

## **Analysis of Large Predator Populations in the Line Islands**

Delamon A. Rego, Stanford University  
Christopher M. Hanson, Stanford University

### ABSTRACT

Human disturbance and reef health are inextricably linked. Specifically, the biomass of predators in a marine ecosystem has been identified as an indicator of overall reef health. Less affected reefs have been shown to have a significantly high percentage of predator biomass relative to the biomass of other fish. Our research examines this trend, observing individual species of predators and accounting for their biomass, species diversity, and functional diversity. All research was conducted in the Pacific Line Islands along a gradient of human disturbance throughout the islands. By using both a line transect and a point count we were able to compile two independent data sets. Both data sets were comparable and indicated that more heavily impacted reefs have lower predator biomass, less predator species diversity, and fewer functional groups represented. With this data we are better able to determine which functional groups are threatened the most and therefore deduce which species face the highest risk of extinction in the respective regions. Furthermore, previous biomass research is reinforced by our findings.

## INTRODUCTION

Anthropogenic factors such as overfishing, pollution, and habitat degradation have taken an increasing toll on coral reef health worldwide (Pandolfi et al., 2003). Due to their long life spans, low fecundity, late age of sexual maturity, and position as the focus of commercial fisheries, predators at the top of marine food webs tend to be most affected by these detriments (Musick et al., 2000; Robbins et al., 2006). As a result of these pressures, worldwide populations of large predators such as sharks, jacks, groupers and snappers have been reduced to an estimated 10% of pre-industrial levels (Myers and Worm, 2003). A strong negative correlation exists between human impact and biomass density of top predators. While predators make up over 50% of fish biomass on relatively undisturbed reefs, they contribute less than 10% of the biomass of more impacted reef systems. (Stevenson et al., 2007).

Selective fishing pressure has been shown to have a significant indirect effect on the overall structure and health of a reef through trophic cascades (Stevens et al., 2000; Pinnegar et al., 2000). A trophic cascade occurs when the removal of one trophic group, typically at the top of the food web, causes an increase in the population of the trophic group directly below it. This increase may, in turn, cause a decrease in the biomass of the subsequent trophic level. The continuation of such effects throughout the food web may ultimately disrupt the healthy functioning of the reef. While several studies (Stevens et al., 2000; Pinnegar et al., 2000) support the claim that predator reduction can lead to trophic cascades, research in Fijian reefs produced results in which there was no significant correlation between total predator biomass density and prey biomass density (Jennings and Polunin, 1996). Thus, trophic cascades cannot be understood by simply studying interactions between trophic levels.

Instead, relationships among functional groups consisting of species occupying similar niches in a food web must be studied in order understand interactions within a food web (Micheli and Halpern, 2005). We suggest that the selective removal of large apex predators can induce negative effects that reverberate throughout a food web through the loss of some functional groups and a reduction in the redundancy (the total number of species in a group) of others.

A correlation exists between animal biomass and species diversity, with high-biomass reefs exhibiting increased species richness and functional diversity (Worm et al., 2006; Micheli and Halpern, 2005). These findings indicate that high-biomass reefs will exhibit greater species diversity and, as a result, more functional diversity than low-biomass reefs. However, while high biodiversity is associated with high functional diversity, selective fishing places even highly diverse reefs in danger of losing specific, underrepresented functional groups (Bellwood et al., 2003). Loss of functional diversity has been shown to underlie low resilience to human pressures and natural disturbances in Caribbean coral reef systems (Micheli et al., 2007), and may have the same effect in the Pacific.

Interaction between species in a food web can be characterized as strong or weak. A weak interaction indicates a secondary predator-prey relationship in which change in one population is not influenced by change in the other. Conversely, a strong interaction is a relationship between a single predator species and a prey species in which a significant change in one *can* lead to change in the other. A study of a Caribbean food web showed that 48% of the strong interactions were accounted for by 10 species of top predators (Bascompte et al., 2005). This demonstrates that while a diverse food web is buffered

against the removal of a random species from a functional group with high redundancy, it is sensitive to selective removal of top predators by overfishing.

Although predator biomass and functional groups within various Caribbean food webs have been identified, no published research had linked biomass to functional diversity in the Central Pacific or analyzed the region's top functional groups. Our research aims to bridge this gap. Our research was conducted in the Line Islands, which are all located within the Inter-Tropical Convergence Zone (ITCZ) and dominated by the NECC. Stevenson et al. (2007) studied the percentage of biomass comprised of high-apex predators throughout the island chain. We want to delve further into the region's ecology by studying the relationship between human impact and biomass, species diversity and functional diversity found in the island chain. The reefs in the chain we studied were located at Christmas Island, Washington Island, Palmyra Atoll and Kingman Reef. The physical properties of the sea are relatively similar throughout the island chain, while human populations and their impacts differ greatly. Thus, it is fair to assume that the majority of biological variance between the reefs can be attributed to anthropogenic factors. The islands are therefore well-suited for a comparative scientific study of anthropogenic influence on predator population structures.

The methodology employed in the study of large predators has varied significantly. While stationary point counts were used by Jennings and Polunin (1996), mobile line transects of fixed lengths were used by Stevenson et al. (2007). The merits and drawbacks of each had not been explored, and it was unknown which method might have been best-suited for ecological fieldwork concerned with the identification of large predators. No study had

combined these methods and compared the data sets obtained by each sampling technique. This study employed both techniques in order to do so.

It is known that an inverse relationship exists between human impact and fish biomass density in the Line Islands, but does this correlation apply to species and functional diversity as well? How does the specific depletion of large predators alter functional diversity within a food web and affect the reef as a whole? In addition, how does continued fishing pressure affect biomass over time? During our surveys of Line Island predators, considering the constant fishing pressure on Christmas Island compared to the continued protection of Palmyra Atoll, we expected to see less biomass density on Christmas Island and similar biomass at Palmyra Atoll compared to data recorded by Stevenson et al. (2007). We expected to record increased diversity in the less affected reefs of Palmyra Atoll and Kingman Reef in comparison to Christmas Island and Washington Island. We also expected to see more species per functional group as well as more total functional groups on Palmyra Atoll and Kingman Reef than on Christmas Island and Fanning Island. Moreover, our use of multiple testing procedures procured varying data sets for each method used, which were analyzed and compared in order to determine which testing procedure was most effective for accurately surveying large predators.

## METHODS

One to three point counts and line transects surveying large piscivorous fish (families Myliobatidae, Carcharhinidae, Lethrinidae, Serranidae, Muraenidae, Carangidae, Sphyrnidae, and Lutjanidae) were taken at five stations on Christmas Island and Palmyra Atoll, and three stations on Washington Island and Kingman Reef (Table 1). In total, 15

surveys were taken at Christmas Island, 12 at Palmyra Atoll, six on Washington Island, and three at Kingman Reef. The stations at Christmas Island and Palmyra Atoll had depths ranging from one-half to eight meters and were the same locations that were tested by Stevenson et al. (2007). Locations with conditions similar to those tested at Palmyra Atoll and Christmas Island (shallow and protected backreef areas) were chosen for Kingman Reef and Washington Island. When possible, all surveys were undertaken by D.R. and C.H., but other trained observers were used at Kingman Reef and certain sites at Palmyra Atoll.

For each line transect, two snorkelers swam shoulder-to-shoulder recording the number and size (to within five cm) of observed predator species while unreeling a 100-m belt behind them. Snorkelers were previously trained to gauge lengths underwater to within five cm. For each transect, four meters were observed on either side of the belt. The starting point for the belt transect was chosen by swimming thirty-five fin strokes off the bow of the transport vessel. The belt was then laid out parallel to the shore. These precautions served both to ensure observation of an environment undisturbed by the transport vessel and also to help reduce human bias in choosing transect location and direction. To avoid counting the same fish twice, families were divided between the two observers, with each observing and recording only their selected predators. All transects were placed randomly at each site and never in continuity to ensure samples independent of replication. Methods were maintained similar to those used by Stevenson et al. (2007), to facilitate comparison.

The center location for point transects was determined by reeling the belt back to the 10-m mark. A snorkel team stationed at this point viewed a cylindrical water column with a radius corresponding to the 10-m distance to the end of the transect belt. For sites with visibility

of <10-m, the maximum observable distance was used for the radius. All fish in the region were observed over a ten-minute period, with the same features examined by scientists testing along the belt transects being noted by those observing at the point transect. Again, each snorkeler identified different species so as to help prevent counting the same fish more than once. The date, time, depth, visibility observer's name, transect number, island, and site number were recorded on the observer's data sheet. Time of observations for both methods was flexible, ranging from morning to late afternoon.

Biomass was calculated using the length (L)—weight (M) equation:

$$M=aL^b.$$

to convert estimated fish lengths into an approximate measurement of biomass. Constants (a, b) vary between species and were obtained from FishBase. Densities were measured by dividing the observed biomass in grams by the total area surveyed in square-meters, adjusted for visibility. Differences in biomass densities between the islands were analyzed using a Kruskal-Wallis Nonparametric One-Way Analysis of Variance. Due to software limitations, graphically depicted 95% confidence intervals were used instead of a pair-wise analysis to qualitatively estimate significance.

Species diversity was described by using our data to calculate the total number of predator species observed in a reef along with each species' proportion of the total count. These two numbers were then entered into the equation for the Shannon-Weiner diversity index, which states that:

$$H = \sum_{x=1}^S p_i \ln p_i$$

where  $H$  is the Shannon-Weiner diversity index,  $S$  is the total number of species in the community, and  $p_i$  is the proportion of  $S$  made up of the  $i$ th species. This index takes into account both species richness (described by  $S$ ) and evenness of distribution (described by  $p_i$ ) to provide an accurate measure of total diversity. Mean diversities were calculated for each island. To analyze the differences in diversity between the reefs, we conducted a one way ANOVA for diversity with a Tukey HSD Multiple Comparison post-hoc test. We tested ANOVA for statistical assumptions and found that variances were equal using Cochran's test ( $p < 0.05$ ). The assumption of normality was not met by the data, but ANOVAs are robust to violations of normality, so we proceeded with the test using a skewed distribution.

Functional groups were chosen based upon the groups used by Micheli and Halpern (2005). Four primary groups were chosen based upon feeding behavior, with each group being further divided into three size categories. The four feeding categories were benthic invertebrate feeders, benthic invertebrate feeders and benthic piscivores, benthic piscivores, and benthopelagic piscivores. Fish were placed into their respective feeding groups based upon diet information obtained from FishBase and *Randall*. The size categories were predators 30-50cm, 50-100cm, and >100cm long. Fish were placed into size categories based upon maximum size as obtained from FishBase. A master list of all functional groups used and the species included in each was then compiled (Table 2). Species information for each island studied was then compared to this master list in order to determine the functional diversity of the reef. Comparisons were made based upon the number of functional groups represented per island as well as the mean redundancy for functional

groups shared between Christmas Island and Palmyra Atoll (the two reefs with the greatest number of samples). Due to the small sample size, an ANOVA test was not taken, and thus significance was not observed for this comparison.

The relative effectiveness of point counts and line transects for gauging biomass were evaluated by comparing the mean densities across all surveys undertaken for each method on Christmas Island and Palmyra Atoll. The differences were analyzed for significance using a single-factor ANOVA test. The relative abilities of the two methods to gauge species richness were determined by qualitatively comparing cumulative species curves across transects taken at Christmas Island and Palmyra Atoll. Each curve included the number of unique species observed on each island as more the number of surveys progressed.

Estimates of human impact were made using available quantitative and qualitative information about each island's population density and fishing pressure. According to word-of-mouth information from immigration officials in the Republic of Kiribati, populations for Christmas Island and Washington Island are ~9,000 and ~1,000, respectively. Washington Island is much smaller with a smaller percentage of reef area and, being more remote, we estimate that it likely relies more on its reef for subsistence. Thus, we propose that the human impact per reef area is greater at Washington Island than at Christmas Island.

Palmyra Atoll and Kingman Reef are both Marine Protected Areas, with Palmyra Atoll having a population of about 20 residents and Kingman Reef having no human presence. Using this information, we assigned the reefs various impact levels with Kingman Reef being the least impacted, followed by Palmyra Atoll, Christmas Island, then Washington Island.

## RESULTS

In the Line Islands a negative relationship was seen between human impact and biomass, species richness, and functional diversity. Mean biomass densities observed at Palmyra Atoll and Kingman Reef ( $101.1\text{g}/\text{m}^2 \pm 29.17$  (SE) and  $91.7\text{g}/\text{m}^2 \pm 52.35$  (SE), respectively) were higher than those observed at Christmas Island and Washington Island ( $7.8\text{g}/\text{m}^2 \pm 2.00$  (SE) and  $0.2\text{g}/\text{m}^2 \pm 0.08$  (SE), respectively (Fig. 1). The mean densities varied significantly from each other (Kruskal-Wallis  $P=0.000$ ) and decreased as human impact levels increased. Compared to data from Stevenson et al. (2007), mean biomass density was 15% of 2005 levels on Christmas Island and 35% of 2005 levels on Palmyra Atoll (Fig. 2). When sharks were removed from the comparison, the figures for Palmyra Atoll were 81% of 2005 levels. This is because their relatively low numbers and large size can cause large variation in biomass estimates when encountered in a transect.

Like biomass, species diversity was lower in reefs with greater human impact. Washington Island, Christmas Island, Palmyra Atoll, and Kingman Reef had mean Shannon-Weiner diversity indices of  $0 \pm 0$  (SE),  $0.24 \pm 0.12$  (SE),  $1.12 \pm 0.22$  (SE), and  $1.21 \pm 0.47$  (SE), respectively (One Way ANOVA for diversity with Tukey HSD Multiple Comparison post hoc test (Table 3)) (Fig. 3). These figures differed significantly and appeared to be related to the varying impact levels of the islands. All reefs with a large number of fish (Christmas Island, Palmyra Atoll, and Kingman Reef) were dominated by the family Lethrinidae, which comprised 56%-85% of observed predators (Fig. 4-6).

Accordingly, functional diversity was reduced on reefs associated with greater human impact. Human impact had a negative correlation with both the number of represented functional

groups, as well as the number of species per functional group. Palmyra Atoll, Kingman Reef, Christmas Island, and Washington Island were represented by six, four, three, and one functional group respectively (Fig. 7). At Palmyra Atoll and Christmas Island (the only two reefs at which the number of transects was roughly comparable), the mean redundancy of the three functional groups represented at both islands was 3.3 and 3.0 species per group, respectively (Fig. 8). Functional groups represented at Palmyra Atoll but missing at Christmas Island were invertebrate feeders 30-50cm, invertebrate feeders 50-100cm, and benthic piscivores 50-100cm.

Estimates of predator biomass and species richness were similar in data sets obtained by line transects and point counts. At Christmas Island, average predator biomass densities were  $8.3 \text{ g/m}^2 \pm 2.5(\text{SE})$  and  $6.7 \text{ g/m}^2 \pm 3.5(\text{SE})$  for line transects and point counts, respectively. On Palmyra Atoll, the respective figures were  $63.6 \text{ g/m}^2 \pm 51.1(\text{SE})$  and  $138.6 \text{ g/m}^2 \pm 81.1(\text{SE})$  (Fig. 8). At both reefs, the differences were not significant (ANOVA  $P=.73$  and  $.46$ , respectively). Total number of species observed was also similar for both line transects and point counts at Christmas Island and Palmyra Atoll. On Christmas Island, seven total species were observed using each survey method. The cumulative species curves were qualitatively similar for both methods as well (Fig. 9). At Palmyra Atoll, ten species were observed during point counts compared to nine observed through line transects. Again, the running count of species observed as more surveys were included was similar for both methods of observation (Fig. 10).

## DISCUSSION

Our findings indicate that there is, in fact, a negative relationship between human disturbance and predator biomass, predator species diversity, and predator functional diversity. For instance, Palmyra Atoll, which experiences no fishing pressure, had a mean predator biomass approximately thirteen times greater than that of Christmas Island, a region with substantial fishing pressure. In addition, Palmyra proved to be five times more diverse than Christmas Island. Six functional groups were represented on Palmyra, while three were present on Christmas Island. Combined with the fact that functional groups tended to be represented by fewer species on more impacted reefs, these findings indicate that there is higher functional diversity on reefs with less human disturbance. Due to their higher diversity among predatory functional groups, these ecosystems are likely more robust and less susceptible to trophic cascade due to predator removal. Findings for Kingman Reef, at which only three transects were taken, were likely skewed lower and it is reasonable to assume that had more time been spent surveying the reef, a functional diversity similar to Palmyra Atoll would have been observed. If true, this would further support the notion that less impacted reefs exhibit higher functional diversity.

On all reefs examined, snappers (Lutjanidae) dominated. Snappers accounted for 56 percent of all predators observed on Christmas Island, 59.3 percent at Palmyra Atoll, and an astounding 85 percent at Kingman Reef. The one exception was Washington Island, where the only predator observed that was greater than thirty centimeters in length was a 30cm foursaddle grouper (*Epinephelus spilotoceps*).

An objective of this experiment was to compare our data with that of Stevenson et al. (2007) from 2005. Both groups sampled many of the same sites at Christmas Island and Palmyra Atoll. We hypothesized that predator biomass on Christmas Island would be less in 2007 due to two more years of continuous fishing pressure, while predator biomass on Palmyra would remain constant as a result of relatively little human disturbance since 2005. Although the Christmas Island data supports our hypothesis by describing a substantial drop in biomass over a two-year period, the figures for Palmyra Atoll, at which fishing pressure is nonexistent, are contrary to what was expected. A significant proportion of this discrepancy, however, can be explained by accounting for the variability of sharks, showing that predator biomass increased at a far greater rate on Christmas Island than at Palmyra Atoll. It is still unclear, however, what other factors besides fishing pressures may be contributing to the downward trajectory of large, predatory fish populations. A few discrepancies existed between methodologies employed by our study and Stevenson et al. (2007) that might have further skewed the data. We observed predators 30cm and above, while Stevenson et al. recorded predators less than 30cm. Thus, their data set should inherently contain more fish per square meter than ours. However, we were also counting more types of fish than Stevenson et al. (2007). We identified all predators 30 cm and above and classified them to the species level. Stevenson et al. (2007) classified predators at a higher taxonomic level (sharks, jacks, snappers, and groupers) and did not account for other predators like emperors and rays.

In addition to surveying different groups of predators, transect specifications also differed. Stevenson et al. (2007) used 50 meter by 4 meter line transects, whereas we utilized 100 meter by 8 meter line transects, as well as a point count with a radius of 10 meters. Point

counts, however, were not included in biomass comparison so as to not confound data. Our decision to use the point count was extrapolated from their data, which suggested that timed transects may be more suitable for viewing large, mobile fish like sharks (Stevenson et al., 2007). However, our results did not support this, and less total predator biomass was observed overall using both line and point transects.

While there are disparities between our data set and Stevenson et al.'s (2007) data set, closer examination reveals much more similarity. Namely, the presence of sharks may greatly skew biomass density estimates. The abundance of sharks at Palmyra in 2005 and the dearth of sharks in 2007 is one such example. Sharks at Palmyra accounted for nearly 76 percent of the biomass density data collected by Stevenson et al. in 2005. If this factor were removed, mean predator biomass density at Palmyra in 2005 would have been  $71.548 \text{ g/m}^2$ , compared to  $56.342 \text{ g/m}^2$ , which is the mean predator biomass density at Palmyra in 2007 if the sharks are removed from the data. These figures are much more comparable.

If sharks are the primary cause of such variability between biomass density figures for Palmyra in 2005 and 2007, one would still be prompted to question why sharks were apparently so much more abundant in 2005 than in 2007. The most logical solution to this quandary, is that the sample size is too small to accurately estimate the abundance of highly mobile predators such as sharks, both in Palmyra and throughout the Line Islands. Thus, it is reasonable to assume that while biomass has declined slightly at Palmyra Atoll, it has declined at a much greater rate at Christmas Island.

This experiment can act as a springboard for anyone interested in reef dynamics or predator populations in the central Pacific Islands. If possible future studies should include an increased sample size to reduce variability. Similar islands could also be tested in the same way for comparison with our data set. To compliment our work with predators, a related study that analyzes herbivorous fish populations would be useful to examine how lower trophic levels are affected by human disturbance. This would be especially useful with regard to support or rejection of the importance of functional diversity.

This study provided important information on the overall status of predator populations in the Line Islands and how these populations are affected by varying levels of human disturbance. At the top of the food web, predators play an irreplaceable role as stewards of the reef. Our study further supports the idea that human impact degrades coral reef ecosystems. Predators are particularly affected since they are often targeted by fisheries and are also frequently slaughtered as bycatch. These animals are a necessary component of a sustainable reef, and their sizes and relative abundance are key factors that denote a sound ecosystem.

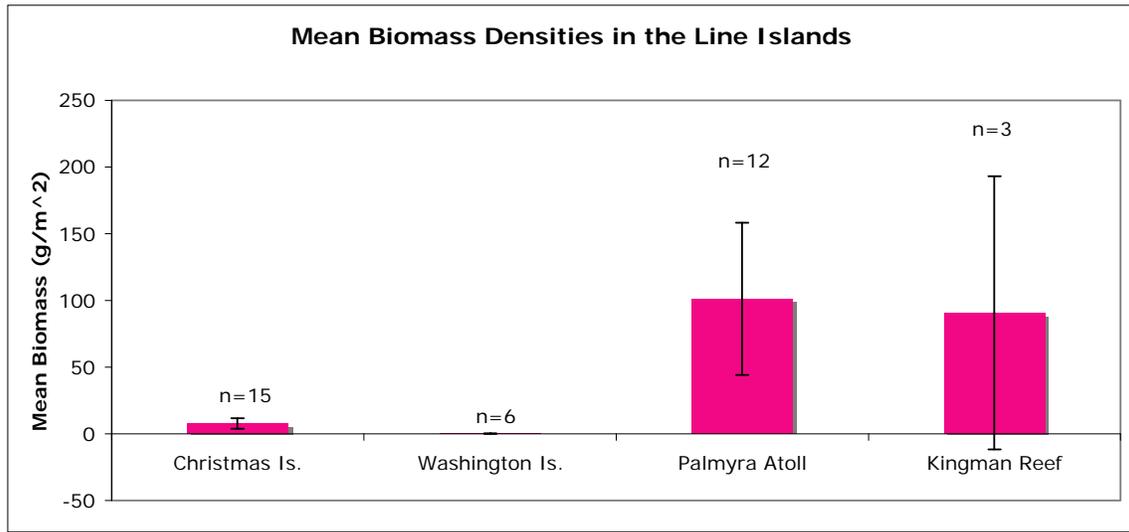
## CONCLUSION

Extended, selective fishing pressure can induce a large decrease in predator biomass over time. As human impact increases along a gradient, predator biomass, species diversity, and functional diversity decrease. Both the number of functional groups represented and the number of species per group are likely reduced by anthropogenic factors. Further research is necessary in order to ascertain with certainty the functional groups most easily affected by human disturbance. It also remains unknown how the removal of these functional groups

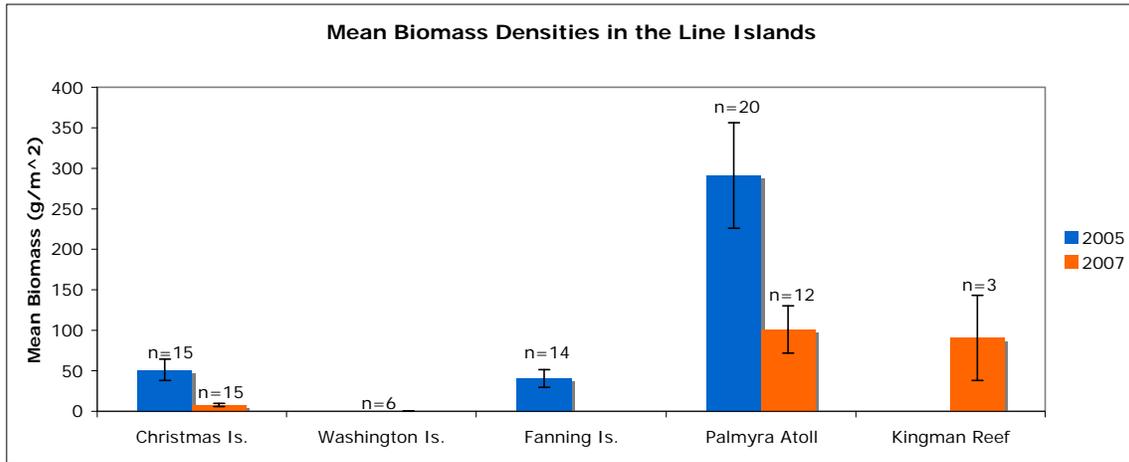
might influence overall coral reef structure in the Line Islands. However, the fact that entire functional groups can be removed through such selective fishing pressure is indicative that certain key aspects of a healthy reef system may be endangered by human activities.

Additionally, our research shows that point counts and line transects are equally effective at measuring both biomass and species diversity for large predators in the Line Islands. Thus, future studies of the region's coral reef ecology may use either method depending on situational or personal preferences without risking diminished quality of the data collected.

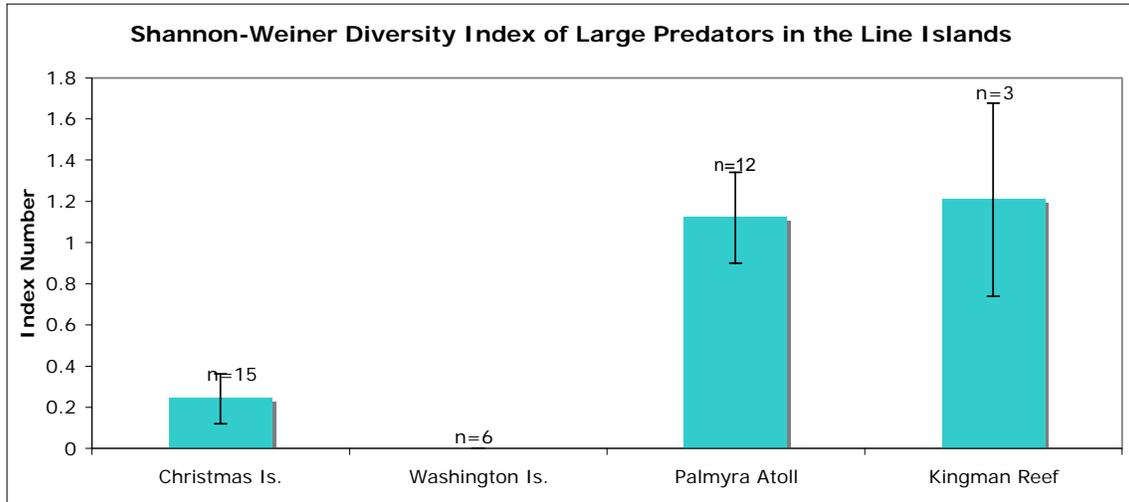
FIGURES



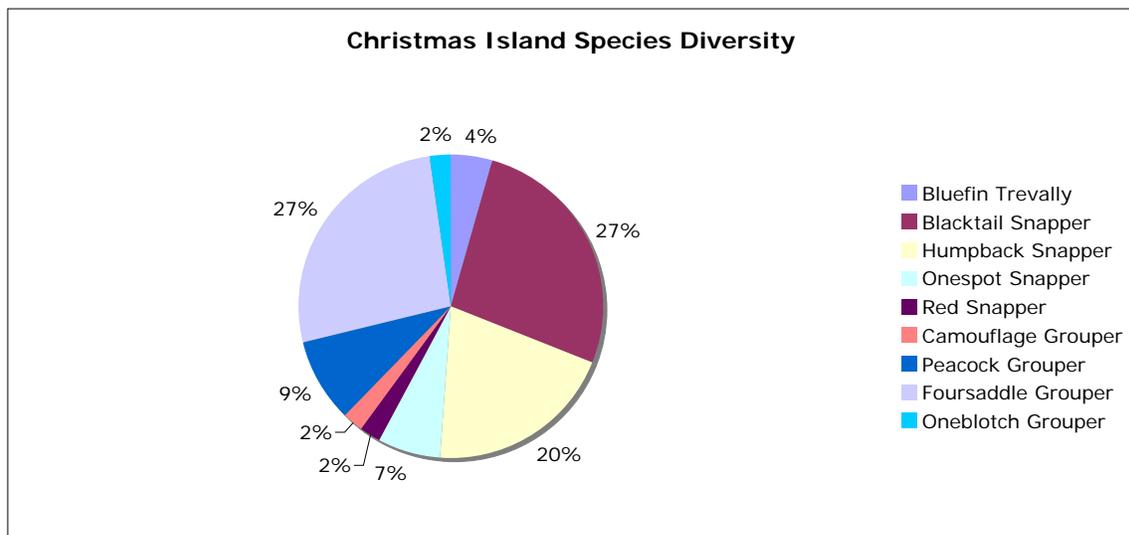
**Figure 1.** Mean biomass densities in the Line Islands with 95% confidence intervals depicted. May 2007, Cruise S211. Thirty-six total transects (n).



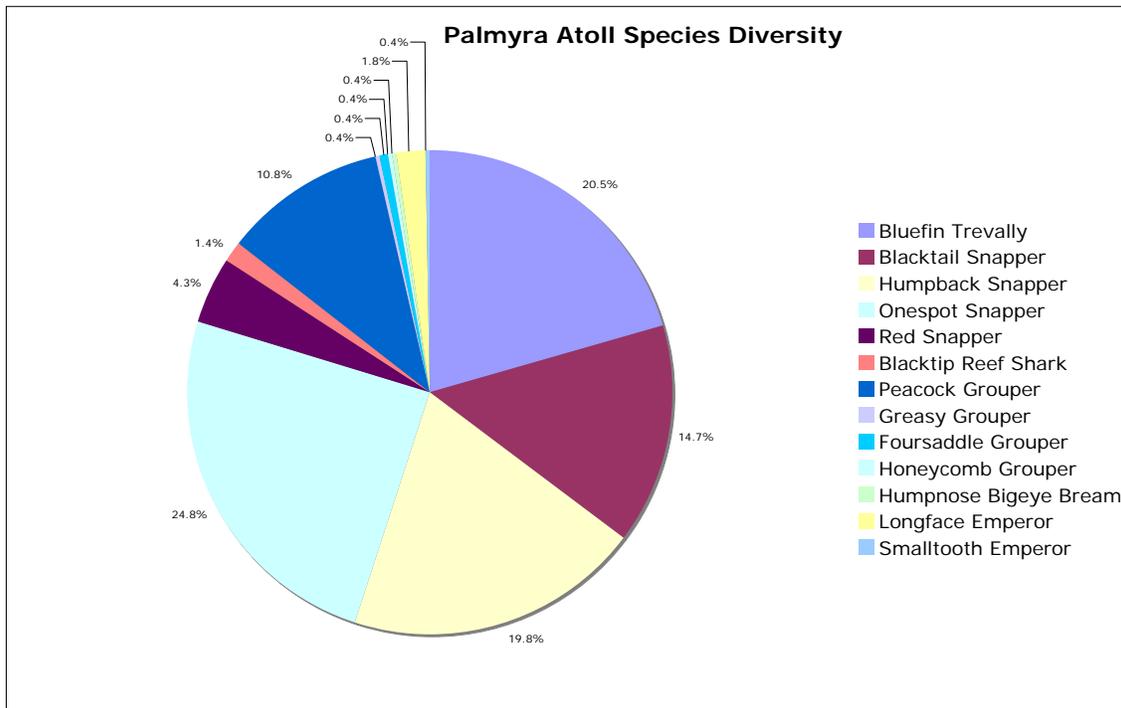
**Figure 2.** A comparison of mean biomass densities (+/- SE) in the Line Islands observed in May 2005 (Cruise S199) and May 2007 (Cruise S211). Christmas Island, Fanning Island, and Palmyra Atoll were examined by Stevenson et al in 2005 (49 total transects), Christmas Island, Washington Island, Palmyra Atoll, and Kingman Reef examined by Rego and Hanson in 2007 (36 total transects).



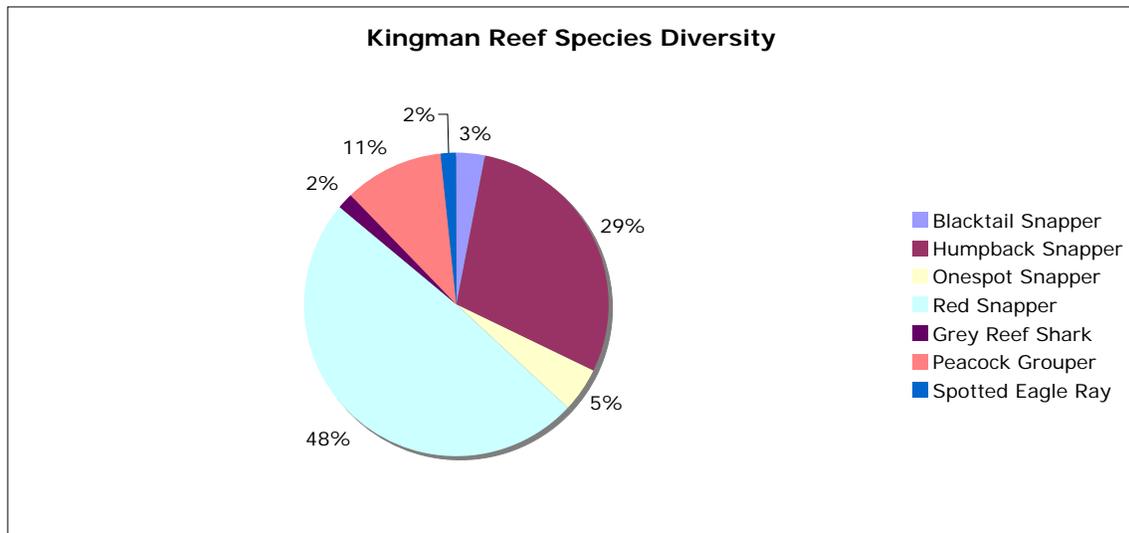
**Figure 3.** Mean Shannon-Weiner diversity index (+/- SE) of large predators in the Line Islands, May 2007, Cruise S211, based on thirty-six total transects (n). A higher index number denotes greater diversity.



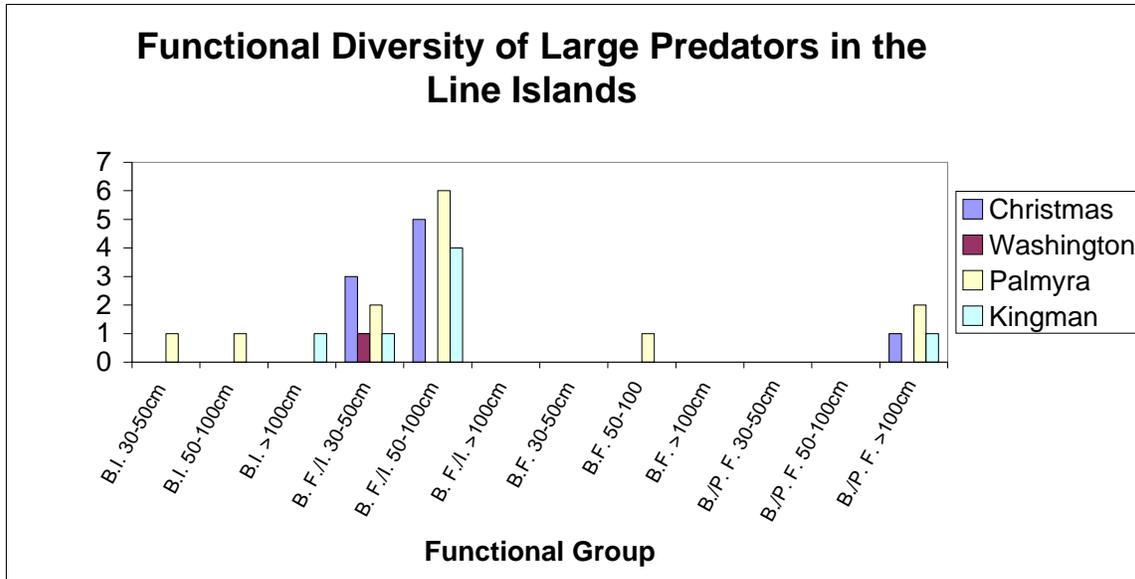
**Figure 4.** Percentages of species observed in Christmas Island transects (cumulative including both line and point). Cruise S211, May 2007.



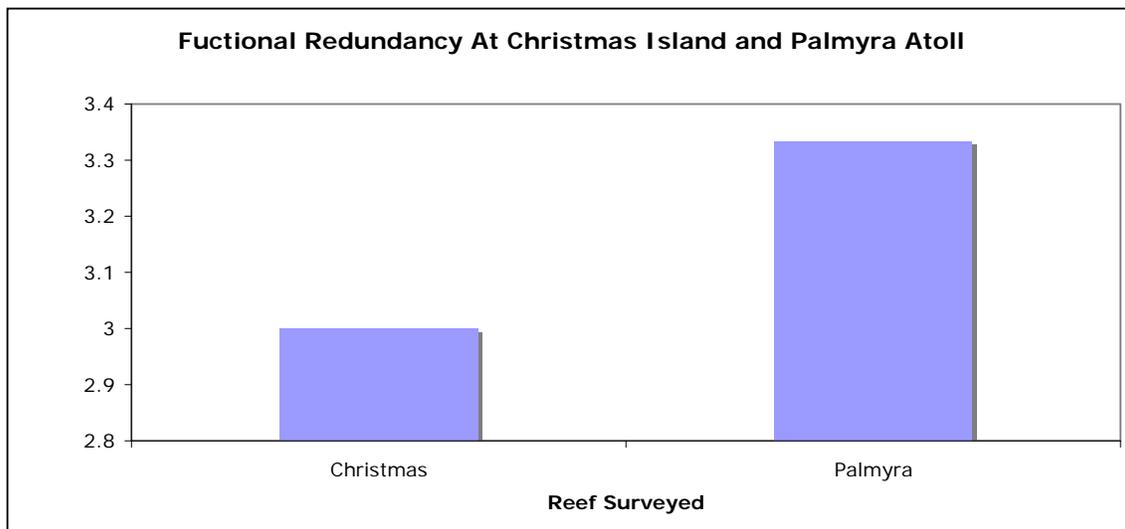
**Figure 5.** Percentages of species observed in Palmyra Atoll transects (cumulative including both line and point). Cruise S211, May 2007.



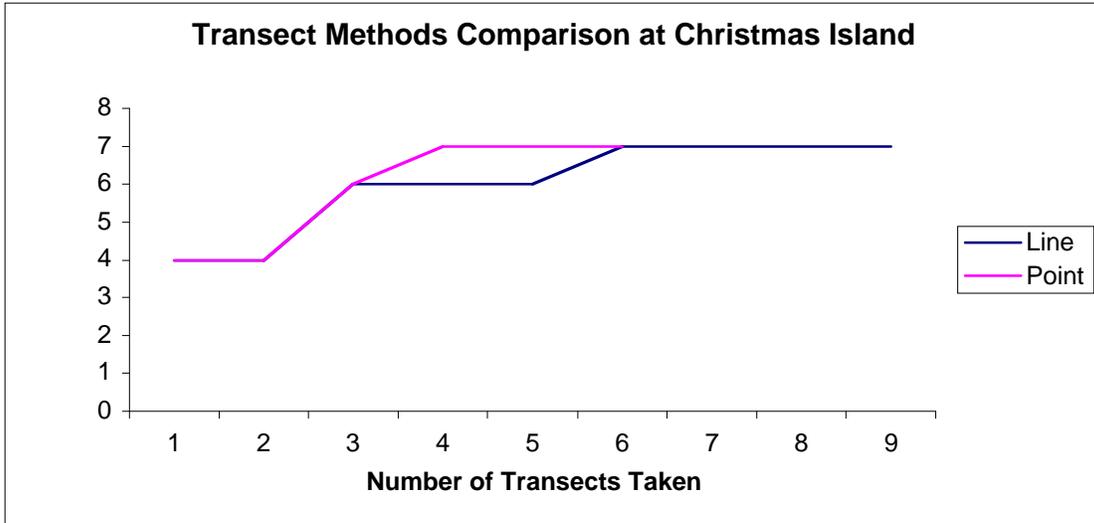
**Figure 6.** Percentages of species observed in Kingman Reef transects (cumulative including both line and point). Cruise S211, May 2007.



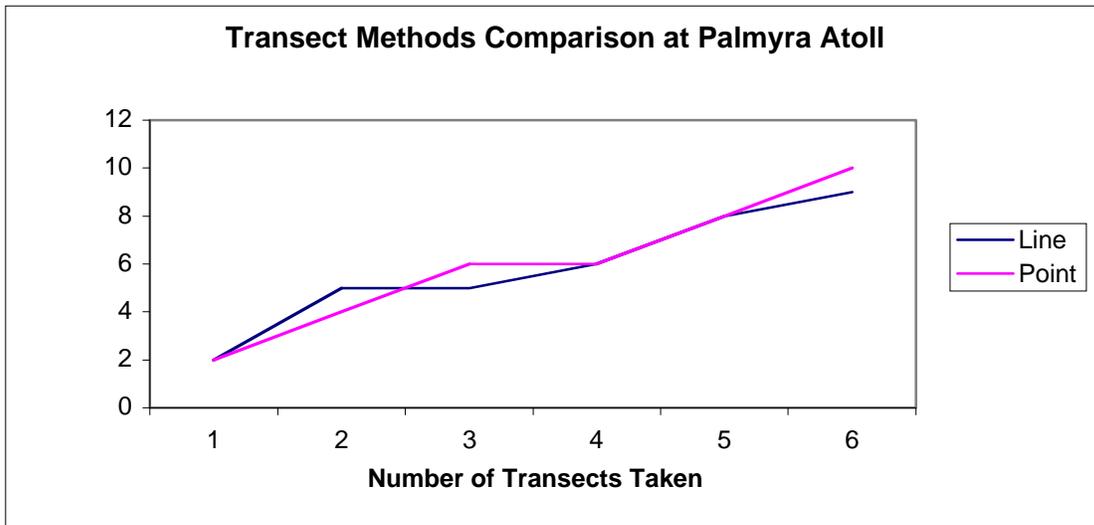
**Figure 7.** Functional diversity of large predators observed in line and point transects combined. Functional groups represented: B.I. (consumers of benthic invertebrates), B.F./I. (consumers of benthic fish and benthic invertebrates), B.F. (consumers of benthic fish), and B.P./F. (consumers of benthic and pelagic fish). Cruise S211, May 2007.



**Figure 8.** Average number of species representing each functional group shared by Christmas Island and Palmyra Atoll. Error bars are not included due to the small sample size of n=3. Cruise S211, May 2007.



**Figure 9.** Cumulative species curves comparing point and line transects at Christmas Island. Number of species is cumulative; as the number of transects increases, the number of species from previous transects is retained. Cruise S211, May 2007.



**Figure 10.** Cumulative species curves comparing point and line transects at Palmyra Atoll. Number of species is cumulative; as the number of transects increases, the number of species from previous transects is retained. Cruise S211, May 2007.

TABLES

**Geographical Coordinates**

Christmas	Washington	Palmyra	Kingman
1°56.4'N X 157°29.2'W	4°40.2'N X 160°23.0'W	5°52.2'N X 162°6.9'W	6°23.6'N X 162°20.7'W
1°56.4'N X 157°29.4'W	4°40.4'N X 160°23.2'W	5°52.3'N X 162°6.6'W	6°23.5'N X 162°20.3'W
1°57.6'N X 157°29.2'W	4°40.6'N X 160°23.7'W	5°52.6'N X 162°6.9'W	6°23.5'N X 162°20.3'W
1°59.1'N X 157°29.0'W **		5°52.28'N X 162°6.4'W	
1°55.4'N X 157°31.2'W**		4°52.9'N X 162°2.3'W**	

**Table 1.** GPS coordinates of all transect sites in the Line Islands. \*\*Coordinates derived from chart, not GPS. Cruise S211, May 2007.

**Master Functional Groups for the Line Islands**

Invertebrate feeder 30-50cm	Invertebrate feeder 50-100cm	Invertebrate feeder >100cm	Benthic Fish/Invertebrate feeder 30-50cm	Benthic Fish/Invertebrate feeder 50-100cm	Benthic Fish/Invertebrate feeder >100cm
	Humpnose Bigeye Bream	Spotted Eagle Ray	Blacktip Coral Grouper	Longface Emperor	Whitetip Reef Shark
	Snowflake Moraya	Golden Trevally	Coral Grouper	Yellowlip Emperor	Giant Moray
	Blue Trevally	Island Trevally	Foursaddle Grouper*	Camo Grouper	Yellowedged Moray
			Honeycomb Grouper	Peacock Grouper	Giant Trevally
			Oneblotch Grouper*	Yellow Edged Lyretail	Yellow Spotted Trevally
			Snubnose Grouper	Humpback Snapper	
			Blacktail Snapper	One Spot Snapper	
				Red Snapper	

**Master Functional Groups for the Line Islands Cont.**

Benthic Piscivore 30-50cm	Benthic Piscivore 50-100cm	Benthic Piscivore >100cm	Benthopelagic Piscivore 30-50cm	Benthopelagic Piscivore 50-100cm	Benthopelagic Piscivore >100cm
Masked Grouper	Greasy Grouper	Green Jobfish		Blackjack	Blacktip Reef Shark
	Smalltooth Jobfish	Leopard Coral Grouper			Grey Reef Shark
					Tiger Shark
					Bluefin Trevally
					Double Spotted Queenfish
					Rainbow Runner
					Blackfin Barracuda
					Great Barracuda

**Table 2.** Master functional groups for the line islands. Groups chosen based upon feeding habits and maximum recorded size. \*Species with no feeding information; grouped based upon general family trends. Cruise S211, May 2007.

**Matrix of Pairwise Mean Differences for Species Diversity:**

	Christmas	Washington	Palmyra	Kingman
Christmas	0.000			
Washington	-0.258	0.000		
Palmyra	0.864	1.122	0.000	
Kingman	0.952	1.209	0.088	0.000

**Table 3.** Analysis of variation in the observed densities of top predators along the Line Islands. Cruise S211, May 2007.

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