How long should marine reserves be monitored for and why?

DOC SCIENCE INTERNAL SERIES 130

Russell Cole

Published by
Department of Conservation
PO Box 10-420
Wellington, New Zealand

DOC Science Internal Series is a published record of scientific research carried out, or advice given, by Department of Conservation staff or external contractors funded by DOC. It comprises reports and short communications that are peer-reviewed.

Individual contributions to the series are first released on the departmental website in pdf form. Hardcopy is printed, bound, and distributed at regular intervals. Titles are also listed in the DOC Science Publishing catalogue on the website, refer http://www.doc.govt.nz under Publications, then Science and Research.

ISSN 1175-6519

ISBN 0-478-22465-6

In the interest of forest conservation, DOC Science Publishing supports paperless electronic publishing. When printing, recycled paper is used wherever possible.

This report was prepared for publication by DOC Science Publishing, Science & Research Unit; editing and layout by Lynette Clelland. Publication was approved by the Manager, Science & Research Unit, Science Technology and Information Services, Department of Conservation, Wellington.

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Russell Cole

National Institute of Water & Atmospheric Research Ltd, PO Box 893, Nelson

ABSTRACT

Monitoring of marine reserves may be done to either assess rehabilitation or preservation within the reserves. For both purposes, it can be argued that monitoring should continue indefinitely. The longevity of marine organisms suggests that monitoring should continue for decades. Based on assessments of CV and rates of change in populations of a few species, at a few marine reserves in New Zealand, it appears that effects for exploited species may emerge over periods of decades, and that 10 years of annual sampling should be regarded as a minimum for monitoring. In the oldest existing reserve, important changes were occurring 20 years after imposition of reserve status. The magnitude of response to marine reserve protection will also depend on the severity of impact that has already occurred. If the impact of harvesting at the reserve site is relatively small, the recovery may be small, and thus the rate of change will be slow. Such changes will require even longer monitoring periods to detect. There is considerable scope for novel approaches to monitoring carnivorous species based on sampling that does not require divers.

Keywords: marine reserves, New Zealand, monitoring.

Cole, R. 2003: How long should marine reserves be monitored for and why? *DOC Science Internal Series 130*. Department of Conservation, Wellington. 20 p.

1. Introduction and discussion

Monitoring flora and fauna in marine reserves is expensive. The goals of monitoring must be weighed against their costs in order to maximise efficiency. This report assesses some of the matters that need to be considered in determining how, where, and when monitoring should be carried out. Why monitoring of marine reserves needs to be done is considered first, then what monitoring means in terms of the biology of organisms. Finally, some numeric reasons determining the duration of monitoring are discussed.

1.1 WHY MONITOR MARINE RESERVES?

The two main objectives for establishing marine reserves are **rehabilitation** and **preservation**. Rehabilitation involves reinstating a former condition that has been modified by humans, whereas preservation focuses on maintaining existing habitats and conditions.

1.2 BIOLOGICAL CONSIDERATIONS

If the goal of monitoring is to understand whether **rehabilitation** has occurred, then monitoring should continue indefinitely. That is because we don't really know what 'rehabilitated' might be, and there is no predictable end point. For example, we do not know whether the Cape Rodney–Okakari Point Marine Reserve at Leigh is now, after 28 years of protection, rehabilitated to its original condition, or whether further changes will occur. One situation where we might know what rehabilitation means is where some impact has occurred in a monitored area, and the—hopefully documented—previous situation represents the endpoint of rehabilitation (which might be judged from similar nearby areas).

If the goal of monitoring is **preservation**, then it is important to confirm that the assemblages or habitats or species of interest are preserved, which requires some form of indefinite surveillance. The question which then arises is whether quantitative data are required to verify that the assemblage or habitat or species is preserved. Kingsford & Battershill (1998: p.35) suggest that presence / absence data provide very little information. However, where those data can be obtained with reasonable spatial / temporal resolution, and in some form of permanently recordable way (e.g. video footage), then monitoring the persistence of a habitat may suffice to indicate that preservation is continuing. Some combination of frequent qualitative checks, combined with less frequent, more detailed (and generally more expensive) quantitative sampling, appears to be the most cost-effective option for ensuring preservation.

Connell & Sousa (1983) provide some theoretical guidelines on the duration of studies required to assess stability of communities. They suggest that to assess stability, studies need to continue for at least twice the lifespan of the species of interest. For kelp *Ecklonia radiata* this means at least 6 years in shallow water (< 6 m) and 10+ years in deeper waters (T.R. Haggitt, University of Auckland, unpubl. data); for kina *Evechinus chloroticus* this means 10+ years (based on Walker 1981), and for snapper *Pagrus auratus* this means 100+ years (Crossland 1981). Paul (1992) summarised ageing data for a variety of coastal fishes, and estimates of maxima are given in Table 1. Based on these data it is clear that to assess stability of communities, monitoring should continue for decades.

TABLE 1. MAXIMUM AGES OF COASTAL FISHES SUMMARISED FROM PAUL (1992).

COMMON NAME	SCIENTIFIC NAME	ESTIMATE OF MAXIMUM AGE (YEARS)
Golden snapper	*Centroberyx affinis*	80
John dory	*Zeus faber*	8
Red gurnard	*Chelidonichthys kumu*	13
Rockfish	*Acanthoclinus fuscus*	9
Jack mackerels	*Trachurus* spp.	22, 28
Koheru	*Decapterus koheru*	5–10
Trevally	*Pseudocaranx dentex*	45
Kahawai	*Arripis trutta*	22–26
Snapper	*Pagrus auratus*	50
Red moki	*Cheilodactylus spectabilis*	59
Tarakihi	*Nemadactylus macropterus*	40
Blue moki	*Latridopsis ciliaris*	33
Grey mullet	*Mugil cephalus*	13
Spotty	*Notolabrus celidotus*	7
Butterfish	*Odax pullus*	10
Opalfish	*Hemerocoetes monopterygius*	3
Blue cod	*Parapercis colias*	17
Variable triplefin	*Forsterygion varium*	4
Barracouta	*Thyrsites atun*	10
Blue mackerel	*Scomber australasicus*	12–15
Lemon sole	*Pelotretis flavilatus*	5+
Common sole	*Peltorhamphus novaezeelandiae*	5
Yellowbelly flounder	*Rhombosolea leporina*	4+
Sand flounder	*Rhombosolea plebeia*	3+

1.3 SHOULD MONITORING STOP?

Monitoring should stop only if management is also going to stop, as the consequences of stopping monitoring are uninformed management decisions. As monitoring of organisms is carried out routinely in relatively few marine reserves, management of those reserves must be done on the basis of managing visitors, rather than managing natural assemblages. There are few studies relevant to such matters (but see Wolfenden et al. 1994; Cocklin et al. 1998). Furthermore, managers need to be aware of how variable natural assemblages are; for example, the loss of most of the northeastern offshore species from the Cape Rodney–Okakari Point Marine Reserve during the later 1980s and 1990s reflected intermittent larval supply, rather than some sort of detrimental impact on the reserve. If most of the kelp forest in a marine reserve (hundreds of ha) can die out from a natural process and be replaced (as occurred at Cape Rodney–Okakuri Point, see Cole & Babcock (1996)), it is unlikely that experimental clearances of a few hundred m^2 will be of lasting consequence. In the real world (and with mobile patchily-distributed species) a count of 0 is quite likely to arise from sampling error (McArdle et al. 1990). The loss of a species from a site or, indeed, an entire reserve, does not mean that it will remain absent, and does not necessarily indicate a problem that requires management. Alterations to monitoring regimes to maximise efficiency appear justifiable.

The relative costs of different types of monitoring obviously influence the schedule and methods of sampling undertaken. If divers are used, a mean value for a site, based on 5 replicate samples, might be 6 person hours. However, BUV (baited underwater video—Willis & Babcock 2000) samples at a single site can really only be replicated over days so that although the costs associated with divers might be reduced, 5 replicate samples might take 10–15 person hours, because the filming would have to be done on different days. Travel costs will differ markedly between sites. Minimising travel costs usually means restricting the number and length of monitoring visits and this can mean that the data collected are inappropriate or less useful than they could be.

1.4 SAMPLING / STATISTICS

Monitoring of marine reserves usually generates data in the form of a series of values for fixed sites. Monitoring is usually carried out annually. This can be because of the funding cycle, because of seasonal differences in the distribution and abundance of target species, or because a particular season is the only one in which conditions are suitable for sampling. However, annual sampling imposes limitations on detecting change. Over the scale at which time series have been sampled to date, only very large changes or effects will be detected. For example, In most reserves < 10 years old, only 5–10-fold changes will be detected. The most accurate results will be attained via high-precision estimates (which will reduce the duration of monitoring), or by extended duration of monitoring.

1.5 SENSITIVITY TO DIFFERENCES IN TIMING OF EVENTS

In establishing long-term monitoring programmes, it is important to consider normal animal movements or life history patterns. Fish may be elsewhere than the area being monitored during a particular season; in deeper or shallower water, for example. Adults of annual species may be visually apparent only at particular times of the year, while their microscopic larval stages may be abundant year-round. Before interpreting monitoring data for any particular species, it is useful to have a good understanding of its ecology. While some data are available on the temporal variability of large common fish species in the north of New Zealand, similar data are lacking for most other parts of New Zealand.

1.6 STATISTICAL METHODS OF DETECTING CHANGE

Gerrodette (1987) provides useful methods for power analyses for detecting trends. Assuming that reserve monitoring data are in the form of quadrats or transects, then Table 1 of that paper indicates that the coefficient of variation (CV) should be proportional to $1/\sqrt{A}$, where A is the abundance. This study gives a series of formulae expressing relationships among the following parameters:

n the number of samples

r the rate of change of the variable being measured

CV the coefficient of variation

α Type 1 error rate

β Type 2 error rate

By setting $\alpha = \beta = 0.05$, Gerrodette (1987) derives a generalised and simplified formula expressing the relationship among 3 parameters:

$r^2n^3 \geq 156CV^2$ Equation 1

The CV may be reduced by taking more than 1 estimate at each interval. In these situations, the CV based on m independent replicate estimates is equal to

$CV / \sqrt{m}$ Equation 2

There are a number of provisos and caveats on those formulae, the most important of which is that the trend is linear through time. Given that we are really only interested in an approximation, that assumption will be made for the remainder of the treatment here. Using calculated values for the CV from various existing monitoring programmes, we can solve Equation 1 for varying rates of change r, to give minimal values of n by rearranging the formula as:

$n^3 \geq 156CV^2 / r^2$

Figure 1 provides this information for varying combinations of CV and duration. A range of CVs for single site surveys of marine reserves is provided in Table 2.

The time series available provide an indication that the r values required to detect change may be attained in the real world. For example, fished spiny lobster populations at Hahei have declined from about 10 in 1996 to 1 in 2001 (Fig. 2A). According to Gerrodette (1987) the decline should be constant, and for the Hahei population, a linear trend looks appropriate. The slope of a model I regression line is calculated to be –1.7 (s.e. = 0.35), whereas that for the increasing spiny lobster population within Cathedral Cove Marine Reserve at Hahei is 0.28 (s.e. = 0.87). (Note that the trend of increase within the marine reserve is nowhere near as clear-cut as for the decline outside). The rate of increase is obviously highly variable, and analysis indicates that, with the estimated rate of increase (0.28) and the calculated CV of 1.0, the reserve would have to be monitored for at least 13 years before the spiny lobster population could be assumed to have reached some sort of stability. Dividing the data up more finely (by depth strata) does not assist the analysis (Fig. 2B). Gerrodette (1987) notes that decreasing trends are easier to detect than increasing ones (p. 1367).

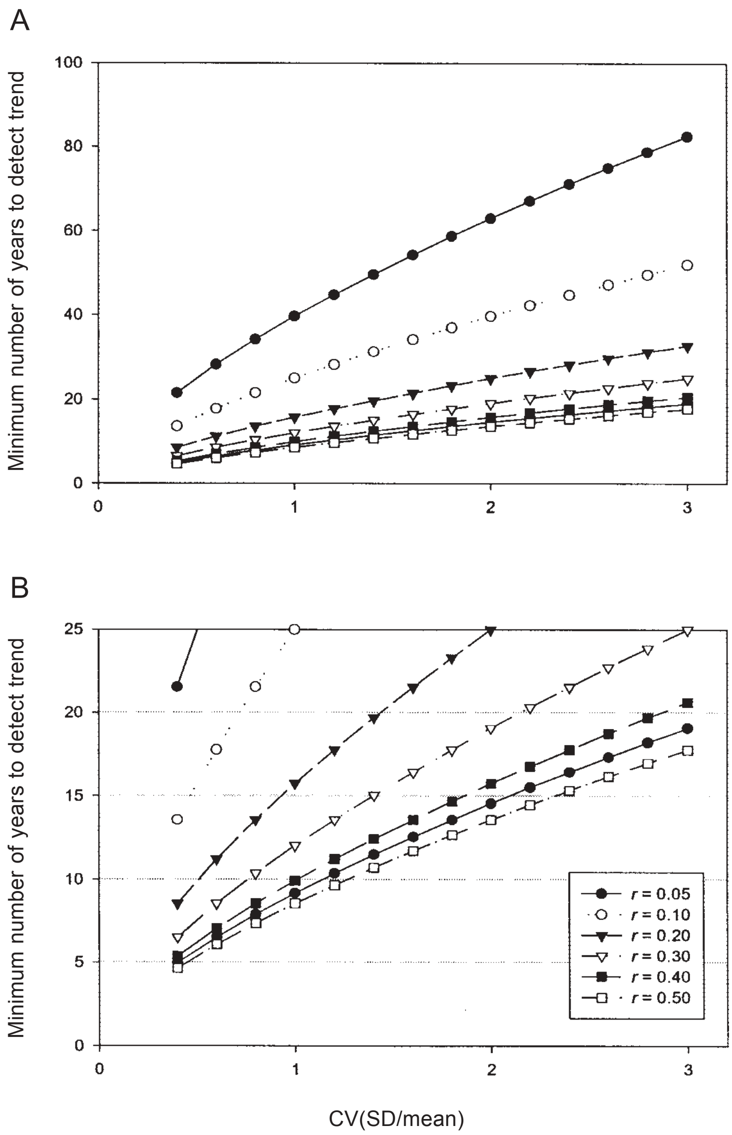

Figure 1. Durations of sampling required to detect change with single annual samples for various CVs and *r* values. Graph B indicates a detail from graph A, with limited y-axis range.

At Te Angiangi Marine Reserve, Hawkes Bay, banded wrasse, scarlet wrasse, spotty (less clearly), and spiny lobster all showed trends in population abundance that could be (loosely) described as linear (Fig. 3). Values of *r* for all species were estimated for fished areas outside the reserve (Control) and within the reserve (Reserve), from regressions of mean abundance on year (Table 3). When NIND (= number of individuals) was excluded (as it is strongly influenced by the most abundant species), the slopes for Reserve and Control (fished) were compared (using the 27 individual species and NSP (= number of species) as replicates). The mean slope for Control was 0.002 (s.e. = 0.02), whereas the mean slope for Reserve was 0.176 (s.e. = 0.104) (a t-test shows the difference in slopes to be not statistically significant at alpha = 0.05). There is clearly a trend for higher values of slope (and thus patterns of increasing abundance) with time. A boxplot (Fig. 4) emphasises the highly skewed nature of the data. Obviously, the duration of the data is short, many of the relationships that have been grouped for that analysis will not be linear, and the analysis represents a

TABLE 2. CVs FOR SITE SURVEYS AT LONG ISLAND – KOKOMOHUA MARINE RESERVE (R.J. DAVIDSON, DAVIDSON ENVIRONMENTAL LTD, NELSON, unpubl. data AND DAVIDSON 2001). WHERE THERE IS MORE THAN 1 SITE, THE AVERAGE CV HAS BEEN TAKEN.

SPECIES	RESERVE		CONTROL
	1999 CV (n = 3 sites)	2000 CV (n = 3 sites)	1999 CV (n = 1 sites)
Blue cod < 10 cm	—	—	—
Blue cod >10, < 30 cm	2.1	2.4	2.2
Blue cod > 30	2.2	2.2	—
Total blue cod	1.6	1.7	2.2
Spotty	0.8	0.7	0.8
Banded wrasse	1.6	1.0	2.3
Butterfly perch	2.4	2.0	—
Scarlet wrasse	1.9	1.7	—
Blue moki < 40 cm	1.4	1.1	1.3
Blue moki > 40 cm	1.7	2.2	3.5
Total blue moki	1.4	2.2	—
Leatherjacket	1.9	2.5	—
Tarakihi	2.1	1.9	—
Butterfish	2.3	1.9	—
Red moki	2.6	2.0	—
Marblefish	3.5	3.5	—
Sweep	3.5	2.3	—
Opalfish	—	—	—
Trumpeter	—	—	—
Seahorse	—	—	—
Goatfish	—	—	—
Copper moki	—	3.5	—
Blue maomao	—	3.5	—
Number of individuals	0.6	0.7	0.8
Number of species	0.3	0.4	0.8

— indicates that species or size class was absent.

gross simplification. However, it is clear that some populations of marine organisms may indeed exhibit patterns of population change sufficiently large to be able to be detected within periods of less than 10 years. Whether that is a general trend for New Zealand reserves awaits a more detailed analysis of a fuller dataset.

In Fig. 5, years required to detect a trend (with single annual surveys) are plotted against various rates of change and CVs. In Fig. 5A, two CVs in keeping with those in Gerrodette (1987) are plotted. Based on the data in Tables 1 and 2, a more realistic range of values is considered in Fig. 5B, along with more realistic ranges of values for *r*. The higher real world CVs are balanced out by the higher absolute values of *r*, and thus the range of durations required for monitoring is not much higher than those in Fig. 5A. Note that the absolute values of rates of increase found at Te Angiangi and Hahei differ between reserve and fished areas; with large negative rates occurring outside Cathedral Cove Marine Reserve at Hahei, and (generally) more moderate positive rates occurring within the reserve; whereas the absolute values for positive rates are much higher at Te Angiangi Marine Reserve. Time to detect change will also

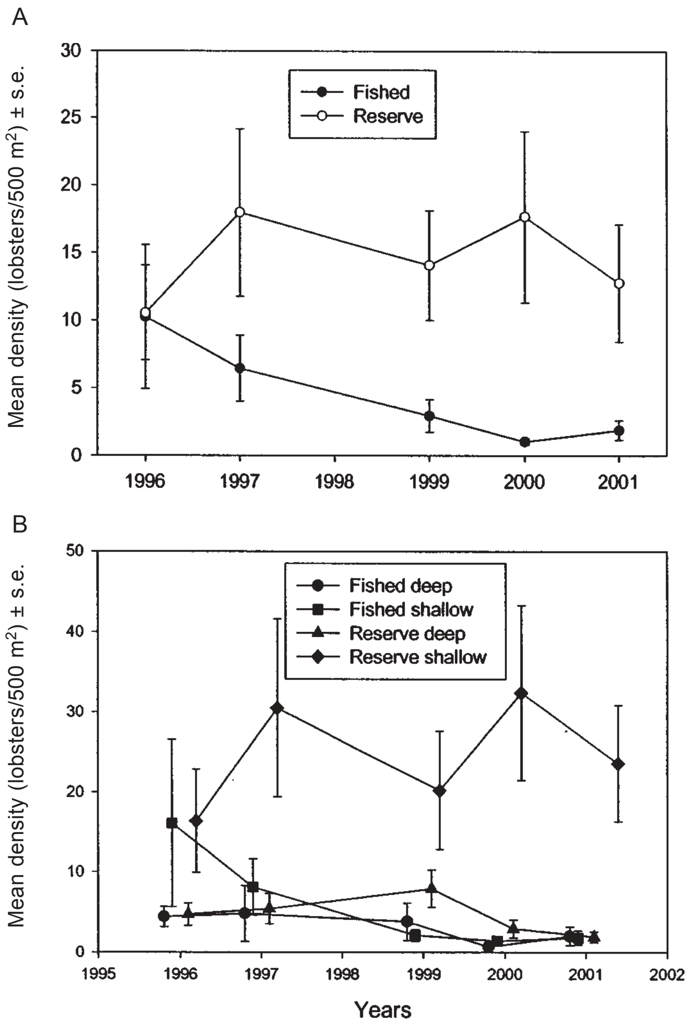

Figure 2. Abundances of spiny lobster at Te Whanganui a Hei Marine Reserve and adjacent fished areas between 1996 and 2001.

depend on how severe any impact is when a reserve is set up and monitoring starts. If the impact is small, recovery may not be observed. In that situation, intensive sampling may be required to reveal the small changes that may occur. It is difficult to discern how impacted populations are in order to decide on a sampling frequency.

Detecting a change in a statistical test derived from a BACI design will take less time than the durations indicated here because any positive trend within a reserve will be complemented and exacerbated by any negative trend outside it. It seems reasonable to expect patterns to emerge after 10 years on the basis of the analyses above, especially if a statistical analysis combines reserve and non-reserve data. However, note that the change from coralline flats to seaweed forest over much of the area between 4 and 10 m depth in Cape Rodney–Okakari Point Marine Reserve occurred in the late 1990s, about 20 years after reserve establishment. When the changes did occur they occurred patchily among sites, and it may be that increased spatial variability is an early indicator of temporal change.

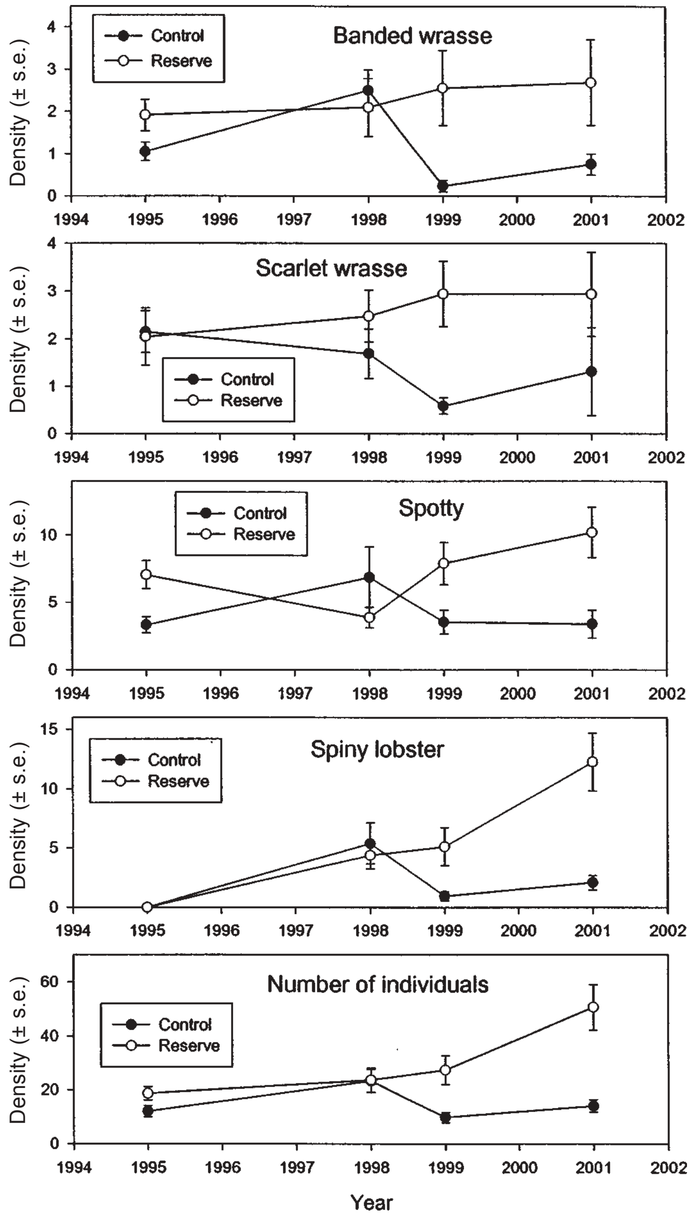

Figure 3. Densities (fish/500 m^2) of abundant fish species at Te Angiangi Marine Reserve, and in adjacent fished areas; *n* varies between 16 and 22 transects per treatment (reserve and control) per year.

Figure 4. Boxplot comparing regression slopes (equivalent to *r*) of mean abundance v. time, from Te Angiangi Marine Reserve and fished control areas.

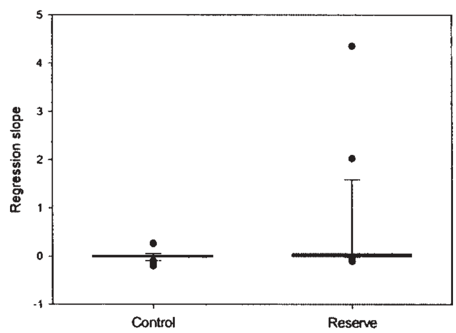

TABLE 3. SLOPES OF REGRESSIONS (AND S.E.) OF ABUNDANCE v. YEAR, FOR TE ANGIANGI MARINE RESERVE, HAWKES BAY, AND FISHED AREAS OUTSIDE THE RESERVE. BASED ON 4 SURVEYS FROM 1995 TO 2001. (SPECIES CODES ARE RETAINED AS PER THE DATA).

SPECIES	FISHED	SE	RESERVE	SE
bwr	−0.09592	0.26611	0.01931	0.14959
scwr	−0.17914	0.13714	0.12067	0.05938
spo	−0.02749	0.49159	0.34258	0.52947
bp	−0.20453	0.16609	0.03422	0.0449
bu	0.05902	0.20715	−0.0876	0.1896
le	−0.02631	0.02776	0.03563	0.02986
bm	0.02337	0.06573	0.0208	0.07591
bc	0.02489	0.12554	−0.10269	0.26871
mao	−0.00867	0.00353	0.05464	0.05677
sp	−0.01733	0.00706	0.02951	0.01996
ta	−0.08648	0.10044	0.01087	0.06487
ma	−0.00413	0.01158	0.01429	0.11859
sc	−0.00277	0.01338	−0.00386	0.04812
rm	0.01537	0.07312	0.05088	0.07618
sw	0.264	0.30059	1.72966	0.66225
cray	0.26848	0.63515	2.02673	0.32401
hi	0.00235	0.00815	—	—
cr	0.0075	0.01607	−0.00433	0.01479
trev	—	—	0.08772	0.04476
jd	0.00235	0.00815	—	—
ray	0.00235	0.00815	0.00042	0.00514
rc	0.00917	0.00601	−0.00824	0.00898
brc	0.00917	0.00601	−0.00475	0.00987
king	0.00917	0.00601	—	—
roc	0.00917	0.00601	0.00042	0.00514
ce	—	—	−0.01974	0.02452
po	—	—	0.00919	0.00516
NIND	0.05359	1.68023	4.35633	1.60218
NSP	−0.01416	0.22451	0.05079	0.24916

NIND = number of individuals, NSP = number of taxa.

1.7 MEASURING CHANGE

It is understood that the Department of Conservation has adopted the sampling practices and analyses recommended in Kingsford & Battershill (1998). That book, and particularly Chapter Three (Kingsford 1998), relies heavily on the methodology summarised in Underwood (1997), and developed through a series of papers prescribing analysis of variance and null hypothesis testing. Although this approach it is widely accepted in southern hemisphere ecological literature, it has been heavily criticised by McBride (1999), Germano (1999, 2001), and Cole et al. (2001). It is even less applicable to the monitoring situation, where the accumulation of data will inevitably lead to null hypotheses being rejected (with fixed Type I error rates), because of large sample sizes. Statistical (and sampling) techniques are always developing. The most robust approach for monitoring marine reserves will be to collect as much data as possible, both inside and outside the reserves, and to focus on the quality of the data, rather than the intricacies of the statistical analysis.

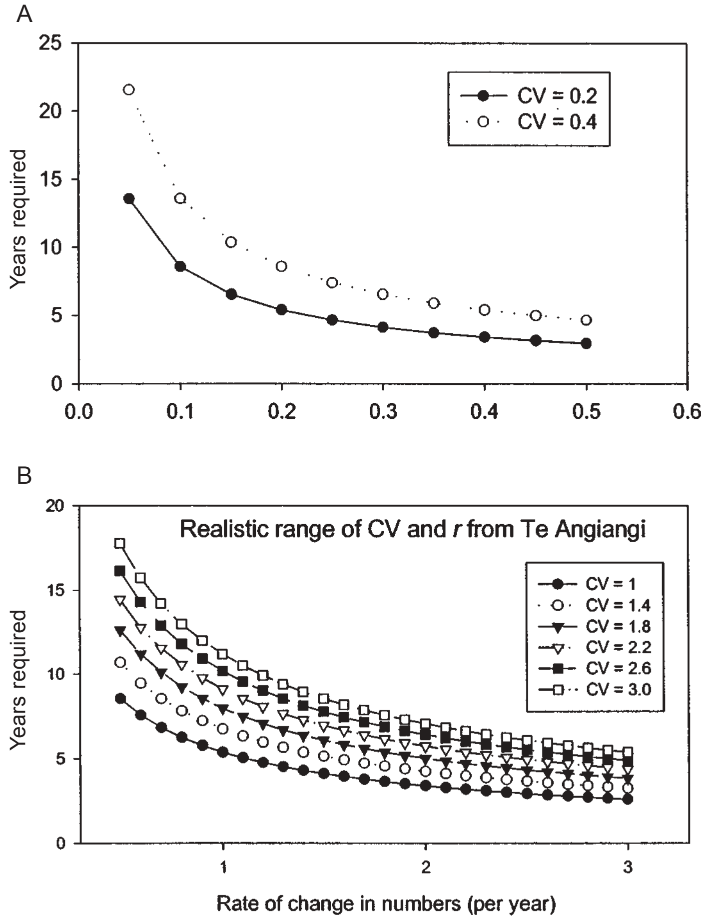

Figure 5. Durations of annual monitoring required to detect various rates of change, for various CV values.

1.8 PRECISION OF ESTIMATES AT A SITE, v. PRECISION OF ESTIMATES FOR RESERVE

Increasing the number of samples increases the precision of an estimate. Gerrodette (1987) provides formulae for the precision of the estimate of a quantity, noting that precision increases (i.e. the s.e. gets smaller) as an inverse square root function of m (Equation 2 above). Studies based on nested analysis of variance usually have replicate sites within reserve and fished (outside reserve) areas, and replicate samples within these sites. Such **hierarchical** designs are usually poor at maximising precision; the highest replication arises for single samples from sites (since that provides greatest *n*). Consider the allocation of 40 samples, from a population which we know has SD = 10. With 4 sites, and 10 replicates at each, the SE = $10/(\sqrt{4})$, or 5. With 20 sites, and 2 replicates at each, the SE = $10/(\sqrt{20})$ or about 2, and if only 1 sample was taken at each site, then the SE would approach 1.6. (This is the format of the data for Te Angiangi Marine Reserve). The most efficient designs are thus those that allocate effort as high up the hierarchy as possible; maximise the number of sites and minimise the number of replicates per site. Diving safety imposes constraints on the allocation of effort in that way, but allocating extensive sampling effort to large sample sizes at each site is inefficient if the aim is to maximise precision for an entire reserve.

The approach favoured by Stewart-Oaten & Bence (2001) is to take repeated samples at a limited number of chosen sites. They provide strong arguments for that approach, but some characteristics of marine reserves suit it poorly. For example, although marine reserve protection may be imposed over 5 km of coast on paper, the fish within 500 m at either end may be subject to fishing, either directly by covert poaching, or because they are attracted to berley outside the reserve. The fish in the centre of the reserve may not be able to be sampled because feeding by divers has changed their behaviour. Thus deciding where to locate sampling sites might not be as straightforward as might be expected. Moreover, there is reason to be concerned about putting too much emphasis on the time-course of events at a limited number of sites. Spatial variability is large, and there is no clear way to express spatial and temporal variability in the same units. Note that in sampling designs with repeated samples taken at a limited number of sites, allocating more effort to replication within sites **will** increase efficiency, in constrast to the situation described in the previous paragraph.

Whichever approach is adopted, its efficiency will depend on the relative magnitudes of spatial and temporal variation. The only recommendation that appears worth making here is to take as many samples as possible, as frequently as possible.

There are several graphical approaches to detecting change in a sampled variable. Kingsford (1998) describes cusums (see Manly 1994). These suffer from not being widely implemented in software packages (but see SAS JMP), and are not generally used in published ecological papers. However, Neill et al. (1992) emphasise that plotting the data by hand is valuable (and may be quicker than some computer-based plotting techniques!). There is a large body of quality control statistical methodology that is little-known in the ecological literature, but very relevant to the monitoring situation. These approaches require frequent samples; qualitative data that could be collected cheaply (e.g. percent cover of kelp canopy sampled by drop camera at five random points within a 20-m radius of a GPS mark) would greatly increase the value of monitoring. Moreover, such approaches would allow DOC field staff to be more readily involved in the collection—and perhaps analysis—of monitoring data (an important consideration given that the people who most routinely visit most marine reserves are DOC field staff).

In Fig. 6, abundances of sea urchins *Evechinus chloroticus* (R. Cole unpubl. data) are plotted as an example of a cusum application. In this example the V-mask bounds are exceeded at a point corresponding to 1991, and remain beyond them until about 1998. Deviations above the control line indicate that the process has undergone a downward shift, and thus it appears that early in 1992 the population declined more rapidly. To obtain maximum utility from this approach it would be necessary to plot the cusums as the data were collected. In that way some form of observational investigation could be started once the departure from statistical control was detected. Note that the data are simply ranked in time.

Formal time-series analysis typically requires hundreds of samples; given that most time series for marine reserves are going to be over 4–5 years to start with, sampling would have to occur weekly or fortnightly to get the required data

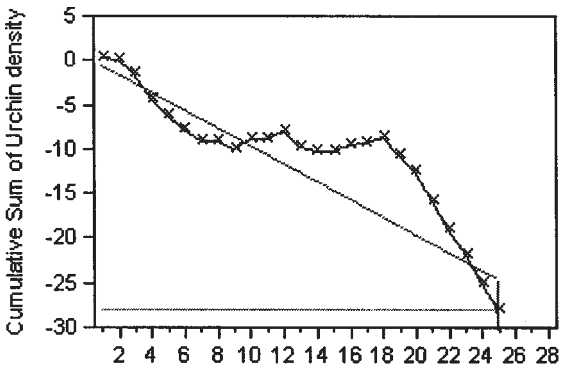

Figure 6. Cusum plot of densities (number/m^2) of the sea urchin *Evechinus chloroticus* at Waterfall Reef, Cape Rodney–Okakari Point Marine Reserve. See Fig. 7 for plot of raw densities.

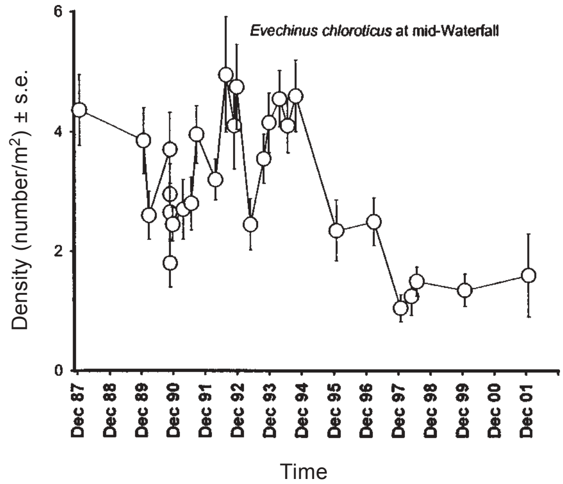

Figure 7. Time series of densities of *Evechinus chloroticus* at Mid Waterfall Rock flats.

volume. The full edifice of time-series analysis is probably well beyond the training of most biologists and advice from statisticians should be sought.

As yet there are no substantial ecological time series from marine reserves in New Zealand, and hence there has been no need to develop or even begin to use large time series for ecological problems. One area where such methods have been developed is in water quality analysis. There are well-established protocols for water quality monitoring run by NIWA (Smith & McBride 1990). As these are used for public health purposes, the methods of analysis that they use would be expected to have suitably stringent standards, and be applicable to ecological monitoring. The emphasis in the water quality program is on data quality. Given that data quality for quantitative samples of mobile animals is likely to be poor (because of variation in underwater visibility, mobility and patchiness of animals, different observers being involved etc.), ways of standardising the data collection process as much as possible are to be encouraged. (Note that the variables of interest in water quality monitoring—abundance of faecal coliforms etc.—are basically densities. It would therefore stand to reason that some type of approach similar to that developed for water quality monitoring would be of value in ecological questions). Removing among-diver variation for counts of mobile organisms is a strong argument for use of baited video, where the actual raw data (i.e. videotapes) can be checked over and used for practice by new workers and archived. This and similar techniques have the disadvantages of only working for carnivores that are commonly found in the open, and rely on reasonable underwater visibility (as do fish counts). Willis & Babcock (2000) and R.G. Cole et al. (unpubl. data) have found them to be suitable for the most common carnivorous fish species in northern areas of both the North and South Islands.

Returning to water quality monitoring, the relatively predictable patterns of seasonal change in some variables (and the long time series) provide opportunities for removal of seasonal trends. For some species (e.g. spiny lobster *Jasus edwardsii*) sufficient ecological information is now available to permit the construction of seasonal trend curves. The water quality literature also provides methods for determining how many samples need to be taken to be confident that values of a variable are at a certain level. For example, there are operating characteristic charts that give the sample size necessary to determine whether a particular standard is being maintained. That approach might be suitable for deciding whether to commence a more expensive quantitative survey with divers. We also note the development of low-cost video for monitoring of deep water areas on the Great Barrier Reef (e.g. Cappo et al. 2000).

1.9 ADDITIONAL IDEAS

1. Imposing a reserve not only removes fishing effort from an area, but probably also increases it in nearby areas via displacement. It may be helpful to include monitoring of fishing effort in neighbouring areas in future monitoring programmes. Department of Conservation field staff could collect such information by simply recording the number of boats fishing in areas that they regularly pass through. There is a wider need for information on the response of fishers to reserves.
2. Develop cheaper ways of monitoring; for example, by using DOC field staff to monitor sites on weekly / monthly basis with low-cost equipment. Costs need not be prohibitive e.g. Drop camera +30 m of cable = $900, digital 8 video = $1500. Videotapes = $20 each, and it should be possible to record 20+ sites per tape.
3. There is some scope for developing existing methodologies to assist monitoring. Aerial photographs have been used for monitoring of studies recently (Andrew & O'Neill 2000), and there may be archival material available.

2. Summary

Marine reserves should be monitored for decades. If management is to be based on biology in the reserve, rather than on behaviour of people, monitoring needs to be an ongoing process. Monitoring of both marine organisms and humans is probably important. Developing ways to sample cheaply and frequently will improve the usefulness of monitoring, and reduce management reaction times. The current emphasis on in situ diver observations cannot be maintained at present costs and with increasing reserve numbers.

3. Acknowledgements

Thanks to Sean Handley for his review of the draft, and all those who collected data.

4. References

Andrew, N.L.; O'Neill, A.L.O. 2000: Large-scale patterns in habitat structure on subtidal rocky reefs in New South Wales. *Marine and Freshwater Research 51*: 255–263

Cappo, M.; Speare, P.; Wassenberg, T.; Harvey, E.; Rees, M.; Heyward, A.; Pitcher, R. 2000: Use of baited remote underwater video stations (BRUVS) to survey demersal fish—how deep and meaningful? In Harvey, E.S.; Cappo, M.: Direct sensing of the size frequency and abundance of target and non-target fauna in Australian Fisheries—a national workshop. 4–7 September 2000, Rottnest Island, Western Australia. Fisheries Research Development Corporation. 187 p.

Cocklin, C.; Craw, M.; McAuley, I. 1998: Marine reserves in New Zealand: use rights, public attitudes, and social impacts. *Coastal Management 26*: 213–231

Cole, R.G.; Babcock, R.C. 1996: Mass mortality of a dominant kelp (Laminariales) at Goat Island, north-eastern New Zealand. *Marine and Freshwater Research 47*: 907–912

Cole, R.G.; McBride, G.; Healy, T.R. 2001: Equivalence tests and sedimentary data: dredge spoil disposal at Pine Harbour Marina, Auckland. *Journal of Coastal Research Special Issue 34* (ICS 2000 New Zealand): 611–622.

Connell, J.H.; Sousa, W.P. 1983: On the evidence needed to judge ecological stability or persistence. *American Naturalist 121*: 789–824

Crossland, J. 1981: The biology of the New Zealand snapper. New Zealand Ministry of Agriculture and Fisheries, *Fisheries Research Division Occasional Publication 33*, 19 p.

Davidson, R.J. 2001: Changes in population parameters and behaviour of blue cod (*Parapercis colias*; Pinguipedidae) in Long Island – Kokomohua Marine Reserve, Marlborough Sounds, New Zealand. *Aquatic Conservation: Freshwater and Marine Ecosystems 11*: 417–435.

Germano, J.D. 1999: Ecology, statistics, and the art of misdiagnosis: the need for a paradigm shift. *Environmental Reviews 7*: 167–190.

Germano, J.D. 2001: Reflections on statistics, ecology and risk assessment. In: Organism – sediment interactions. Aller, J.Y.; Woodin, S.A., Aller, R.C. (Eds). University of South Carolina Press, Columbia. Pp. 33–42.

Gerrodette, T. 1987: A power analysis for detecting trends. *Ecology 68(5)*: 1364–1372.

Kingsford, M.J.; Battershill, C.N. (Eds) 1998: Studying temperate marine environments. A handbook for ecologists. Canterbury University Press, Christchurch. 335 p.

Kingsford, M.J. 1998: Analytical aspects of sampling design. In: Studying temperate marine environments. A handbook for ecologists. Kingsford, M.J.; Battershill, C.N. (Eds) Canterbury University Press, Christchurch. Pp. 49–83.

Manly, B.F.J. 1994: CUSUM methods for detecting changes in monitored environmental variables. Pp. 239–253 in Fletcher, D.J.; Manly, B.F.J. (Eds): Statistics in Ecology and Environmental Monitoring. University of Otago Press, Dunedin., Dunedin, New Zealand.

McArdle, B.H.; Gaston, K.J.; Lawton, J.H. 1990: Variation in the size of animal populations: patterns, problems and artefacts. *Journal of Animal Ecology 59*: 439–454.

McBride, G.B. 1999: Equivalence tests can enhance environmental science and management. *The Australian & New Zealand Journal of Statistics 41*: 19–29.

Neill, A.; Harper, S.; Dale, J.; Krouse, D. 1992: Of cusums and quality: a practical guide to using and installing cusums. *DSIR Physical Sciences Report 43*. DSIR Physical Sciences, Wellington. 36 p.

Paul, L.J. 1992: Age and growth studies of New Zealand marine fishes, 1921–90: a review and bibliography. *Australian Journal of Marine and Freshwater Research 43*: 879–912.

Smith, D.G.; McBride, G.B. 1990: New Zealand's national water quality monitoring network—design and first year's operation. *Water Resources Bulletin 26*: 767–775.

Stewart-Oaten A.; Bence, J.R. 2001: Temporal and spatial variation in environmental impact assessment. *Ecological Monographs 71*: 305–339.

Underwood, A.J. 1997: Experiments in ecology: their logical design and interpretation using analysis of variance. Cambridge University Press, Cambridge. 504 p.

Walker, M.M. 1981: Influence of season on growth of the sea urchin *Evechinus chloroticus*. *New Zealand Journal of Marine and Freshwater Research 15*: 201–205.

Willis, T.J.; Babcock, R.C. 2000: A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research 51*: 755–763.

Wolfenden, J.; Cram, F.; Kirkwood, B. 1994: Marine reserves in New Zealand: a survey of community reactions. *Ocean and Coastal Management 25*: 31–51.