opal Reef, which is classified as and outer shelf crescentic reef with an area of 24.7 square kilometers (Sweatman et. al 2008). Fish counts were performed first at South North Opal (SNO), a green zone, then at Blue Boy, an adjacent yellow zone. Each of the seven groups visited SNO then Blue Boy on two consecutive days, resulting in a collection of data on a total of 14 days between May 31 and July 8, 2011.

SNO reef is located on a reef flat which slopes to small wall and this area is full of small channels and pinnacles, which make for excellent fish habitat (Markley & Schappy 2010). Opal Reef has been extensively sampled since 1986. The reef had an active crown-of-thorns starfish outbreak in 2000, which lasted until 2002, when the reef was reclassified as “recovering” (Sweatman *et al*. 2008). As a result of this outbreak, median live coral cover diminished to low levels (1-10% of reef cover) by 2003. Opal Reef was still classified as recovering in 2007, although coral cover had returned to levels classified as moderate (11-20%). The percentage of hard coral cover in particular, declined from 24% in 1995 to 19% in 2007 and soft coral cover declined as well from 43% in 1995 to 34% in 2007 (Sweatman *et al*. 2008). Data on crown of thorns outbreaks by study site were unavailable.

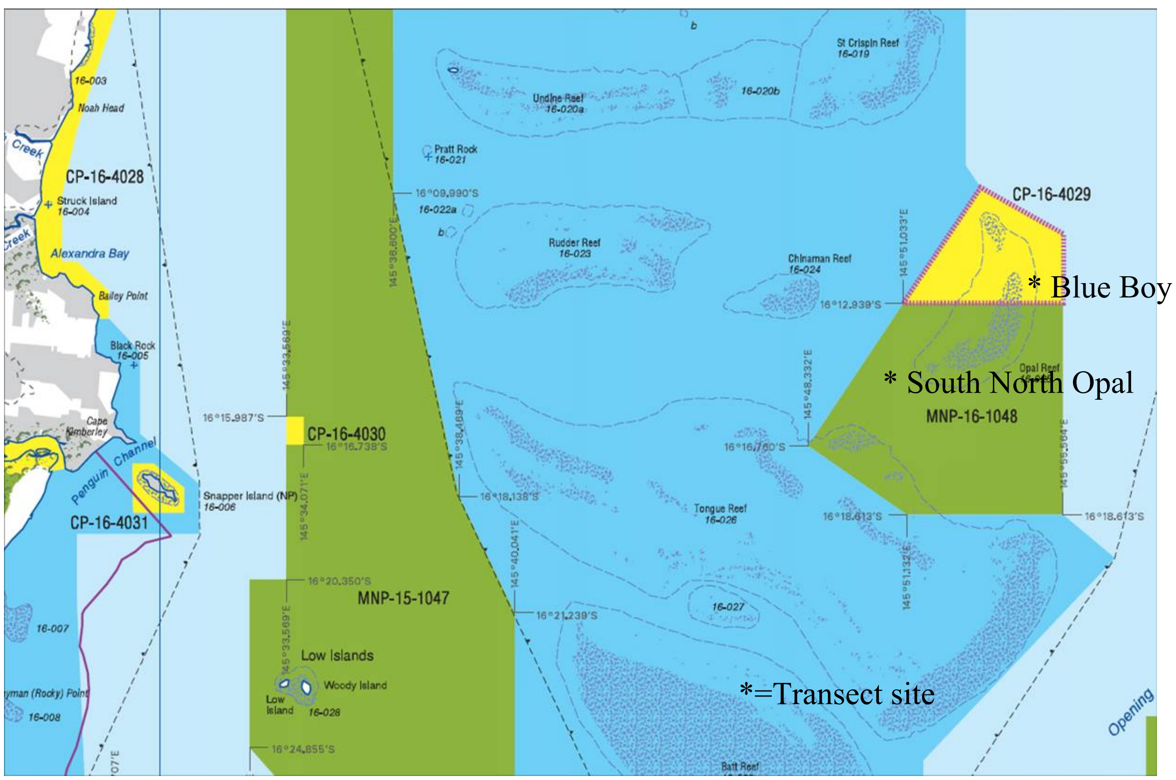

Figure 1: Map of Opal Reef, Great Barrier Reef, (right) with yellow no-take zone and green limited-fishing zone indicates. Source: GBRMPA

Data Collection Methods for AUIP Groups

Student groups from seven different schools participating in a study abroad program with American University International Programs (AUIP) collected the data, titled “AUIP.”

These student groups of approximately 30 students each conducted visual surveys along two different transect lines at Opal Reef (latitude 16°13S longitude 145°53E). Each of the seven groups visited the reef two times on separate days, resulting in a collection of data on a total of 14 days between May 31 and July 8, 2011.

When the school groups arrived at the transect site, they were divided into a red group and a blue group, and each student was given a partner within their own group. They were also given snorkeling equipment and a lesson on the technique used for collecting data, which was an underwater visual census (UVC). UVC is a typical technique for monitoring reef fish distribution and abundance, where a SCUBA diver or snorkeler swims along a transect line and counts fish along this line (GBRMPA 2009). In this case, students snorkeled along a 50 meter by approximately 3 meter transect line running through depths of approximately 2 to 8 meters. The red group conducted a census along a red transect line and the blue group conducted a census along a blue transect line. Each pair within the group was directed to count all fish belonging to a particular family (or in one case species -- the humphead wrasse) within each transect. This resulted in each fish group being counted by two pairs of snorkelers along different transect lines. The data were reported as the mean fish count per transect.

Families and species were studied in field guides before snorkeling to ensure proper visual recognition. In addition, four groups recorded the type of substrate and coral underneath each transect line. These transects spanned the back reef flat and the reef crest. Each group also had a waterproof clipboard, which they used to tally the number of fish they observed from each zone. Sampling was consistent between zones in terms of time of day, weather conditions, and tidal patterns.

Data Collection Methods for Reef Check Australia

This report also analyzes supplemental data from Reef Check Australia's monitoring program. Reef Check Australia (RCA) is a non-profit organization, which began gathering data in 2001 with the help of highly trained volunteer divers. This program is meant to aid the Australian Institute of Marine Science (AIMS) long-term monitoring program described above because it expands to sites that are not included within the scope of the AIMS monitoring program (Andrews et al. 2007). The AIMS program surveys a few sites biannually in high detail and RCA's program surveys a more widespread array of sites in medium detain annually (Andrews et al. 2007). The RCA program is meant to foster a sense of community stewardship of the reef because community members choose which sites they would like to survey and the data are used for policymaking and zoning purposes.

The RCA data were gathered by volunteer divers, who began by laying 100m of transect tape along the reef floor. The transect line was then further divided in to 4 × 20 meter sections with 5m between each section (Stella et al. 2008). In places where the reef was not continuous, the transect subsections were separated by 5 meters or more of area that was not surveyed (Stella et al. 2008). Fish surveys were performed using a modified belt-transect method. For this method, a diver swam in an S-shape along the transect line and aimed to move at a speed of about 20 meters in 10 minutes. The surveyor records selected fish species in 2.5 meter belts on either side of the transect line and 5 meters above the transect line (RCA 2012).

RCA conducts research at several reefs and these reefs are further subdivided into survey/dive sites, and then research sites. This report examines the surveys from Opal

Reef and the survey/dive sites for Opal Reef were located at five different green zone locations– (South-North Opal (SNO), The Wedge, Two Tone, Bashful Bommie, and Split Bommie) and one yellow zone site (Cathedrals). There may be several different transect lines or research sites at each dive site. In summary, research sites are contained within dive sites, which are contained within the reef. The habitat type and depth vary by individual research site, and this is information reported with the data. Data from the same dive site with the same habitat type and depth were averaged between research sites. The fish counts were originally reported as an average per 100 square meter transect, therefore numbers are aggregated for the 4 × 20m transects. In this study, the counts were then reported as mean fish counts per dive site. Therefore, if there were multiple transects at a given dive site, the fish counts for each transect at that site were averaged to get the mean count per dive site. It is important to note that RCA data were reported for different transect sizes than the AUIP data, so the results could not be aggregated.

Statistical Analysis

Fish counts were evaluated at the family (in one case, species) and feeding guild level and compared between management zones. The distribution of mean fish counts by species was developed using *R* statistical software (Fig 2). These plots showed that the mean fish counts by species did not have equal variances, indicating that the assumptions for a one-way ANOVA test were not met (Ramsey and Shafer 2013, pp. 121). The pattern of unequal variances still occurred even after a log transformation of the data. In addition, a non-parametric test would not have been appropriate in this case because there were different shaped distributions between families, and the non-parametric alternative for ANOVA, the Kruskal Wallis test, relies on the assumption that all the groups have the same shaped distribution (McDonald 2009).

Therefore, a Welch's t-test was used to compare the differences in counts for each individual species between the green zones and the yellow zones. A Welch's t-test was chosen, because it does not require the samples being compared to have equal variance. The assumption of equal variance is often violated with count data of species and families. For example, lower counts for the large and rare humphead wrasse had lower variance compared to counts for butterflyfish, which were commonly seen. The Welch's t-test works well in this case because each individual sample standard deviation is used to estimate its own population standard deviation, rather than pooling all the samples together to find one estimate of the population standard deviation (Ramsey and Shafer 2013, pp. 98).

Sample sizes in this study were small, so the assumption that populations were normally distributed was likely not met. Due to possible violations of the normality assumption, the 95% confidence intervals for the Welch's t-test are approximate and not exact.

A quasi-Poisson regression analysis was used to estimate the effect of zoning designation on the mean counts of fish by feeding guild. The Poisson regression is appropriate for this study because it has features that make it a good estimate for problems that involve "count data where the spread increases with the mean" (Ramsey and Shafer 2013, pp. 676). The model showed evidence of over-dispersion, which meant the residuals that were still larger than what is typically acceptable for Poisson regression. A quasi-Poisson model was then developed to account for the over-dispersion. This model accounted for the larger than normal residuals by adjusting the standard errors. Without the Quasi-Poisson correction for over-dispersion, the standard errors would be too low, thus the chances of Type 1 error, or falsely concluding that there was a difference between zones if one did not exist, would exceed the test level α.

Reef Check Australia supplied data sets for five different no-take zones and one conservation zone. After discussion with a statistician, it was decided that statistical tests should not be run on the RCA data because the unequal ratio of no-take zones to conservation zones would make comparisons difficult. In addition, there were no fish from the target families observed in the conservation zone during RCA's dives, so this would have also made comparisons challenging. Therefore, the Reef Check Australia data were included only as anecdotal evidence in this study.

Results

Table 1: AUIP student group mean fish counts per transect and Welch's t-test comparing no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef, by family, species, and feeding guild.

Species/Family	Mean per transect count no-take	Mean per transect count conservation	Welch's t-stat	95% CI	P-value
AUIP Grouper (Serranidae)	7.86	3.29	(z stat) 2.0743	(3.38x10^-5, 9)	0.038**
AUIP Jacks (Carangidae)	2.00	0.43	-1.58	(-3.92, 0.77)	0.158
AUIP Snapper (Lutjanidae)	14.6	22.4	0.50	(-26.86, 42.57)	0.628
AUIP Snapper (Lutjanidae no Ohio)	17.0	13.0	-0.27	(-36.67, 28.67)	0.791
AUIP Humphead Wrasse (*Cheilinus undulates*)	1.29	1	N/A	N/A	N/A
AUIP Surgeonfish (Acanthuridae)	48.3	42.6	-0.46	(-41.08, 26.80)	0.6541
AUIP Parrotfish (Scaridae)	61.0	95.1	0.71	(-72.61, 140.89)	0.494
AUIP Butterflyfish (Chaetodontidae)	33.86	57.71	1.79	(-5.14, 52.85)	0.098*
AUIP Piscivores with Ohio	25.75	27.12	N/A	N/A	N/A
AUIP Piscivores without Ohio	28.15	17.72	N/A	N/A	N/A
AUIP Herbivores	109.29	136.29	N/A	N/A	N/A
AUIP Corallivores	33.86	57.71	N/A	N/A	N/A

Results

There was moderate evidence that there was a difference between the no-take zone and the conservation zone for groupers (Family Serranidae) of the piscivore guild (Welch's T-test, P-value= 0.038). The estimated difference in mean counts for the no-take zone was 4.57 higher than the conservation zone (Table 1). There was also evidence that there were different mean fish counts in the no-take zone and the conservation zone for butterflyfishes of the corallivore guild (Family Chaetodontidae) (Welch's t-test, P-value=0.098). The estimated difference in mean counts for the no-take zone was 23.85 higher than the conservation zone (Table 1).

Family Level

The results were mixed for the difference in number of fish for each zone, categorized by families and species within feeding guilds. There were higher fish counts in the no-take zone than the conservation zone for two piscivore families -- groupers and jacks -- and one piscivorous species, the humphead wrasse (Fig. 3). There were lower fish counts in the no-take zone than the conservation zone for one piscivore family, snappers. There is also some concern about the snapper data from Ohio State University school group because the pair of snorkelers that were counting snapper found a very large school at the end of the transect in the conservation zone. They had to estimate the number of fish in the school and they believed that their estimation was inaccurate. Because this one school of fish was seen in an isolated, single event, it was treated like an outlier and tests for snapper were run both with and without the Ohio data. Running the Welch's t-test and quasi-Poisson regression without the Ohio snapper data point did not result in statistical significance (Table 1). However, removing the Ohio group from the snapper data did result in the aggregate number of piscivores being higher in the no-take zone than the conservation zone (28.15 no-take zone vs. 17.72 conservation zone) (Fig. 4).

In regards to fish at the lower trophic levels, the mean numbers of herbivorous fish also showed mixed results across families. Mean fish counts were greater in the no-take zone than the conservation zone for surgeonfishes and lower in the no-take zone than the conservation zone for parrotfishes (Table 1). The overall numbers of herbivores were greater in the conservation zone (total mean of 137.7), than the no-take zone (total mean of 112.3) (Fig. 5). The butterflyfishes were the lone family measured from the corallivore feeding guild.

Feeding Guild Level

The quasi-Poisson regression analysis showed that the difference in zoning regulations between the no-take zone and conservation zone was not associated with a change in the mean fish counts by feeding guild for piscivores, herbivores, or corallivores (quasi-Poisson regression, P-values: 0.263, 0.314, 0.263, respectively) (Table 2). The only explanatory variable of significance was for feeding guild alone and it showed that the overall number of fish in the piscivore feeding guild was significantly less than for the herbivore or corallivore feeding guild, after accounting for zoning effect, which is characteristic of most coral reef ecosystems and is to be expected (Quasi-Poisson Regression, P-value= 1.26e-10) (Table 2).

RCA Data

It is important to note that no fish were observed in the conservation zone in this data set (Table 3), which made it difficult to run a statistical analysis.

Discussion

This study was limited by the fact that there were only seven measurements for each category of fish (family [in one case, species] or feeding guild) and large variances in many of the fish counts. This data structure made it very difficult to detect any real differences if they existed between management zones. In addition, the fact that the data included in this study were from only one reef out of the world's largest reef network limits our inference about the effects of marine zoning on fish community composition. Yet despite these limitations, the results still showed significantly higher numbers of grouper in the no-take zone than the conservation zone, significantly lower numbers of butterflyfish in the no-take zone than the conservation zone, and some indication of lower numbers of parrotfishes in the no-take zone than the conservation zone. Given that small butterflyfishes and parrotfishes are prey of large groupers, these findings may be indicative of a trophic cascade caused by the cessation of fishing in the marine reserve.

The fact that the only family that had significantly higher numbers in the no-take zone than the conservation zone was the piscivorous grouper family suggests that the elimination of fishing in the no-take zone in 2005 may be contributing to a trophic cascade. According to this hypothesis, fishing in the conservation zones of Opal Reef cause decreases the abundance of target species, which are mostly piscivores. In the years since the zoning changes, the commercially targeted piscivores likely became more abundant in the no-take zones relative to the conservation zones due to the protection afforded by stronger fishing regulations. Increased piscivore abundance in the no-take zones resulted in higher rates of predation that caused greater reductions in abundances of herbivore and corallivore prey species relative to conservation zones.

The statistical significance of higher numbers of grouper in the no-take zone is especially important to recognize. Fish from the grouper family, especially coral trout, are popularly and commercially caught in Australia and have been victims of overfishing in the past. This could indicate that the higher numbers of grouper in the no-take zone vs. the conservation zone may be a result of protection from overfishing (Coleman and Williams 2002). Grouper grow very slowly, live a relatively long time, and are easy for fishermen to locate, which make them easy targets for overexploitation. Grouper can grow up to 10 cm per year for their first two years. Their growth rate then levels out to around 3 cm per year after two years and decreases to less than 1 cm per year after 7 years (Ferreira and Russ 1994). Typical fisheries management strategies based on catch sizes have often been unsuccessful for grouper due to their complex life histories (Coleman and Williams 2002). The fact that marine reserves provide a more ecosystem-based approach to management may have helped contribute to the higher numbers of grouper observed in the no-take zone.

The somewhat lower numbers of parrotfish and significantly lower numbers of butterflyfish in the no-take zone compared to the conservation zone may be explained by increased predation from the grouper and other piscivores, which is also indicative of a trophic cascade. Previous studies have shown that the prey species of coral trout commonly include parrotfish and occasionally include butterflyfish (St. John 2009). Decreases in local populations of grouper cause a release from predation on such prey species and can result in a local population increase for these prey species (Mumby et. al 2012).

This shift towards higher numbers of grouper and lower numbers of prey species in the no-take zone is in its relatively early stages and can be expected to develop over time, a trend that could be demonstrated if AUIP continues to gather data over many summers at Opal Reef. Graham et al. (2003) looked at the effects of marine reserve designations on coral trout and its prey in no-take and limited-fishing zones that had been established for 14 years in the Whitsunday and Palm Islands. They found that the abundance of coral trout was 3-4 times greater in no-take zones than fished sites. After controlling for habitat variations, they also found that eight out of nine prey species had higher densities in the fished zones than the no-take zones. There was also significant evidence that the density of all prey species combined was twice as high in the fished zones as the no-take zones (Graham 2003). The results for Opal Reef may be showing the start of similar trends to those for the Whitsunday and Palm Islands, yet more time and more observations are necessary to test this prediction.

It is important to take into account the time horizons when considering the effects of the marine reserve designation. There are several examples of other reserve sites where the magnitude of the reserve effect increased with time (Halpern 2003). The direct effects of marine reserve designation on target species can take 3 to 7 years to be detected, but the first detection of indirect effects usually take from 11 to 15 years (Babcock et. al 2010). Because changes in management structure of the GBR began only in 2004, many of the indirect effects of the reserve implementation most likely have not yet been detected.

This study can be used in conjunction with surveys at other reefs in the GBR to make conclusions about the impacts of marine reserves on the reef network as a whole, but it cannot be used alone to draw these conclusions. In addition, it is important to note that this study makes comparisons only between “no-take” and limited-fishing zones, so results may have been more significant if it took into account surveys of areas with no fishing regulations. It would have been helpful to have enough data to run a Before-After- Control-Impact comparison (Stewart-Oaten et al. 1986) on areas that were open to fishing before and then became green zones in 2004, but unfortunately such data were not available for Opal Reef. A BACI would provide a better understanding of the exact cause of changes in fish abundance between zones, because this type of design controls for variability between the two sites. Most importantly, this study could be improved by adding additional replication, both with more transects and at the same transects but during different times of the year.

The comprehensive data collection programs from organizations like GBRMPA, AIMS, and Reef Check Australia will allow for the monitoring of long-term trends in reserve effects. However, many of the issues in determining reserve effects lie in the fact that much of the data have yet to be analyzed. Some of these organizations were unwilling to share data with others looking to do analyses. In the cases where the organizations were willing to share data, the format of the data collection often made it difficult for comparison between zones. Efforts should be made to streamline data management and distribution so that the impacts of zoning can be analyzed efficiently.

Conclusion

Although the Great Barrier Reef is one of the world’s best managed and least threatened coral reefs, its biodiversity is still at risk of declining. Recent findings show that if major changes are not made in regards to rates of disturbance and coral declines, then coral cover in the central and southern regions of the GBR is projected to decrease to 5–10% by 2022 (Dea'th et al. 2012). GBR Management strategists have started to address these rates of disturbance by focusing on reducing anthropogenic risk factors, such as overfishing, coastal development, and pollution because such threats can be managed on a local level (Dea'th et al. 2012).

Although the impetus behind reducing fishing intensity has typically been to protect target species, the evidence of a trophic cascade in the no-take zone of Opal Reef demonstrates that the broader effects of fishing on the Great Barrier Reef can extend beyond target species to entire ecosystems. This evidence suggests that the 2004 decision to set aside 33% of the Great Barrier Reef as no-take marine reserves is already changing the trophic structure of reef ecosystems. Therefore, fisheries managers should continue to implement management strategies like marine reserves, which have beneficial effects on entire ecosystems rather than just a single species.

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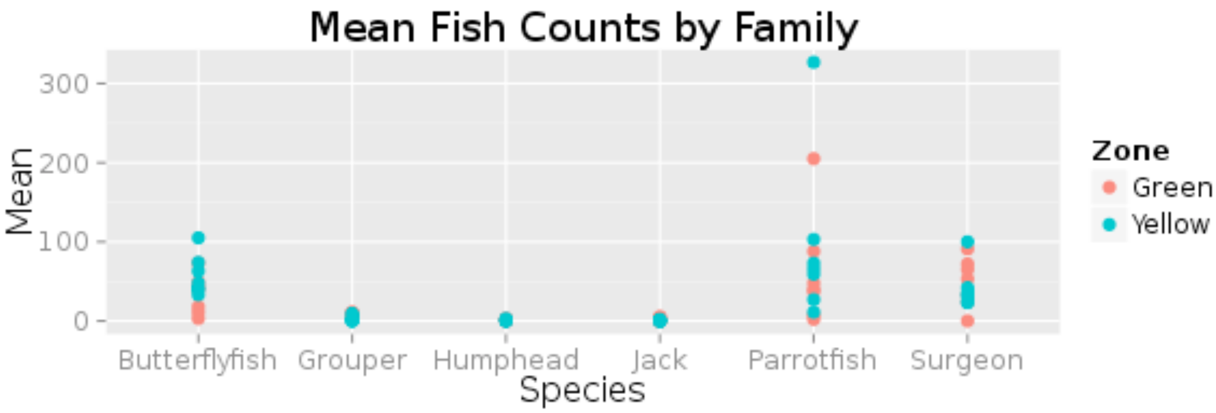
Appendix A- Plot of Mean Fish Counts

Fig 2. Mean fish counts by family



Appendix B: Quasi Poisson Analysis on Feeding Guilds

Table 2: Regression Coefficients by Guild

	Estimate	P-value
Corallivores	1.3097	0.290
Herbivores	1.8042	0.627
Piscivores	0.1765	0.2900184

Estimates for median differences of counts between feeding guilds for limited-fishing (yellow) zones and no-take (green zones) on Opal Reef, Great Barrier Reef. The differences between zones are not significant. Estimates are multiplicative.

Quasi Poisson Regression Output

Poisson Regression by

Coefficients:

```
             Estimate   Std. Error t value Pr(>|t|)
(Intercept)    3.6800     0.2966   12.407  < 2e-16 ***
Yellow Zone    0.2698     0.2537    1.063    0.290
Herbivore      0.2933     0.3039    0.965    0.337
Piscivore     -1.9358     0.4288   -4.515 1.84e-05 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

(Dispersion parameter for quasipoisson family taken to be 43.12791)

Null deviance: 4908.3  on 97  degrees of freedom
Residual deviance: 2570.2  on 94  degrees of freedom
```

Actual Adjusted Parameters

Estimates for herbivores and piscivores had to be added to the estimate for yellow zone, to get the actual effect of feeding guild by zone.

Appendix C: Reef Check Australia Data

Table 3. RCA mean and standard error (SE) of fish counts per transect for no-take (green) zones and limited-fishing (yellow) zones by feeding guild, family, and species.

Family or Species	Mean Fish Count Green	SE Green	Mean Fish Count Yellow	SE Yellow
RCA Snapper (Lutjanidae)	22.1	21.6	0	0
RCA Humphead Wrasse (*Cheilinus undulatus*)	0.0125	0.0125	0	0
RCA Coral Trout (*Plectropomus leopardus*)	0.167	0.0684	0	0
RCA Parrotfishes (Scaridae)	2.971	0.650	0	0
RCA Butterflyfishes (Chaetodontidae)	2.725	0.422	0	0
RCA Piscivores	5.58	5.40	0	0

Appendix D: Graphs of Results

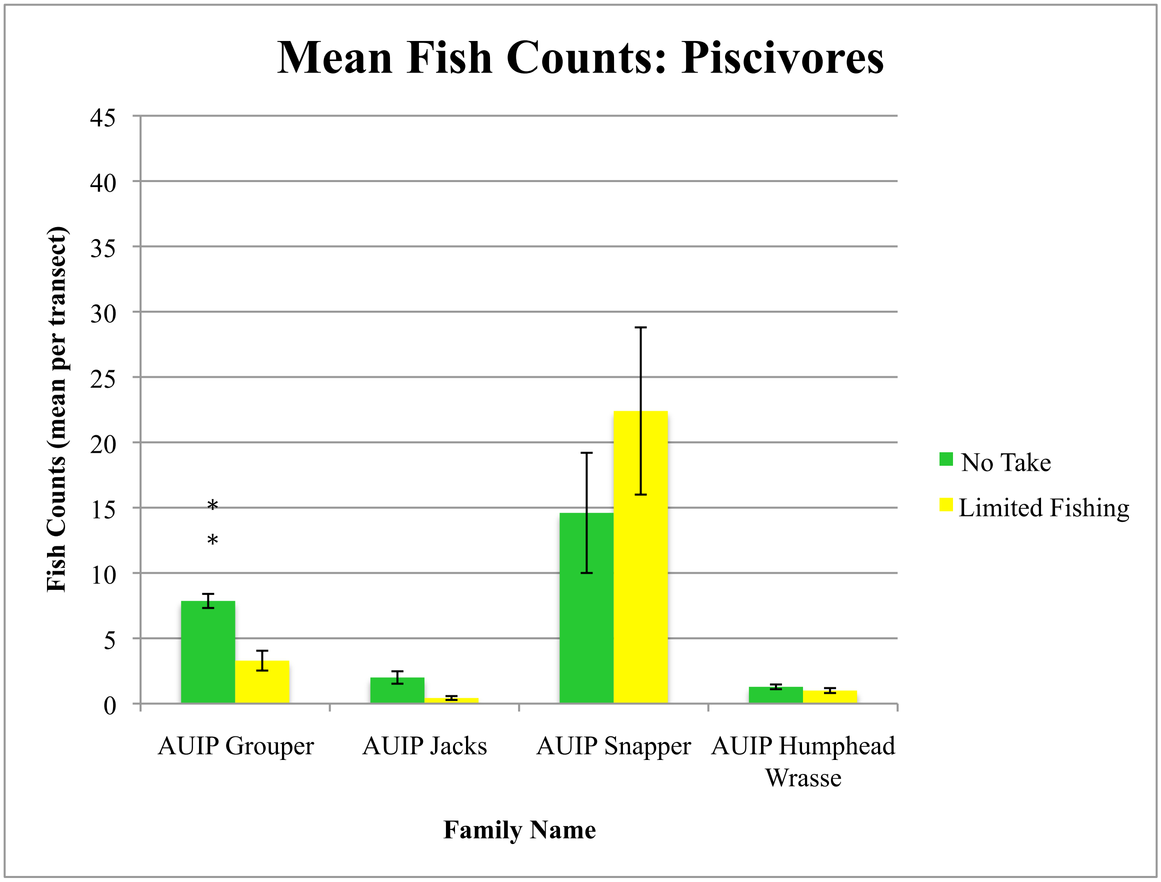


Fig 3. Mean (±SE) counts of piscivorous fish in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. Double Asterisks indicate significant differences between zones (P<0.05).

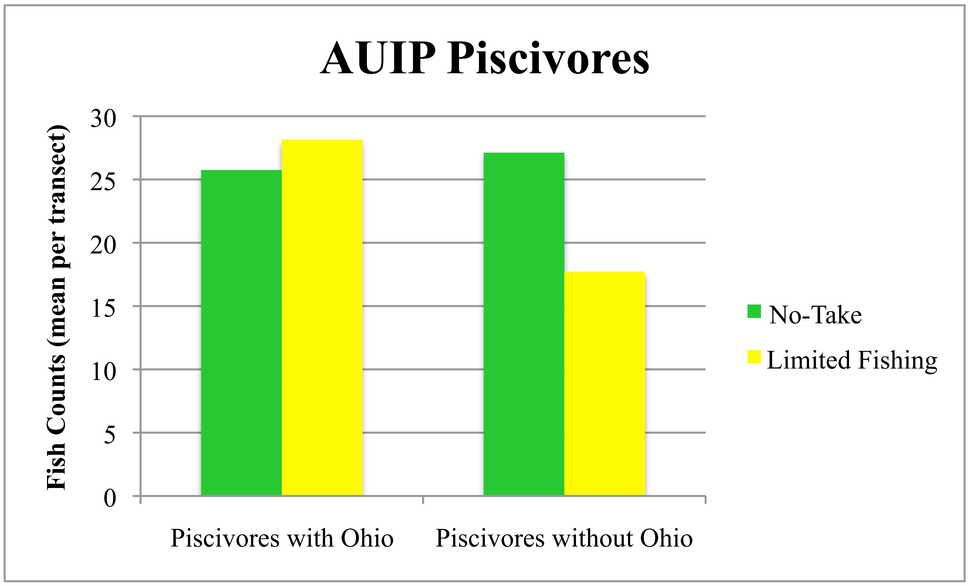


Fig 4. AUIP sum of mean fish counts for all families and species for the piscivore feeding guild (with and without Ohio) in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. The difference between zones is not significant.

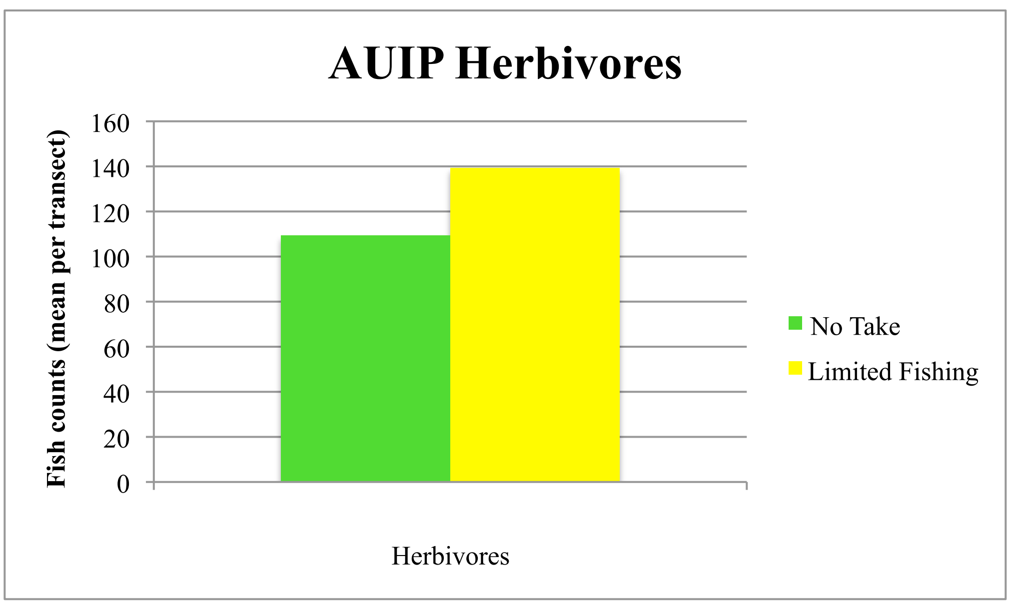


Fig 5. AUIP sum of mean fish counts for all families and species for the herbivore feeding guild in no-take (green) zones and limited- fishing (yellow) zones on Opal Reef, Great Barrier Reef. The difference between zones is not significant.

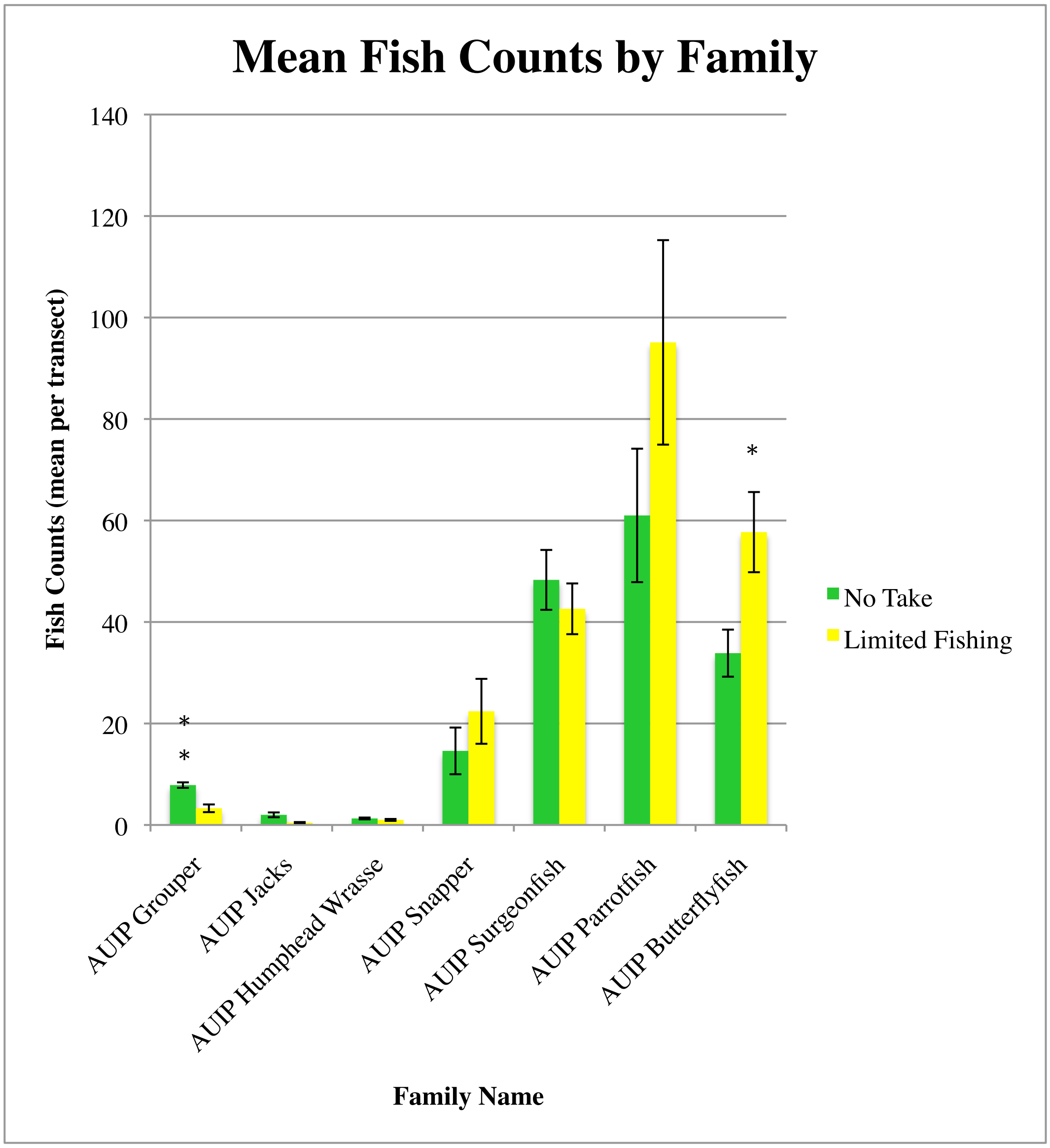


Fig 6. Mean (±SE) fish counts by family in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. Asterisks indicate weak significant differences between zones (P<0.10). Double Asterisks indicate significant differences between zones (P<.05)

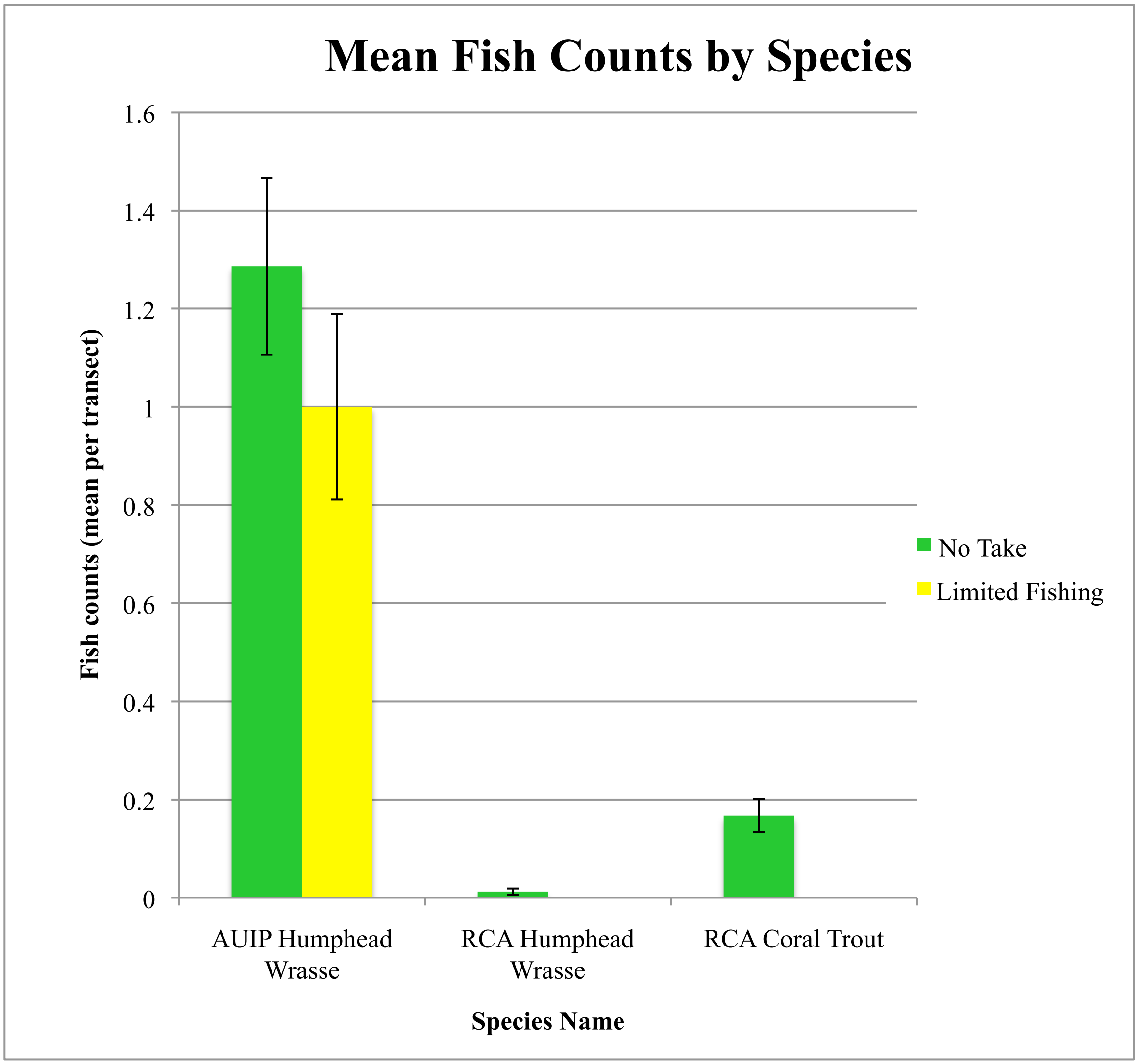


Fig 7. Mean (±SE) fish counts by species in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. No differences between zones are significant.

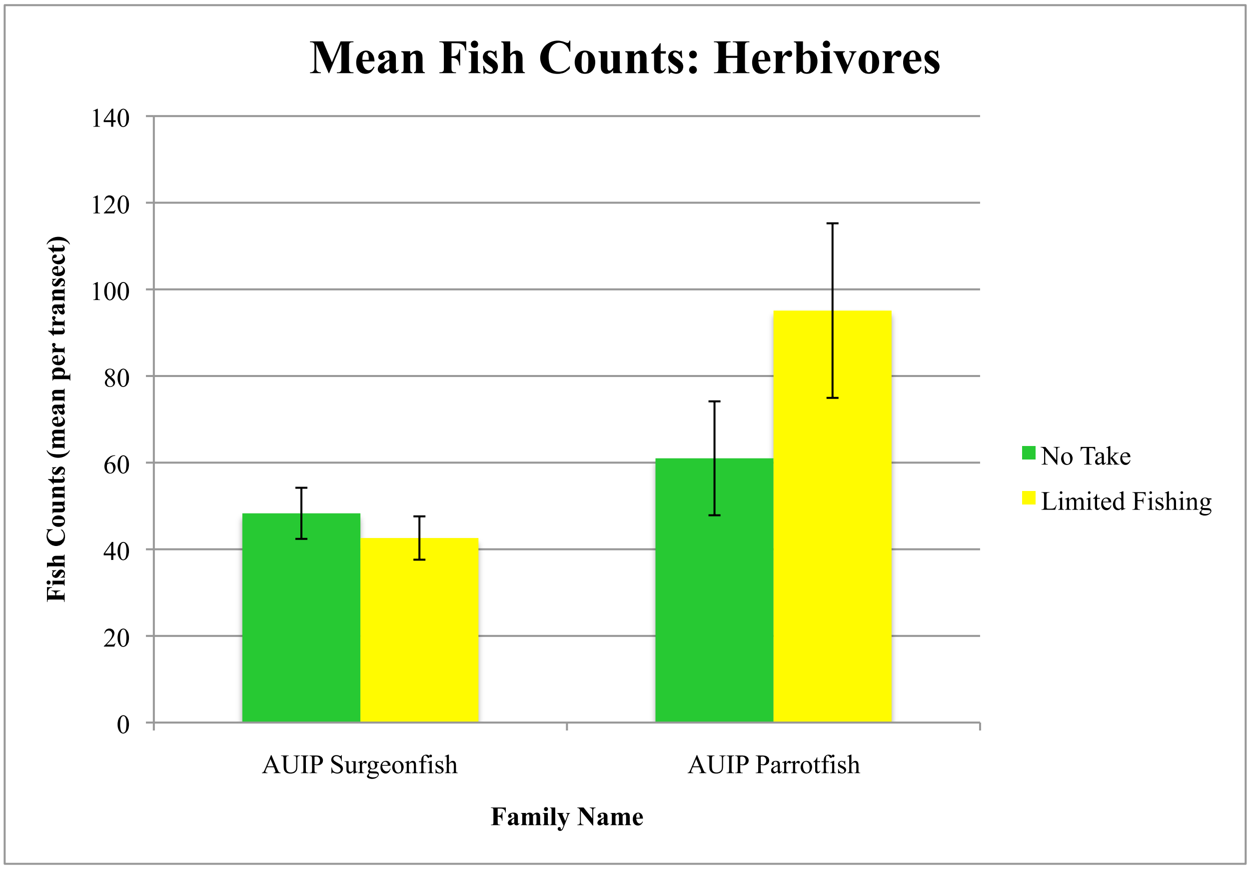


Fig 8. Mean (±SE) counts of herbivorous fish by family in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. No differences between zones are significant.

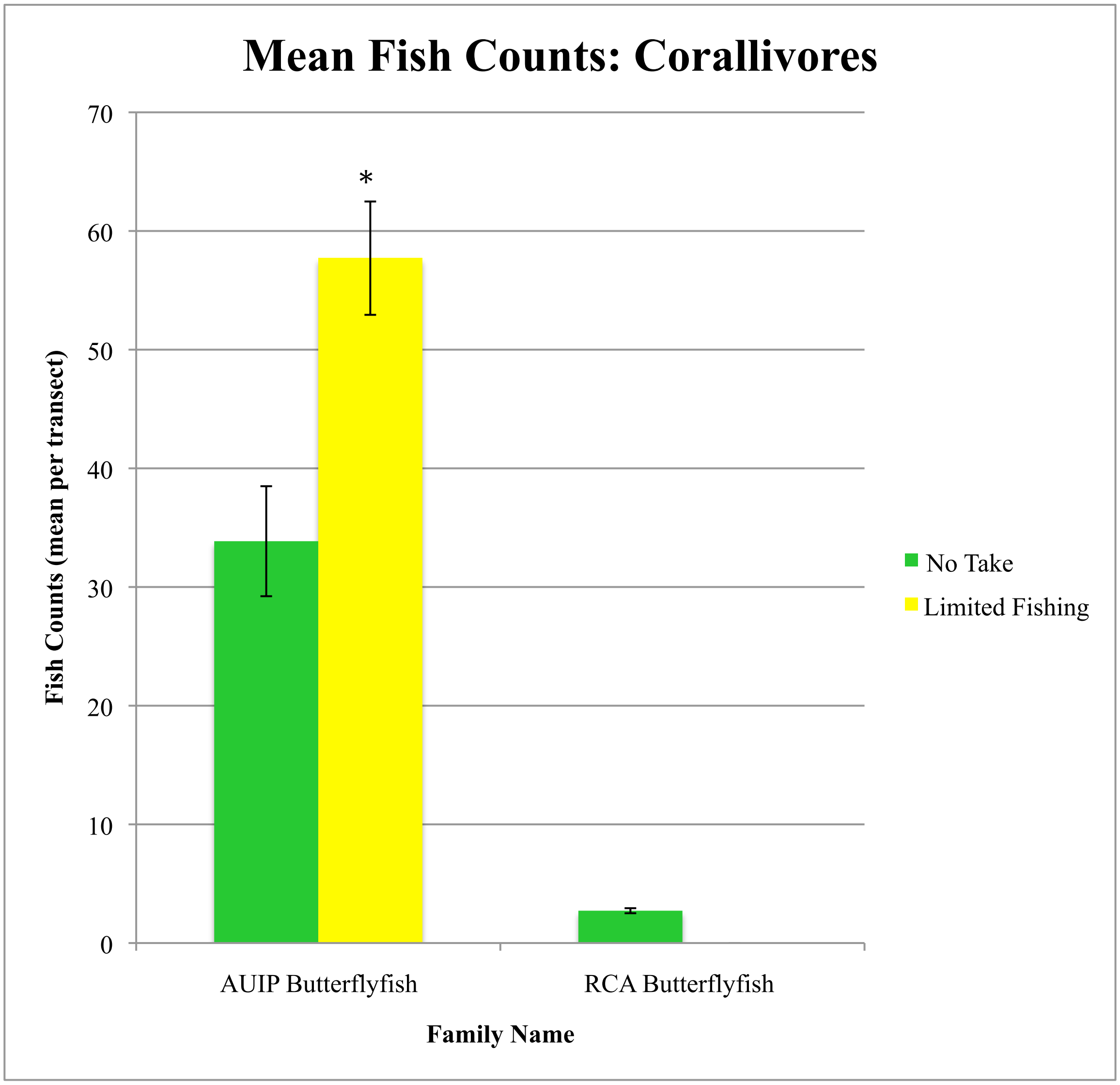


Fig 9. Mean (±SE) counts of corallivorous fish in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. Asterisks indicate weak significant differences between zones (P<0.10).

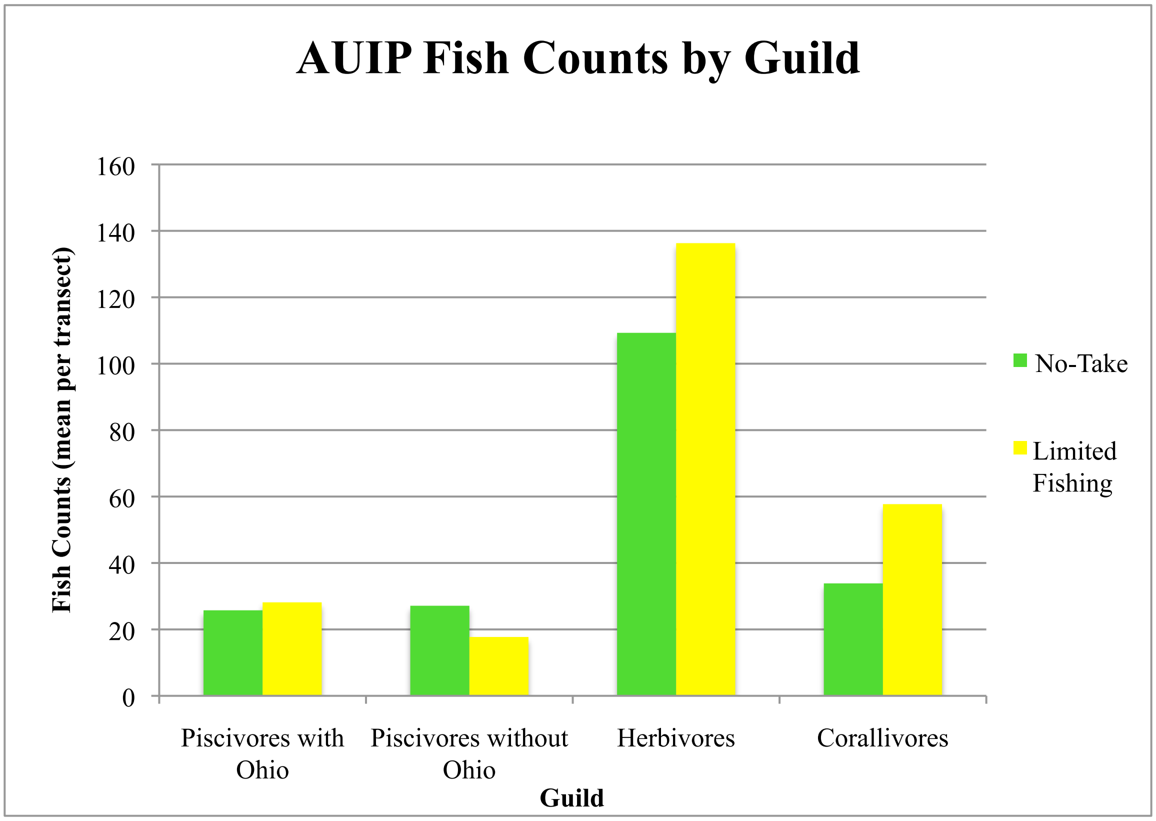


Fig 10. AUIP sum of mean fish counts for all families and species for each feeding guild (with and without Ohio) in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. No differences between zones are significant.

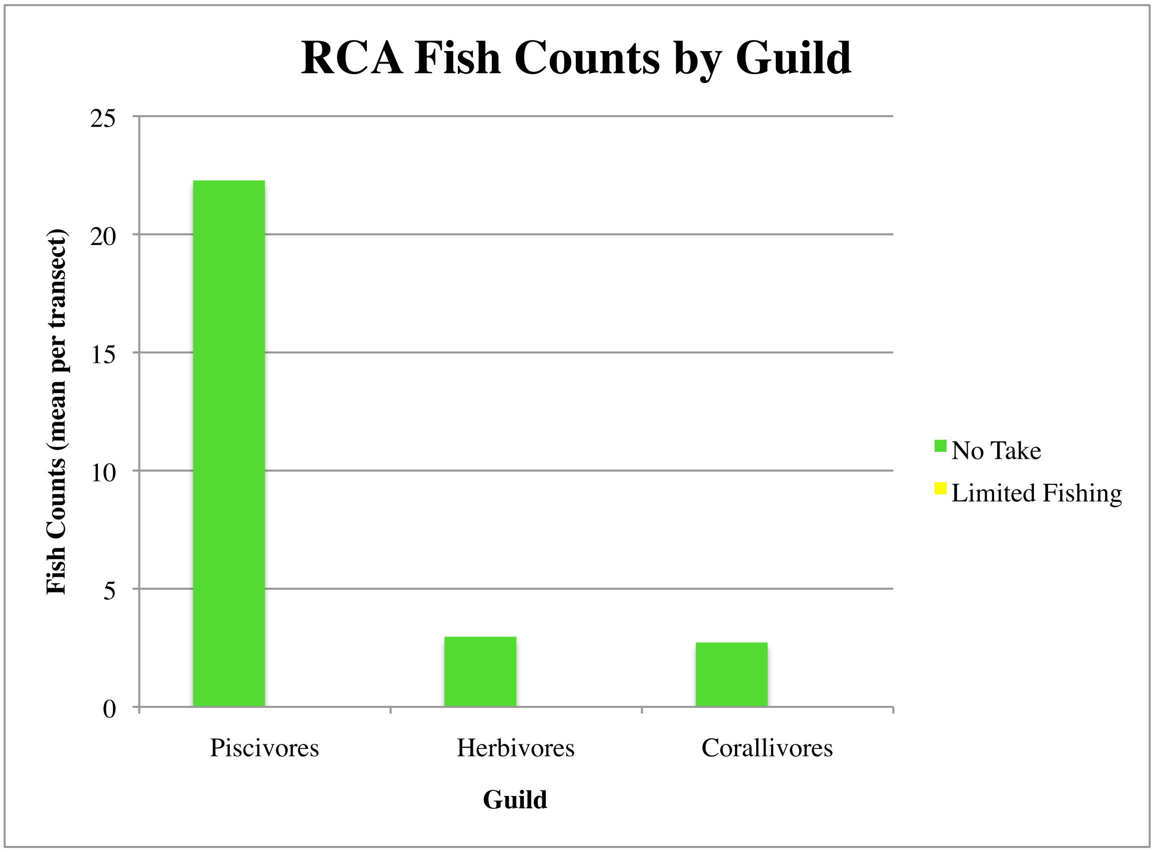


Fig 11. RCA sum of mean fish counts for all families and species for each feeding guild, in no-take (green) zones and limited-fishing (yellow) zones on Opal Reef, Great Barrier Reef. No differences between zones are significant.

Appendix E: Activity Regulations by Zone

Table 4: Activity Regulations for Conservation Zones

Activity	**Permitted?**
Aquaculture	Permit
Bait netting	Yes
Boating, diving, photography	Yes
Crabbing (trapping)	Limited
Harvest fishing for aquarium fish, coral and beachworm	Permit
Harvest fishing for sea cucumber, trochus, tropical rock lobster	No
Limited collecting	Yes
Limited impact research	Yes
Limited spearfishing (snorkel only)	Yes
Line fishing	Limited
Netting (other than bait netting)	No
Research (other than limited impact)	Permit
Shipping (other than a designated shipping area)	Permit
Tourism program	Permit
Traditional use of marine resources	Permit (or TUMRA)
Trawling	No
Trolling	Yes

Source: GBRMPA

Table 5: Activity Regulations for No-Take Zones

Activity	**Permitted?**
Aquaculture	No
Bait netting	No
Boating, diving, photography	Yes
Crabbing (trapping)	No
Harvest fishing for aquarium fish, coral and beachworm	No
Harvest fishing for sea cucumber, trochus, tropical rock lobster	No
Limited collecting	No
Limited impact research (non extractive)	Yes
Limited spearfishing (snorkel only)	No
Line fishing	No
Netting (other than bait netting)	No
Research (other than limited impact-non extractive)	Permit
Shipping (other than a designated shipping area)	Permit
Tourism program	Permit
Traditional use of marine resources	Permit (or TUMRA)
Trawling	No
Trolling	No

Source: GBRMPA