Effects of the Poor Knights Islands Marine Reserve on demersal fish populations

Photo: Phil Bendle

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Executive Summary

Fish surveys were conducted at the Poor Knights Islands before (September 1998), and after (March and September 1999) implementation of full marine reserve status. Results indicate that despite restrictions on angling methods and gear in those areas of the Poor Knights Islands open to fishing prior to September 1998, fishing pressure may have had significant impacts on fish assemblage structure. Three reef associated species known to be targeted by anglers, the snapper (*Pagrus auratus*), tarakihi (*Nemadactylus macropterus*) and pink maomao (*Caprodon longimanus*), increased in estimated density by 302%, 101% and 129%, respectively, after one year of protection. By September 1999, snapper were significantly larger and more abundant at the Poor Knights Islands than at the reference sites, the Mokohinau Islands and Cape Brett. However, because we could not obtain data from the reference sites concurrent with the earlier Poor Knights surveys, we cannot be certain that increases in fish density at the Poor Knights was due to release from fishing pressure alone.

The rapidity of recolonisation of these species to shallow reefs was surprising, but may be explained by seasonal movements of fishes and the underwater topography at the Poor Knights. The islands rise steeply from the continental shelf, and so may be more quickly colonised from adjacent deep-water habitats. Coastal locations have a shallower bathymetric profile, requiring fish to cross extensive soft-bottom habitats to reach reefs.

Closure of the islands to fishing coincided with two years of above-average sea-surface temperatures, which precipitated pulses of successful recruitment of fishes with subtropical affiliations. There were marked increases in the density of the wrasses *Coris sandageri*, *Pseudolabrus luculentus* and

Suezichthys aylingi. In addition, several rarer species were detected in the 1999 surveys, including *Anampses elegans*, *Coris dorsomacula*, *Epinephelus daemelii*, *Thalassoma amblycephalum*, and *Plagiotremus rhinorhynchus*. Significant declines occurred in the density of the clown toado *Canthigaster callisterna*.

Significant differences were found between the assemblage structure of Cape Brett, the Mokohinau and the Poor Knights Islands, reflecting their locations and the relative strength of the influence of the East Auckland Current. Successful understanding of the response of fishes to protection at the Poor Knights therefore depends not on direct comparison of fish densities among locations, but on comparison of population trends over time.

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1. Introduction

The Poor Knights Islands lie 24 km east of the coast of Northland. The islands are separated from the mainland by relatively deep waters and rise abruptly into the waters of the East Auckland current. The geography of the Poor Knights has given rise to a rich diversity of marine life with varied and ecologically unique floral and faunal communities. Pelagic and schooling fishes are present in high abundance in the waters of the Poor Knights terrace, and the reef areas of the islands themselves support a high diversity of demersal fishes. In recognition of the ecological importance of the Poor Knights, the area was established as a marine reserve in 1981. The intent of the marine reserve legislation was to protect the area's unique reef life and demersal fish populations. In recognition of this intent, fishing restrictions within the reserve area (800 m surrounding the islands) made provision for recreational fishing targeting species deemed to be not directly part of reef system ecology (migratory and pelagic species). Demersal reef fishes, considered part of the area's unique ecology, were protected by gear method restrictions (trolling and unweighted lines only) and small no-take ‘exclusion zones’; the intent being to prevent the incidental catch of these animals by recreational fishing within the reserve area.

The provision under the Marine Reserves Act, which allowed for limited recreational fishing within the Poor Knights (reviewed every five years), expired in December 1995. Continuation of recreational fishing access within the Poor Knights Islands Marine Reserve (PKIMR) was subject to Ministerial discretion. In line with provisions laid down in the Act, the Department of Conservation called for submissions regarding recreational access to the PKIMR. Representatives of the environmental sector argued strongly that recreational fishing activity within the reserve had adversely impacted upon the ecological ‘character’ of the area. The counter submissions from the recreational fishing sector were likewise strong. Unfortunately, evidence either for or against the continuation of recreational fishing within the reserve was scant and inconclusive. Allegations forwarded by the respective sector groups were based largely on ‘perceptions’ rather than demonstrable scientific ‘fact’. Following the decision by the Minister not to renew the right of recreational fishing access to the reserve, recreational fishing groups sought and won an injunction. In May 1997 the Minister of Conservation announced plans to change the fishing regulations in the Marine Reserve due to take effect on 1 October 1997 (partial closure), with a complete fishing exclusion extending to 800m from the islands coming into effect from 1 October 1998.

One of the major reasons for both public and legal indecision regarding the continuation of fishing in the PKIMR was a lack of scientific data regarding demersal finfish abundance over time and in various sections of the reserve. Ecological studies of reef fish abundance, largely based on diver visual fish count techniques, have been carried out intermittently at the Poor Knights Islands since 1972 (Schiel 1984, Choat et al. 1988). However, no single study or group of studies provided information of sufficient detail to allow an assessment of the impacts of recreational fishing activity on reef fish ecology. There were two reasons for this. Firstly, each of the reef fish studies was designed to investigate different specific ecological questions, and consequently methods, places, seasons and species of interest varied between studies. Secondly, the majority of studies were undertaken prior to the Poor Knights being declared a marine reserve in 1981. Only four fully documented Poor Knights reef fish studies have occurred since 1981 (Schiel 1984, Choat & Ayling 1987, Choat et al. 1988, Kingsford & MacDiarmid 1988).

This project was designed to provide quantitative scientific information on the effects of full marine reserve status on the reef fish communities of the Poor Knights Islands. To date, three surveys have been completed: the first immediately prior to complete closure (September 1998), and the others at six-month intervals following closure (March and September 1999). The surveys used both traditional scuba-based underwater visual census (UVC) methods and baited underwater video (BUV) (Willis & Babcock 1997, 1998, 2000, Willis et al. 2000). The latter method allows sampling of carnivorous species that are not amenable to visual methods, and sampling at depths outside diver operating range. For long term monitoring purposes (currently funded to 2002), two reference localities (which continue to be fished) were included in the survey programme in September 1999: Cape Brett and the Mokohinau Islands. Both are influenced by similar hydrographic regimes to the Poor Knights, being subject to regular cross-shelf intrusions of subtropical oceanic water (Sharples 1997), although one is a mainland site (Cape Brett) with correspondingly shallower underwater topography.

As part of this survey programme, we examined methodological factors that might bias long term monitoring results: UVC transect size and the effect of time of day on fish density estimates. We compared density estimates from two UVC transect methods: the long (50 × 10m) used by Schiel (1984) and Choat & Ayling (1987) with the shorter (25 × 5m) transects used in concurrent monitoring programmes at the Leigh and Hahei marine reserves (Willis & Babcock 1997, Willis et al. 2000, Willis unpubl. data). Shorter transects can be done with higher levels of replication and are less likely to straddle several different habitats, and therefore would tend to return less within-site variability. Larger transects, if constrained to particular habitats should give higher means (and hence more statistical power), but require more field time to obtain suitable levels of replication.

Large changes in the composition and/or density of reef fish assemblages may occur where fish behaviour varies with the time of day. This has the potential to bias large-scale survey programmes where sites are examined only once, and of necessity are surveyed at differing times of day. Little is known about the daily activity patterns of New Zealand reef fishes, and we therefore tested whether density estimates of major groups of reef fishes changed at the same sites during the day.

It is difficult to quantify changes through time in fish assemblage structure or the density of individual species that may occur due to some putative human impact (in this case, cessation of fishing at the Poor Knights). This is because fishes are generally mobile (meaning that ‘density’ estimates can vary at small temporal and spatial scales), and variability due to natural processes such as recruitment, natural mortality and large-scale climatic change can obscure or completely mask anthropogenic effects. Detection of such effects can therefore only be definitively made where a suitable time series of data (long enough to encompass natural variability) exists both at the ‘impact’ site and at least one independent ‘control’ or reference site. This report should therefore be regarded as preliminary to the long term monitoring programme currently being carried out at Cape Brett, the Mokohinau and Poor Knights Islands.

2. Methods

Sampling effort at all sites and times is summarised in Appendix 1. Maps of the Poor Knights Islands (36º28’ S, 174º45’ E), Mokohinau Islands (35º55’ S, 175º07’ E) and Cape Brett (35º11’ S, 174º20’ E) study locations, with sampling area boundaries and UVC and BUV sampling sites marked, are given in Appendix 2. Specific locations referred to in the text are marked on these maps.

2.1 Effect of underwater visual census transect type on density estimates

To compare density estimates of reef fish using 25 × 5 m and 50 × 10 m transects, we selected shallow (5 – 12 m, inner Nursery Cove) and deep (15 – 26 m, The Labyrinth) sites to survey with both methods. Both sites were surveyed with 50 × 10 m transects mid-morning on 20/9/98, and then again with 25 × 5 m transects at the same time of day on 21/9/98. We assumed that fish density would not change significantly between the two days. The 50 × 10 m transects were done in a similar fashion to the methods of Choat & Ayling (1987) and Choat et al. (1988). A diver laid out a 50 m tape on the reef, and then swam back along it counting fish 5 m either side of the tape. In contrast, the 25 × 5 m transect counts were done by attaching the tape to the substratum and swimming the tape out while counting. A 5 m ‘lead-in’ was swum before commencing counts to avoid including fish attracted to the diver while the tape was being attached. Thus, the two transect types differed in both area censused and counting protocol. Several planktivorous species were omitted from all UVC transect counts, because they were either so common or so patchily distributed that their inclusion would have caused decreases in the accuracy of counts of the rest of the assemblage. Schooling species were usually recorded as ‘present’. These species were: demoiselle (*Chromis dispilus*), trevally (*Pseudocaranx dentex*), koheru (*Decapterus koheru*), and jack mackerel (*Trachurus* spp.).

The precision (p) of mean estimates is defined as the ratio of the standard error to the mean, and is a useful means of comparing the amount of variance around means of varying magnitude, and hence the ability of the researcher to detect differences between those means, should they exist (Andrew & Mapstone 1987). In this study, fish density estimates made using 25 × 5 m and 50 × 10 m transects were calculated using the formula

$$p = \left(s / \sqrt{n}\right) / \bar{x}$$

where s = sample standard deviation, n = number of replicates, and x = mean density (McCormick & Choat 1987, Andrew & Mapstone 1987, Riddle 1989). The replication (n) necessary for a given precision (in this case, p = 0.15, 0.2) was calculated by rearrangement of the above equation so that

$$n = \left(s / p\bar{x}\right)^2$$

An arbitrary sampling precision of 0.2 or less is often regarded as appropriate for ecological studies (e.g. Downing 1989, Riddle 1989), and we follow this convention. Limited time in the field meant that the sample sizes we could obtain for this comparison were low (50 m transects: n = 6 per site, 25 m transects: n = 9 per site) and hence precision of mean density of individual species was correspondingly low, excepting the black angelfish (*Parma alboscapularis*). Consequently, we compare the precision of the transect types by pooling the remainder of the sample species into two groups: the Labridae, and other non-cryptic reef fishes.

2.2 Effect of time of day on underwater visual census density estimates

If levels of fish activity vary during the day, it may be possible to obtain significant differences between sites which are in fact similar, due merely to counts being conducted at different times. In a large-scale survey, this effect (if it exists) may be reduced by ensuring that sites nested within survey areas are not consistently sampled at given times. For example, it may be convenient, but poor practice, to survey the closest sites to home base first thing in the morning. An *a priori* randomisation of the order in which nested sites are surveyed is appropriate, unless the fish assemblage is known not to vary throughout the day. This is unlikely unless only specific components of the assemblage are to be examined and crepuscular species are deliberately excluded.

Two locations were chosen (Nursery Cove and Labrid Channel) which we knew *a priori* possessed high densities of reef fishes. Two divers completed six replicate 25 × 5 m transects (as described above) at deep (>15 m) and shallow (3 – 12 m) sites at both locations at three times (0730 hrs, 1030 hrs and 1400 hrs) on 24 September 1998.

The factors Location (Nursery and Labrid), Depth (Deep and Shallow) and Time were treated as fixed factors in a three-way Analysis of Variance (ANOVA). The most common species (*Parma alboscapularis* and *Coris sandageri*) were examined separately, and the remaining species were pooled into Labridae (including *C. sandageri*) and reef fish (excluding Labridae and *P. alboscapularis*).

We employed backward model fitting, where saturated models (with all factors and interactions) were fitted in the first instance, and non-significant interactions progressively removed to release degrees of freedom for examination of the main effects. Homogeneity of variance was checked by examination of residual plots and the Shapiro-Wilks test. Natural log [$\ln(x+0.01)$] transformation of the data was performed to help variances approach normality where necessary.

2.3 Underwater visual census site/habitat comparisons

2.3.1 Within the Poor Knights Islands

UVC surveys of reef fish were done using 125 m^2 (25 × 5 m) transects as described above. Each site was surveyed by three divers, who each completed three transects, giving n = 9 transects per site. Sites were nested within larger survey areas (Appendix 2) for geographical comparisons, and broad habitat comparisons were made on the basis of depth and topography (e.g. shallow sloping reef, wall, archway or cave). It was originally intended to stratify reef fish counts by more specific habitat characterisations (e.g. *Ecklonia radiata* forest, *Carpophyllum* forest, urchin-grazed rock flat), but habitat patches were frequently so small that even the short transect length employed would straddle more than one habitat type.

The three surveys generally repeated the same sites, although some (eg. Te Paki Point) were surveyed only once due to poor weather conditions on subsequent trips.

Changes in the relative density of selected species were estimated using a generalised log-linear model (see BUV methods, Section 2.4.2). Ratios of change in density after one year were generated, and depth effects estimated where they occurred consistently. Statistical significance of ratios does not necessarily imply that the observed changes are biologically meaningful, given the inherent variability of density data obtained from mobile species such as fishes. We therefore considered increases or decreases of a magnitude greater than 100% (ie at least a doubling or halving of abundance) to be the minimum biologically significant change.

The level of similarity between fish assemblages at different sites and times was assessed using non-metric multidimensional scaling (MDS, Clarke 1993). Transect data were pooled at the site level, then square-root transformed before generating a Bray-Curtis similarity matrix (Bray & Curtis 1957). The resulting matrix was agglomeratively clustered using the group-average method (Clarke 1993), and this was graphically presented by MDS ordination using the PRIMER statistical package (Clarke & Warwick 1994). Effects of site, broad-scale habitat type, and survey were examined using non-parametric multivariate analysis of variance (NP-MANOVA, Anderson 2001). Count data very rarely conform to the assumptions of traditional parametric MANOVA. NP-MANOVA is a new analytical method, based on permutation tests using distance matrices such as Bray-Curtis similarities. It surpasses previously widely employed non-parametric analyses (eg. Clarke 1993) in its ability to partition variation across multiple factors which form part of the experimental design (Anderson 2001).

2.3.2 Between Cape Brett, the Mokohinau and Poor Knights Islands

Comparison of reef fish abundance and assemblage structure was made from surveys of Cape Brett, the Mokohinau and Poor Knights Islands using similar analyses to those described above, except that the level ‘location’ was added. The three locations were surveyed during September and October 1999.

2.4 Patterns in abundance of carnivorous fishes from Baited Underwater Video

Baited underwater video (BUV) surveys of carnivorous fish species were conducted concurrently with the UVC surveys. The system (described in Willis & Babcock 1998, 2000) was deployed from the research vessel to depths up to 50 m at sites at least 1 km from diving activities (so the presence of divers would not interfere with fish responses to the bait). Each sequence was recorded for 30 min from the time the video assembly reached bottom. At least five replicate BUV deployments were done in each of the four survey areas (Appendix 1), but perfectly balanced replication between areas was not possible as weather conditions influenced specific sites surveyed.

2.4.1 Analysis of video footage

At the laboratory, 8mm videotapes were copied to VHS tapes for analysis and archiving. Videotapes were played back with a real-time counter, and the maximum number of each species of fish observed during each minute was recorded (thirty counts made during each 30-minute sequence). Only fish visible on the screen at any one time were recorded to avoid counting the same fish twice. The lengths of snapper (*Pagrus auratus*) and tarakihi (*Nemadactylus macropterus*) were obtained by digitising video images using the Mocha® image analysis system (Jandel Corporation), and obtaining a three-point calibration (to compensate for wide-angle distortion) for each image using the marks visible on the stand. Measurements were only made of those fish present when the count of the maximum number of fish of a given species in a sequence (eg MAXsna) was made. While this means that some fish moving in and out of the field of view may not have been measured, it also avoids repeated measurements of the same individuals. It is likely that this approach results in more conservative abundance estimates in high density areas than low density areas, and therefore observed relative differences between sites are also likely to be conservative.

2.4.2 Statistical analysis of video data

Changes in the abundance of snapper at the Poor Knights was examined using a generalised log-linear model (Millar & Willis 1999, Willis et al. 2000), as the data are counts and hence do not satisfy the assumptions of normality and homogeneity of variance that are required by ANOVA. Such data are best modelled using the Poisson distribution, or more generally, as Poisson with possible overdispersion due to the fact that fish may not behave independently of each other. The log-linear model was fitted using maximum likelihood by the SAS procedure GENMOD. This expresses the fish counts, Y, as

$$Y \sim Poisson(\lambda)$$

where $Poisson(\lambda)$ denotes a (possibly overdispersed) Poisson distribution with expected value of λ, and $\log(\lambda)$ is modelled as a linear function of the effects. For example, the expected count of fish in replicate j in area i at time (or survey) k is modelled (where i and k are fixed effects) by

$$\log(\lambda_{ijk}) = \alpha_i + \delta_k$$

where α and δ are area and time parameters, respectively, to be estimated. In this way, estimates of changes in abundance (relative to the pre-reserve survey in September 1998) are calculated as a ratio, and log-scale error estimates around each ratio are provided for selected fish species. In this case snapper (*Pagrus auratus*), yellow and grey moray eels (*Gymnothorax prasinus* and *G. nubilis*), tarakihi (*Nemadactylus macropterus*), northern scorpionfish (*Scorpaena cardinalis*), and red pigfish (*Bodianus unimaculatus*) exhibited consistency in response to bait and occurred in sufficient numbers for this analysis. A saturated model was run for each species, and if there was no interaction between time and survey area, the model was run again with main effects only.

Mean snapper size was compared between the three Poor Knights surveys, and between the September 1999 Poor Knights survey and the Mokohinau Islands and Cape Brett surveys. This was done using one-way analysis of variance (ANOVA) on natural log-transformed snapper length data (to homogenise variances). Pairwise Tukey (highest significant difference) comparisons were performed when ANOVA was significant, to determine which of the samples differed from the others.

Size and abundance measures of snapper were combined by modelling changes in estimated relative biomass (expressed as kg per BUV deployment). Snapper weight was estimated from measured lengths using the equation (from Taylor & Willis 1998):

$$W = 0.00007194 \times L^{2.793}$$

where W is estimated weight (g) and L is fish length (mm). Weights were modelled in the same way as snapper numbers, that is, with a Poisson distributed log-linear model.

3. Results

3.1 Effect of underwater visual census transect type on density estimates

Density data from 500 m^2 and 125 m^2 transects were expressed as fish 100 m^{-2} for comparison. Estimated densities of labrids and reef fish in 125 m^2 transects were at least two times and 4.5 times that estimated from 500 m^2 transects in both depth strata, respectively (Table 1). No *Parma alboscapularis* were counted in deep transects, but density in the shallow stratum from 125 m^2 transects was 3.8 times the 500 m^2 transect estimate (Table 1).

Precision of 125 m^2 estimates was generally better than 500 m^2 estimates (Table 1), and where the value of p was lower for 500 m^2 means, it could be partly attributed to the distribution of values around a lower mean (ie values are bounded by a lower limit of zero). The arbitrarily specified precision of 0.2 could generally be achieved by n = 9 to 12 125 m^2 transects, whereas a logistically difficult number of 500 m^2 transects (eg. for Labridae at deep sites) would be required for the same level of precision. Assuming (conservatively) that 500 m^2 transects take twice as long to complete as 125 m^2 transects, in all but one case it would take less field time to obtain estimates of comparable precision using 125 m^2 transects. In practice, small scale habitat patchiness at the Poor Knights made it very difficult to constrain 50 m transects within a particular gross habitat type.

3.2 Effect of time of day on underwater visual census density estimates

Elimination of non-significant terms in ANOVA models did not alter the statistical significance of any main effects, so the full (saturated) models are presented in this section. Depth and location were the main factors that significantly affected the density of *Parma alboscapularis*, with the highest densities at both sites occurring in the shallow stratum, and with Labrid Channel generally supporting greater numbers than Nursery Cove. (Table 2). There was a slight, but non-significant, trend for shallow densities to decrease at both sites during the day and for the Labrid deep densities to increase slightly (Fig. 1a). Time was not a significant factor overall and interaction terms were also not significant. The fitted model accounted for 76.8% of the observed variance.

Table 1. Comparison of fish density estimates from 25 × 5 ($n = 9$) and 50 × 10m ($n = 6$) transects in shallow (5-12 m depth) and deep (15-26 m depth) habitats at Nursery Cove. *Parma* is the black angelfish (*Parma alboscapularis*), Labridae = pooled mean of all wrasses counted, and Reef fish = all other non-cryptic reef fishes pooled. Means are expressed as fish 100 m^{-2}, SE = standard error of the mean, p = precision, and n(x) = the number of replicate transects required to achieve $p = x$.

Group	Depth	Transect size (m^2)	Mean (fish 100 m^{-2})	SE	*p*	n(0.2)	n(0.15)
Parma	Shallow	125	1.78	0.37	0.21	10	18
		500	0.47	0.15	0.33	16	29
	Deep	125	0				
		500	0				
Labridae	Shallow	125	5.42	1.05	0.19	9	15
		500	2.20	0.30	0.14	3	6
	Deep	125	3.55	0.71	0.20	9	16
		500	1.70	0.63	0.37	21	38
Reef fish	Shallow	125	3.11	0.71	0.23	12	21
		500	0.67	0.22	0.32	16	29
	Deep	125	5.07	0.87	0.17	7	12
		500	1.07	0.16	0.15	4	6

Density of *Coris sandageri* was also generally higher in shallow strata, and higher at Labrid Channel than at Nursery Cove (Fig. 1b). There was a significant interaction between depth and time (Table 3), caused by a distinct increase in measured density with time in the Labrid Channel deep site, and a similar decrease at the Nursery Cove shallow site (Fig. 1b). Since these trends are not consistent within depths or locations, interpretation of the main effects is confounded and we cannot say whether time of day might affect *C. sandageri* density. The model accounted for 55.2% of the variance.

Examination of all species of Labridae (including *C. sandageri*) as a functional group gave similar results to those obtained for *P. alboscapularis*. There were no significant interactions between factors, but location and depth were significant main effects (Table 4). The model accounted for 62.4% of the variance, and the largest component of this was differences between sites, with Labrid Channel again having higher densities (Fig. 1c).

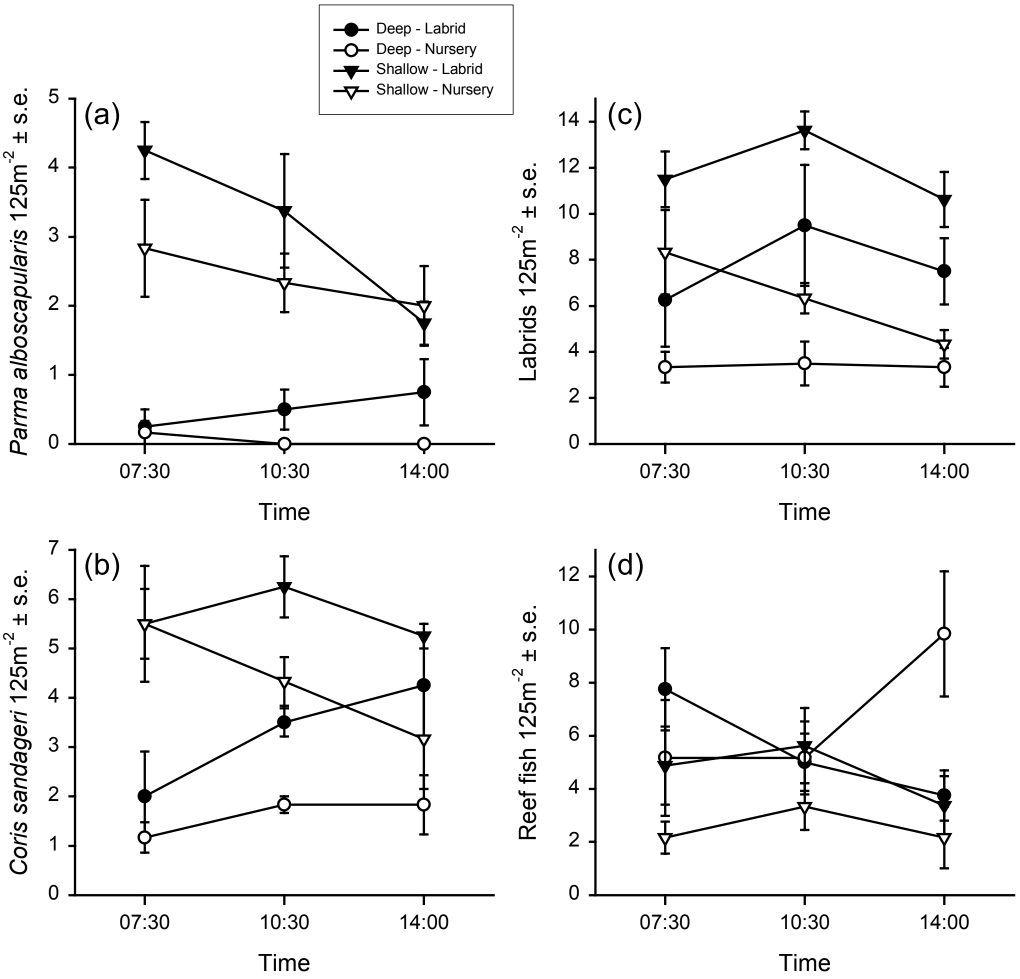

Figure 1. Mean UVC estimates at three times of day of (a)*Parma alboscapularis*, (b) *Coris sandageri*, (c) all Labridae, and (d) all other reef fish within deep and shallow sites at Nursery Cove and Labrid Channel.

Table 2. Time of day comparison: black angelfish *Parma alboscapularis*, ln(x+0.01) transformed, W=0.919, p<0.01 due to one outlier on residual plot.

Source	DF	SS	MS	F	Sig.
Location	1	19.70	19.70	10.33	**
Depth	1	301.62	301.61	158.09	***
Time	2	1.68	0.84	0.44	ns
Loc×Depth	1	7.06	7.06	3.70	ns
Loc×Time	2	3.58	1.79	0.94	ns
Depth×Time	2	4.61	2.30	1.21	ns
Loc×Depth×Time	2	4.26	2.13	1.12	ns
Error	60	114.47			
Total	71	493.97			

Table 3. Time of day comparison: Sandager's wrasse *Coris sandageri*, raw data W=0.972, ns

Source	DF	SS	MS	F	Sig.
Location	1	37.41	37.41	13.13	***
Depth	1	111.85	111.85	39.26	***
Time	2	2.44	1.22	0.43	ns
Loc×Depth	1	0.39	0.39	0.14	ns
Loc×Time	2	10.28	5.14	1.80	ns
Depth×Time	2	21.38	10.69	3.75	*
Loc×Depth×Time	2	0.83	0.41	0.15	ns
Error	60	170.92	2.85		
Total	71	381.50			

Table 4. Time of day comparison: Labridae, raw data W=0.969, ns

Source	DF	SS	MS	F	Sig.
Location	1	418.84	418.84	46.14	***
Depth	1	214.17	214.17	23.59	***
Time	2	36.25	18.13	2.00	ns
Loc×Depth	1	6.33	6.33	0.70	ns
Loc×Time	2	37.24	18.62	2.05	ns
Depth×Time	2	226.53	13.26	1.46	ns
Loc×Depth×Time	2	2.49	1.24	0.14	ns
Error	60	544.67			
Total	71	1447.28			

The remainder of the non-cryptic reef fishes occurred in densities of 2.2 - 9.8 fish 125 m^{-2} (Fig 1d). Depth was the only significant main effect (Table 5), which was probably due to high numbers of goatfish (*Upeneichthys lineatus*) that occurred in groups at the deep sites. Time of day had no significant effect. The lack of pattern was reflected in the low proportion of variance (28.1%) accounted for by the model.

Table 5. Time of day comparison: all reef fishes, raw data W=0.974, ns. *$p<0.05$, **$p<0.01$, ***$p<0.001$

Source	DF	SS	MS	F	Sig.
Location	1	3.04	3.04	0.23	ns
Depth	1	107.65	107.65	8.12	**
Time	2	0.65	0.33	0.02	ns
Loc×Depth	1	45.89	45.89	3.46	ns
Loc×Time	2	76.42	38.21	2.88	ns
Depth×Time	2	34.43	17.21	1.30	ns
Loc×Depth×Time	2	37.72	18.86	1.42	ns
Error	60	795.62	13.26		
Total	71	1106.44			

3.3 Patterns in reef fish assemblages - UVC site/habitat comparisons

3.3.1 At the individual species level – univariate analyses

Twelve fish species were examined for changes in relative density over time at the Poor Knights Islands using UVC data. Mean density (fish 125 m^{-2} transect ± standard error) of each fish was plotted within three depth strata (Shallow, ≤ 10 m; Mid, 11-20 m; Deep > 20 m) for each survey (Figs 2-4). Ratio estimates of change in density at the Poor Knights between September 1998 and September 1999 (Table 6), and between the Poor Knights, Mokohinau, and Cape Brett in September 1999 (Table 7) were obtained using log-linear modelling. The saturated model was fitted first, and then the model was run again with non-significant interactions between survey and depth removed. Reef fish density estimates can be affected by small-scale spatial and temporal variability, caused in part by habitat patchiness as well as their mobility. The statistical significance of a difference between two samples therefore does not necessarily imply a real biological change. As a result, in this report we only regard changes of the magnitude of 100% (ie a doubling or halving of density) as being indicative of biologically significant difference.

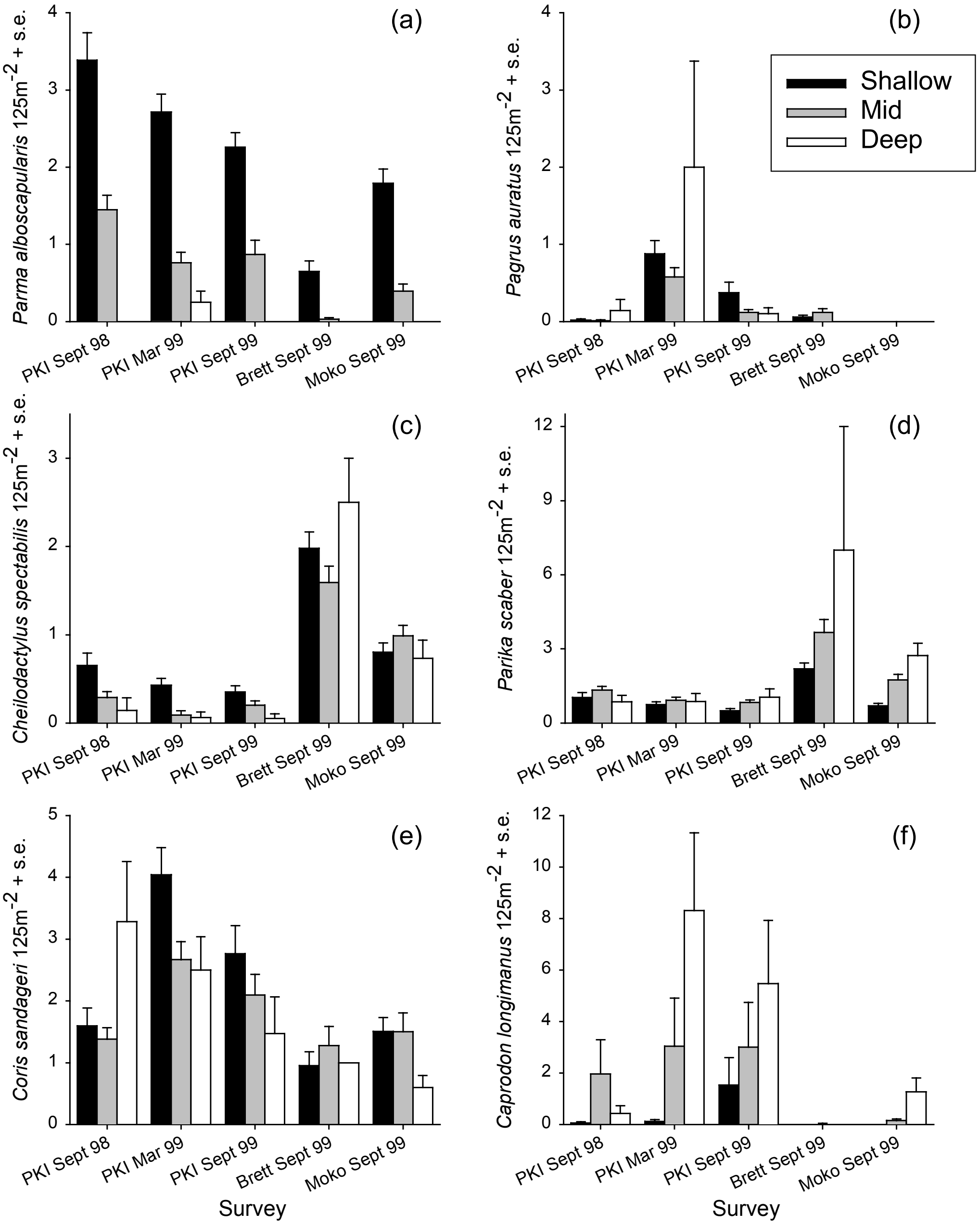

Figure 2. Mean density (fish $125m^{-2}$ + standard error) of fishes in three depth strata (Shallow: ≤ 10m, Mid: 11-20m, Deep: > 20m) for each survey of the Poor Knights Islands, Mokohinau Islands, and Cape Brett. (a) *Parma alboscapularis*, (b) *Pagrus auratus*, (c) *Cheilodactylus spectabilis*, (d) *Parika scaber*, (e) *Coris sandageri*, (f) *Caprodon longimanus*.

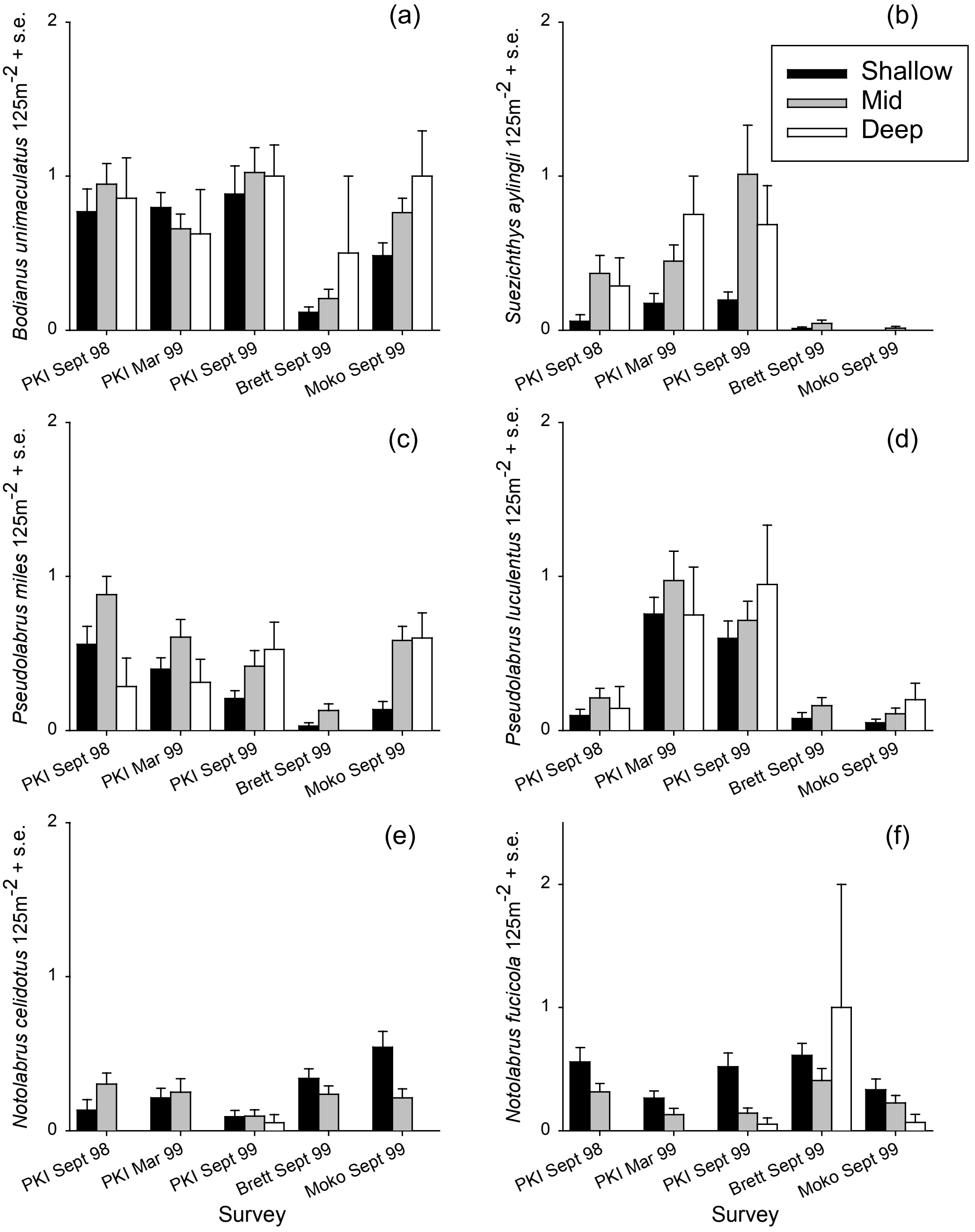

Figure 3. Mean density (fish 125m^{-2} + standard error) of fishes in three depth strata (Shallow: ≤ 10m, Mid: 11-20m, Deep: > 20m) for each survey of the Poor Knights Islands, Mokohinau Islands, and Cape Brett. (a) *Bodianus unimaculatus*, (b) *Suezichthys aylingi*, (c) *Pseudolabrus miles*, (d) *Pseudolabrus luculentus*, (e) *Notolabrus celidotus*, (f) *Notolabrus fucicola*.

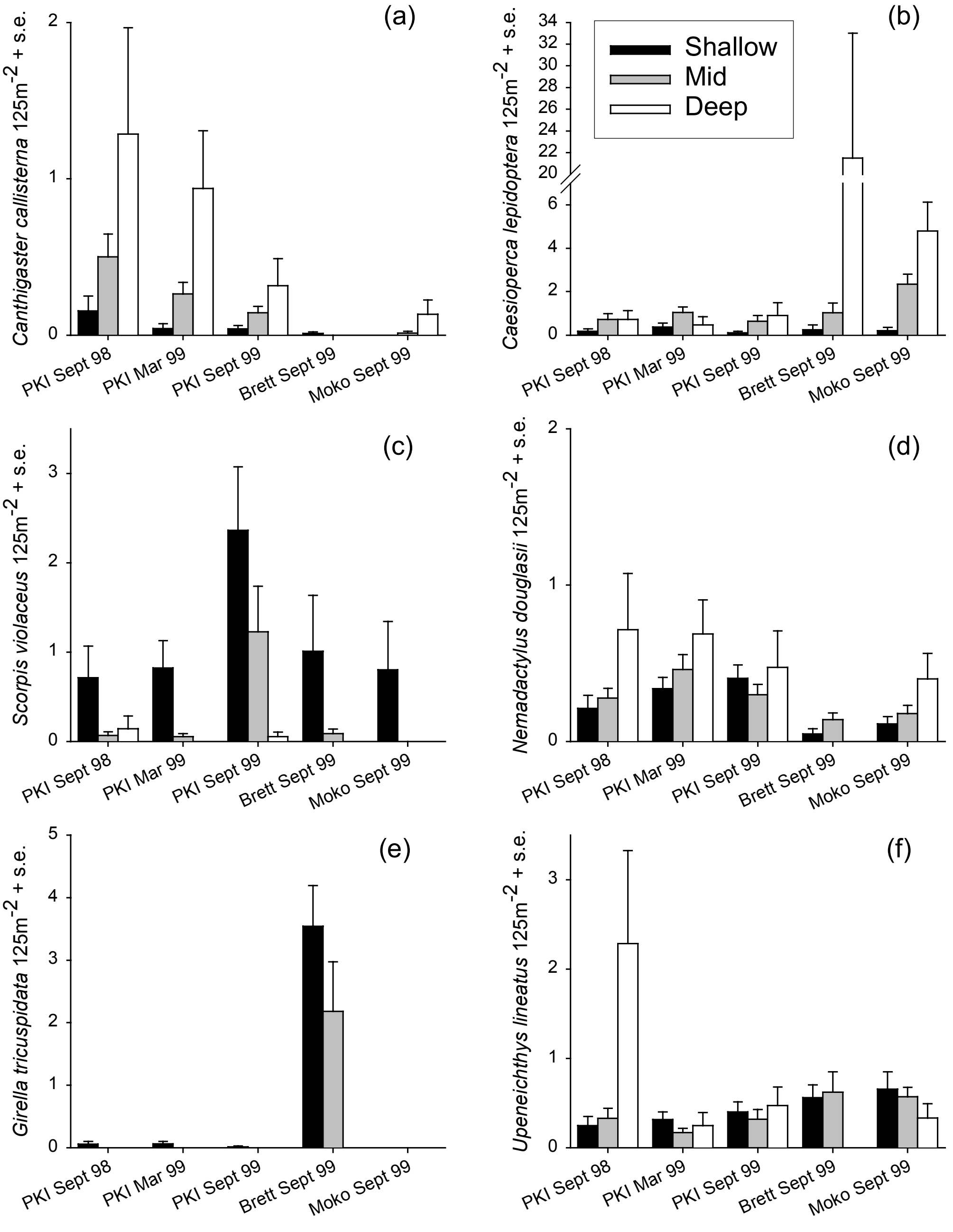

Figure 4. Mean density (fish 125m^{-2} + standard error) of fishes in three depth strata (Shallow: ≤ 10m, Mid: 11-20m, Deep: > 20m) for each survey of the Poor Knights Islands, Mokohinau Islands, and Cape Brett. (a) *Canthigaster callisterna*, (b) *Caesioperca lepidoptera*, (c) *Scorpis violaceus*, (d) *Nemadactylus douglasii*, (e) *Girella tricuspidata*, (f) *Upeneichthys lineatus*.

Table 6. Log-linear model estimates of annual change in fish density at the Poor Knights estimated by UVC, between September 1998 (before complete reserve closure) and September 1999. χ^2 values in bold type indicate a statistically significant change ($p < 0.05$) in relative density. Ratios in bold type we consider to be biologically significant (see text). 'No fit' means that the model algorithm did not converge.

Species	Ratio	Lower 95% CI for ratio	Upper 95% CI for ratio	χ^2
Bodianus unimaculatus	1.11	0.83	1.47	0.48
Caesioperca lepidoptera	0.90	0.55	1.48	0.18
Canthigaster callisterna	**0.26**	**0.16**	**0.42**	**31.39**
Caprodon longimanus	**2.29**	**1.25**	**4.21**	**7.21**
Cheilodactylus spectabilis	0.64	0.51	0.80	**8.53**
Coris sandageri	1.46	1.09	1.96	**6.36**
Girella tricuspidata	No fit			
Nemadactylus douglasii	1.28	0.85	1.92	1.36
Notolabrus celidotus	**0.43**	**0.26**	**0.73**	**10.00**
Notolabrus fucicola	0.74	0.52	1.06	2.70
Pagrus auratus	**9.31**	**2.53**	**34.30**	**11.26**
Parika scaber	0.60	0.46	0.79	**12.98**
Parma alboscapularis	0.64	0.51	0.80	**14.69**
Pseudolabrus luculentus	**4.27**	**2.51**	**7.28**	**28.54**
Pseudolabrus miles	0.49	0.34	0.69	**16.73**
Scorpis violaceus	**5.14**	**2.32**	**11.41**	**16.26**
Suezichthys aylingi	**2.71**	**1.69**	**4.33**	**17.17**
Upeneichthys lineatus	No fit			

For the time comparison (within the Poor Knights) there were significant interactions between survey and depth for *Pagrus auratus*, *Caprodon longimanus*, and *Coris sandageri* (Fig. 2b,e,f) due to high variability obtained from counts within the deep depth stratum, and for *Scorpis violaceus* because it only occurred at shallow sites at the Mokohinau Islands. As the deep sites constituted a relatively small part of the dataset (because of safety constraints on deep diving), and were therefore not likely to contribute much to the variance of the overall model, we examined the main effect time regardless of the significant interaction. Seven of the 18 species examined changed in density from September 1998 to September 1999, five of which were increases and two (*Notolabrus celidotus* and *Canthigaster callisterna*) decreased (Table 6). Increases in density of *Pseudolabrus luculentus*, *Suezichthys aylingi* and *Coris sandageri* appear to be attributable to a strong pulse of recruitment over the summer of 1998/99, which is illustrated for *C. sandageri* by the proportionally greater contributions of juveniles to counts in 1999 (Fig. 5). It is possible that the drop in density of *N. celidotus* is the result of either competitive interactions with *P. luculentus*, or an inverse relationship in larval supply between the two species dictated by the prevailing water mass (ie the East Auckland Current vs water of coastal origin). As the source of *C. callisterna* recruits is likely to be subtropical, it is possible that the observed decrease in the density of this species (during a time when it may be expected to have increased) may be due to a relative increase in natural mortality coupled with local recruitment failure.

Table 7. Log-linear model estimates of differences in fish density between the Poor Knights, Mokohinau Islands, and Cape Brett estimated by UVC in September 1999. Ratios are estimated for the two island locations (Poor Knights and Mokohinaus) relative to the mainland location (Cape Brett). χ^2 values in bold type indicate a statistically significant change ($p < 0.01$) in relative density. Ratios in bold type we consider to be biologically significant (see text). 'No fit' means that the model algorithm did not converge.

	Mokohinau Islands				Poor Knights Islands			
Species	Ratio	Lower 95% CI for ratio	Upper 95% CI for ratio	χ^2	Ratio	Lower 95% CI for ratio	Upper 95% CI for ratio	χ^2
Bodianus unimaculatus	**3.94**	**2.59**	**6.00**	**40.79**	**5.72**	**3.81**	**8.60**	**70.47**
Caesioperca lepidoptera	1.40	0.97	2.02	3.25	**0.35**	**0.21**	**0.58**	**16.19**
Canthigaster callisterna	2.57	0.89	7.45	3.04	**16.88**	**6.55**	**43.47**	**34.28**
Caprodon longimanus	15.23	0.62	375.9	2.77	**216.6**	**9.59**	**4895.01**	**11.43**
Cheilodactylus spectabilis	**0.50**	**0.40**	**0.62**	**38.90**	**0.14**	**0.10**	**0.20**	**112.85**
Coris sandageri	1.34	0.97	1.84	3.15	**2.19**	**1.64**	**2.94**	**27.73**
Girella tricuspidata	No fit				No fit			
Nemadactylus douglasii	1.73	1.04	2.89	4.56	**3.71**	**2.35**	**5.86**	**31.66**
Notolabrus celidotus	1.30	0.94	1.80	2.60	**0.35**	**0.21**	**0.57**	**17.55**
Notolabrus fucicola	0.54	0.37	0.77	**11.27**	0.62	0.44	0.88	**7.20**
Pagrus auratus	No fit				**2.88**	**1.93**	**4.30**	**26.81**
Parika scaber	**0.42**	**0.34**	**0.53**	**56.16**	**0.22**	**0.16**	**0.29**	**109.11**
Parma alboscapularis	**3.14**	**2.25**	**4.38**	**45.44**	**4.53**	**3.29**	**6.24**	**86.00**
Pseudolabrus luculentus	0.73	0.41	1.31	1.10	**5.58**	**3.69**	**8.43**	**66.48**
Pseudolabrus miles	**4.67**	**2.84**	**7.67**	**36.96**	**4.03**	**2.43**	**6.67**	**29.37**
Scorpis violaceus	0.71	0.37	1.36	1.05	**3.26**	**2.05**	**5.16**	**25.28**
Suezichthys aylingi	0.21	0.03	1.38	2.65	**23.14**	**10.51**	**50.96**	**60.82**
Upeneichthys lineatus	1.02	0.72	1.45	0.01	0.64	0.43	0.97	4.36

The three other increases were exhibited by species exploited by recreational fishers: snapper *Pagrus auratus* (831%) blue maomao *Scorpis violaceus* (414%) and pink maomao *Caprodon longimanus* (129%). The estimated nine-fold increase in the former we regard with some caution, given the large confidence intervals surrounding the estimate and known biases associated with visual censuses of this species (Willis et al. 2000).

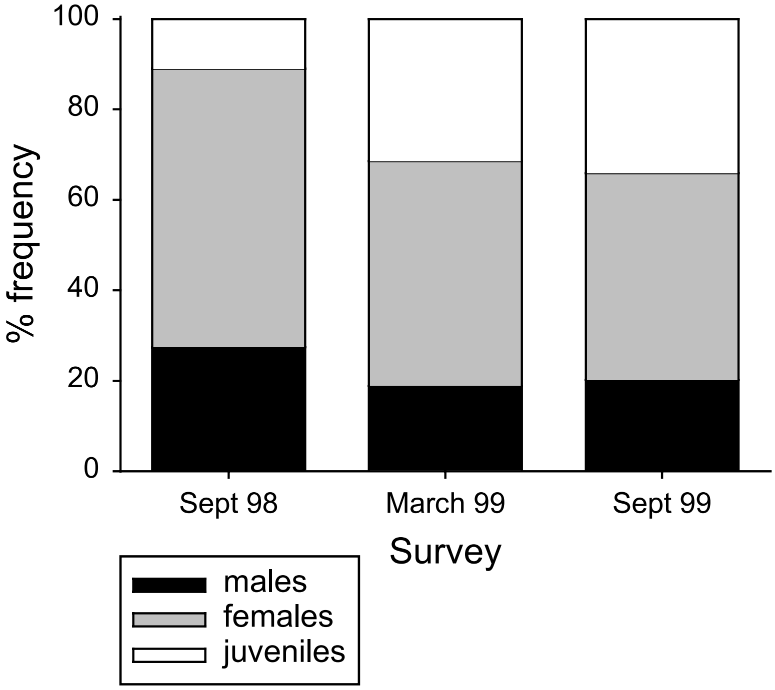

Figure 5. Percentage contribution of male, female, and juvenile *Coris sandageri* to UVC counts at the Poor Knights.

In September 1999, density of all species except *Notolabrus fucicola* and *Upeneichthys lineatus* differed between Cape Brett and the Poor Knights (Table 7). Generally, species conformed to the large-scale distributional patterns previously described (Choat & Ayling 1987, Choat et al. 1988, Willis 1995). Both island locations had significantly lower densities of red moki *Cheilodactylus spectabilis* and leatherjackets *Parika scaber* than the mainland site, but higher densities of red pigfish *Bodianus unimaculatus*, scarlet wrasse *Pseudolabrus miles*, clown toado *Canthigaster callisterna* and the black angelfish *Parma alboscapularis* (Table 7). There could be no ratio estimate for *Pagrus auratus* at the Mokohinau Islands because no snapper were counted in transects at that site. Similarly, *Girella tricuspidata* occurred at high densities at Cape Brett, but was all but absent from the other surveys. Despite the overlap in species composition, the relative densities of fishes at the Poor Knights Islands and Cape Brett are fundamentally different, with the Mokohinau Islands lying somewhere in between.

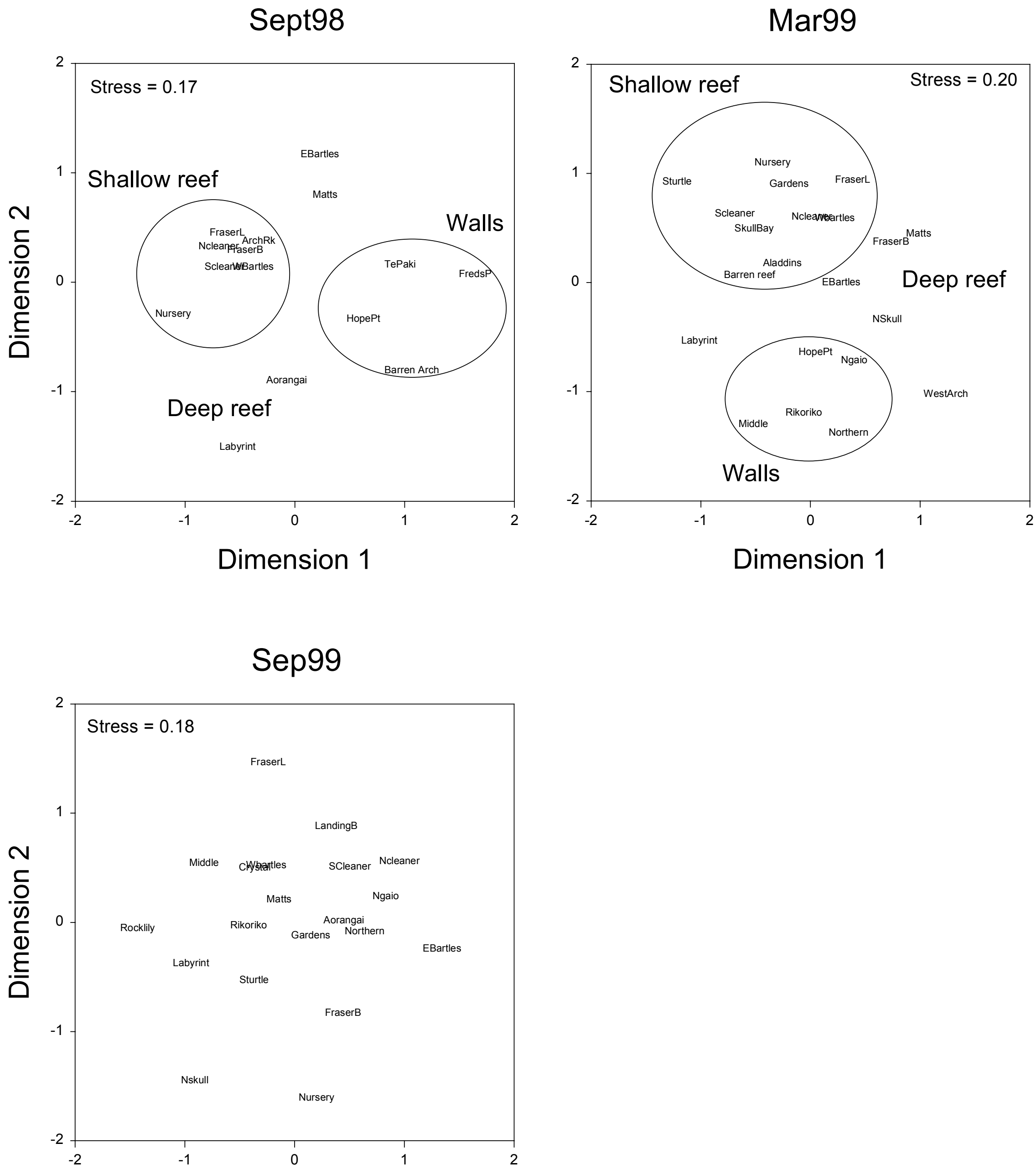

Figure 6. MDS ordination of sites surveyed by UVC at the Poor Knights Islands, by survey.

3.3.2 At the assemblage level – multivariate analyses

Cluster analysis and MDS ordination of sites from the September 1998 and March 1999 Poor Knights surveys differentiated general habitat types (deep reef, shallow reef, walls and overhangs) well (Fig. 6). Sites were not well separated in the September 1999 survey, which may be due to reductions in the densities of species characteristic of particular general habitats in this survey e.g. black angelfish *Parma alboscapularis* on shallow reefs, and the clown toado *Canthigaster callisterna* around walls and deep reefs. MDS ordination of the three Poor Knights surveys did not appear to clearly separate them (Fig. 7a), although a one-way NP-MANOVA did detect a significant difference ($F_{2,42} = 2.47$, $p = 0.003$), which pairwise *a posteriori* comparisons showed was due to differences between the pre-reserve survey (September 1998) and the subsequent two surveys (which were not significantly different from each other).

Ordination of sites sampled at the Poor Knights, Mokohinaus and Cape Brett (using 62 fish species) in September 1999 separated the three locations well (Fig. 7b), and these differences were statistically significant (one-way NP-MANOVA, $F_{2,57} = 10.56$, $p = 0.003$). The lack of overlap between locations in the ordination suggests that some elements of the fish assemblage are consistently different (either through composition or density) at the three survey locations. Examination of the depth-stratified plots (Figs. 2-4) shows that parore *Girella tricuspidata*, red moki *Cheilodactylus spectabilis* and leatherjackets *Parika scaber* are much more abundant at Cape Brett than either of the island locations. Conversely, *Pseudolabrus miles*, *Bodianus unimaculatus*, *P. alboscapularis*, *Nemadactylus douglasii* and *Coris sandageri* are more abundant at the islands than Cape Brett. The Poor Knights are differentiated from the Mokohinau Islands by relatively low densities of the spotty *Notolabrus celidotus*, and higher densities of *Caprodon longimanus*, *Pseudolabrus luculentus* and *Suezichthys aylingi*, as well as low numbers of subtropical species (e.g. *C. callisterna*, *Coris picta*) that are absent or very rare at the Mokohinaus and Cape Brett.

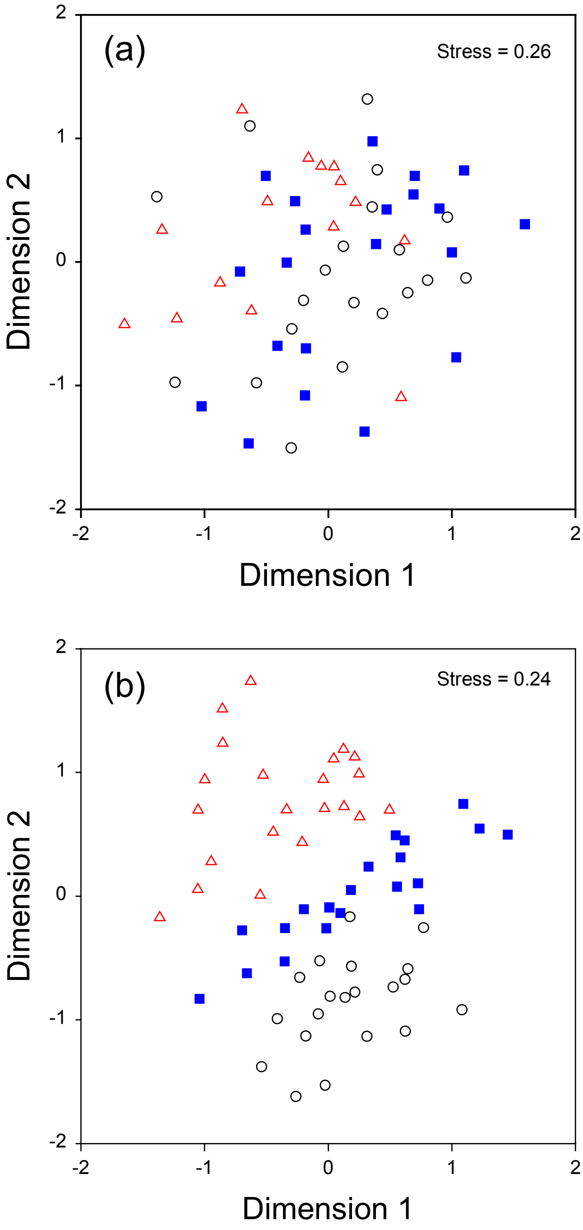

Figure 7. MDS ordination of sites (pooled replicates), based on UVC surveys of 62 species of reef fish (a) at the Poor Knights Islands in September 1998 (triangles), March 1999 (squares) and September 1999 (circles), and (b) during September/October 1999, at the Poor Knights Islands (circles), Mokohinau Islands (squares) and Cape Brett (triangles).

3.4 Comparison of carnivorous fish density using BUV

3.4.1 Within the Poor Knights Islands

Changes in the relative density of five carnivorous species were modelled from BUV data. Many other species occurred on the video footage, but snapper, tarakihi, red pigfish, northern scorpionfish and moray eels showed strong responses to the bait, and are species known to be vulnerable to angling. Relatively low numbers of snapper were detected in the September 1998 survey, but six months after complete closure, relative abundance was estimated to have risen by a factor of 6.3 (or 530 %). The following September, abundance had dropped, but was still over four times greater than the September 1998 estimate, an annual increase of 300% (Fig. 8). Conversely, tarakihi numbers dropped between the first and second surveys, but in September 1999 recovered to twice the density of a year earlier (Fig. 8), whereas pigfish significantly decreased (by 30 %) in estimated density. There was no significant change in the relative density of scorpionfish, or moray eels (Fig. 8). Table 8 gives ratios of September 1998 (pre-closure): September 1999 fish abundance with estimates confidence intervals for the five species. It should be noted that confidence intervals around ratios and density estimates generated from log-linear models are calculated on the log scale, and hence are multiplicative on the arithmetic scale (therefore the error bars are of unequal length). With the exception of tarakihi, there was no significant survey × area interaction for any species, so ratios were obtained from fixed effects (survey and area) models without the interaction term.

Table 8. Log-linear model estimates of change in carnivorous fish density at the Poor Knights, estimated by BUV between September 1998 (before complete reserve closure) and September 1999. χ^2 values in bold type indicate a statistically significant change ($p < 0.05$) in relative density.

Species	Ratio	Lower 95% CI for ratio	Upper 95% CI for ratio	χ^2
Pagrus auratus	4.02	2.25	7.18	**22.13**
Nemadactylus macropterus	2.01	1.05	3.85	**4.51**
Bodianus unimaculatus	0.69	0.55	0.88	**8.78**
Scorpaena cardinalis	1.21	0.69	2.10	0.45
Gymnothorax spp.	0.72	0.48	1.09	2.42

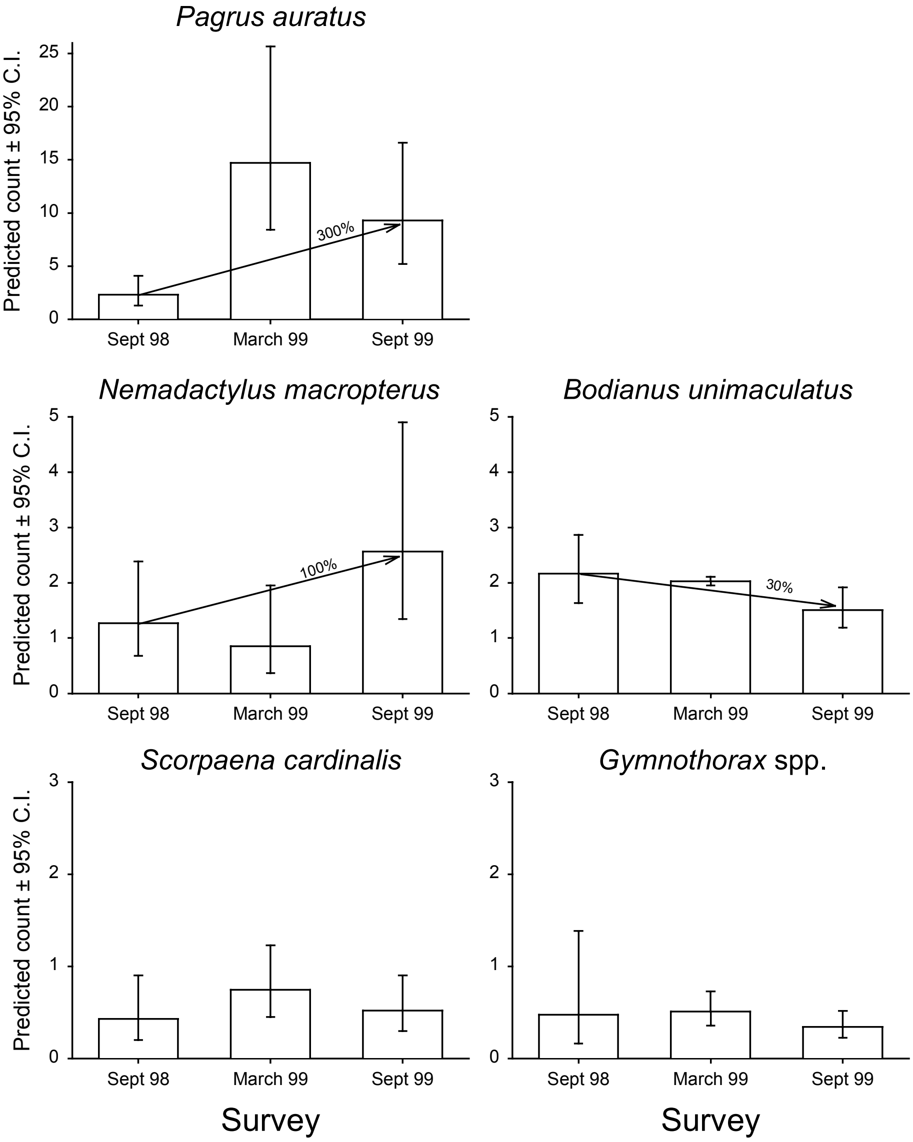

Figure 8. Model estimates (expressed as the expected number of fish per BUV deployment) of abundance of carnivorous species over time at the Poor Knights Islands. Significant increases in abundance are indicated on the plots.

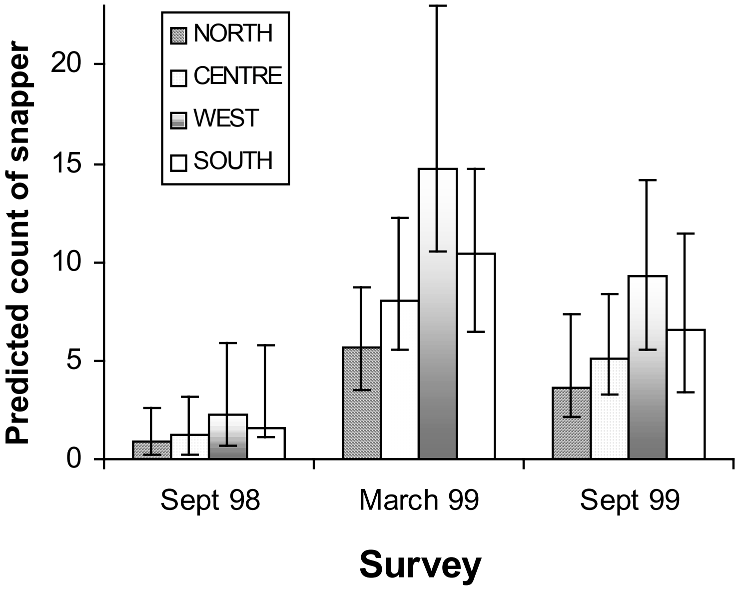

Figure 9. Snapper relative density in the four survey areas, estimated from generalised linear modelling of BUV data. Error bars are 95% confidence intervals.

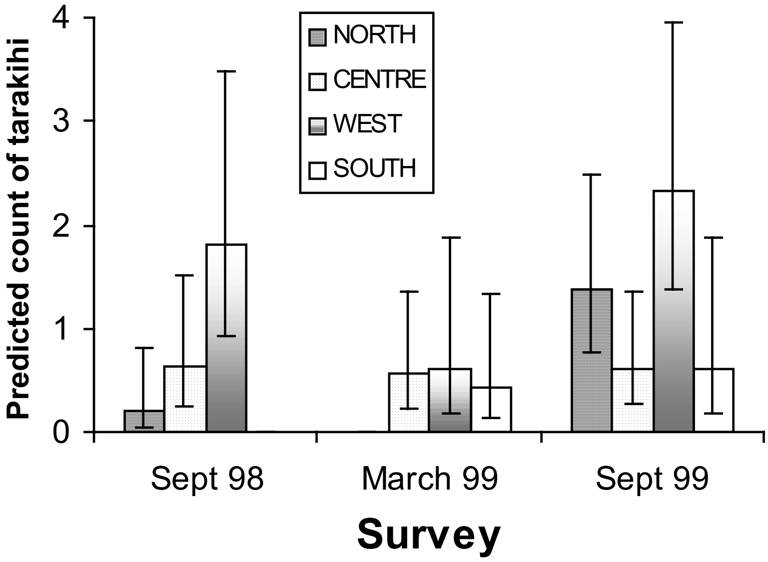

Figure 10 Tarakihi relative density in the four survey areas, estimated from generalised linear modelling of BUV data. Error bars are 95% confidence intervals.

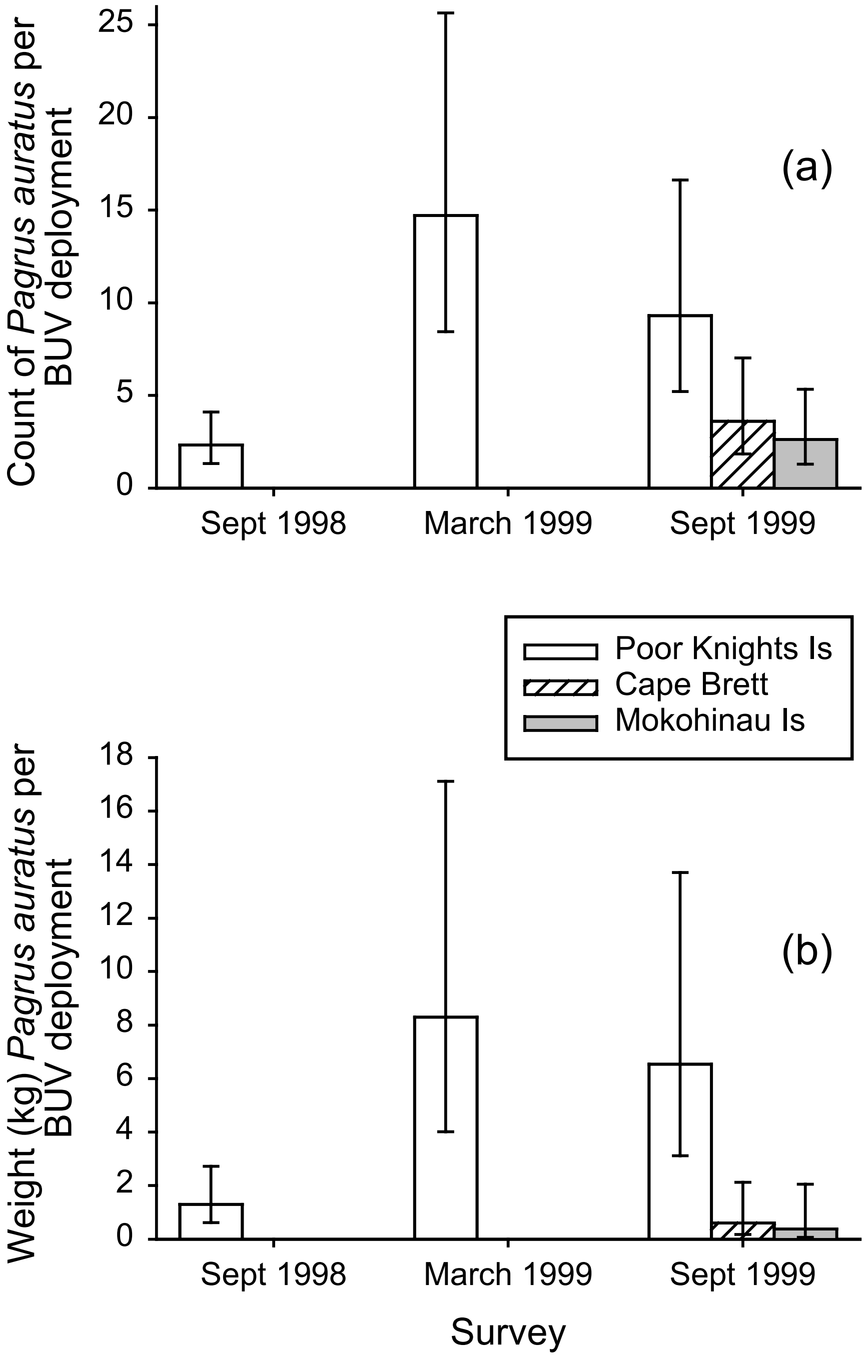

Figure 11. Modelled mean (a) count and (b) weight (kg) of *Pagrus auratus* per BUV deployment (± 95% C.I.) at the Poor Knights, Cape Brett, and the Mokohinau Islands.

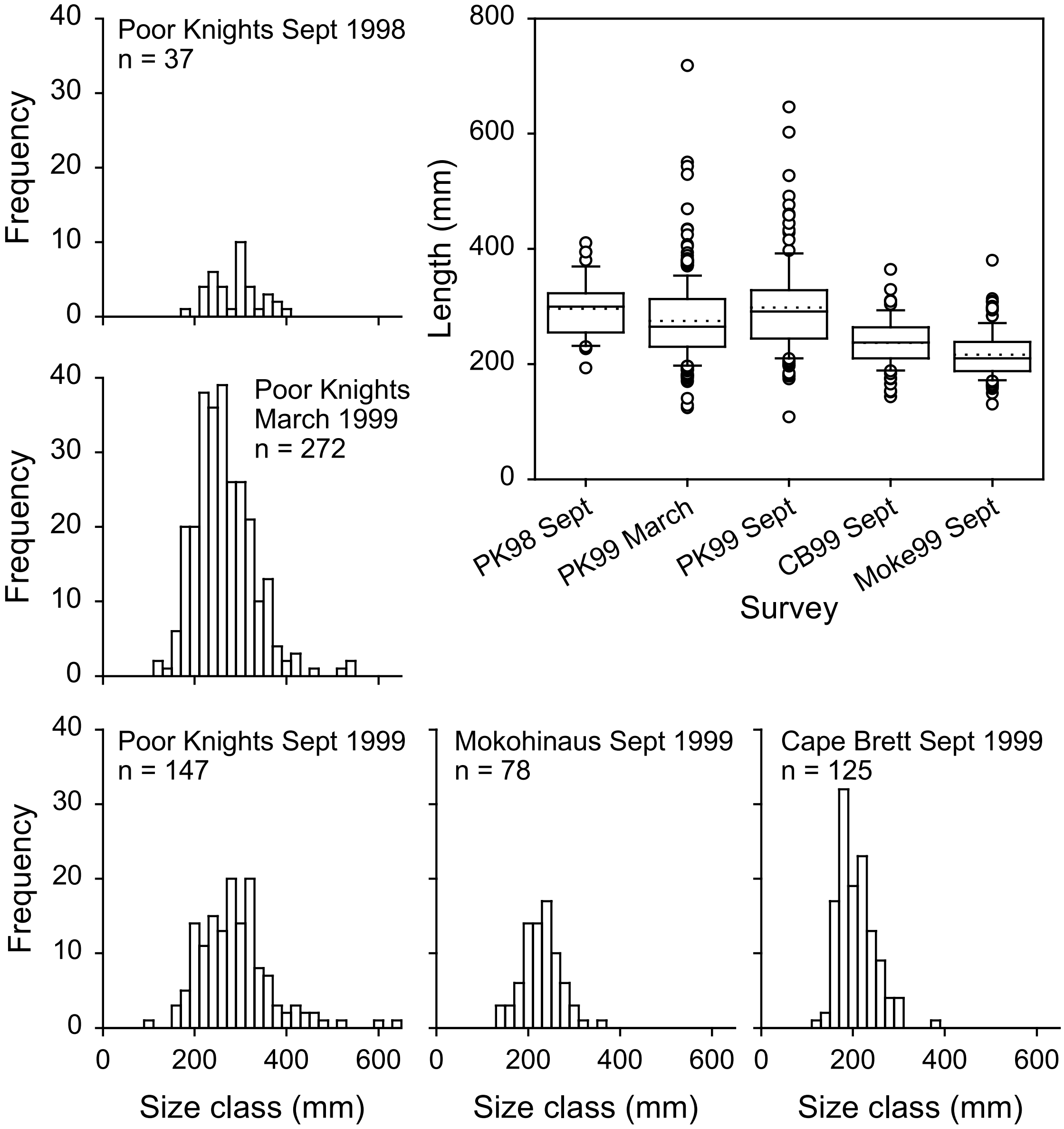

Figure 12. Size frequency distributions (20 mm classes) of *Pagrus auratus* in BUV samples from each survey. The box-plot at top right summarises each of the size distribution plots. Solid and dashed horizontal lines are medians and means, respectively. The upper and lower box limits are 25th and 75th percentiles, and error bars represent the 10th and 90th percentiles. Data points outside the 10th and 90th percentiles are plotted individually.

Relative density of snapper and tarakihi were modelled by survey area to determine site-related variation in abundance estimates. Snapper increased in all areas with time, but were consistently more abundant in the West and South survey areas (Fig. 9), which coincides with the areas previously closed to fishing around Aorangi Island. Tarakihi numbers were low overall, but the highest counts were obtained from the West area (Fig. 10), which we attribute to substratum type. The West area contains the greatest proportion of relatively shallow (< 40 m depth) soft sediment. The sea floor of the other three areas is dominated by reef.

3.4.2 Between Cape Brett, the Poor Knights and Mokohinau Islands

Densities of *Pagrus auratus* at Cape Brett and the Mokohinaus in September 1999 were estimated to be 3.6 ($\chi^2 = 7.7, p < 0.01$) and 1.6 ($\chi^2 = 2.9, p > 0.05$) times, respectively, the density of Poor Knights snapper in September 1998 (Fig 11a). Densities at both unprotected sites were significantly lower than at the Poor Knights ($p < 0.01$) in September 1999. The same analysis conducted on snapper biomass did not change the ratios between the Poor Knights surveys markedly, except that the September 1999 survey mean increased slightly relative to March 1999 (Fig. 11b). This indicates that the mean size of snapper may be increasing at the Poor Knights. Conversely, the weight estimates from both Cape Brett and the Mokohinaus reduced relative to all Poor Knights surveys, implying that populations at these sites are composed of generally smaller fish.

This is borne out by examination of the size structure of snapper in each survey (Fig. 12). Within the Poor Knights surveys, the mean size of snapper in March was significantly lower than either September survey (ANOVA, $F_{2,453} = 5.89, p < 0.01$). However, the fitted model accounted for very little of the observed variance ($R^2 = 0.025$), and should be treated with caution, especially given the inequalities in sample size. Conversely, comparison of the three September 1999 surveys showed real differences between all three sites (ANOVA, $F_{2,347} = 70.58$, $p < 0.001$, $R^2 = 0.29$). Snapper at the Mokohinau Is averaged only 216 mm (± 40 s.d.), whereas Cape Brett had a mean size of 237 mm (± 41 s.d.) and the Poor Knights 298 mm (± 80 s.d.).

3.5 Subtropical fish immigrants

The Poor Knights Islands are well known for the subtropical element of their fish fauna. During this survey, several species were positively identified which were new to the Poor Knights, or infrequently seen. Voucher specimens could not be collected, but sightings were collated for future reference (Table 9). Of interest is *Coris dorsomacula*, which has not been previously recorded from the Poor Knights, and in New Zealand is known from one juvenile specimen caught in the Bay of Islands (Francis et al. 1999). Both *Plagiotremus rhinorhynchus* and *Aulostoma chinensis* are new sightings for New Zealand.

Table 9. Fish species of tropical or subtropical origin recorded during these surveys.

Species	Common name	Family	Location (date)
Mobula japonica	manta ray	Mobulidae	Bartle's Bay (3/3/99)
Acanthistius cinctus	yellow-banded perch	Serranidae	Northern Arch (on BUV, 22/9/99)
Epinephelus daemelii	spotted black grouper	Serranidae	Bartle's Bay (3/3/99)
Parupeneus spilurus	Black-spot goatfish	Mullidae	southern Cleanerfish Bay (3/3/99)
Anampses elegans	elegant wrasse	Labridae	The Gardens (16/2/00)
Coris dorsomacula	pinked-lined wrasse	Labridae	The Gardens (estimated length: 130mm, 14/2/00)
Coris picta	combfish	Labridae	Nursery Cove (2/3/99), Bartle's Bay (3/3/99), South Harbour
Thalassoma amblycephalum	bluntheaded wrasse	Labridae	southern Cleanerfish Bay, Motukapiti Is. (5/3/99, 10/8/99), Labrid Channel (2/3/99), Skull Bay (5/3/99), Bartles Bay (17/9/99)
Plagiotremus rhinorhynchus	bluestriped fangblenny	Blenniidae	Rikoriko Cave (4/3/99)
Aulostoma chinensis	trumpetfish	Aulostomidae	southern Cleanerfish Bay (3/3/99)

Sea surface temperatures (SST) recorded at Leigh usually provide a good representation of trends in SST for Northland as a whole. During 1998-99, Leigh SST was 1-1.5°C higher than the 30-year average (Fig. 13). Under these conditions, it might be expected that recruitment of subtropical fishes to the Poor Knights would be greater than in previous years (Francis & Evans 1993), and future surveys will continue to document sightings (and attempt to obtain photographic records) of such species.

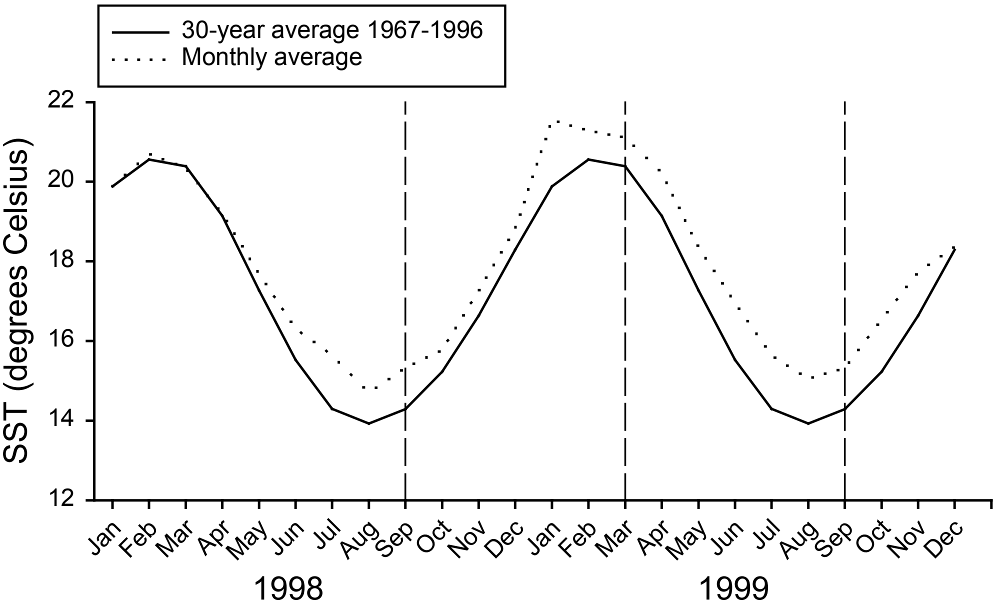

Figure 13. Sea surface temperature (recorded at Leigh) monthly averages for 1998-99, along with the Leigh 30-year monthly average. Vertical dashed lines represent the timing of surveys analysed in this report. Note that SST at the Poor Knights Islands is usually c. 2°C warmer than the mainland.

4. Discussion

4.1 Effect of UVC transect type on density estimates

The shorter transect (125 m^2) with higher replication levels performed more effectively than fewer, larger (500 m^2) transects. Increasing replication had the effect of reducing the error estimate associated with the mean, although smaller sampling units would usually be expected to produce lower means and a greater number of zero counts in the dataset. This was true in this case, however when data was converted to a density estimate (ie fish 100 m^{-2}) the smaller transects gave higher means. This is likely to be due to two factors: first, the ability of smaller transects to be more easily constrained to target habitats or depth strata, and second, the increase in diver activity necessary to conduct the large transects was likely to have caused some species of fish to flee the area, while attracting other species to the diver. At sites often frequented by divers (e.g. Nursery Cove and the Gardens) we noted that the wrasses, in particular, would follow divers while transects were being swum. These fish (usually *Coris sandageri* and *Bodianus unimaculatus*) would have been omitted from counts because they seemed to deliberately approach from behind the diver. Such problems, caused by the dependence of UVC methods on fish behaviour, are well known (e.g. Cole 1994, Jennings & Polunin 1995, Kulbicki 1998, Willis et al. 2000), but there does not appear to be any simple solution at this stage.

Unless water visibility is consistently good, wide transects cannot be consistently applied. Use of 10 m wide transects requires at least 5 m horizontal visibility, and these conditions are seldom found at many locations elsewhere in New Zealand. For example, at the Long Island – Kokomohua Marine Reserve in the Marlborough Sounds, counts of blue cod *Parapercis colias* are conducted using 2 m wide transects, as visibility is frequently less than 3 m (R.G. Cole, pers. comm.). Use of narrower transects thus enables us to obtain data comparable with that obtained from other localities, and this, coupled with the higher replication levels obtainable for similar effort, would make the use of smaller transects a more sensible option.

4.2 Effect of time of day on UVC density estimates

There was no consistent effect of time of day on reef fish density estimates, but there were some changes recorded at particular sites or particular depths. Overall there did not appear to be any systematic bias, but this work was done by repeatedly surveying the same site, and we cannot discount the possibility of changes in the fish response to divers caused by regular visits to the same places. Also, we conducted this experiment on only one day due to logistic limitations, and therefore the data do not encompass between-day variability in fish behaviour. There is a need for a more extensive study to be made of diel changes (both within and between days) in fish activity patterns.

4.3 UVC site/habitat comparisons

The depth-dependent distributions evident in some species support the conclusions drawn by Schiel (1984). Species that predominantly occur on shallow reefs include *Parma alboscapularis*, *Notolabrus celidotus*, *Notolabrus fucicola*, and *Scorpis violaceus*. Conversely, *Caprodon longimanus*, *Canthigaster callisterna*, and *Caesioperca lepidoptera* characterise deeper reefs. Other common species surveyed tended to be distributed fairly evenly throughout the depth range we could dive (< 30 m).

The most marked increases in fish density during the first year of monitoring could be attributed to recruitment or the absence of fishing. Two wrasse species (*Suezichthys aylingi* and *Pseudolabrus luculentus*) increased in abundance dramatically following strong recruitment episodes during the summer of 1998/99. Slight, but non-significant increases in *Coris sandageri* appeared also to be due to recruitment, as juveniles contributed substantially to estimates in March 1999. Significant increases were also detected in the density of snapper (*Pagrus auratus*) and pink maomao (*Caprodon longimanus*), which we attribute to the cessation of fishing around the islands.

Differences in the composition of fish assemblages in different broad-scale habitat types were illustrated by ordination. Shallow reefs were well separated from wall and overhang sites, partly due to depth distributions mentioned above, and partly because steep wall sites tended to be subject to stronger current regimes and therefore possessed proportionally more planktivores. Deep reefs tended to possess elements of both wall and shallow reef faunas, and hence fell between them in the ordination. When the three surveys of the Poor Knights were examined together, the pre-reserve survey was significantly different to the two post-reserve surveys, although this was not visually apparent in the MDS plot. The MDS stress value was quite high, indicating that relationships between the samples were not easily represented in two-dimensional space. There was slightly better separation between the surveys using a three-dimensional ordination (not presented), but this was not clear. We interpret this to mean that, despite large changes in the density of some species, there was no overall alteration to community structure following reserve establishment.

Between Cape Brett, the Mokohinau and Poor Knights Islands

Univariate comparisons between the Mokohinau Islands, Poor Knights, and Cape Brett illustrated that intrinsic differences exist between the three locations in terms of the relative abundance of individual species and composition of the assemblage. Despite the obvious influence of the East Auckland Current (EAC) at all three places, Cape Brett has a characteristic mainland fauna, the Poor Knights are dominated by subtropical influences, and the Mokohinau Islands are intermediate. Although the latter location is the furthest offshore, it receives less of the main flow of the EAC than the Poor Knights because eddies push the current offshore at around 35°S (Stanton et al. 1997). The large differences between locations were well illustrated by MDS ordination of sites sampled in September 1999 (Fig. 7b). The Mokohinau Islands and Cape Brett were chosen as reference sites to examine fishing effects at the Poor Knights because we thought them likely to be the most similar marine habitats available. This study effectively demonstrates the uniqueness of the Poor Knights fish fauna. Future assessment of the dynamics of offshore island fish communities would be aided by the inclusion of Tuhua (Mayor Island) at a survey location for comparison, especially given that half of Tuhua’s waters are protected by a marine reserve.

4.4 Comparison of carnivorous fish density using BUV

An estimated increase in snapper abundance of 300% (from BUV) in one year following closure to fishing is a surprising result, but may be explainable by the offshore position of the Poor Knights. Previous surveys at Leigh, Hahei and Tawharanui protected areas indicate that snapper abundance in these reserves changes seasonally, with large increases in density through summer and autumn relative to winter/spring (Willis & Babcock 1998). If seasonal migrations across the continental shelf are made in response to changing water temperatures, snapper may seek deep water during the winter months as temperature fluctuations are lessened there relative to the surface. As the Poor Knights islands are surrounded by deep water, snapper need travel relatively short distances to move between shallow and deep-water environments, hence the Poor Knights can be rapidly recolonised. We note here that the change in snapper density from the pre-reserve survey to the March 1999 survey was a ratio of 6.54 (lower 95% CI = 3.54, upper = 11.99), or a 550% increase in six months. This extremely marked change in density supports the hypothesis of seasonal migration in snapper.

The estimated 30% decrease in *Bodianus unimaculatus* from BUV at the Poor Knights was not supported by UVC data. It is possible that increasing densities of snapper deterred pigfish from approaching the bait in the latter surveys. Large snapper frequently exhibit both inter- and intraspecific aggression in the presence of bait (Willis unpubl. data). Snapper guarding behaviour is most often directed at conspecifics, but may have been sufficient to deter *B. unimaculatus*, which is not an aggressive species (pers. obs.).

4.5 Immigration of subtropical species

Choat et al. (1988) documented the decline of several formerly common Poor Knights fish species, including the crimson cleanerfish *Suezichthys aylingi*, and argued that populations in northern New Zealand originated from (and are dependent upon) larvae transported from source populations at the Three Kings Islands. Francis & Evans (1993) suggested that recruitment events of subtropical species are linked to the El Niño-Southern Oscillation (ENSO) cycle, and Sharples (1997) confirmed that intrusions of subtropical water across the continental shelf are driven by periods of southeasterly winds, most prevalent during the La Niña phase of ENSO. The 1998/99 period was subject to La Niña conditions, and indeed, many new subtropical species have been recorded from northern New Zealand during this period (Francis et al. 1999). If mean water temperatures remain high in the near future, the Poor Knights may establish adult populations of species such as *Thalassoma amblycephalum*, which have not been recorded in New Zealand as adults before.

4.6 Conclusions

For monitoring programmes, it is of more interest to determine the trajectory, rate and magnitude of any change attributable to some putative environmental impact, rather than merely establish the magnitude of difference between given points in space or time. The successful detection of differences in fish density (or, for that matter, any element of the marine biota) between two sites is, in a sense, a trivial result unless the magnitude of the difference (or 'effect size') is very large. Intrinsic biological variability will always provide statistically significant differences between sites where a suitable number of replicates have been obtained (Edgar & Barrett 1997), and the probability of obtaining statistical significance will increase with sample size.

This point is well illustrated by our use of the Mokohinau Islands and Cape Brett as reference sites for examining change at the Poor Knights. Whilst the biotic composition at all three locations is influenced to some extent by the East Auckland Current (EAC), their geographic separation means that there are obvious differences (even at the largest scale), so direct comparison of abundances at one point in time is meaningless. The key to determining whether cessation of fishing at the Poor Knights will have long-term benefits rests on the acquisition of a time series of several years' data at the three sites. If fishing pressure at the Mokohinau Is and Cape Brett is maintained or increases in future, and the Poor Knights fish populations continue to recover, there should be a convincing upward slope to the PKI time series relative to the two fished sites. In this context, the inability to conduct pre-reserve surveys at reference sites from September 1998 is unfortunate, and has seriously affected the level of confidence with which we can attribute change in fish populations to marine reserve implementation.

Long time series of data also provide the opportunity to investigate the dynamics of fishes whose abundance is primarily affected by processes other than fishing. The coincidence of these surveys with years of above average sea-surface temperature (SST) has allowed the documentation of large pulses of recruitment of some subtropical wrasses. These species were formerly common at the Poor Knights (Schiel 1984, Choat et al. 1988), but declined during the early 1980s. The orange wrasse (*Pseudolabrus luculentus*), crimson cleanerfish (*Suezicthys aylingi*) and Sandager's wrasse (*Coris sandageri*) in particular appeared to benefit from a large influx of juveniles during the summer of 1998/99. It is as yet unknown whether high SST facilitates larval survival and transport on the EAC, or whether larval supply via the EAC is fairly consistent and high SST enhances post-settlement survivorship. It is unlikely that implementation of full marine reserve status affected the increases in abundance of these wrasses.

In the case of the Poor Knights, estimates of the effects of fishing are likely to be conservative, as the half of the Poor Knights had already been closed for one year before our first survey. The PKI areas 'North' and 'Centre' were continuously subject to fishing during this time, and had similarly low densities of snapper to the currently fished Cape Brett and Mokohinau sites. By contrast, areas 'West' and 'South' consistently returned the highest snapper density estimates. It is difficult to determine whether this is due to intrinsic differences in habitat type, or reflects effects of the earlier closure to fishing.

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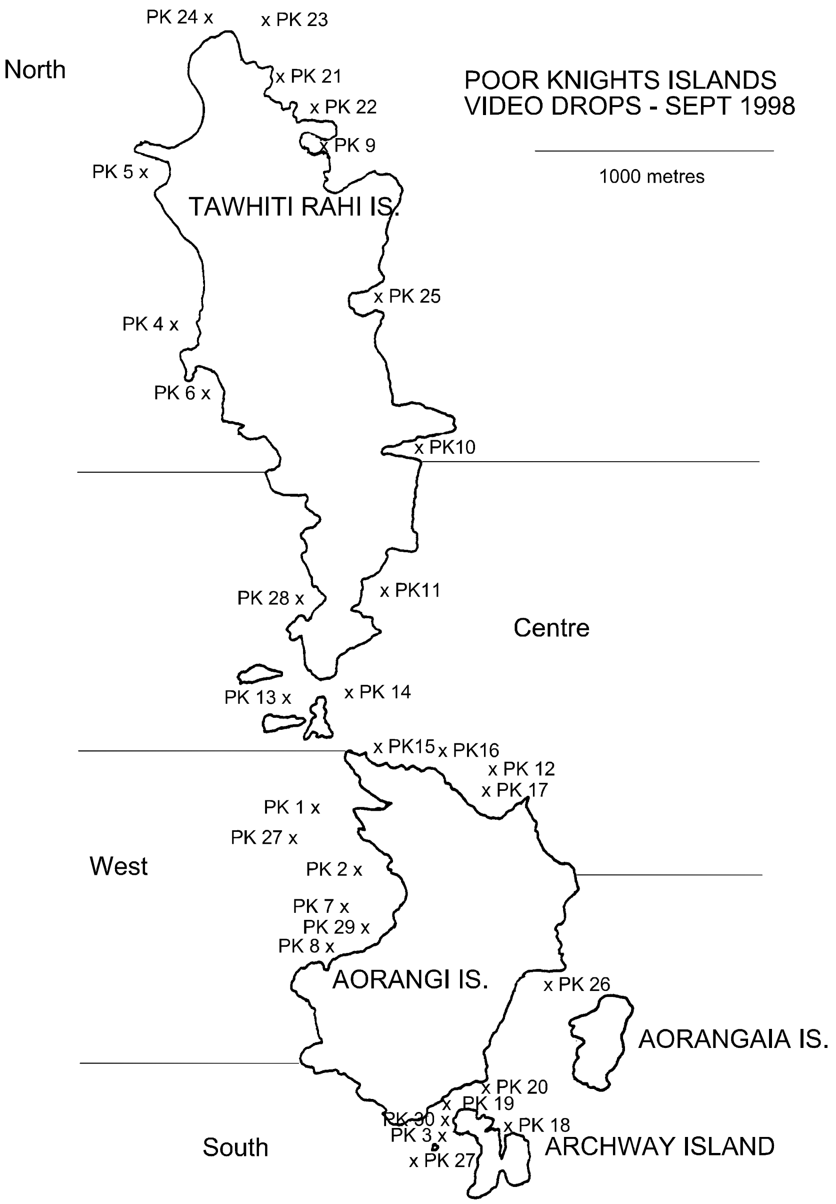
Appendix 1.

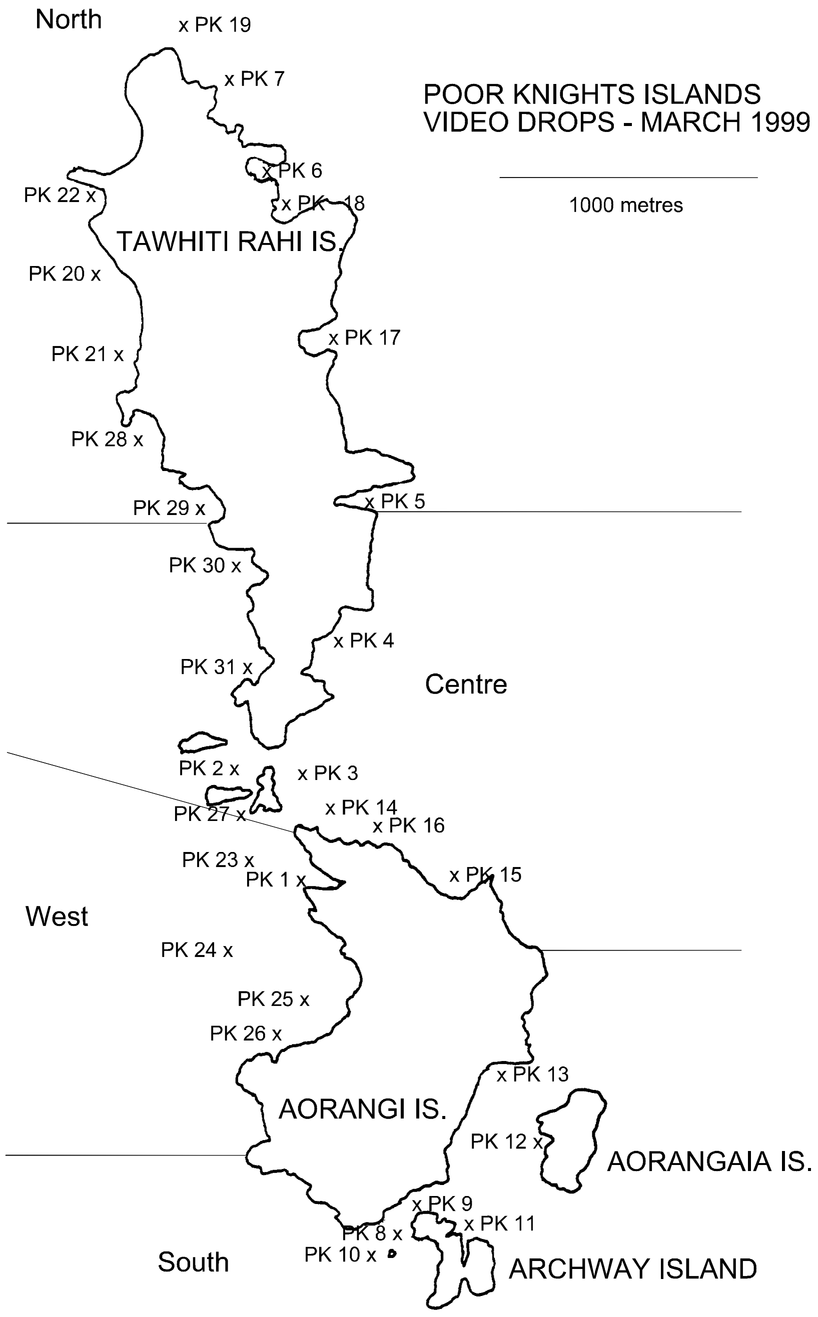
Sampling design and levels of replication for underwater visual census and baited underwater video sampling

		September 1998			March 1999			September 1999		
Location	Survey area	# UVC sites	Total UVC transects	BUV replicates	# UVC sites	Total UVC transects	BUV replicates	# UVC sites	Total UVC transects	BUV replicates
Poor Knights	North	5	46	10	5	45	10	6	54	8
	Centre	4	35	8	5	45	9	5	45	10
	West	2	18	5	5	45	5	5	45	6
	South	4	36	7	5	45	7	4	36	5
	Total	*15*	*135*	*30*	*20*	*180*	*31*	*20*	*180*	*29*
Cape Brett	West							7	63	11
	North							7	63	9
	East							3	45	8
	South							5	27	7
	Total							*22*	*198*	*35*
Mokohinau	Trig							7	63	9
	Burgess							6	54	10
	Fanal							7	63	12
	Total							*20*	*180*	*31*

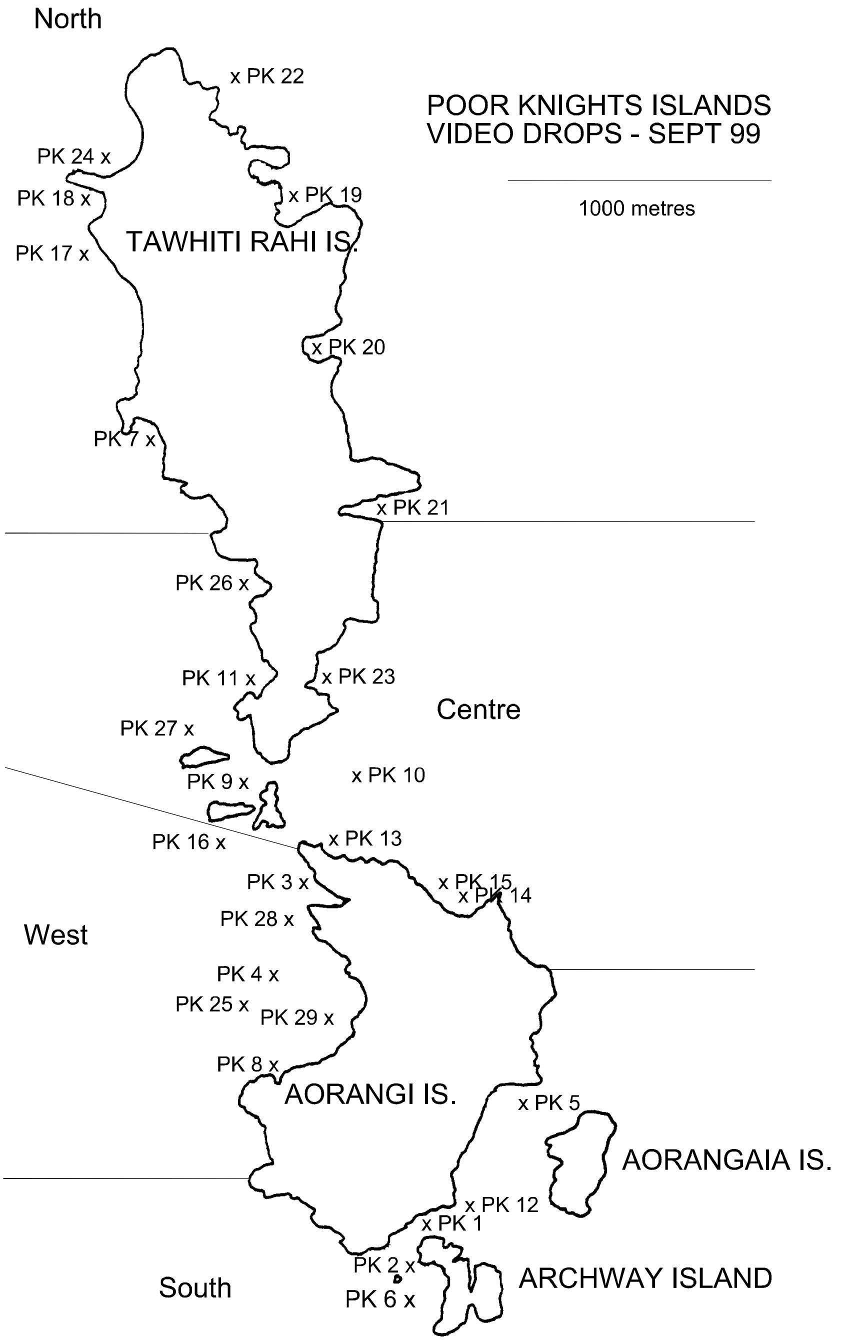
Appendix 2.

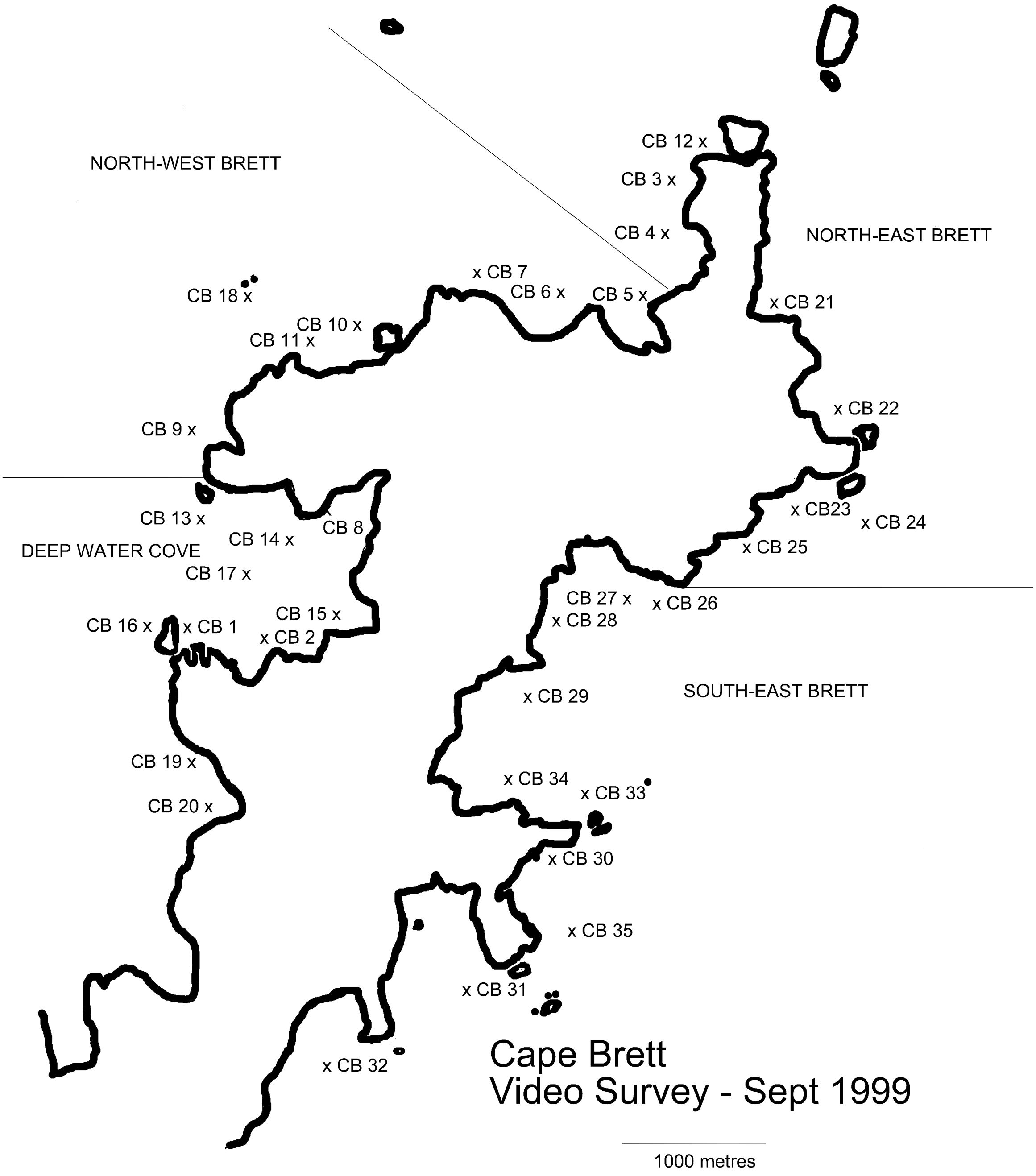
Maps of Cape Brett, the Mokohinau and Poor Knights Islands with locations of BUV stations.



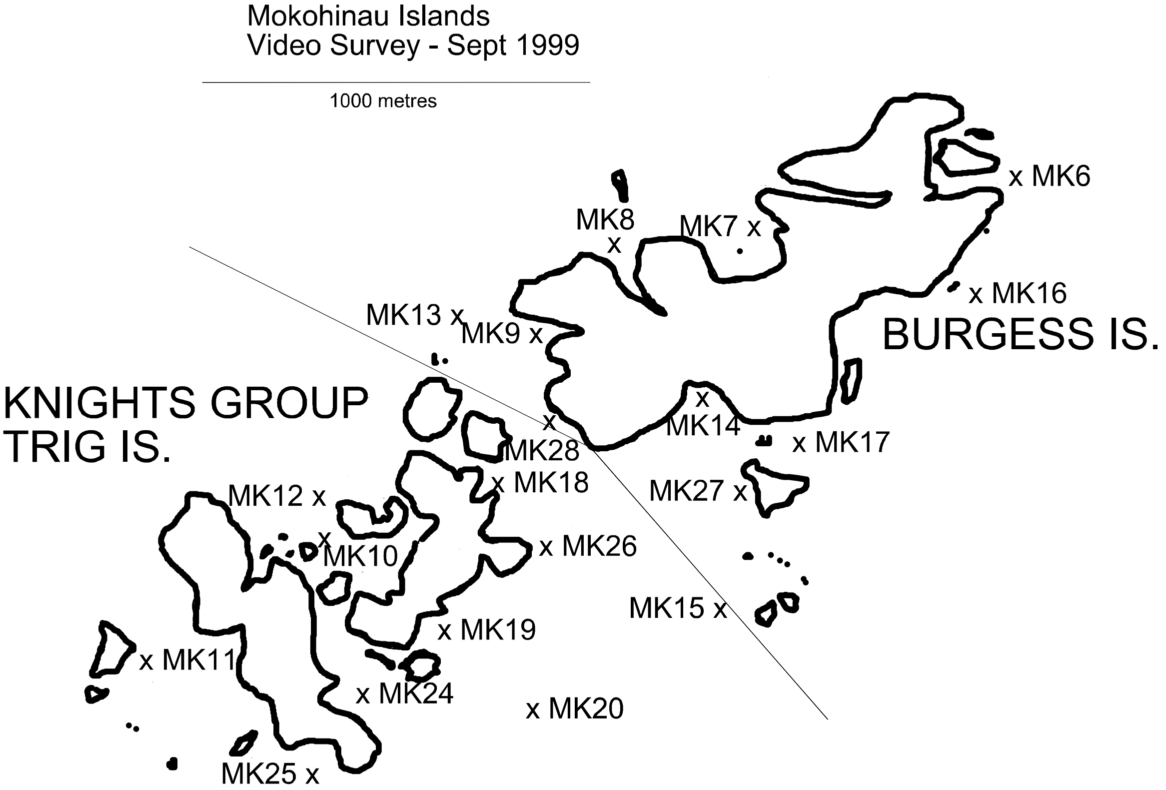


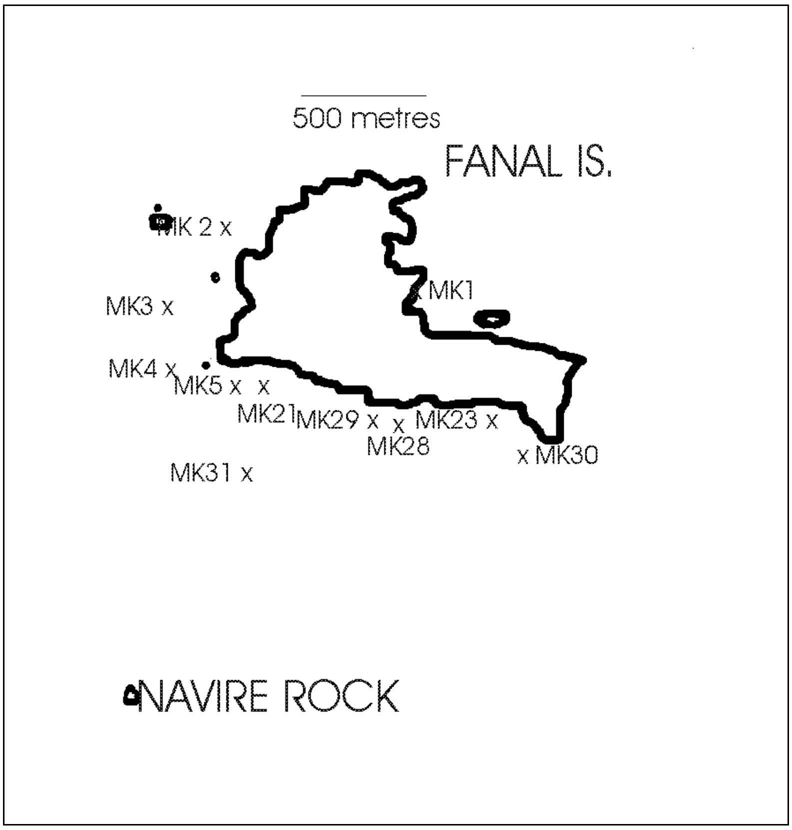
POOR KNIGHTS ISLANDS
VIDEO DROPS - MARCH 1999


North
POOR KNIGHTS ISLANDS
VIDEO DROPS - SEPT 99
1000 metres
x PK 22
PK 24 x
PK 18 x
x PK 19
PK 17 x
TAWHITI RAHI IS.
x PK 20
PK 7 x
x PK 21
PK 26 x
PK 11 x
x PK 23
Centre
PK 27 x
PK 9 x
x PK 10
PK 16 x
x PK 13
PK 3 x
x PK 15
x PK 14
PK 28 x
West
PK 4 x
PK 25 x
PK 29 x
PK 8 x
AORANGI IS.
x PK 5
AORANGAIA IS.
x PK 12
x PK 1
PK 2 x
PK 6 x
ARCHWAY ISLAND
South



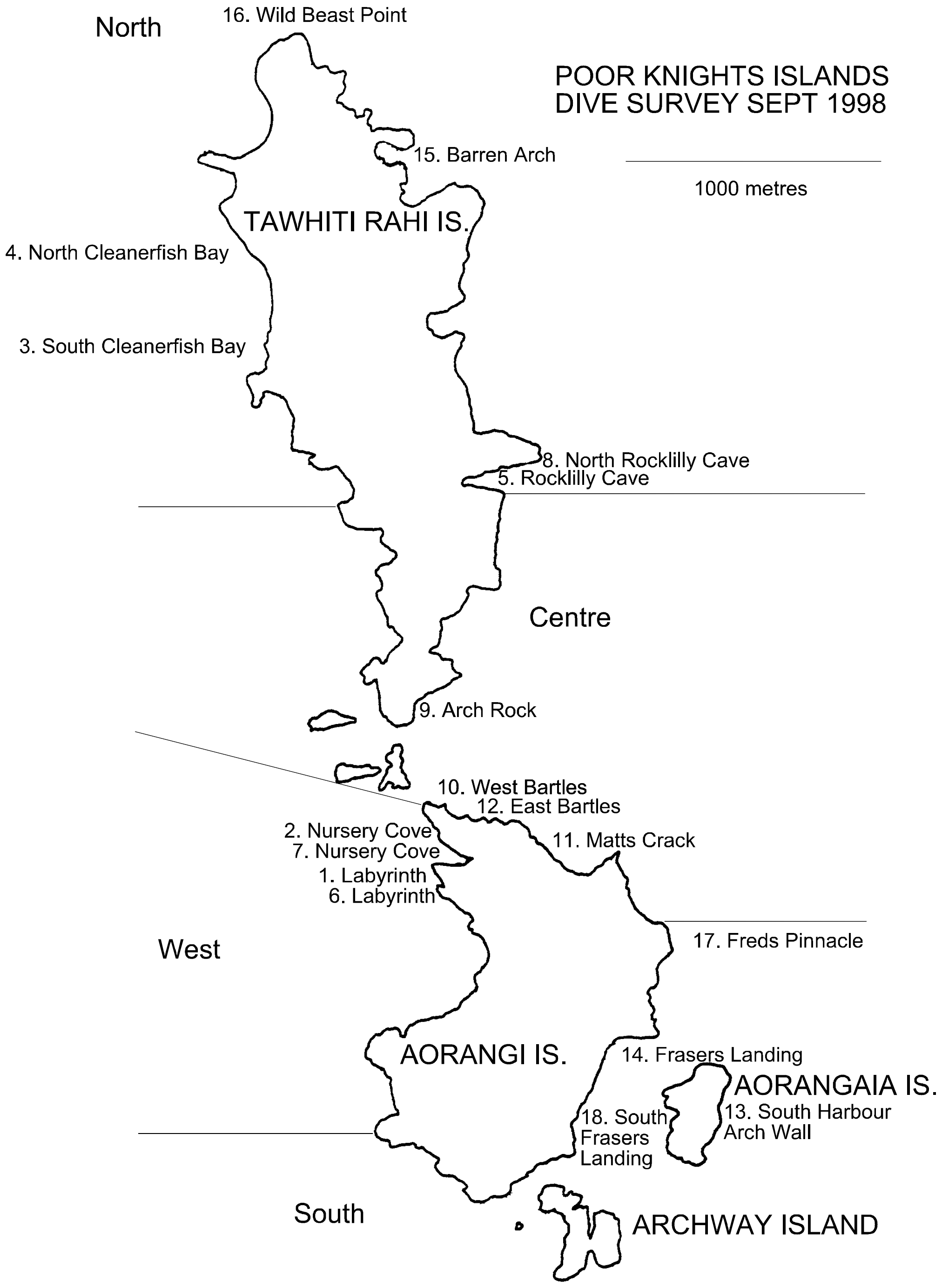
Cape Brett
Video Survey - Sept 1999

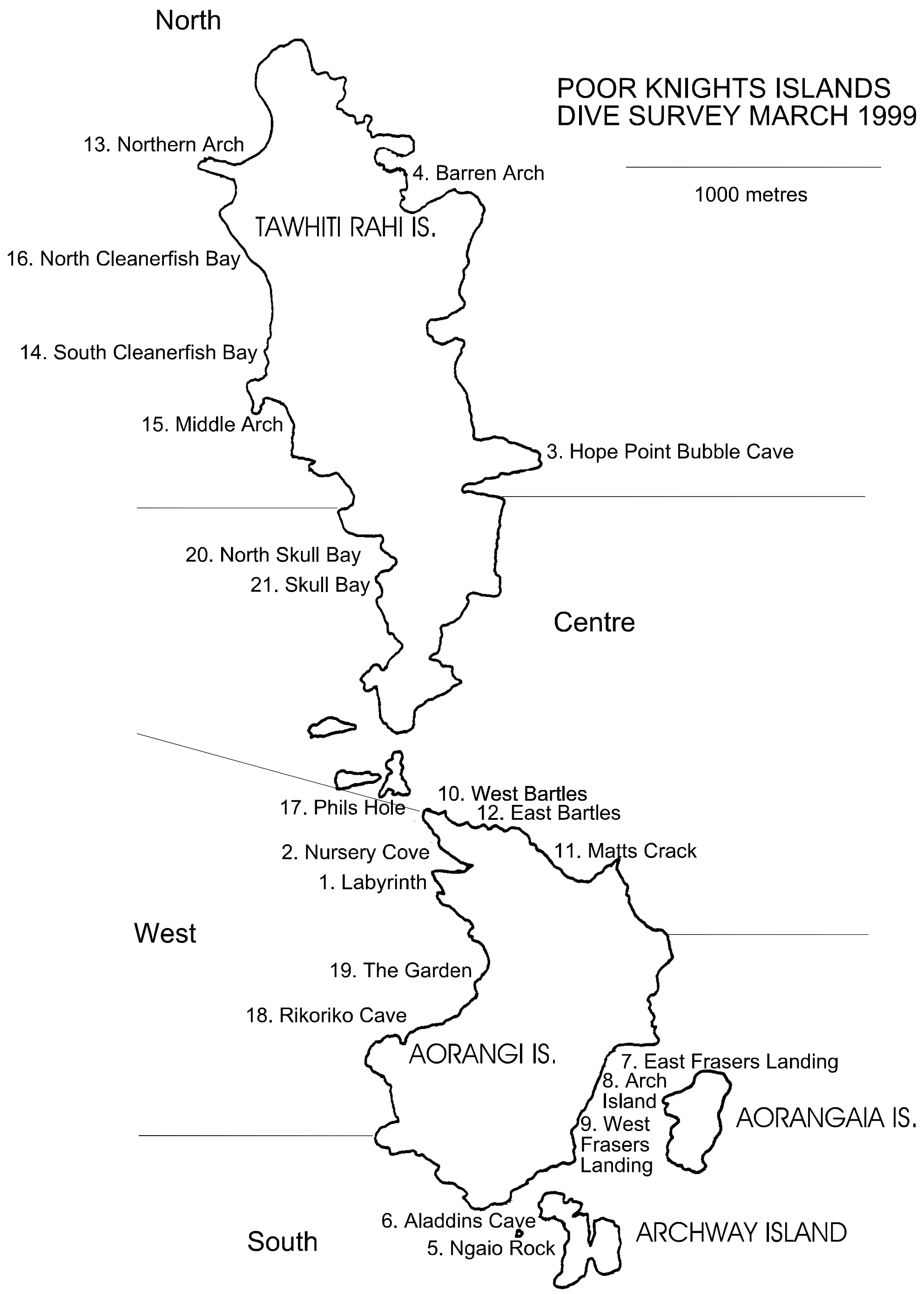

Mokohinau Islands
Video Survey - Sept 1999
1000 metres
x MK6
MK8 x
MK7 x
x MK16
BURGESS IS.
MK13 x
MK9 x
KNIGHTS GROUP
TRIG IS.
x MK14
x MK28
x MK17
MK12 x
x MK18
MK27 x
x MK10
x MK26
MK15 x
x MK19
x MK11
x MK24
x MK20
MK25 x

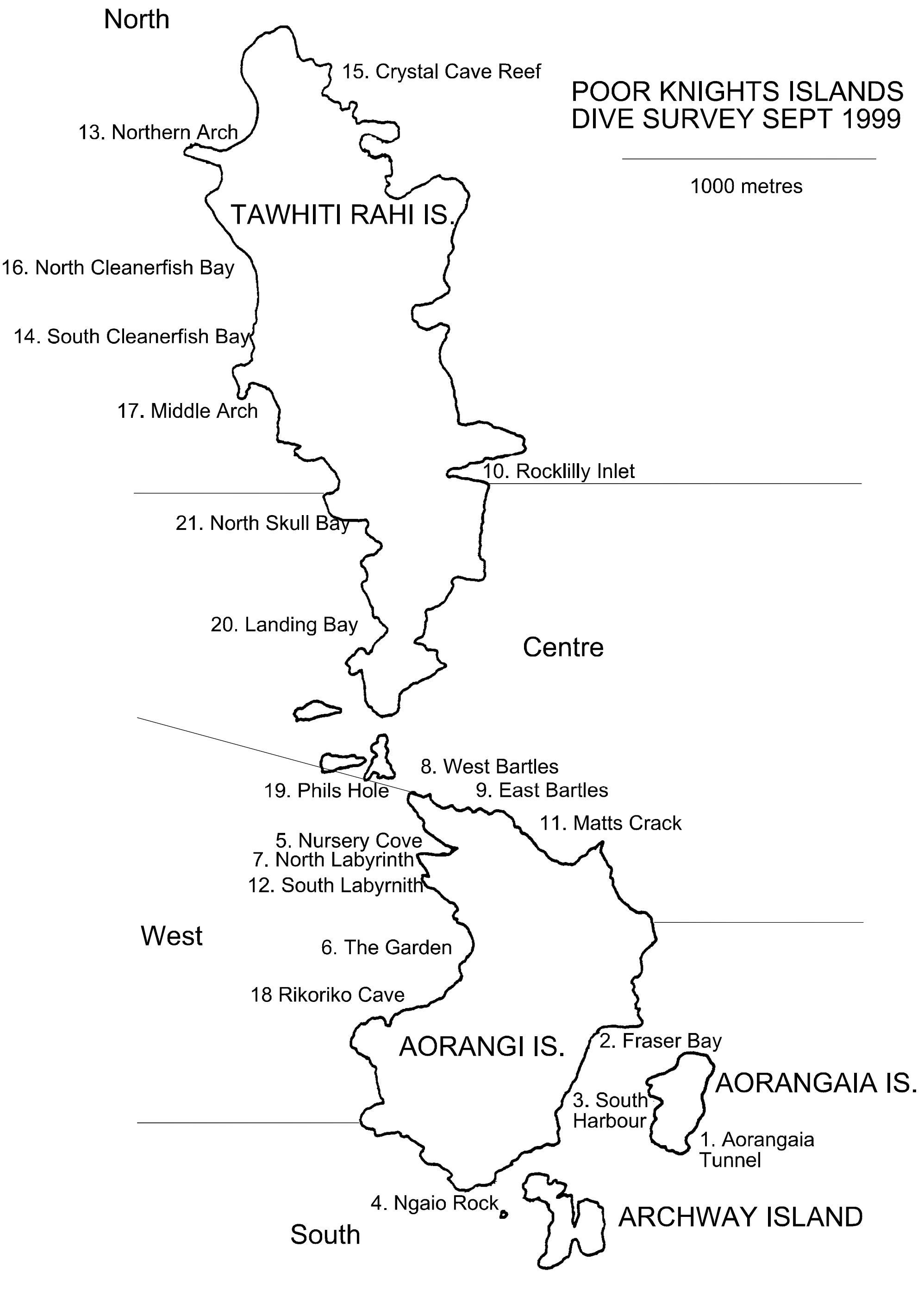

500 metres
FANAL IS.
MK 2 x
MK1
MK3 x
MK4 x
MK5 x
x MK21
MK29 x
x MK28
MK23 x
x MK30
MK31 x
NAVIRE ROCK

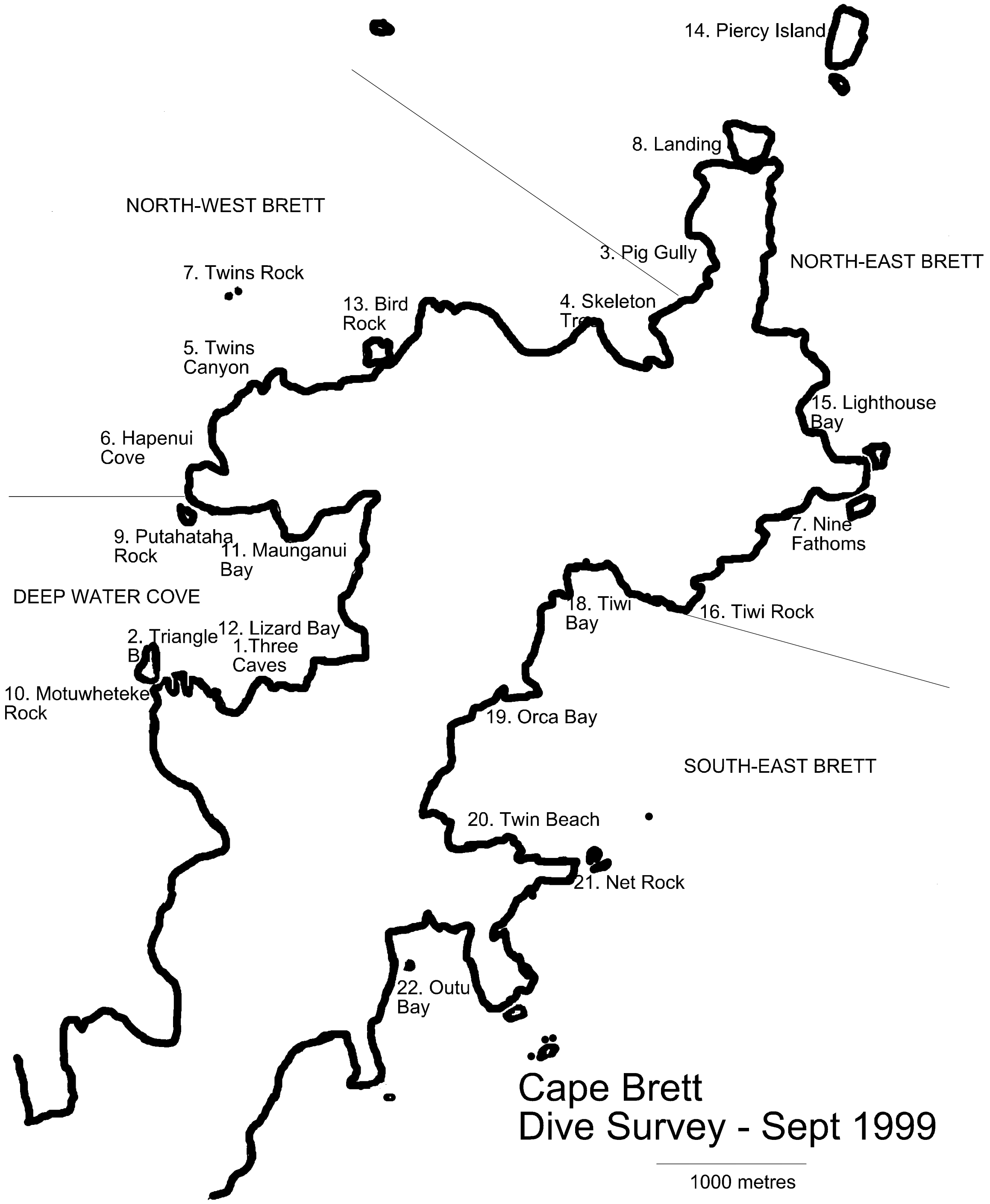
Appendix 3

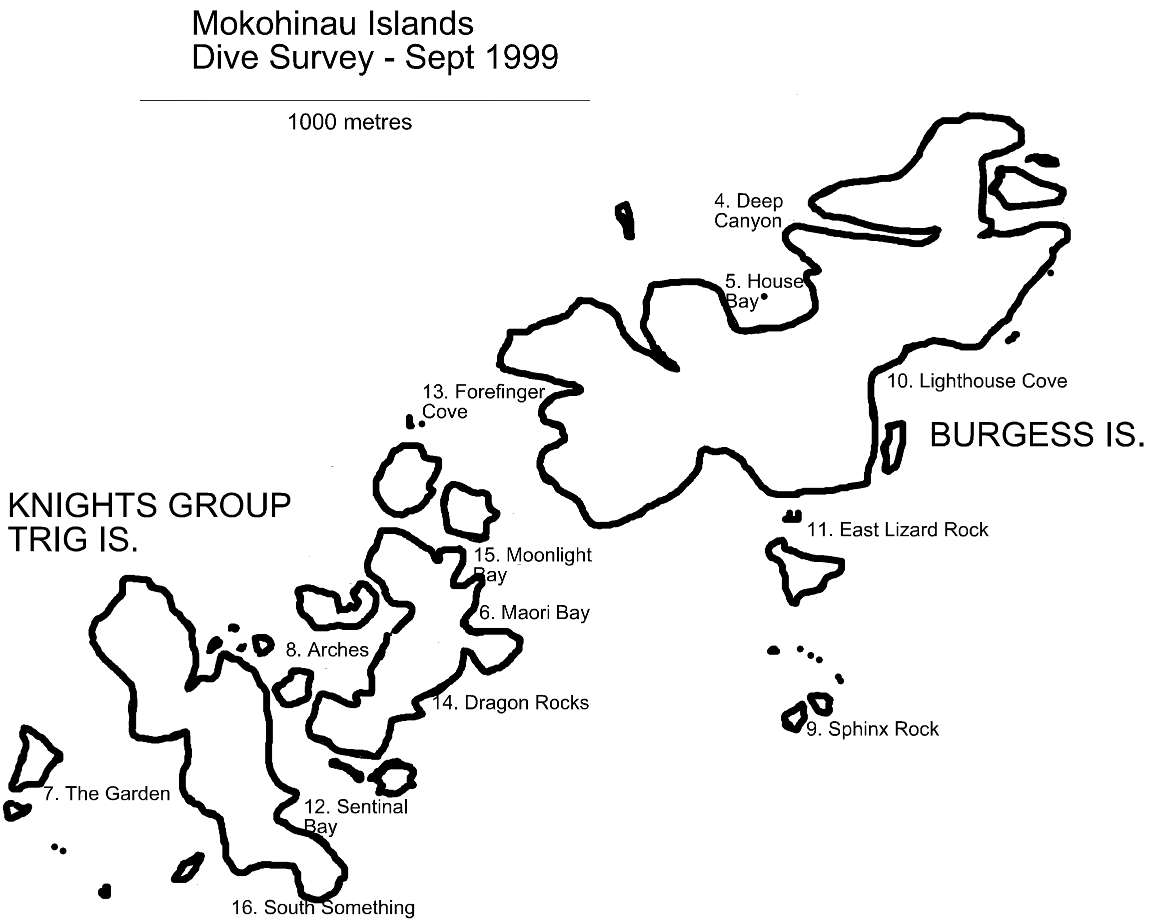
Maps of Cape Brett, the Mokohinau and Poor Knights Islands with locations of UVC (diver census) stations.

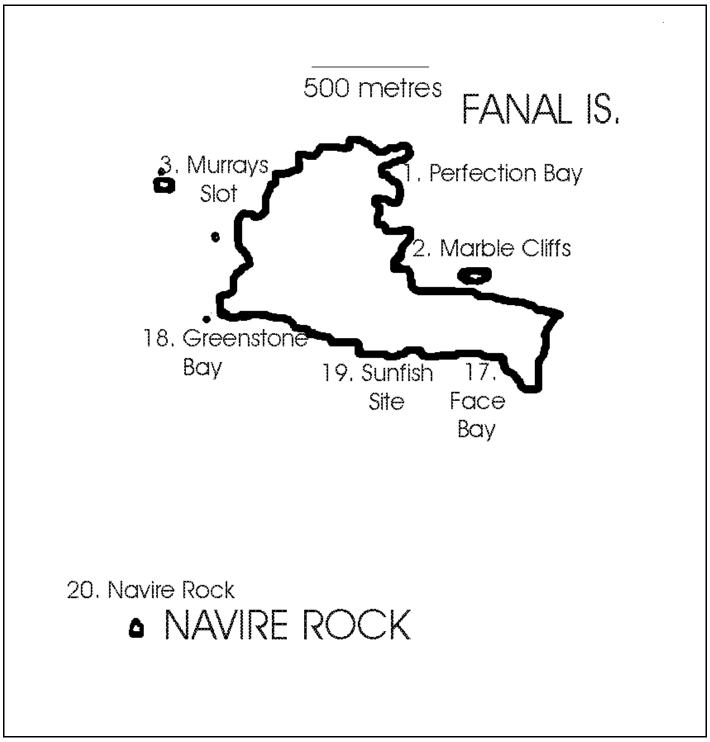



POOR KNIGHTS ISLANDS
DIVE SURVEY MARCH 1999
1000 metres
North
Centre
West
South
TAWHITI RAHI IS.
AORANGI IS.
AORANGAIA IS.
ARCHWAY ISLAND
1. Labyrinth
2. Nursery Cove
3. Hope Point Bubble Cave
4. Barren Arch
5. Ngaio Rock
6. Aladdins Cave
7. East Frasers Landing
8. Arch Island
9. West Frasers Landing
10. West Bartles
11. Matts Crack
12. East Bartles
13. Northern Arch
14. South Cleanerfish Bay
15. Middle Arch
16. North Cleanerfish Bay
17. Phils Hole
18. Rikoriko Cave
19. The Garden
20. North Skull Bay
21. Skull Bay


North
15. Crystal Cave Reef
POOR KNIGHTS ISLANDS
DIVE SURVEY SEPT 1999
1000 metres
13. Northern Arch
TAWHITI RAHI IS.
16. North Cleanerfish Bay
14. South Cleanerfish Bay
17. Middle Arch
10. Rocklilly Inlet
21. North Skull Bay
20. Landing Bay
Centre
8. West Bartles
19. Phils Hole
9. East Bartles
11. Matts Crack
5. Nursery Cove
7. North Labyrinth
12. South Labyrnith
West
6. The Garden
18 Rikoriko Cave
AORANGI IS.
2. Fraser Bay
AORANGAIA IS.
3. South
Harbour
1. Aorangaia
Tunnel
4. Ngaio Rock
ARCHWAY ISLAND
South


14. Piercy Island
8. Landing
NORTH-WEST BRETT
3. Pig Gully
NORTH-EAST BRETT
7. Twins Rock
13. Bird Rock
4. Skeleton Tree
5. Twins Canyon
15. Lighthouse Bay
6. Hapenui Cove
7. Nine Fathoms
9. Putahataha Rock
11. Maunganui Bay
DEEP WATER COVE
18. Tiwi Bay
16. Tiwi Rock
2. Triangle B
12. Lizard Bay
1. Three Caves
10. Motuwheteke Rock
19. Orca Bay
SOUTH-EAST BRETT
20. Twin Beach
21. Net Rock
22. Outu Bay
Cape Brett
Dive Survey - Sept 1999
1000 metres


Mokohinau Islands
Dive Survey - Sept 1999
1000 metres
4. Deep Canyon
5. House Bay
10. Lighthouse Cove
BURGESS IS.
13. Forefinger Cove
KNIGHTS GROUP
TRIG IS.
11. East Lizard Rock
15. Moonlight Bay
6. Maori Bay
8. Arches
14. Dragon Rocks
9. Sphinx Rock
7. The Garden
12. Sentinal Bay
16. South Something


500 metres
FANAL IS.
3. Murrays Slot
1. Perfection Bay
2. Marble Cliffs
18. Greenstone Bay
19. Sunfish Site
17. Face Bay
20. Navire Rock
NAVIRE ROCK