Poor Knights Islands Marine Reserve

Baited underwater video monitoring Winter 2010

New Zealand Government

Department of Conservation
Te Papa Atawhai

Poor Knights Islands Marine Reserve

Baited underwater video monitoring winter 2010

Paul Roux De Buisson

Published by:
Department of Conservation
Northland Conservancy
P.O. Box 842
Whangarei 0140
New Zealand

Review process

This document has been submitted to one internal and one external expert reviewer. Reviewers were selected because of their expertise in marine science and marine conservation. This review process was undertaken to ensure that methodologies and recommendations within this report are robust and in line with best practice. This report has been reviewed by:

Dr Debbie Freeman
Dr Chris Denny

© Copyright November 2011, New Zealand Department of Conservation

ISBN: 978-0-478-14921-0

Cover photo: Baited video monitoring equipment in action at the Poor Knights Islands Marine Reserve.
Photo: Paul Roux De Buisson.

In the interest of forest conservation, we support paperless electronic publishing.

CONTENTS

Executive summary

Baited underwater video (BUV) and diver surveys have been used to monitor the response of reef fish to protection at the Poor Knights Islands Marine Reserve (PKIMR) since 1998. These surveys have recorded increases in the size and abundance of fish species previously targeted by fishing, particularly snapper, which have increased by an estimated 14 times. In the summer of 2009 BUV was used to measure fish abundance and size at the PKIMR. Notably there was no significant difference between snapper abundances in 2002 and 2009 and diver surveys around the baited underwater video suggest that BUV may have reached saturation point. Saturation occurs when the number of fish attracted to the bait pot exceeds the capacity of the cameras field of view.

Past surveys have recorded much higher abundances of snapper in summer than in winter surveys. Summer counts appeared to have reached a plateau possibly as a result of the BUV being saturated however winter counts up until 2002 have continued to increase since the marine reserve was established. The last winter BUV survey at the PKIMR was carried out in 2002 and this investigation reports on a winter BUV sample in 2010. If winter counts had continued to increase it could be inferred that summer populations could also have increased and that saturation is limiting BUV methodology to detect further increases in abundance. If winter counts had reached a plateau then either the population had stabilised or saturation is also occurring in winter (albeit at a different level to summer).

Snapper counts in the winter of 2010 were not significantly different from 2002, the last time a sample was taken in winter. Diver observations of the BUV indicated that saturation is also a factor in winter. Divers observed that snapper response to bait and fish schooling behaviour were much different in winter than in summer. In winter snapper were observed to be less vigorous around the bait, to swim more slowly and to form less dense feeding aggregations than in summer. The density of feeding aggregation affects the average maximum snapper (ams) indices because the cameras field of view is limited. Because of these behavioural differences saturation density is lower in winter than in summer. Snapper populations at the PKIMR may be continuing to increase in abundance and size. The current baited underwater video methodology has a limited capacity to capture any further increases because of saturation.

Snapper size and weight recorded on BUV surveys has continued to increase at the PKIMR after 12 years of protection. This biomass increase is driven by greater numbers of recorded fish in the 400-600mm size range. Fish greater than 600mm are known to be present in considerable numbers (pers. obs) but were not observed interacting with the baited underwater video. This size related behavioural factor has implications for the effectiveness of BUV to monitor snapper populations once populations have recovered significantly. The current BUV methodology may have reached its limit to detect any future trends in key species at the PKIMR. It is suggested that side view stereo video equipment be tested in the future. Stereo systems have a larger field of view which typically results in more species being sampled. Stereo systems can collect size-structure information for a variety of species. Given the wider field of view, this system may solve the saturation problem currently limiting monitoring of reef fish.

The Poor Knights data set is now over twelve years old and provides important information about the performance of a successful no-take marine reserve over a medium time frame in New Zealand. The continuation of this data set is vital to assist with the adaptive management of the reserve itself as well as to support the design of successful marine reserve networks both nationally and internationally in the future. Alternative methodologies are required to record future trends in reef fish assemblages at the Poor knights Islands Marine Reserve.

Introduction

The Poor Knights Islands group situated 24 km off the coast of Whangarei, north eastern New Zealand supports a diverse range of habitats and species. The islands are separated from other land masses and reef systems by a boundary of deep water in excess of 100m. Rare subtropical fishes, invertebrates and algae can be seen here along with large predatory species such as bronze whaler sharks (*Carcharhinus brachyurus*), kingfish (*Seriola lalandi*) and snapper (*Pagrus auratus*). Since a full no take marine reserve was established in 1998 reef fish numbers have been monitored by the University of Auckland and the Department of Conservation (Denny and Shears 2004; Roux De Buisson 2010).

Scientific studies have documented a dramatic recovery of previously harvested species. Baited underwater video surveys have found that snapper were 14 times more abundant at the Poor Knights in 2009 than in 1998, when the area was not fully protected (Roux De Buisson 2010). Diver surveys have also reported a large increase in the numbers and size of snapper over this period (Denny 2008; Denny and Shears 2004). Large snapper well over 10kg are now commonly seen by divers.

Protected natural areas like marine reserves provide important data to differentiate the impacts of fishing from natural environmental variables. By comparing marine reserves with areas open to fishing we can get some insight into the impacts of fishing without the need for complex mathematical models that are prone to error. Roux De Buisson (2010) showed that recorded snapper abundance at unprotected North Cape, Cape Karikari and Cape Brett were 9.3%, 18.7% and 11.1% that of the PKIMR respectively. Investigations have also shown that in addition to targeted species, other components of marine ecosystems, such as seaweeds and invertebrates may also respond to protection. This is because there are multitudes of relationships among bottom dwelling organisms and associated fish populations (Shears, Babcock et al. 2008). Marine reserves have been shown to affect entire reef systems due to close links between fish, algae and associated invertebrates (Babcock, Kelly et al. 1999; Shears and Babcock 2003).

Marine protected areas can conserve representative, rare, outstanding and biologically important ecosystems. In north eastern New Zealand some species in no take marine reserves at the PKIMR, Tawharanui, Leigh and Hahei have radically recovered after protection, however the nature and magnitude of responses to protection throughout this region have varied (Freeman 2008; Roux De Buisson 2010; Shears 2006; Shears, Grace *et al.* 2006; Sivaguru 2007; Taylor, Anderson *et al.* 2004). Knowledge about the capability of marine protected areas to adequately protect biodiversity across a range of environments is crucial to enable the design of an effective network of marine protected areas in the future.

At the PKIMR initial rapid recovery of snapper populations following implementation of no-take status was due to an influx in migratory fish rather than recovery of a resident population (Denny & Babcock 2003). Surveys conducted in 2009 (Roux De Buisson 2010) indicate that snapper numbers may be levelling out after an initial strong increase. There was no significant difference in average snapper counts between summer 2001 and summer 2009 surveys. The apparent lack of change in snapper abundance since 2001 may be explained by a limitation of the monitoring method used. The baited underwater video (BUV) system used in previous surveys may have limited capability to measure any further increases in snapper abundance at the PKIMR. Divers have observed that in most locations in summer, the baited underwater video system is saturated with fish. When the camera is saturated there are more snapper present around the bait than can fit within the cameras field of view. In this situation the camera is unable to capture the large biomass of fish outside the camera frame.

If saturation is occurring sampling may therefore be underestimating the size of the snapper population at the Poor Knights. Camera saturation is not an issue at the fished reference locations where there are much lower numbers of fish, and therefore competition for space around the bait does not result in fish being excluded from the video camera's field of view. As a result differences in abundance estimates between the Poor Knights Islands Marine Reserve and fished reference locations are potentially greater than has been indicated by previous reports (Denny 2008; Denny and Shears 2004; Roux De Buisson 2010).

In past investigations average snapper counts have been much lower in winter than summer. Winter counts have been increasing over time but have not yet reached the apparent saturation level shown for summer counts. The last winter BUV survey at the PKIMR was carried out in 2002 and this investigation reports on a winter BUV sample in 2010. If winter counts have continued to increase it could be inferred that summer populations have also increased in abundance and that saturation is limiting BUV methodology. If winter counts have reached a plateau then either the population has stabilised or saturation is also occurring in winter (albeit at a different level to summer). This investigation is designed to improve understanding of the performance and the limitations of BUV in recovered environments where fish abundance is high such as the PKIMR.

Aims

1. To determine whether winter snapper counts at the Poor Knights Island Marine reserve have continued to increase since 2002, have reached a plateau or have declined.
2. To determine whether winter snapper size and biomass has continued to increase since 2002, has reached a plateau or has declined.
3. To determine whether fish saturation of BUV is occurring in winter surveys.
4. To determine if BUV methodology is suitable to monitor snapper abundance and biomass at the Poor PKIMR.

Methods

Baited underwater video

A baited underwater video system (BUV) has been used to monitor reef fish populations in New Zealand for more than ten years (Willis and Babcock 2000). This baited underwater video system has been used to assess the abundance and size distribution of snapper and blue cod *(Parapercis colias)* inside and outside marine reserves in New Zealand (Davidson and Richards 2005; Davidson 2001; Denny and Shears 2004; Roux De Buisson 2010; Sivaguru 2007; Taylor, Anderson *et al.* 2003; Willis, Millar *et al.* 2003). Over the years, changes have occurred in the design and construction of the baited underwater video system. The majority of these changes have been restricted to materials used for the tripod and small refinements in design (Denny and Shears 2004; Denny, Willis *et al.* 2003;

Langlois, Anderson *et al.* 2008). The evolution of the BUV system used in the present study is described in (Langlois, Anderson et al. 2008). The BUV used in 2009 sits on the substrate and is held upright by a pressure buoy. Previous baited underwater video sampling methods and analyses at the PKIMR were repeated (Denny and Babcock 2003; Denny and Shears 2004; Roux De Buisson 2010; Willis and Babcock 2000).

BUV sampling involves dropping a camera attached to a frame (figure 1) into the water and filming fish as they are attracted to a bait pot (Willis and Babcock 2000). Baited underwater videos were submerged 28 times at the PKIMR for a 30 minute sampling period. Bait pots were filled with fresh bait (pilchards) before each deployment and drops were placed a minimum distance of 50m from another deployment location to avoid potential effects on fish behaviour. The recorded video was played back and the maximum number of each fish species recorded in a frame over the 30 minute film sequence was recorded. The maximum number of each species was then averaged across all drops to give the average maximum snapper (ams) count. The frame containing the maximum number of snapper was analysed further. Each maximum snapper frame was saved from the video sequence and calibrated against a scale bar of known length and a bait container of known length within the baited underwater video's field of view (figure 1). Because the camera is not bi-focal care was taken to accurately measure fish length. Fish were measured using three point calibration and were only measured when they were at the same level as a calibration point of known length, usually the bait container. Snapper length from the video frame with the maximum number of snapper was converted to wet weight biomass using $W = aL^b$ where W is weight (g), L is length (mm) , a is 7.194×10^{-5} and b is 2.793 (Taylor and Willis 1998).

Sampling locations were fixed GPS waypoints typically on sand immediately adjacent to rocky reefs (Appendix 1). A research vessel approached a fixed historical waypoint, the bottom was confirmed to be sand immediately adjacent to rocky reef with a Humminbird, transducer mounted side scan sonar (figure 2) and then the baited underwater video was dropped. In situations where the sampling site was over rocky reef substrate an additional quick release camera with a live video feed to the research vessel was attached to the baited underwater video (figure 1). In this way the operators could check that the field of view was not obscured by seaweed prior to finalising the drop.

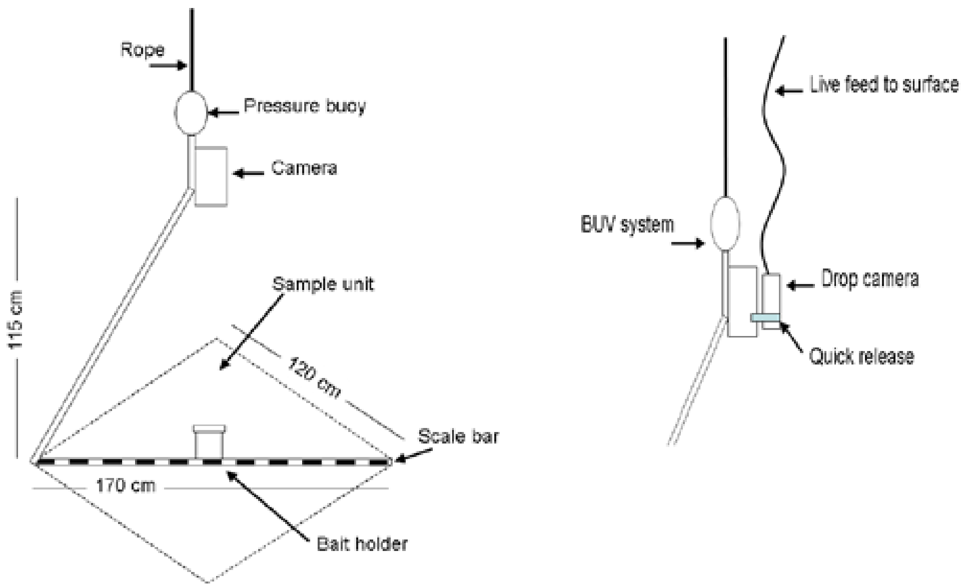
Patterns in reef fish abundance may vary according to variation in recruitment, change in habitats, physical forces (e.g. sea condition, water temperature), food resources, ontogenetic changes in habitat requirements, time of day, reproductive behaviour, variation in mortality rates through predation, inter-specific interactions and the degree of fidelity fishes have to reefs (Kingsford and Battershill 1998). All of these factors vary through space and time and may affect baited underwater video counts. The response of carnivorous species to the baited underwater video apparatus at any point in time may also be affected by tidal cycles, depth and the position of the sampling station in relation to the reef structure and current patterns. In order to avoid bias, sampling locations include a range of depths and positions in relation to reef structure and current flow. Sampling was undertaken throughout all tidal cycles and sand habitat adjacent to reef has been targeted at all locations where possible. By sampling using consistent methods over long time scales we can detect the relative influences of protection and fishing on reef fish populations, particularly snapper.

Scuba divers with video cameras and still cameras observed fish behaviour around the baited underwater video. Divers would remain motionless and conceal themselves behind rocks and seaweed. Divers were instructed to observe fish behaviour and schooling density inside and outside the cameras field of view. Diver observations were a key approach to establish whether saturation is an issue in winter BUV surveys.

Statistical analysis

BUV is particularly well suited to measuring the relative abundance of large mobile predators such as snapper. BUV data are counts and therefore do not satisfy the assumptions of normality and homogeneity of variance that are required by ANOVA. Therefore, the BUV data was analysed using the Poisson distribution using the GENMOD procedure in SAS to obtain unbiased estimates of relative abundance for dominant carnivorous species. See (Willlis, Millar et al. 2000) for a more detailed description of this analysis. Size data are continuous variables and satisfy assumptions of normality and homogeneity of variability required by ANOVA. Size data were analysed using a one way ANOVA procedure in sigmaplot.

The average maximum snapper count (ams) is used throughout this report and is calculated by averaging the maximum count of snapper from each camera drop across all 30 drops at each location.



(a) (b)

Figure 1: Baited underwater video system (BUV). (a) This system sits on the substrate and is kept upright by a pressure buoy. The camera films fish attracted to the bait. (b) A quick release camera supplying real time footage to the surface is attached allowing the baited underwater video to be placed in complex habitats such as kelp forests and rocky reef systems. The umbilical camera is removed once the baited underwater video is in place.

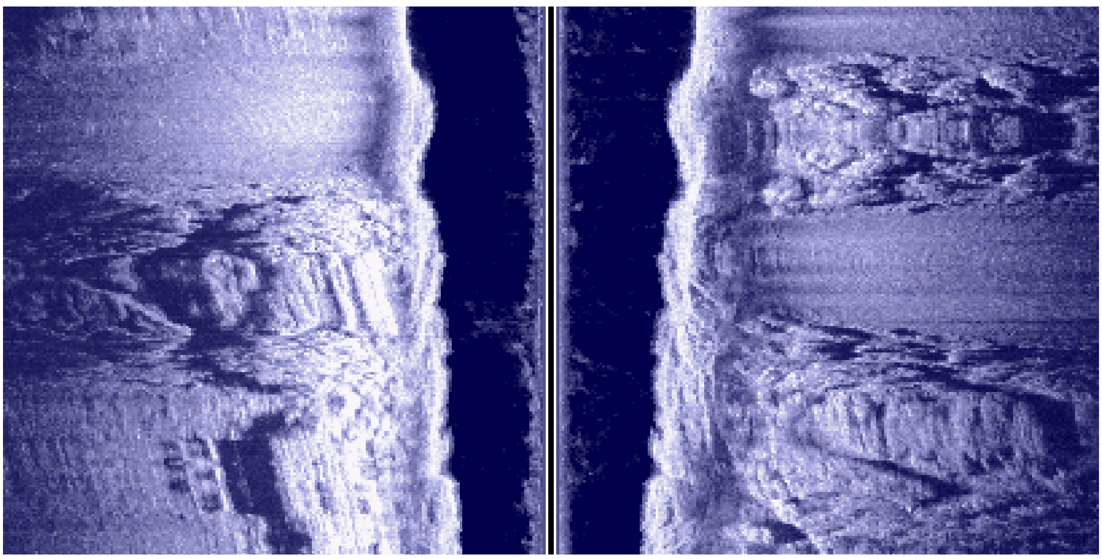


Figure 2: Humminbird sidescan sonar image used to determine the location of sampling locations on sand immediately adjacent to rocky reefs. Smooth areas are sand and rough areas are reef.

Results

The average maximum snapper counts (ams) from winter and summer samples at the PKIMR are shown in figure 3. Winter counts have been statistically compared using ratios and the results are displayed in table (1). From 1998 (before no take protection) to 1999 winter average maximum snapper (ams) counts increased significantly from 1.5 ams ± 0.3 to 5.8 ams ± 0.8 ($p<0.0001$). In subsequent years winter snapper counts have not changed significantly at the Poor Knights. In winter 2010 the average maximum snapper count was 8.1 ams ± 0.81 which was not significantly different from the last winter sample in 2002 (9.7 ams ± 1.1).

The average length of snapper recorded on baited underwater video cameras has increased significantly and is shown in figure 4. Average fork length of snapper at the Poor Knights has increased significantly from 296mm ± 8.5 in 1998 to 384 ± 16 in 2010. The average length of snapper was slightly higher in 2010 than 2009 although this difference was not significant ($p = 0.0878$). Figure 5 compares the size frequency distribution of recorded snapper populations at the Poor Knights in winter from 1998 to 2010. Recorded snapper size has changed since full no take protection. Snapper smaller than 200mm and larger than 600mm have almost never been recorded on baited underwater video cameras (Denny and Shears 2004; Denny, Willis *et al.* 2003; Roux De Buisson 2010). Therefore any changes we can detect in snapper size frequency and abundance using BUV are restricted to fish between 200 and 600mm in length.

In 1999, one year after the Poor Knights Islands was closed to fishing there was an influx of snapper in the 200-400mm range (figure 5). Only one snapper greater than 400mm in length was recorded in this year. Snapper numbers in the smaller 200-300mm size range have not increased in abundance since the initial influx in 1999. Snapper in the 300-400mm size range increased steadily from 1999 reaching a peak in 2002 after four years of no take protection. Snapper in the 200-300m and 300-400mm size range were recorded in lower numbers in 2010 than in 2002.

The number of recorded fish over 400mm has increased gradually over the period of no take protection. In 1998 one 400-500mm snapper was recorded. In 1999 after one year of protection 14 snapper were recorded in this size range. In 2002 after 4 years of closure 32 snapper were recorded in the 400-500mm size range and in 2010 after 12 years of no take protection 68 snapper in the 400-500mm size range were recorded. In the 500-600mm size range snapper counts have increased but at a slower rate than smaller size classes. In 1998 before no take protection zero snapper were recorded in this size range. This count increased to five by 2002 and in 2010 after 12 years of protection 18 snapper were recorded in this size range.

Very few snapper over 600mm have been recorded with baited underwater video at the Poor Knights. In 1998 zero were recorded, in 2002 after 4 years of protection 1 was recorded, in 2010 after 12 years of protection 4 snapper greater than 600mm in length were recorded. As we move through time from 1998 to 2010 larger fish make up a higher percentage of snapper recorded at the PKIMR.

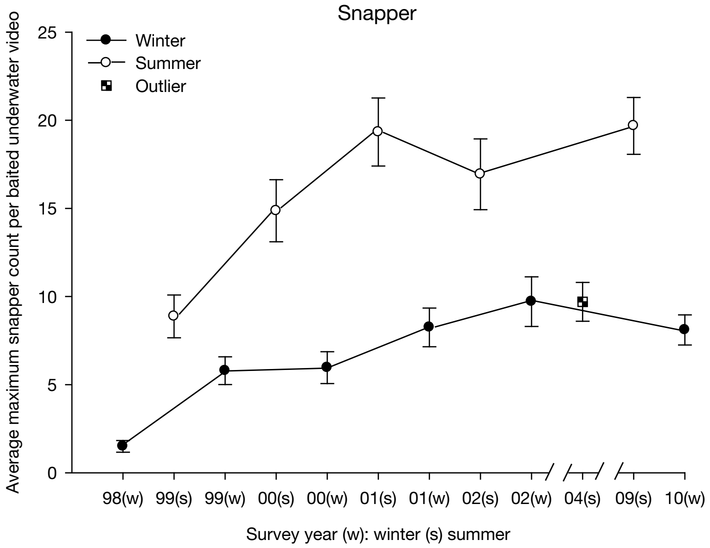


Figure 3: Average maximum number of snapper per baited underwater video at the Poor Knights and standard error bars. The 2004 sample is treated as an outlier as it was undertaken 3 months later than previous summer samples and therefore does not represent a summer or a winter sample.

Table 1: A statistical comparison of winter recorded snapper abundance at the Poor Knights. The ratio indicates the proportion of snapper present in that year compared with 2010. A p value of less than 0.05 indicates a statistically significant difference.

SURVEY	RATIO	UPPER CL	LOWER CL	P-VALUE
1998	0.172769	0.086783	0.343952	**<.0001**
1999	0.714909	1.082776	0.472023	0.1131
2000	0.736313	1.106918	0.489789	0.1411
2001	1.018061	1.471644	0.704279	0.9241
2002	1.198176	1.713435	0.837864	0.3218
2010	1	1	1	

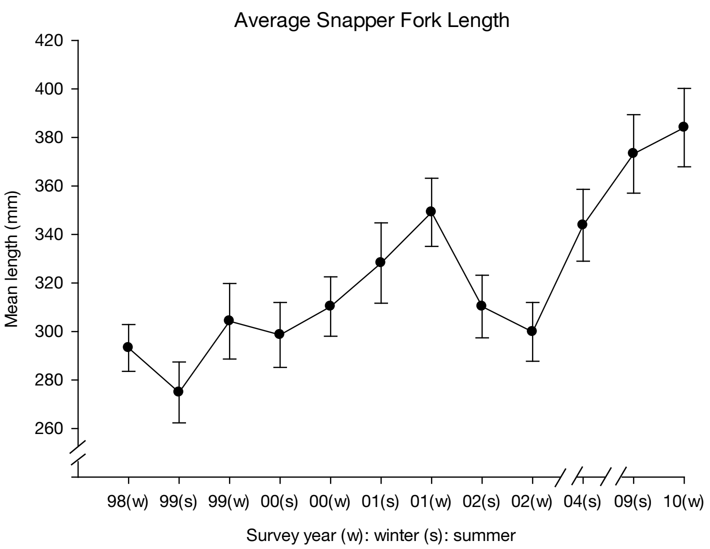


Figure 4: Average maximum snapper length (fork length) at the Poor Knights Islands from summer and winter surveys from 1998 to 2010.

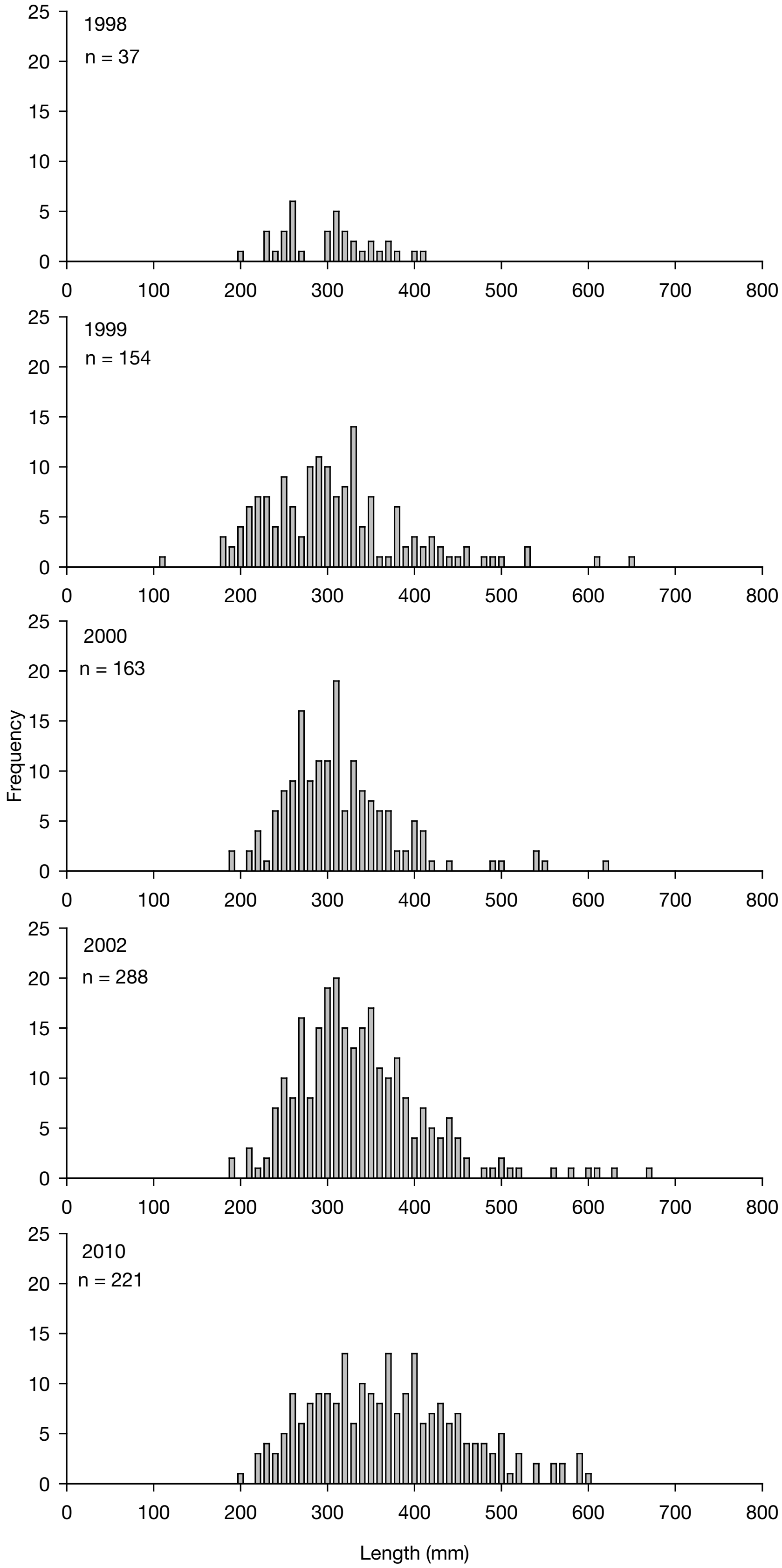


Figure 5: Size frequency distributions of snapper at the PKIMR from winter samples.

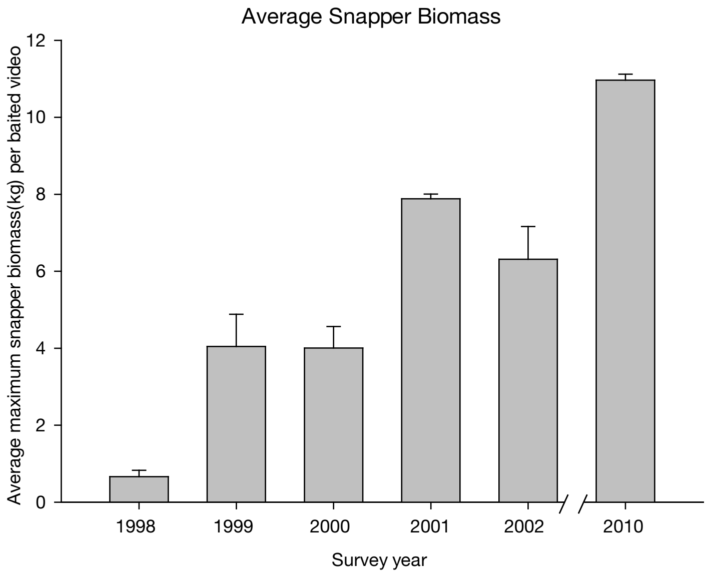


Figure 6: Average maximum snapper biomass per baited underwater video from all winter samples at the PKIMR and standard error bars.

Snapper length was converted to biomass (weight (g)) and results from winter samples are displayed in figure 6. Although snapper counts have reached a plateau (figure 3) biomass has increased significantly over the past 8 years from 6.31kg ± 0.85 in 2002 to 10.96kg ± 0.16 in 2010. This increase in biomass has been driven by an increase in the size of fish recorded on baited underwater video cameras (figure 4).

Kingfish counts have increased 6 fold over the past 8 years (figure 7). The average maximum number of kingfish (amk) increased from 0.1 ± 0.05 (amk) in 2002 to 0.59 ± 0.19 (amk) in 2010. In 2002 one kingfish was recorded for every ten camera drops. In 2010 six kingfish were recorded for every ten camera drops. Average baited underwater video counts for pigfish, sandagers wrasse, porae, northern scorpion fish, trevally and leatherjacket are illustrated in figures 8 -12.

Divers observed the behaviour of snapper and other species around the bait and made comparisons with summer observations. In this winter survey large numbers of snapper were present outside the BUV's field of view. Smaller fish would approach the bait more quickly and spend more time at the bait than larger fish. Large fish would circle the bait outside the cameras view, some would enter the sampling frame and others would not. These behavioural factors resulted in a lower proportion of large fish being captured by the BUV than were actually present, as a significant proportion of the largest fish observed did not swim into the cameras field of view at all.

Some key behavioural differences were observed between summer and winter surveys. In winter snapper were less vigorous than in summer. In winter snapper would swim more slowly and would spend less time at the bait and more time swimming slowly around the baited underwater video outside the cameras field of view. The aggregations formed at the bait in winter were therefore less dense than in summer. In winter snapper would wait for an opening around the bait and then would move in and nudge the bait under the camera. At times larger individuals would guard the bait and would actively chase smaller fish away from the bait. The guarding behaviour observed in large fish was more pronounced in winter than in summer. Guarding resulted in low numbers of fish within the cameras field of view at any one time.

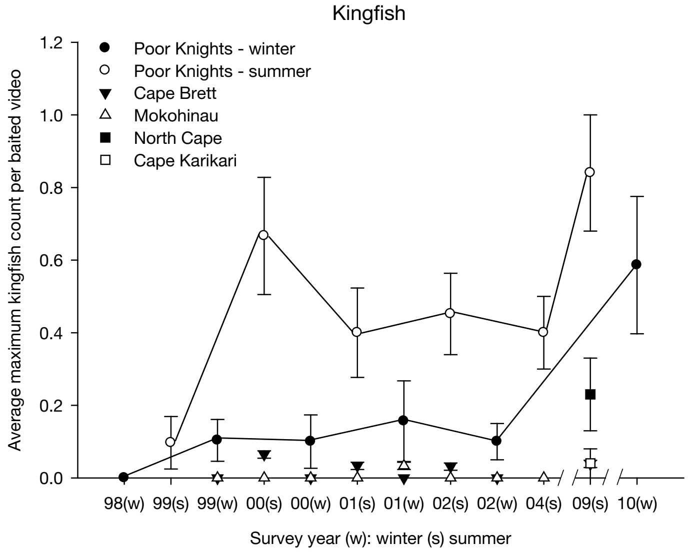


Figure 7: Average maximum number of Kingfish per baited underwater video from all locations and standard error bars. The Poor Knights have been separated into winter and summer samples.

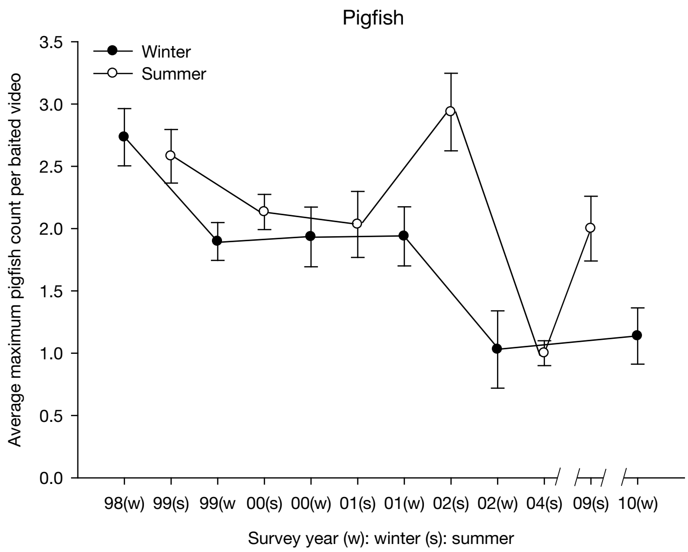


Figure 8: Average maximum number of pigfish per baited underwater video from the Poor Knights and standard error bars in winter and summer.

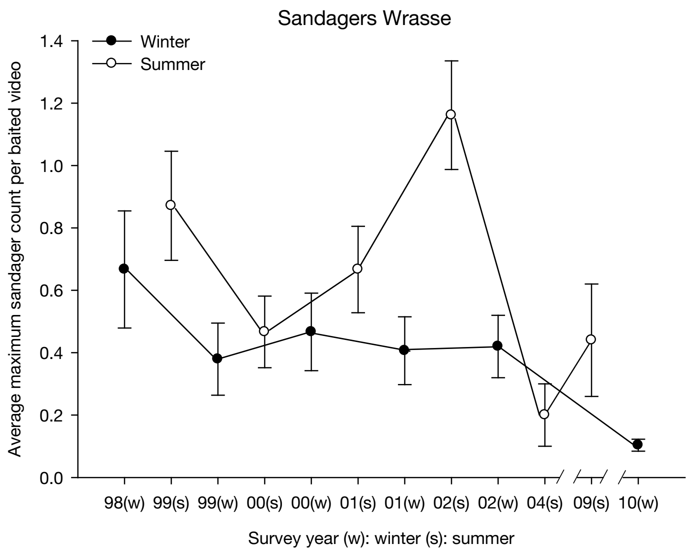


Figure 9: Average maximum number of sandagers wrasse per baited underwater video from the Poor Knights and standard error bars in winter and summer.

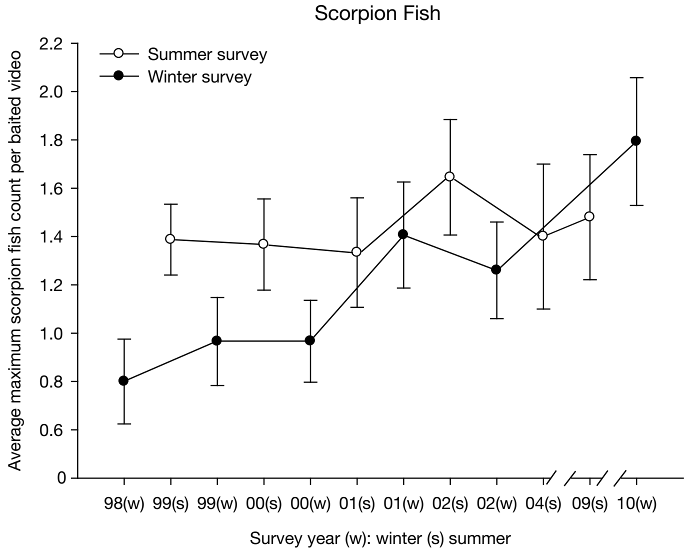


Figure 10: Average maximum number of scorpion fish per baited underwater video from the Poor Knights and standard error bars in winter and summer.

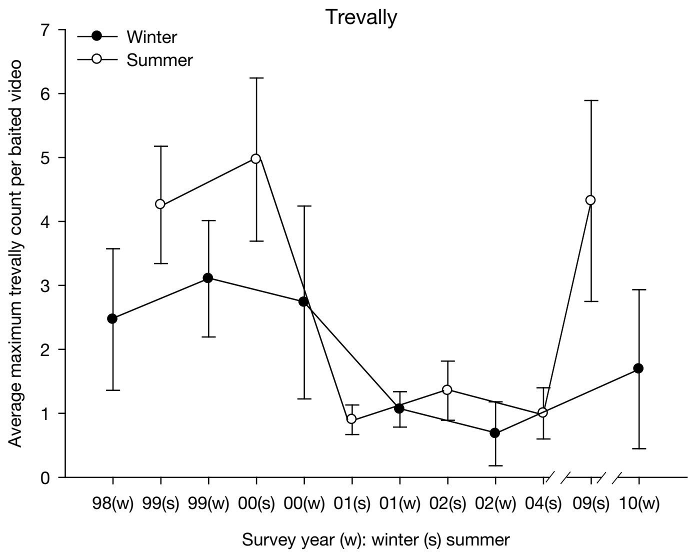


Figure 11: Average maximum number of trevally per baited underwater video from the Poor Knights and standard error bars in winter and summer.

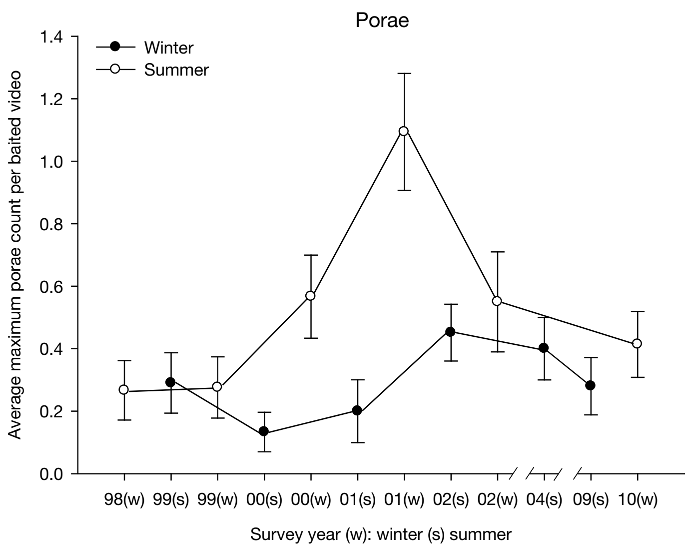


Figure 12: Average maximum number of porae per baited underwater video from the Poor Knights and standard error bars in winter and summer.

Discussion

Snapper counts in winter 2010 were not significantly different from 2002, the last time a sample was taken in winter. Diver observations of schooling behaviour, competitive exclusion and the presence of fish outside the cameras field of view indicate that this lack of change in abundance reflects a limitation of the method to capture fish outside the cameras field of view (saturation) rather than a true levelling off in abundance. When snapper populations become more abundant and larger in size, BUV will become less affective as a tool to measure ongoing changes because of this saturation effect along with changes in behaviours such as schooling, feeding and guarding.

BUV is particularly effective in the early stages of protection because snapper are generally BUV positive increasing our power to detect differences when differences are small. In situations where snapper numbers are low due to fishing or during the early periods of marine reserve protection BUV may reveal significant results when data from alternative methods such as diver survey is highly variable and unable to detect statistically significant differences. However this study has shown that once snapper populations have recovered in abundance and size the current BUV system is limited and alternative methods should be used to monitor further changes beyond the initial recovery.

This study found that BUV saturates at different levels in summer than winter due to seasonal behavioural differences in snapper. The winter and summer snapper population at the Poor Knights may be continuing to increase in abundance and size, however the current BUV methodology may not be capable of capturing any further increases in abundance. An alternative baited underwater video method widely used in Australia is based on the stereo-video systems developed by Mark Shortis and Euan Harvey for the Department of Conservation for use in Fiordland (Harvey and Shortis 1996). The method uses two horizontally positioned cameras with converging fields of view in which a stereo calibration can be made. The horizontally facing cameras give a much greater field of view over the substrate limited only by the visibility of the water. This larger field of view typically results in many more species being sampled and so these systems can collect size-structure information for a variety of species (Langlois, Harvey et al. 2010). Given the wider field of view this system may solve the saturation problem currently limiting monitoring of reef fish at the Poor Knights.

Although snapper counts have not increased snapper biomass has continued to increase after 12 years of protection. This biomass increase is driven by greater numbers of fish in the 400-600mm size range at the Poor knights. Fish greater than 600mm are known to be present in considerable numbers at the Poor Knights (pers. obs) but are not interacting with the BUV for unknown reasons. These larger fish have been observed swimming outside the cameras field of view and do not swim under the camera to get to the bait. Willis et al. (2000) compared BUV with long line fishing in the same area around Leigh marine reserve. Fishing methods recorded snapper up to 1m in length and substantial numbers of fish over 600mm were caught while BUV recorded only one fish over 600mm. Fish size clearly effects behaviour and snapper over 600mm in length are less likely to interact with the BUV bait than smaller fish. This size related behavioural factor has implications for the effectiveness of BUV to monitor snapper populations once populations have recovered significantly in size and abundance.

For carnivorous species other than snapper it is more difficult to determine trends in abundance from the baited underwater video data. These species are present at all locations in relatively low numbers and the data is often highly variable between locations, sites and years. At the Poor Knights, average maximum kingfish counts have increased from 1998 to 2009. This study recorded a much higher kingfish count than previous winter samples which may be an indication of a positive increase in this species. Results are still highly variable and further sampling will reveal a clearer result for kingfish at the Poor Knights. The downward facing baited underwater video camera used in this study also has a limited capability to capture large schools of kingfish and is probably not a good method for monitoring kingfish abundance.

Pigfish and sandagers wrasse may have dropped off in numbers at the Poor Knights since monitoring began in 1998. A reduction in numbers may be a result of increased competition with snapper for food or increased direct predation pressure. An alternative theory is that declines may be due to natural mortality after a recruitment pulse in 1998 due to warmer sea surface temperatures (Denny & Shears 2004). However abundance counts have remained low and have not changed much over the sampling period and a longer time series of data is required to resolve the effect of the marine reserve on these two species. Porae and northern scorpion fish are both vulnerable to some forms of fishing. However, neither species has increased significantly in abundance at the Poor Knights. Baited underwater video is probably not an effective monitoring method for trevally because of this species' tendency to school, resulting in highly variable data.

Conclusion

BUV has generated remarkable data sets in north eastern New Zealand describing changes in abundance and size of snapper after protection. BUV is a cost effective and statistically powerful method to detect changes between protected and unprotected areas when differences are small. However this investigation has indicated saturation and behavioural changes in snapper limit the effectiveness of BUV to monitor patterns in abundance beyond the initial recovery. Additional methods such as stereo cameras and diver surveys should be combined with BUV to further our understanding of how the snapper population will respond to protection in the future.

The Poor Knights data set is now over twelve years old and provides important information about the performance of a successful no-take marine reserve over a medium time frame in New Zealand. The continuation of this data set is vital to assist with the adaptive management of the reserve itself as well as to support the design of successful MPA networks both nationally and internationally in the future. Not all ecosystems respond in the same way to fishing or conservation attempts. By studying these variations and understanding the reasons for mixed successes we increase our ability to make appropriate and informed management decisions. Both within New Zealand and internationally it is clear that there is a high degree of variability and ongoing change in the abundance of targeted species over long time scales (Babcock, Shears et al. 2010). Implementation of successful marine protected area networks needs to be supported by long term data collection to test MPA network performance. Internationally, quality data sets measuring the performance of MPAs over long time scales are rare (Babcock, Shears et al. 2010).

Recommendations

1. Stereo baited underwater video should be trialled at the Poor Knights. Paired standard baited underwater video and stereo baited underwater video should be used to determine if the stereo system is capable of measuring a greater range of relative abundance than the standard baited underwater video system.

2. Under water visual counts (UVC) or diver transects have proven to be an effective method to measure fish community response to protection. A downward facing camera is limited to capturing fish that are attracted to a bait pot while diver surveys sample the range of fish diversity including herbivorous and more cryptic species. The last UVC was carried out at the Poor Knights in 2007. A further survey would provide useful information about the response of the entire fish community to protection at the Poor Knights.

3. Little is known about the effects such a large increase in predatory biomass will have on other components of the ecosystem at the Poor Knights. Further investigations into habitat utilisation and the diet of snapper are necessary to resolve any potential effects on other species and habitats at the Poor Knights Islands Marine Reserve.

Acknowledgements

I would like to thank Fiona Watson for analysing the baited underwater video camera footage and inputting all of the raw data. For help on the boat dropping cameras I would like to thank Debbie Freeman and Callum Lilley for battling difficult conditions to get the job done. Marine ranger Leon Candy (skipper), Ben Barr and Blair Jones also helped in the field. I would like to thank Amy Macdonald, Debbie Freeman and Chris Denny for reviewing the draft version.

References

Babcock RC, Kelly S, Shears NT, Walker JW, Willis TJ (1999) Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series* **189**, 125–34.

Babcock RC, Shears NT, Alcala AC, Barrett NS, Edgar GJ, Lafferty KD, McClanahan TR, Russ GR (2010) Marine Reserves Special Feature: Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *PNAS , 18256-18261*.

Davidson R, Richards L (2005) Comparison of fish at reserve and control sites from Long Island-Kokomohua and Tonga Island Marine Reserves using baited underwater video (BUV), catch, measure, release (CMR) and underwater visual counts (UVC). *Prepared by Davidson Environmental Limited for Department of Conservation, Nelson. Survey and Monitoring Report No. 466.*

Davidson RJ (2001) 'Changes in population parameters and behaviour of blue cod (*Parapercis colias*; Pinguipedidae) in Long Island – Kokomohua Marine Reserve, Marlborough Sounds, New Zealand.'

Denny CM. (2008) The effect of alternate management strategies on reef fish populations in northeastern New Zealand. *Department of Conservation, Northland Conservancy*, 53.

Denny CM, Babcock RC (2003) Do partial marine reserves protect reef fish assemblages? *Biological Conservation* **116**, 119–129.

Denny CM, Shears NT (2004) 'Effects of five years of no take marine reserve protection on reef fish populations at the Poor Knights Islands.' Leigh Marine Laboratory, University of Auckland, Warkworth, New Zealand. Report to the Department of Conservation, Northland.

Denny CM, Willis TJ, Babcock RC (2003) Effects of Poor Knights Islands Marine Reserve on demersal fish populations. In. (Department of Conservation Science Internal Series 142)

Freeman DJ, MacDiarmid, A.B. (2008) Healthier lobsters in a marine reserve: effects of fishing on disease incidence in the spiny lobster, *Jasus edwardsii*. *Marine and Freshwater Research* **60(2)**, 140-145.

Harvey ES, Shortis M (1996) A system for stereo-video measurement of subtidal organisms. *Journal of the Marine Technological Society* **29**, 10-22.

Kingsford M, Battershill B (1998) Studying temperate marine environments a handbook for ecologists. *Canterbury University Press*, 334.

Langlois TJ, Anderson MJ, Roux De Buisson PE (2008) 'Baited underwater video: Comparison of a benthic versus a suspended system.' Prepared for Northland Conservancy, Department of Conservation, Auckland Uniservices Ltd, The University of Auckland.

Langlois TJ, Harvey ES, Fitzpatrick B, Meeuwig JJ, Shedrawi G, Watson DL (2010) Cost-efficient sampling of fish assemblages: comparison of baited video stations and diver video transects. *Aquatic biology* **9**, 155-168.

Roux De Buisson PE (2010) Poor Knights Islands Marine Reserve and Mimiwhangata Marine Park fish monitoring 2009. *Department of Conservation, Northland Conservancy*, 34.

Shears NT, Babcock R (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series* **246**, 1–16.

Shears NT, Babcock RC, Salomon AK (2008) Context-Dependent Effects Of Fishing: Variation In Trophic Cascades Across Environmental Gradients. *Ecological Applications* **18**, 1860-1873.

Shears NT, Grace RV, Usmar NR, Kerr V, Babcock RC (2006) Long-term trends in lobster populations in a partially protected vs. no-take marine park. *Biological Conservation* **132**, 222–231.

Shears NT, Usmar, N.R. (2006) 'Response of reef fish to partial and no-take protection at Mayor Island (Tuhua).' DOC Research and Development Series 243, Science and Technical Publishing.

Sivaguru K (2007) 'Cape Rodney to Okakari Point Marine Reserve and Tawharanui Marine Park fish (baited under water video system) monitoring 2007.' Department of Conservation.

Taylor BT, Willis TJ (1998) Relationships amongst length, weight and growth of north-eastern New Zealand reef fishes. *Marine & Freshwater Research* **49**, 255–260.

Taylor RB, Anderson MJ, Egli D, Willis TJ (2003) 'Cape Rodney to Okakari Point Marine Reserve fish monitoring 2003: Final Report.' Prepared for Auckland Conservancy, Department of Conservation. Leigh Marine Laboratory, Department of Statistics, University of Auckland.

Taylor RB, Anderson MJ, Usmar N, Willis TJ (2004) 'Te Whanganui a Hei Marine Reserve fish monitoring 2004: Final Report.' Prepared for Department of Conservation, Auckand Uniservices Ltd, The Universtity of Auckland.

Willis TJ, Babcock RC (2000) A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research* **51**, 755–63.

Willis TJ, Millar RB, Babcock RC, Tolimieri N (2003) Protection of exploited fish in temperate regions; high density and biomass of snapper *Pagrus auratus* (Sparidae) in northern New Zealand marine reserves. *Journal of Applied Ecology* **40**, 214–227.

Willlis TJ, Millar RB, Babcock RC (2000) Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. *Marine Ecology Press Series* **198**, 249-260.

Appendix 1: Poor Knights Islands Marine Reserve monitoring site information

Depth	Name	Date	Temperature	Habitat	Max Snapper
36	Rikoriko	9.9.2010	15.8	sand	8
29	The Labrynth	9.9.2010	15.8	sand	7
23	The Gap	9.9.2010	15.8	sand	15
42	Cairneys rock	8.9.2010	15.8	reef	7
32	Maomao arch	8.9.2010	15.8	sand	5
42	Skull Bay	10.9.2010	15.8	sand	10
29	Shag Bay	8.9.2010	15.8	sand	6
19	Nursery Cove	9.9.2010	15.8	sand	3
38	Middle Arch	10.9.2010	15.8	sand	8
46	South Cleanerfish	10.9.2010	15.8	sand	1
34	Aorangaia Island	9.9.2010	15.8	reef	9
18	Southern Arch	9.9.2010	15.8	reef	13
31	Chris's Area	9.9.2010	15.8	sand	20
29	Ngaio rock	9.9.2010	15.8	sand	2
23	Matts Crack	15.9.2010	15.8	reef	2
26	West Bartles	10.9.2010	15.8	reef	7
38	Arch Rock	8.9.2010	15.8	reef	6
38	RockLilley Inlet	8.9.2010	15.8	mixed	8
40	Cave Bay	8.9.2010	15.8	reef	11
29	The Gardens	9.9.2010	15.8	sand	4
36	North Frasers Bay	15.9.2010	15.8	reef	8
28	South Frasers Bay	9.9.2010	15.8	sand	14
38	North Cleanerfish	8.9.2010	15.8	reef	10
36	Northern Arch	10.9.2010	15.8	reef	4
35	Barren Arch	8.9.2010	15.8	reef	2
25	Blue Mao Mao	9.9.2010	15.8	reef	13
11	Labrid Channel	9.9.2010	15.8	sand	16
30	Freds pinnacle	15.9.2010	15.8	reef	8
35	East bartles	10.9.2010	15.8	reef	8