Rapid high-precision monitoring of coral communities to support reactive management of dredging in Mermaid Sound, Dampier, Western Australia.

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Abstract

Embarking on a major dredging program with a lack of agreement on the potential risk to adjacent coral communities, there was a need to develop a rapid-turnaround monitoring program for adaptive management of the dredging activity. Triggers for management were agreed to be measures of decline in coral community status. Practical considerations made the decline of cover of live coral the only viable indicator for this area.

A program was designed to support fortnightly assessment of any decline in coral cover above 10%. A fortnightly monitoring program was implemented at 19 sites for a period of 12 months to cover 2 dredging projects in the Dampier Harbour, Western Australia.

Between November 2003 and December 2004, the majority of change seen in these coral communities resulted from natural phenomena, including freshwater inundation, wave action and competition with macroalgae. Mortality from dredging activities was apparent at one site only, where a coral community occurred within a few hundred metres of intense dredging. Mortality appeared to be due to direct smothering of corals by coarse and fine sediments. Other populations within 1km of dredging and spoil disposal sites were exposed to greatly elevated turbidity for weeks at a time, but showed no resulting change in coral cover.

While video or photographic recording of coral cover using line transects is commonly used in surveys of coral abundance, accuracy and precision are rarely made explicit for these techniques. The survey technique used here employed 5 replicated transects at each monitoring site, assessed by 10 images from fixed intervals along the transect. Using 50 intercept points per image allowed this technique to test the hypothesis of 10% change with statistical power between 0.35 (for sites where coral cover was low, <30%, and patchy) and 0.95 (for sites where coral was abundant, >50%, and more even).

An assessment of the effect of changing the sampling intensity within images (using 5, 10, 25 & 50 points) showed that at sites where coral cover was low and patchy, increasing sampling intensity greatly increased precision, but at sites where coral cover was high and even, it had little effect.

Keywords: sediment, turbidity, coral, monitoring, dredging

Introduction

Monitoring Impacts of Dredging/Disposal on Coral Reefs

Dredging and the subsequent disposal of spoil result in the suspension of bottom sediments into the water column at sites of uplift and disposal. Ignoring the issue of liberation of chemical toxicants or nutrients from contaminated sediments, the principal factors likely to impact corals will be limited to direct sedimentation and decreased light availability. Even with this simple set of impacts, predicting potential responses and how to monitor them is not straightforward. Sediment and turbidity can impact on a variety of ecological and physiological process including growth, mortality, reproduction and recruitment (Rogers 1990) as well as competition with other biota (Nugues & Roberts 2002).

A large number of studies describe change (or lack thereof) in response to increased sedimentation or turbidity in corals for differing parameters. Some examples include:

- community structure (Loya 1976, Dodge & Vaisnys 1977, Cortes & Risk 1985) - generally a long term issue this is more relevant where dredging may change the sediment/turbidity pattern over several years;
- short term mortality (Brock et al 1966; Dodge & Vaisnys 1977, Stafford-Smith et al 1993) – the most commonly assessed and described response;
- sub lethal indicators of stress – including growth rate and reproduction (Brown & Howard 1985, Stafford Smith et al 1993, Jones 1997; Bourke 2004)

Ideally, monitoring conducted as an input to adaptive management of dredging operations should be directed at sublethal effects whenever practical. However, in many cases this is not practical as the measurement of these indicators can require time frames unsuited to operational dredge management or their interpretation can be confounded by the indicator's response to factors unrelated to dredging. Bleaching of coral tissues through the expulsion of zooxanthellae has been used as an indicator for adaptive management of dredging programs on the Great Barrier Reef, where it has formed part of a broader decision-making scheme (Stafford-Smith et al 1993). While there are suggestions that the technique is not sufficiently sensitive to isolate dredge-related impacts (Hoegh-Gulberg et al 1996), its use as a monitoring tool continues around dredging projects.

Similar concerns would apply to the use of pulse amplitude modulation (PAM) fluorescence measurement from zooxanthellae in corals (Jones et al 1999) as a sublethal indicator of dredging effect. Again, this is primarily an indicator of the relationship between coral and zooxanthellae, which may be affected by many other factors unrelated to sediment stress (Hoegh-Gulberg et al 1996).

Recent studies on the ratio of structural and storage lipids in corals show some promise of providing sub-lethal indication of stress (Harriott 1993; Ward 1995). However techniques currently in use may not be practical in providing rapid assessments and lipid metabolism may be confounded by differing responses from coral taxa with differing levels of resilience to sedimentation (Bourke 2004, Saunders et al. in prep).

Monitoring of the Dampier Dredging Programs

During 2003, two major dredging programs proposed in Mermaid Sound, Dampier Harbour, Western Australia (see Stoddart & Anstee, this volume) were referred to the Western Australian Environmental Protection Authority (EPA) for assessment. Coral communities were known to exist within distances where water quality was likely to be affected by these sediments and the potential for impact on these populations was considered a significant factor (EPA 2003 a+b).

While dredging and sea dumping had occurred previously within similar areas of the Harbour without documented impacts on these coral populations, the intensity of monitoring undertaken for those programs was deemed insufficient to allay fears of impact for these large, possibly contemporaneous, projects. Therefore the EPA assessment recommended implementation of a program of adaptive management for dredging and disposal, governed by the results of coral impact monitoring.

Initial attempts to develop a monitoring program reviewed the efficacy and practicality of using coral bleaching as a sublethal indicator. In addition to the uncertainty in interpreting results, there were substantial practical problems that mitigated against such a technique. The Dampier Harbour marine environment is a naturally turbid area (Stoddart & Anstee, this volume), and visibility can be reduced to effectively zero in waters adjacent to dredging operations. In such an environment, visual data can be unavailable for long periods.

Further, partial mortality of coral colonies resulting from sedimentation occurs when fine sediments smother corals. In this case, bleaching is only visible once the sediment covering is brushed from the coral to reveal the underlying tissue. Such an intervention would invalidate any sampling design using repeated sampling of regularly brushed corals to reflect the broader population.

It was decided to use estimates of actual mortality as indicators of impact. For monitoring to be used as a trigger for change in management of the dredging, the program needed to be able to identify the occurrence of dredging-related mortality before unacceptably large proportions of coral communities were destroyed.

Using Connell (1997) as a basis, the EPA determined that a 30% mortality event would represent an unacceptable level of impact for these populations. As an arbitrary management trigger and to allow for the routinely dynamic nature of corals, it was agreed that a 10% mortality effect would represent an indication that dredging- induced mortality had started and the management should be applied to avoid breaching the 30% criterion. Based on a similar mix of practical constraints and biological processes, the frequency of mortality monitoring was set to a fortnightly cycle.

Previous studies of coral communities in the Harbour have confirmed that the amount of live coral cover can change substantially in response to weather-related and other phenomena (Blakeway, this volume). To avoid unnecessary restrictions on dredging and disposal that could result from mortality caused by the impacts of a widespread factor unrelated to dredging, Net Mortality (i.e. Impact less Reference level) was used as the management trigger.

In practice, the assessment of mortality usually involves repeated measures of live cover over time. Monitoring coral cover in benthic communities has a long history from very fine scale observations (sensu Connell et al 1997) to large scale surveys (sensu Carleton & Done 1995). Studies using the latter methods are often more concerned with a broad description of the status of coral reefs as a whole (English et al. 1997) rather than in precise, repeatable measures appropriate to assessing coral mortality. The majority of past studies allow little assessment of their precision or accuracy and thus their statistical power to detect change.

Previous studies of change in coral cover over time have been conducted within the Dampier Harbour as part of long-term monitoring of coral communities for Woodside Energy (see summary in Blakeway, this volume) or the assessment of dredging impacts (ECS 1998). All used fixed location adaptations of the Australian Institute of Marine Science (AIMS) video assessment technique developed for rapid assessment of reef status (Abdo et al. 2003). The methodology of the former surveys has been criticised for a very low statistical power to detect change (Harvey et al., 2000) and the latter would have had similar characteristics (unpublished data).

While there are no accepted standards in setting the level of power for an analysis of change (see Methods for explanation), the EPA considered that the low level (or unspecified level) of power in previous studies was unacceptable in the present case and required that the current study explicitly consider power. The design of the monitoring program was to provide a solution with a level of statistical power appropriate to the 10% mortality effect trigger (EPA 2003a&b).

Under management plans developed in response to the EPA assessments, the task for the monitoring program to be developed for the Dampier Port Authority (DPA) and Hamersley Iron Pty Limited (Hamersley) dredging programs was to

- Assess the extent of coral mortality at sites likely to be impacted by dredging or disposal
- Assess the extent of coral mortality at sites which would suffer similar background mortality but avoid dredging-related mortality

- Assess the above at a level of precision capable of detecting a 10% shift in coral cover with an acceptable level of statistical power;
- Complete the assessment of all monitoring sites throughout the Harbour from starting a field survey to finalising the statistical test within the fortnightly period.

Methods

Site Locations

Monitoring sites were selected to represent the major coral populations adjacent to dredging and disposal (Table 1, Figure 1). They provided a uniform distribution of monitoring at varying distances and directions from the source of the sediment plume at the dredging and disposal areas. This allowed for an assessment of the influence of tidal currents, wind direction and other meteorological variables on the migration of a sediment plume away from the dredging operations.

Reference sites were selected on the basis of similar bathymetry and weather aspect wherever possible and as sites outside the immediate impact of dredging. As the most similar biotic communities were usually located close to each other, there was a concern that the 'Near Reference' sites might be impacted by the same factors influencing the Impact sites. To provide confidence that a data set from unimpacted sites would be available, a second set of Reference sites was selected on the basis of being distant from any impacts. These 'Far Reference' sites were less similar to Impact sites than the 'Near Reference' set.

Table 1. Monitoring sites and their function for the two dredging programs.

Site	Function	Program
Angel Island (ANGI)	Reference (Near)	DPA/HI
Conzinc Bay North (COBN)	Impact	DPA/HI
Conzinc Island (CONI)	Impact	DPA/HI
Dampier Wharf North (DPAN)	Impact	DPA
East Lewis Island 1 (ELI1)	Impact	HI
East Lewis Island 2 (ELI2)	Impact	HI
East Lewis Island 3 (ELI3)	Impact	HI
Gidley Island (GIDI)	Reference (Near)	DPA/HI
High Point (HGPT)	Reference (Far)	DPA/HI
Holden Point (HOLD)	Impact	DPA
King Bay (KGBY)	Impact	HI
Malus Island (MALI)	Reference (Far)	DPA/HI
North Withnell (NWIT)	Reference (Near)	DPA/HI
South Withnell (SWIT)	Reference (Near)	DPA/HI
Supply Base (SUPB)	Impact	DPA
Tidepole Island (TDPL)	Impact	HI
West Intercourse Island (WINI)	Reference (Far)	DPA/HI
West Lewis Island 1 (WLI1)	Reference (Far)	DPA
West Lewis Island 2 (WLI2)	Reference (Far)	DPA

Sampling Methods

Constraints on the sampling included the need to:

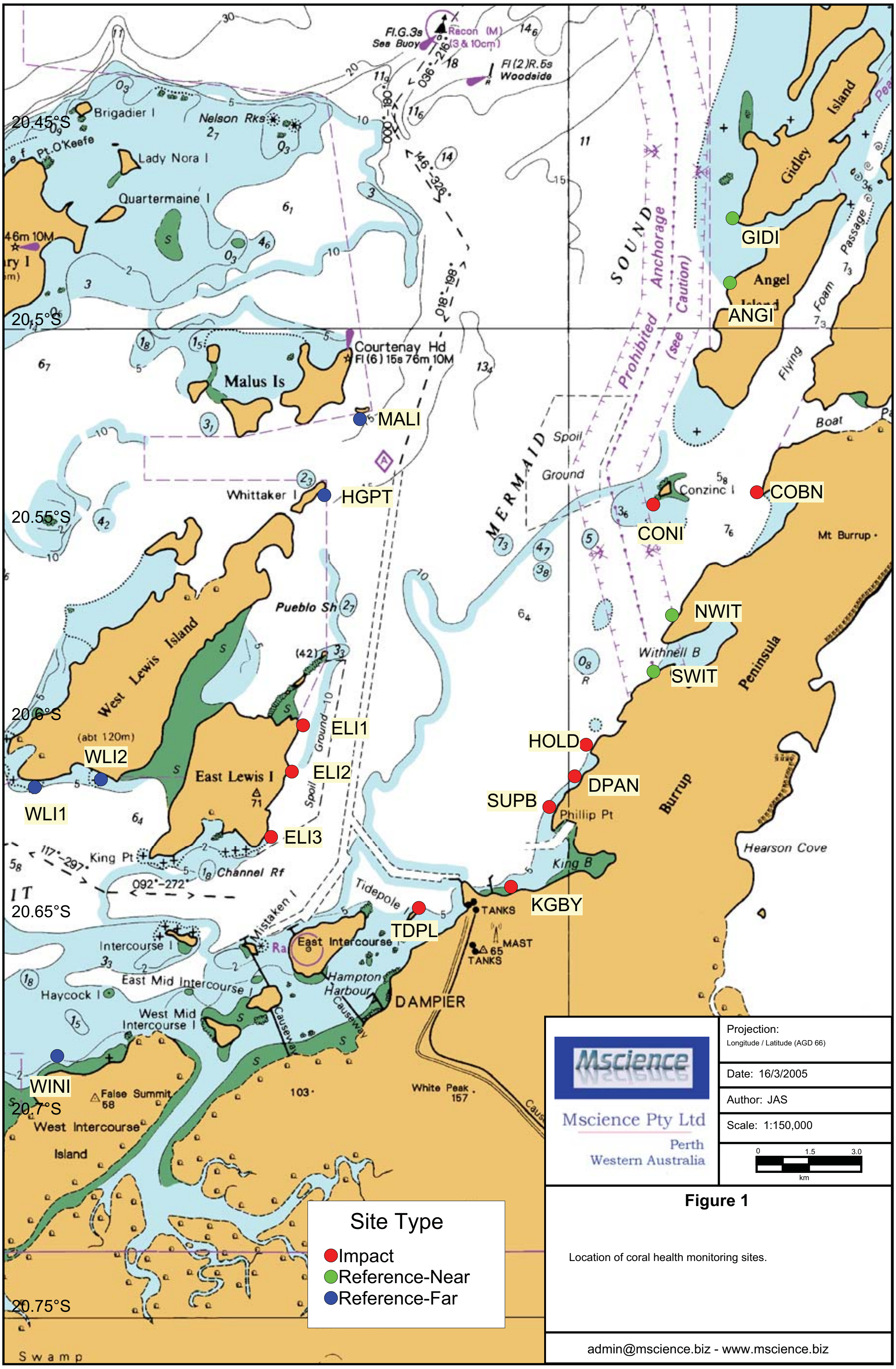
- cover an ecologically (or socially) meaningful amount of coral;
- cover many sites
- allow rapid turnaround
- be measurable in poor visibility
- remove as much subjectiveness as possible
- cover over 12 months at fortnightly intervals.

Based on the above needs, a design was developed as an adaptation of the Coral Reef Assessment & Monitoring Program of the University of Hawaii (Brown et al 2004). This design used 5 x 10m transects to represent coral at each site – 10m was chosen as a suitable length to represent local cover while allowing precise relocation of the transect. Cover for each transect was estimated as the mean of 10 frames. Variance between times for individual frames was not estimated as this would be swamped by error terms from variation in exact placement of the frame.

Initially, images of frames were captured from video recordings. A digital 3 CCD underwater video camera was swum slowly (3-4m/min) along the transect perpendicular to the bottom at a distance yielding a frame size of around 50 x 70 cm. Subsequently, the recording was transferred to a compressed digital file and the time taken to record the 10m transect noted. Frames were then captured from the file at the beginning of the transect and at even time spacing thereafter to provide 10 frames in total (covering about 50% of the area under the transect). Variability in replicating exact frame locations between sampling events (Trips) was problematic in this technique due to currents and swell making it difficult to swim at a constant speed on each occasion. To reduce that variability and improve the quality of images obtained for analysis, recording methods were altered to use a still camera on a leg at 1m distance intervals on the transect line. Trips 1-9 used video and trips 10-30 used the still camera.

Most coral communities around Mermaid Sound exist as relatively narrow bands fringing shorelines wherever suitable rocky habitat exists at depths between extreme spring low tide and around 6m below MLW (Blakeway & Radford, this volume). Thus the 5 transects were placed along an essentially linear arrangement parallel to the shoreline. The initial transect was placed in an area visually assessed as having amongst the highest coral cover at the site. Following transects were started approximately 5m distance at 45° from the end post of the prior transect.While not providing fully random spatial replicates, this technique covers a broad section of the longitudinal and depth range of coral communities.

Figure 1. Location of coral monitoring sites in Dampier Harbour.



Scoring of Coral Cover within Images

Digital images were colour balanced using commercial photo-enhancement software. In turbid conditions, this was essential to allow recognition of benthos. Images were then processed using Photogrid 1 (courtesy of C.Bird, University of Hawai'i) to overlay 50 points using the program's Stratified Random option. Points were then assigned by scorers to one of 10 categories: 6 coral groups, other fauna, flora, sand-rock-rubble, or unknown. The project used two scorers; each trained using the AIMS C-Nav system. Over the 14 months of scoring, it was necessary to cross-calibrate scorers regularly by standardising their scoring of several standard transects. Comparison of data from standardisation exercises suggested that inter-scorer effects could show consistent differences of 5-10% in coral cover in the absence of regular cross calibration.

Standard scoring methods use a category of Unknown to hold points obscured by transect equipment or unable to be identified from the image. Points in this category are excluded from analysis (eg in a transect with 100 Unknowns, averages will be calculated from 400 points). However, a consistent bias in assignment of benthic category to Unknown can bias results. In highly turbid conditions (visibility <1.5m) the frequency of Unknown points rises.

Working in very turbid waters can bias results. Within our digital images, the distribution of light returned to the camera lens was not uniform. Bulky complex light sources were unsuited to working in shallow conditions where swell was frequently present and simple light sources were negated by backscatter from suspended solids. Recording under incident light was most effective, but meant that surfaces closer to the camera were better lit (and thus more easily discriminated) than surfaces on the bottom substrate (sometimes in shadow). Coral morphology and size were key factors, such that large arborescent, plate or massive colonies were resolved much more readily than small encrusting corals. Estimates of cover by the former can be biased upwards from sampling trips where turbidity was high, while estimates for transects dominated by the latter may bias downward. Consistent bias can also occur if observers differentiate coral from substrate by recognising regular complex structures in coral. In low light conditions it is difficult to identify structure and corals may be assigned to unknown or incorrectly identified as 'abiotic'.

Not only are images captured from video of much lower acuity than routinely used still image cameras (around 640x480 pixels vs 2600x2000 pixels), but they frequently suffer from motion-blur. The frequency of Unknown points was significantly reduced when image capture techniques switched from video frames to digital still frames (mean Unknown 27.1 vs 14.8, p<.001).

The intensity of sampling within each image affects the precision and accuracy of the estimate of cover for each frame. To date, the most common technique has been to follow the AIMS method (Carleton & Done 1995) using 5 points per frame. Harvey et al (2000) point to the inability of this level of image sampling to support levels of precision and power appropriate to testing small changes in coral communities over time and suggest 40-60 dots per frame are required on the basis of Monte Carlo simulations. Brown et al (2004) provide some hypothetical and some empirical data to support a similar level of intensity in frame sampling.

Here, we used 50 points per frame as recommended by Brown et al (2004), but conducted trials using Monte Carlo and empirical scores to investigate the effects of frame sampling intensity for two monitoring sites – one with a high level of coral cover evenly spread between transects (ELI2) and one with a low level of cover and higher between-transect variance (HOLD). For both sites the Monte Carlo sampling involved selecting random points from transects scored for 50 points to produce estimates of means and variance based on 5, 10, 25 and 50 points. The empirical tests used two scorers scoring each transect three times for each of intensity levels.

Statistical assessment

The sampling design was constructed with the intention of supporting a BACI analysis using Repeated Measures Analysis of Variance (RMANOVA). Brown et al (2004) provide estimates of power analysis for RMANOVA in detecting change over time at a site using the Transect as the base subject and frames within a transect as a nested measure. They report the results of a study using statistical bootstrap and actual studies to investigate the effects of varying numbers of points per frame, frames per transect and transects per site on the power of this technique to detect change over time at their Hawaiian sites. Based on their analyses, the present design (50 points per transect; 5 transects per site) was estimated a priori to provide a statistical power in excess of 0.8 in detecting a 20% change or 0.7 for a 15% change in coral cover at any site.

The primary task for monitoring was to test whether any Impact site had suffered more than 10% mortality of its baseline coral abundance after adjustment for any mortality at Reference sites (derived from EPA 2003 a & b). In practice, this was translated into estimates of decline in coral cover with only Impact sites monitored routinely. Where Impact Sites showed a significant decline Reference sites could then be evaluated to provide any offset in calculating net change.

The working hypothesis for monitoring was

H_0 – live coral cover has not declined more than 10% of the baseline value, with

H_1 – live coral cover has declined more than 10% of the baseline value

The spatial abundance of corals is typically highly patchy at scales of metres to tens of metres. Variation in coral cover between frames within a transect and between transects within a site was high – usually well above the 10% relative cover trigger. To allow the test statistic to concentrate on change in cover, it was decided to test the above hypothesis with a paired t-test for each Impact site using individual transects paired between the baseline and the current trip result.

Power of a test is defined as the ability to reject the null hypothesis when it should be rejected. It is the inverse of β, the probability of a Type II error. It is dependent on the effect size (in this case a decline of 10% of baseline) and its relationship to error variance terms which can obscure real shifts in the mean. While the significance level for rejecting null hypotheses, α, is conventionally set at 0.05, there is no general level accepted for β. Rather, power should be set on the basis of the consequences of the test missing a positive result.

Underwood and Chapman (2003) suggest that the power of detection of effects of Pulse impacts (here mortality from short-term elevation in suspended sediments) can be increased most effectively through the use of multiple sampling periods, rather than large numbers of control sites. While the present program used many sampling periods, that was principally for the early detection of impacts and would not have improved the power of tests as the hypothesis tested here related to change between the baseline period and any single survey trip. Multiple estimates of impact, such as in repeated measures ANOVAs, are thus not applicable.

Power analysis for the monitoring program used here was calculated using NCSS-PASS software (Hintze 2001). Power was estimated using the assumption that for sites where there was little obvious change over the monitoring period, between- trip differentials should reflect error terms alone. Power was estimated for a number of such sites with a range of coral cover, using at least 10 between-trip comparisons to provide an average variance estimate to compare against an effect size of 10% change against the baseline for each site.

Sampling events

Sampling was conducted prior to dredging commencing to provide an estimate of the baseline level of cover of Impact and Reference Sites. Where possible, baseline values were calculated from the average of 2 or 3 monitoring trips conducted over Nov/Dec 2003. As details of the dredging program evolved during that period and later, some site baselines are calculated from a single trip. As the Hamersley dredging program did not start until mid 2004, after the impacts of Cyclone Monty in 1-3 March, it was decided to re-estimate the baseline from those Impact and Reference sites during April.

Following the commencement of dredging, monitoring trips were conducted fortnightly. The DPA dredging program ran from 8 January 2004 until 20 May 2004 and the Hamersley program from 8 May to 23 October 2004. Monitoring of each program's Impact Sites continued for 2 months beyond the cessation of dredging.

Results

Change in coral communities over 12 months of dredging

Coral cover in communities surveyed in the Dampier Harbour ranged from 18% to almost 80% (Table 2). In most cases, the standard error of the mean was around 10%, supporting the decision to use a test of paired transects rather than a RMANOVA which would retain this inter-transect variance. In hindsight, several of the Reference sites would not have served as adequate controls for impacts as the composition of coral community was different from their targeted Impact Sites.

When the change against baseline cover levels was assessed by site for the entire monitoring period there was no clear indication of dredging and disposal impacts yielding an outcome where Impact sites declined and Reference sites did not (Fig. 2). Indeed, only 1 Impact Site dropped by over 10% of coral cover (SUPB) while several Reference sites showed a strong decline.

Table 2. Baseline means of coral cover and standard error of the mean (ie between transect variation).

Site	DPA Base	S.E.	HI Base	S.E.	Dominant Corals
ANGI	39.1	3.65	38.6	4.21	Acropora
COBN	47.0	3.50	39.5	3.33	Other/Faviid
CONI	33.8	9.99	30.6	10.80	Porites
DPAN	37.2	3.95	-	-	Faviid/Turbinaria
ELI1	77.3	3.39	69.3	5.26	Pavona
ELI2	74.5	3.84	70.2	4.49	Pavona
ELI3	-	-	30.1	3.97	Pavona
GIDI	41.3	3.97	32.1	3.20	Acropora/Faviid
HGPT	55.2	5.39	49.5	3.31	Pavona
HOLD	18.2	4.59	-	-	Faviid/Other
KGBY	46.1	3.69	40.9	4.42	Faviid/Other
MALI	41.1	7.51	39.0	6.49	Porites
NWIT	32.4	4.45	31.2	5.11	Faviid/Turbinaria
SUPB	49.7	4.95	-	-	Turbinaria
SWIT	34.1	1.10	37.2	2.26	Faviid/Other/ Turbinaria
TDPL	49.7	3.44	37.8	5.49	Pavona/Faviid
WINI	23.1	2.46	10.2	3.18	Turbinaria/ Porites/Faviid
WLI1	39.0	3.36	-	-	Porites
WLI2	46.9	10.5	-	-	Porites/Other

Closer examination of how sites changed over time (Fig.3 a & b) together with diver observations provides a means of interpreting these changes. Clearly, there were many different causes of mortality which impacted one or a small number of sites:

- Dredging related mortality: the only site where a clear impact of dredging was seen was SUPB where coral cover declined by around 80% - this was immediately adjacent to the DPA dredge operation and corals were smothered by sediments which remained in place for the remainder of monitoring;
- Seasonal competition with plants: At WINI and TDPL coral cover declined substantially between November 2003 and April 2004: scores for benthic cover of macroalgae and diver observations demonstrated that this was due to strong growth of *Sargassum* spp. overtopping the corals. Although some mortality occurred, once the *Sargassum* had disappeared (around June/July) coral cover figures recovered to close to original levels;

- Seasonal wave conditions: prior to the onset of Cyclone Monty (1-2 March 2004) and as a result of the cyclone, many sites experienced strong wave and swell conditions. At GIDI and ANGI coarse sediments and plant material liberated by the wave action was seen settling on corals – especially tabulate *Acropora* (which was only found in any abundance at these sites). Subsequent mortality of these individuals caused these Reference sites to fall by almost 10%.
- Cyclonic rainfall: during the passage of Cyclone Monty the Dampier Harbour experienced very heavy rainfall and runoff. Salinity of surface waters dived sharply (Stoddart & Anstee, this volume) and remained low for some time. Surveys immediately after the cyclone showed that corals exposed to low salinity surface waters and terrigenous sediments suffered very high mortality. Live coral at WLI2 was almost totally lost and the shallow transects at WLI1 were also affected. Some recovery of corals occurred with bleached colonies regaining zooxanthellae over a few months. Initially this was included in mortality estimates as it is not possible to identify whether bleached corals are alive or dead from recorded images.

$$p = \frac{s/\sqrt{n}}{\bar{x}}$$

Figure 2. Change in site means for coral cover from baseline values over the year (mean and 95% confidence intervals). The dotted line represents a 10% decline against baseline.

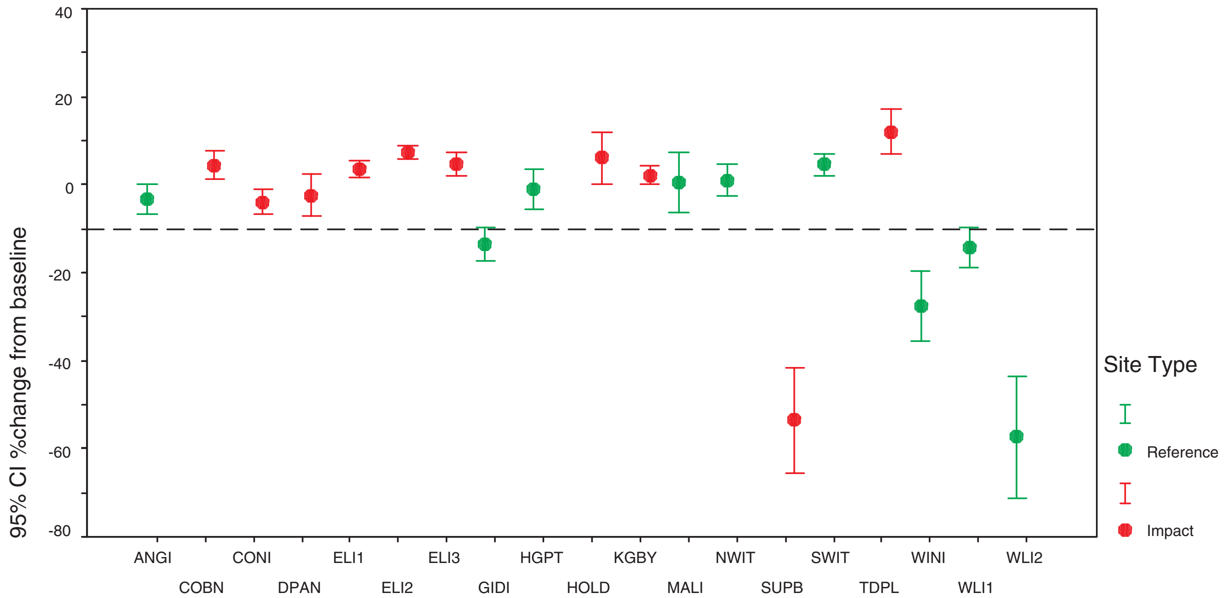
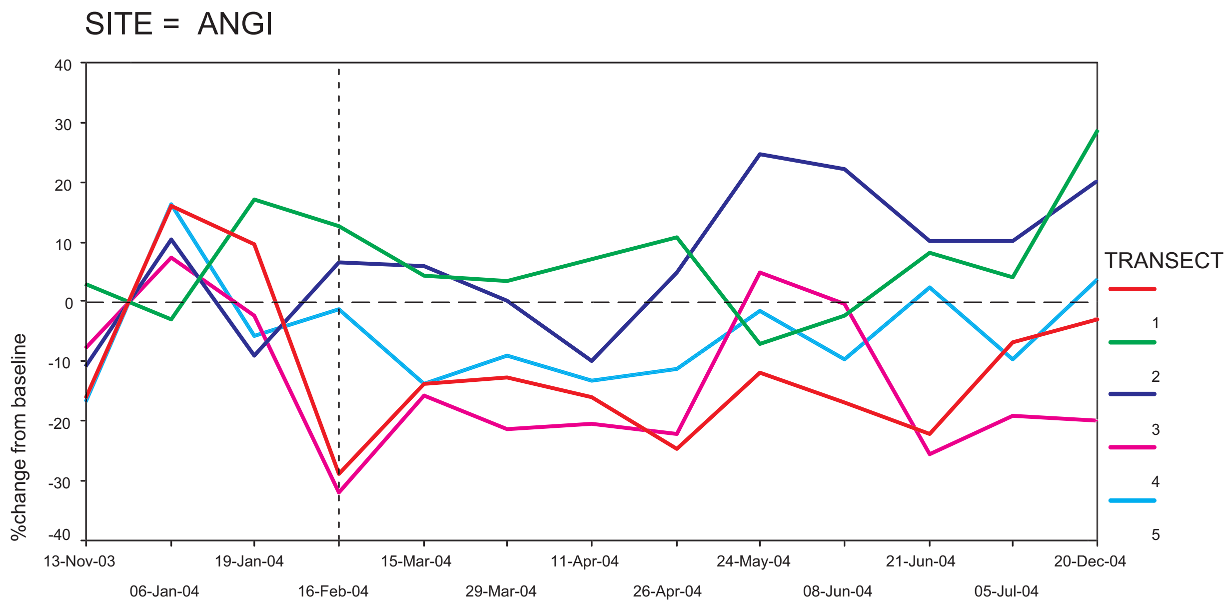
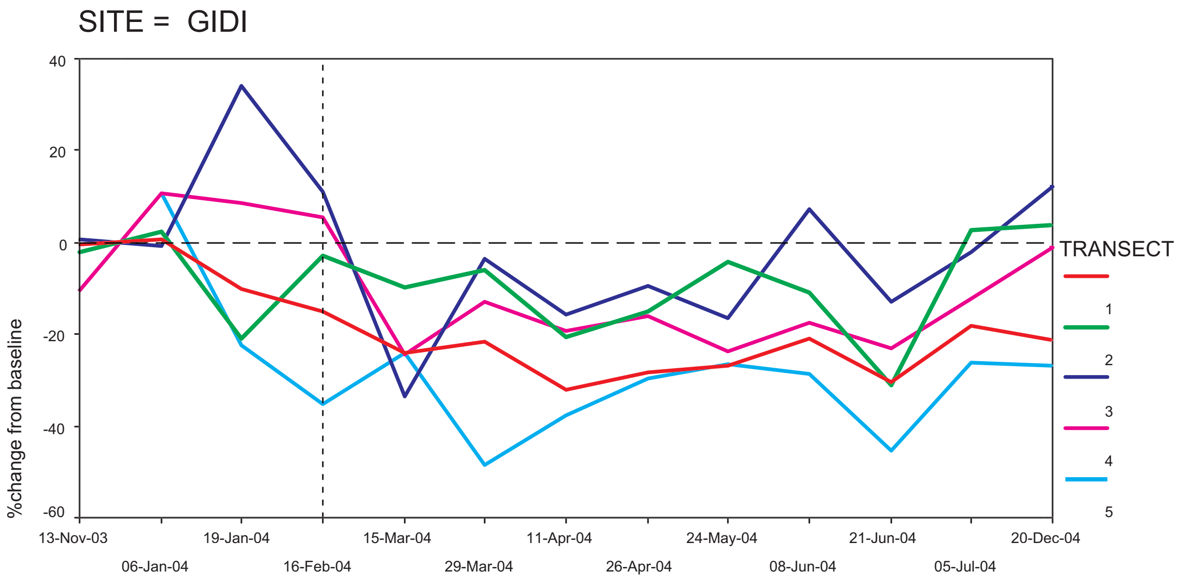


Figure 3. Differential from baseline during coral monitoring. Dotted line represents the last period before the impact of Cyclone Monty.
a) Reference sites

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20
0
-20
-40
-60
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06-Jan-04
19-Jan-04
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15-Mar-04
29-Mar-04
11-Apr-04
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4
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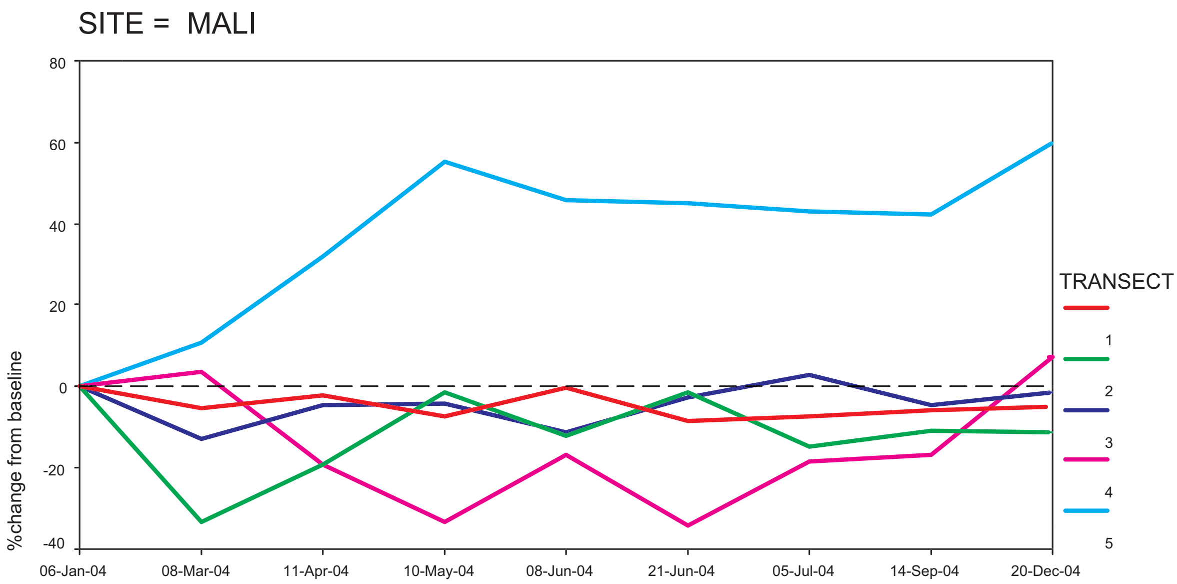
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%change from baseline
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0
-20
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06-Jan-04
08-Mar-04
11-Apr-04
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08-Jun-04
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05-Jul-04
14-Sep-04
20-Dec-04
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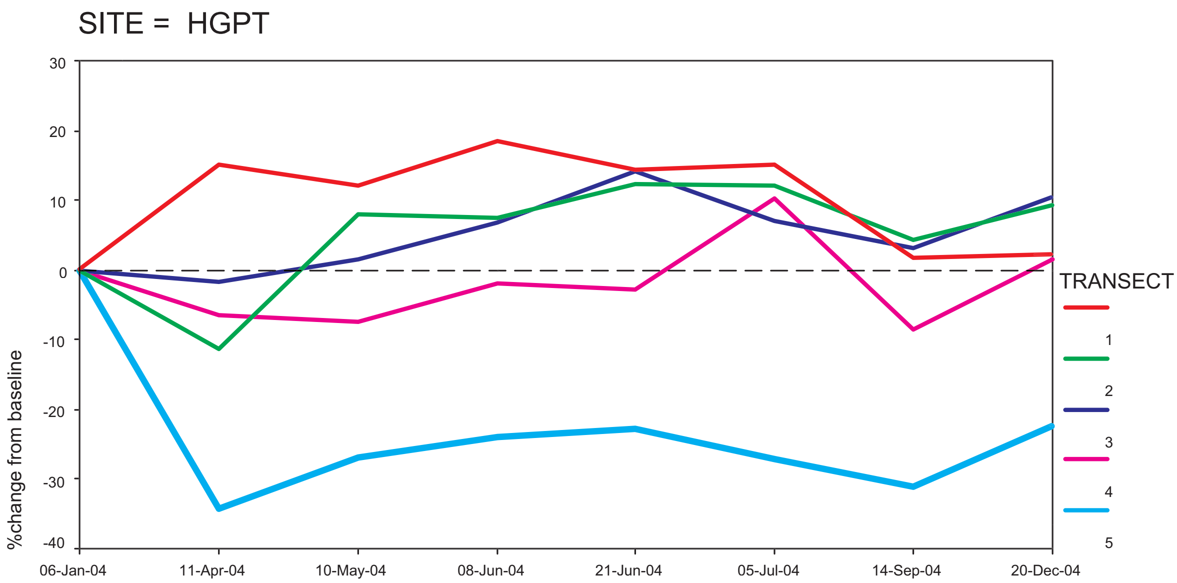
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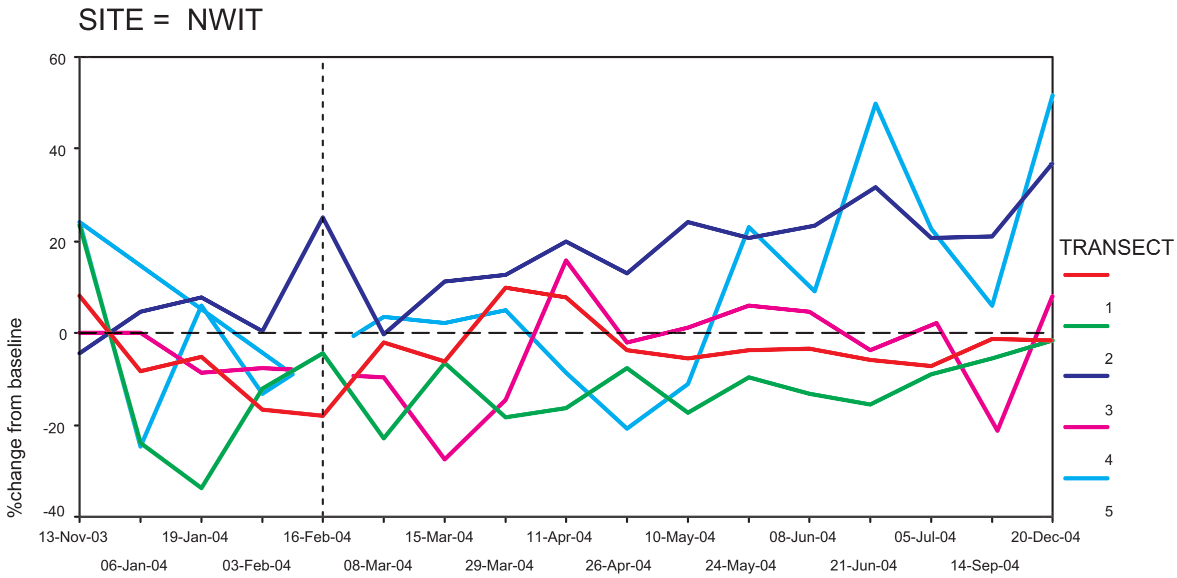
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29-Mar-04
11-Apr-04
26-Apr-04
10-May-04
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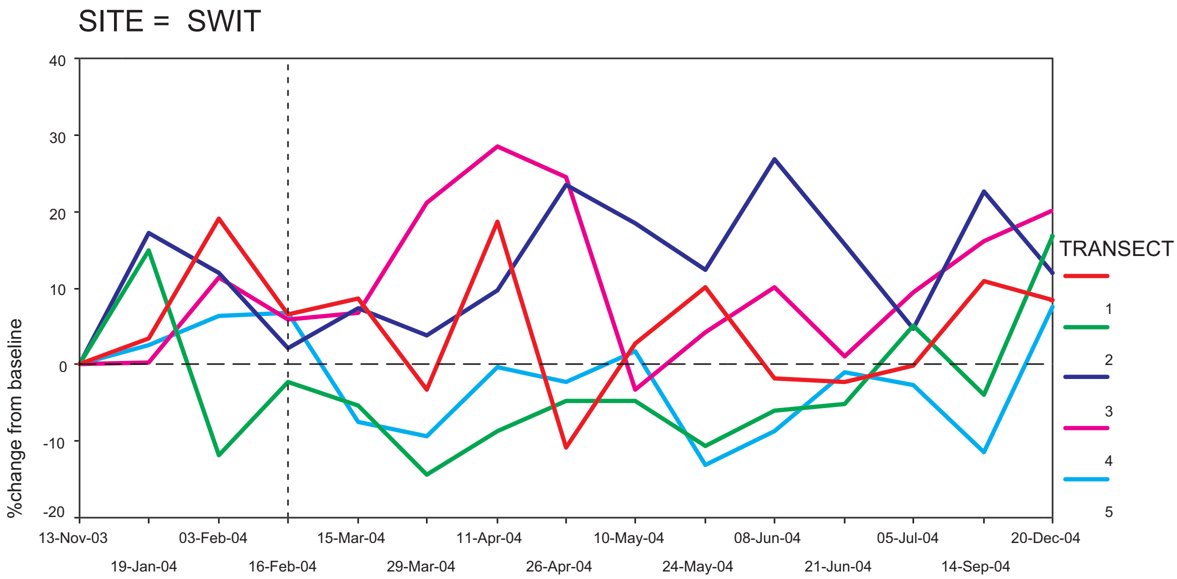
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08-Jun-04
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05-Jul-04
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TRANSECT
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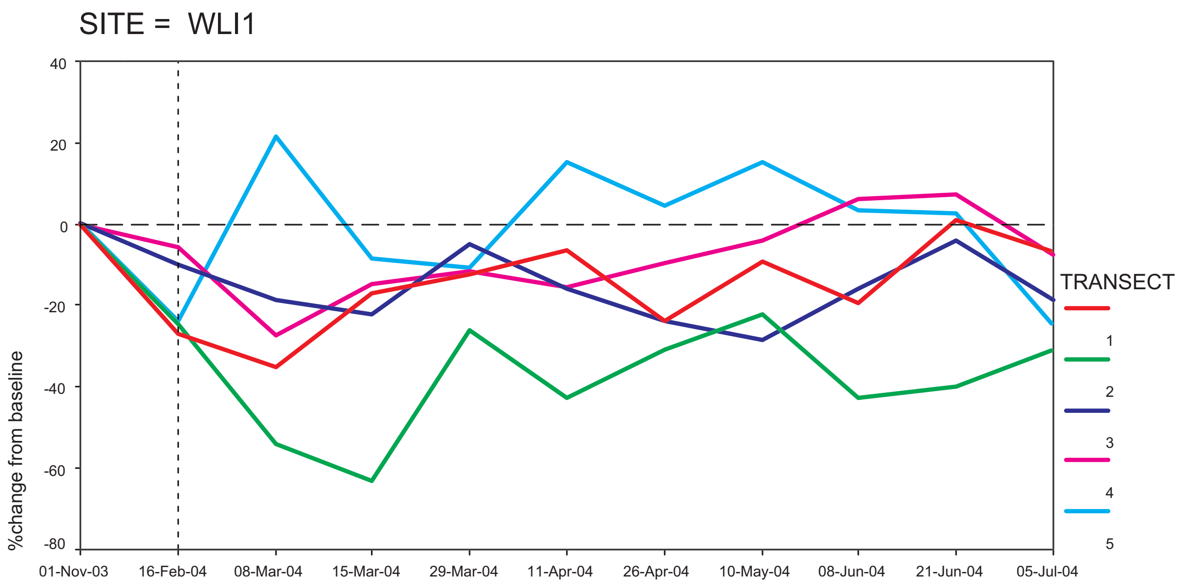
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-80
01-Nov-03
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15-Mar-04
29-Mar-04
11-Apr-04
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10-May-04
08-Jun-04
21-Jun-04
05-Jul-04
TRANSECT
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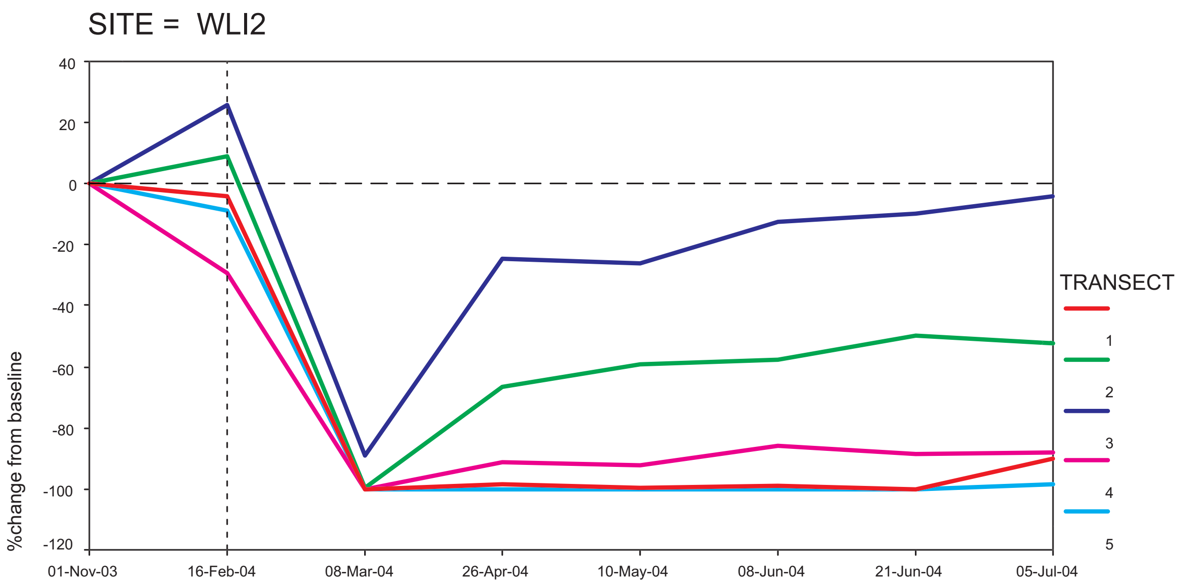
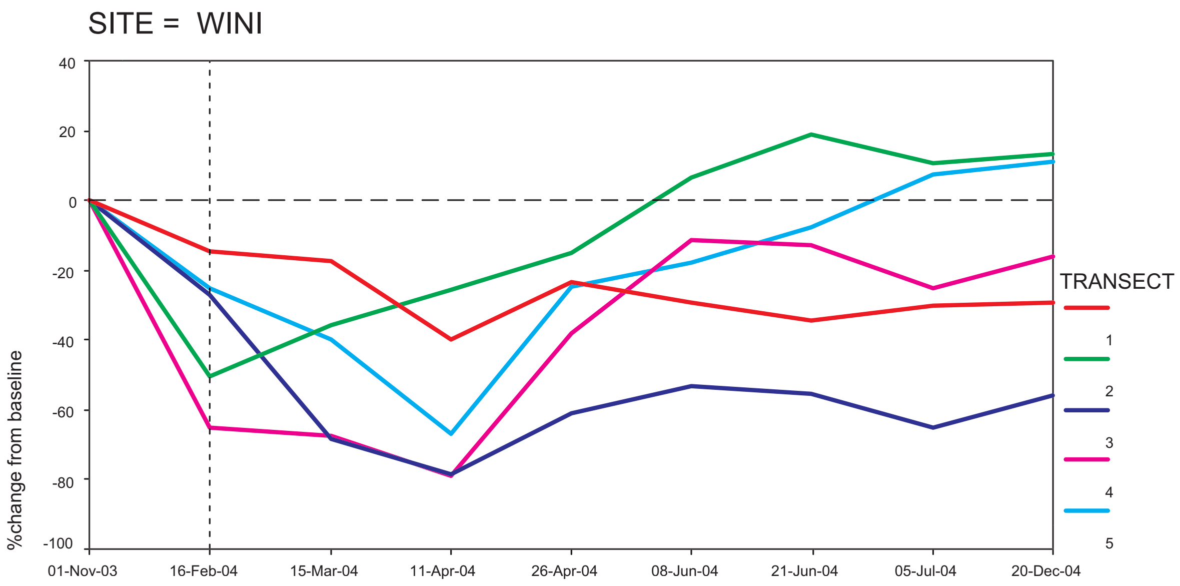
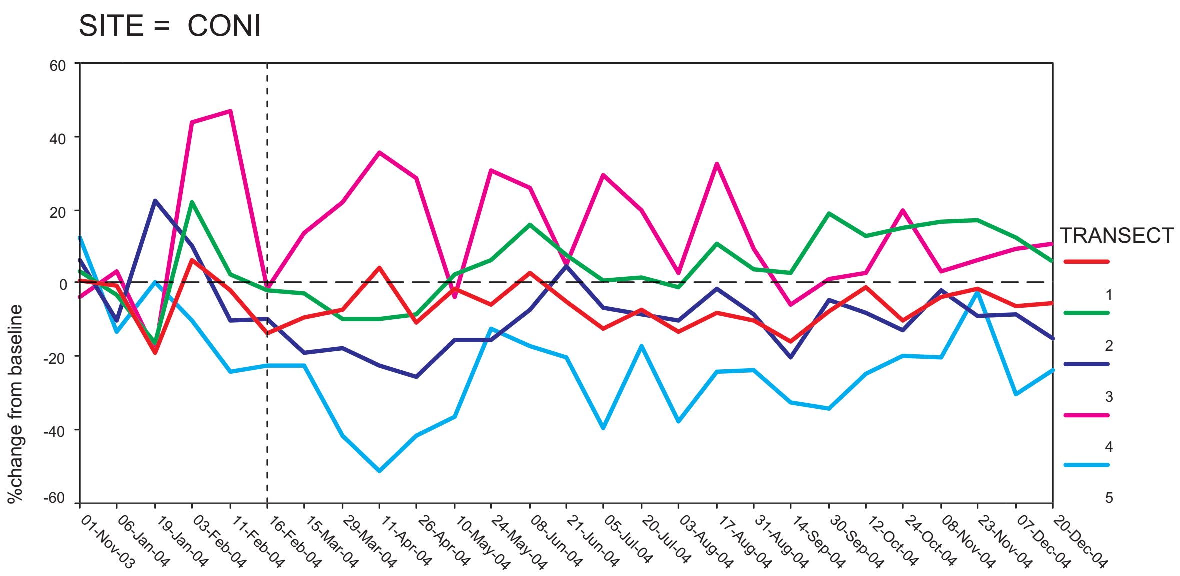
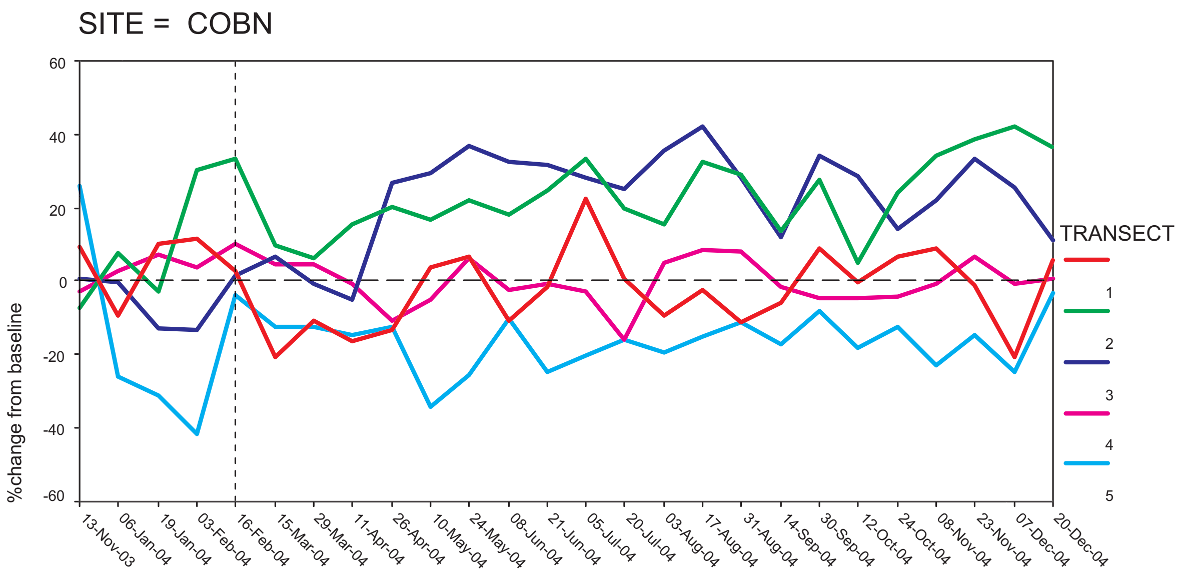




b) Impact sites

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06-Jan-04
19-Jan-04
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10-May-04
24-May-04
08-Jun-04
21-Jun-04
05-Jul-04
20-Jul-04
03-Aug-04
17-Aug-04
31-Aug-04
14-Sep-04
30-Sep-04
12-Oct-04
24-Oct-04
08-Nov-04
23-Nov-04
07-Dec-04
20-Dec-04
TRANSECT
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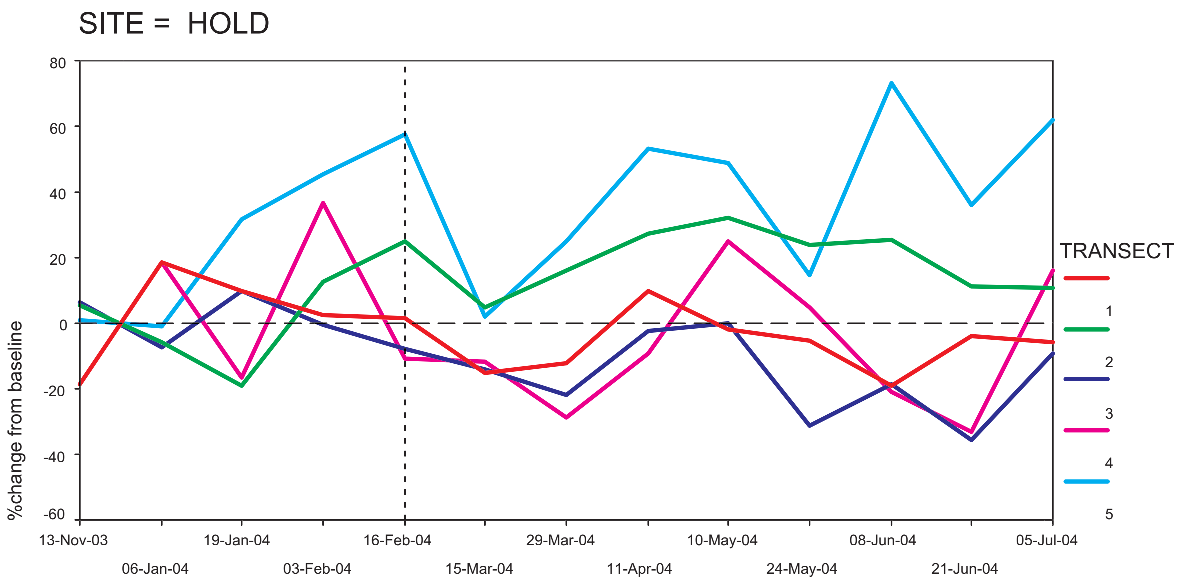
SITE = HOLD
80
60
40
20
0
-20
-40
-60
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13-Nov-03
06-Jan-04
19-Jan-04
03-Feb-04
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15-Mar-04
29-Mar-04
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TRANSECT
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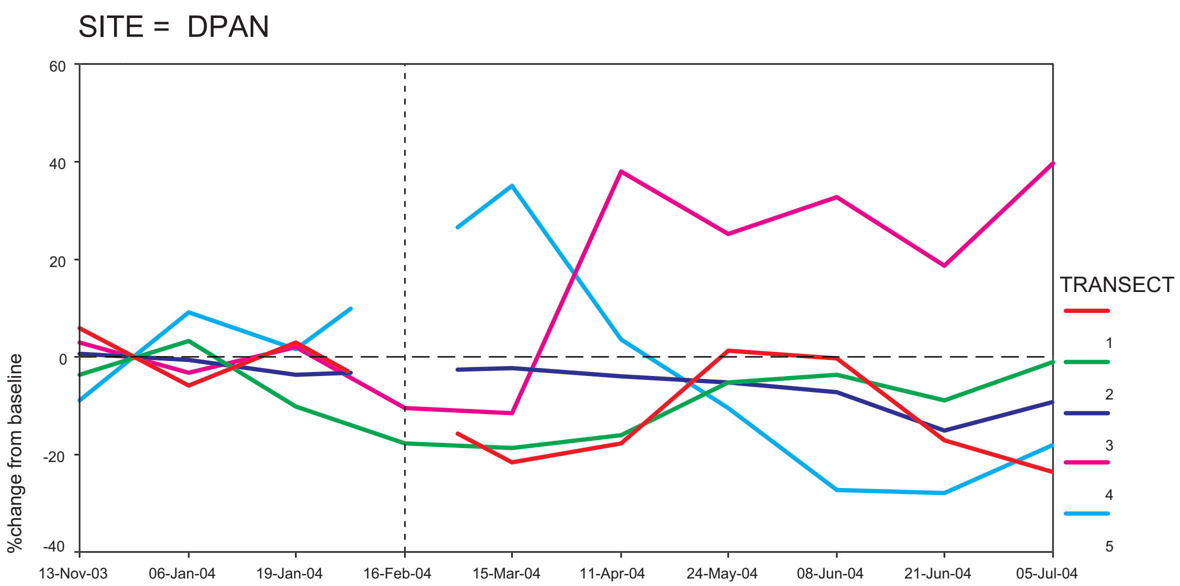
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-40
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06-Jan-04
19-Jan-04
16-Feb-04
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24-May-04
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05-Jul-04
TRANSECT
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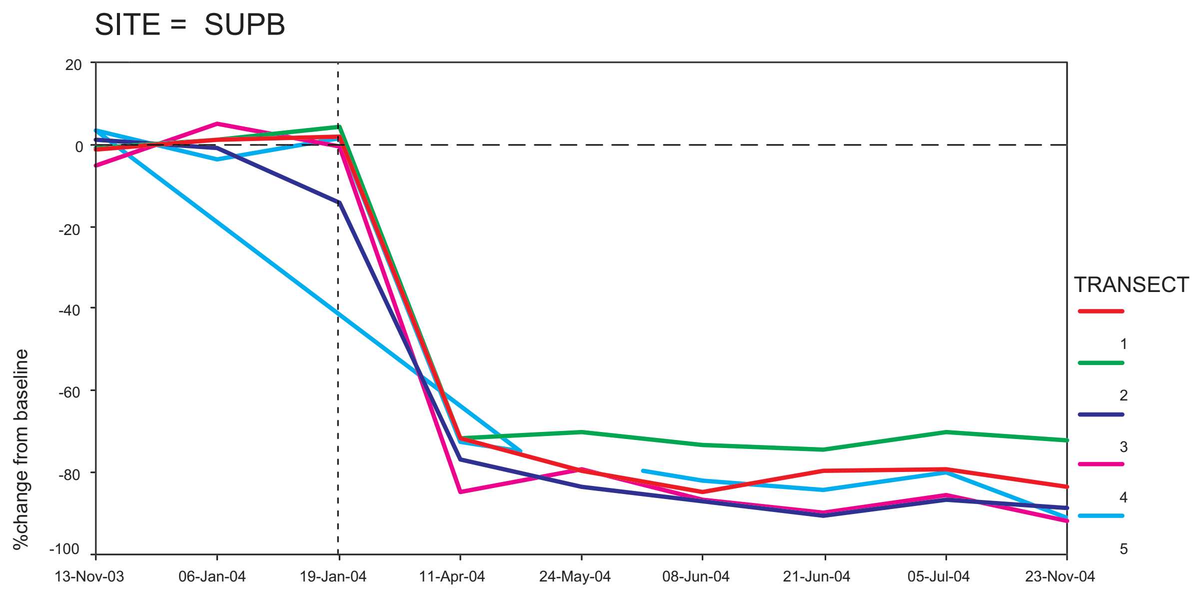
SITE = SUPB
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-40
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13-Nov-03
06-Jan-04
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TRANSECT
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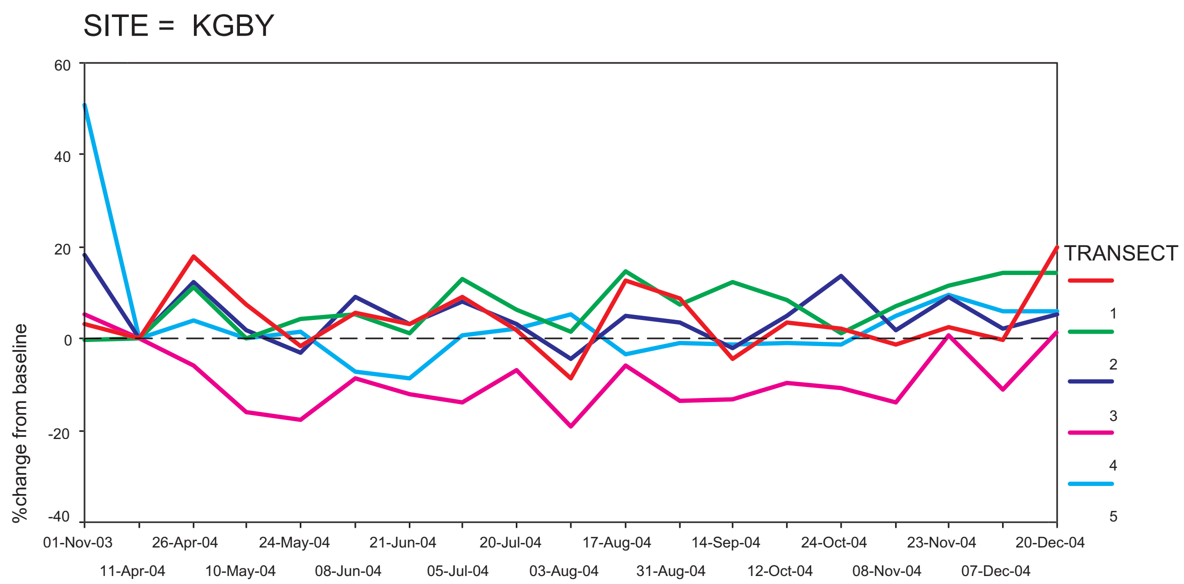
SITE = KGBY
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26-Apr-04
10-May-04
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21-Jun-04
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03-Aug-04
17-Aug-04
31-Aug-04
14-Sep-04
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20-Dec-04
TRANSECT
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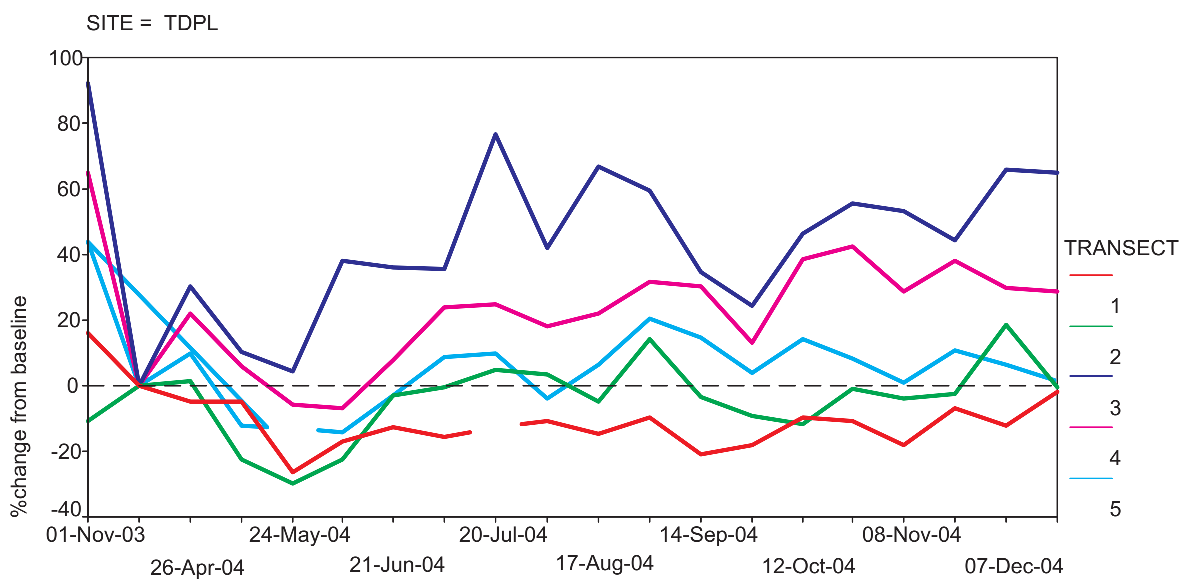
SITE = TDPL
%change from baseline
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40
20
0
-20
-40
01-Nov-03
26-Apr-04
24-May-04
21-Jun-04
20-Jul-04
17-Aug-04
14-Sep-04
12-Oct-04
08-Nov-04
07-Dec-04
TRANSECT
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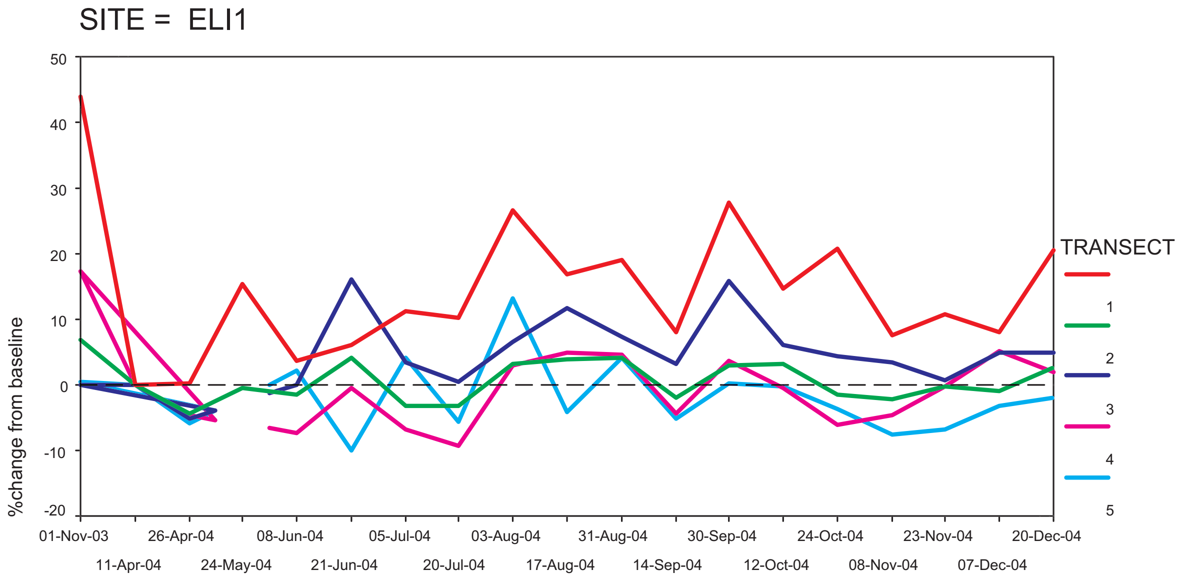
SITE = ELI1
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50
40
30
20
10
0
-10
-20
01-Nov-03
26-Apr-04
08-Jun-04
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03-Aug-04
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30-Sep-04
24-Oct-04
23-Nov-04
20-Dec-04
11-Apr-04
24-May-04
21-Jun-04
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07-Dec-04
TRANSECT
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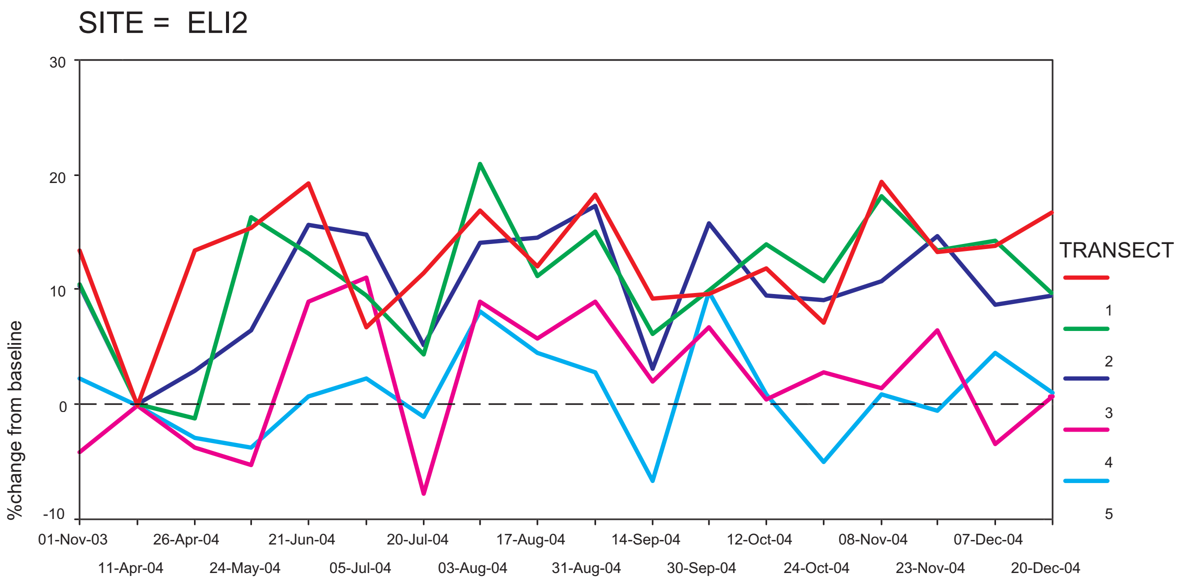
SITE = ELI2
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10
0
-10
01-Nov-03
26-Apr-04
21-Jun-04
20-Jul-04
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24-May-04
05-Jul-04
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20-Dec-04
TRANSECT
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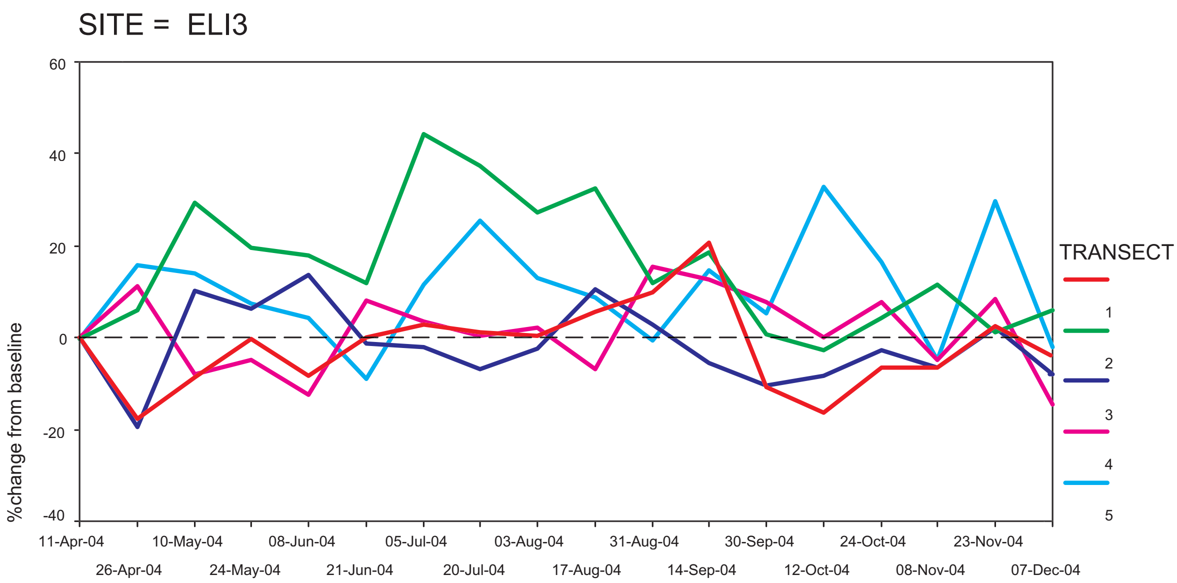
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0
-20
-40
11-Apr-04
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08-Jun-04
05-Jul-04
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24-Oct-04
23-Nov-04
26-Apr-04
24-May-04
21-Jun-04
20-Jul-04
17-Aug-04
14-Sep-04
12-Oct-04
08-Nov-04
07-Dec-04
TRANSECT
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Sampling intensity & Power analyses

For comparability with earlier studies such as Harvey et al (2000), assessment of the effects of increasing the intensity of points sampled used precision (*p*) estimated as

where s = standard deviation and n = the number of estimates (here 6) – i.e. lower *p* values are more precise (higher power). The precision of estimates for the HOLD site with low (18%) and variable cover was substantially improved by raising the number of points sampled, but for the ELI2 site with higher (70%) and more even cover, increasing point sampling intensity was of marginal return (Fig. 4). Examining the confidence intervals for individual transects shows a similar outcome (Fig.5).

Examination of estimates of the power of this technique to detect a 10% decline in coral cover shows that power was heavily dependent on absolute level of coral cover at a site (Fig.6). This pattern resulted from a similar level of error variance for sites independent of the actual mean cover.

Figure 4. Effects of sampling intensity within images on precision.

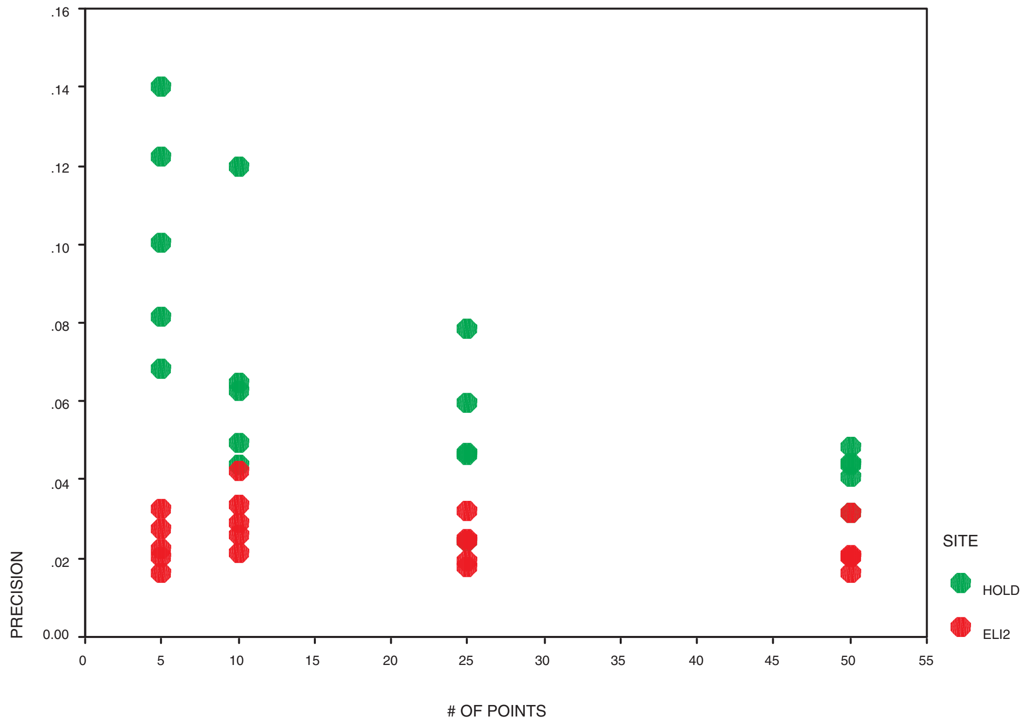


Figure 5. Confidence intervals and means for two sites with varying sampling intensity.

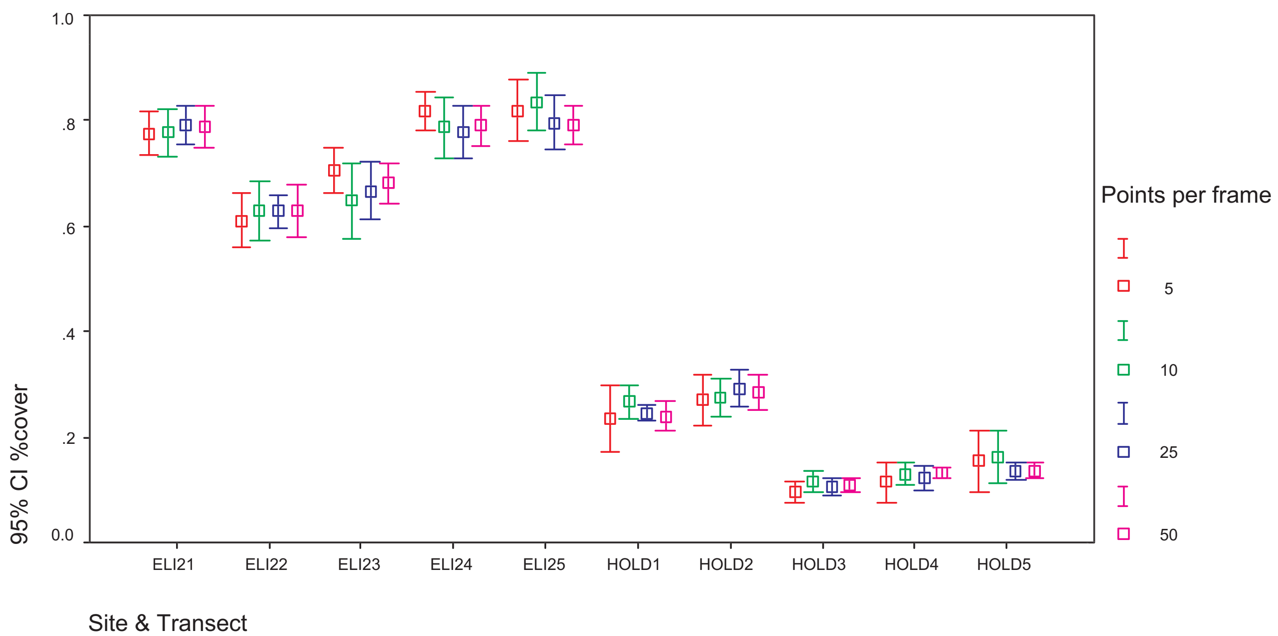
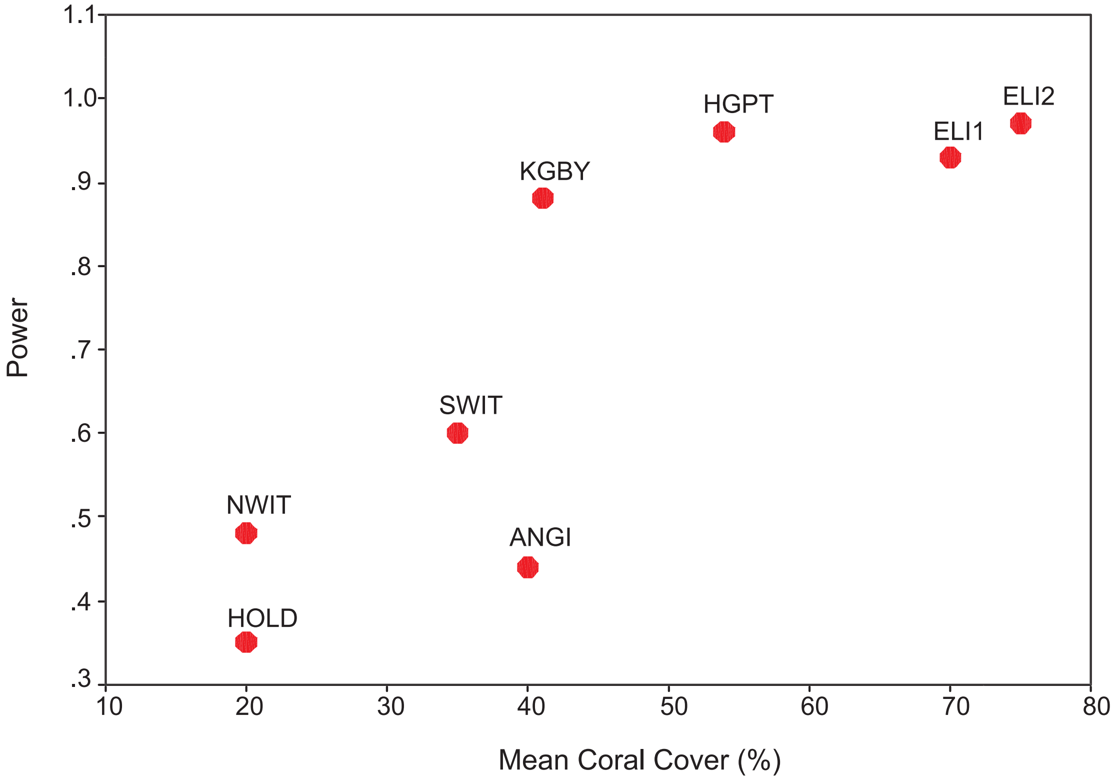


Figure 6. Power analysis for paired-tests of 10% decline.



Discussion

Changes in Dampier Harbour coral communities

The dynamic nature of coral communities has been well established and their continual cycles of disturbance and recovery documented in detail (Connell 1997). Coral communities are resilient to short-term impacts such as cyclones or predator outbreaks, from which they recover relatively rapidly (eg Connell et al 1997). However, chronic impacts such as overfishing or habitat destruction can force ecosystems into permanently altered states where coral communities disappear entirely (Pandolfi et al 2003).

Dredging impacts such as the resuspension of sediments during uplift or disposal are relatively brief in timescales relevant to coral ecology. Unless these impacts result in loss of habitat or in indirect long-term change to the environment such as changing the hydrodynamic or sedimentary environment, coral communities should recover to pre-impact states. Past monitoring of dredging effects in Dampier Harbour has suggested little or no change has occurred in coral cover as a result (Leprovost et al 1990; ECS 1998).

Over the 14 months of monitoring conducted for this program, the changes seen in coral abundance were principally the result of natural phenomena. The occurrence of Cyclone Monty at the beginning of March had a profound impact on the coral communities. Heavy swells and wind waves from the Category 4 cyclone (winds up to 280 km/hr) affected many of the sites, but the most damaging impact came from the rainfall (>300mm in 24hrs) and associated runoff. Surface salinity in many areas dropped dramatically (see Stoddart & Anstee, this volume) and a considerable amount of terrigenous sediment was deposited on corals.

Many of our monitoring sites showed a decline associated with the cyclone, but effects were most pronounced at WLI1 and WLI2 where the Maitland River freshwater plume overtopped the communities and mortality in the shallows along the entire southern shore of West Lewis Island was over 75%. Extensive mortality was also observed by divers in shallow coral communities around other islands where freshwater collecting in onshore catchments would have spilled into the sea. While initial visual estimates of coral mortality at many of these sites by divers were close to 100%, many individual corals were only bleached and regained their zooxanthellae (and thus colour) over the following months.

Following the extensive late summer rains associated with the cyclone, almost permanent freshwater seeps were observed around the WINI site and microalgal growth often obscured visibility or covered corals, leading to a decline in cover. Of greater impact at this site was the seasonal growth of the macroalgae *Sargassum* spp.. At WINI and at the TDPL site, summer growth of this species overtopped about 40% of coral – leading to an apparent decline in cover (with intercept points in images assigned to algae rather than coral). Although live coral cover rose rapidly following disappearance of the *Sargassum* over winter, it did not return to the level of spring 2003, suggesting that some coral had died as a result.

Elsewhere, wave action during strong weather has been documented as a natural cause of coral mortality. Despite the unusually high waves and swell recorded over many of our sites during Cyclone Monty, there was little evidence of physical damage. At Gidley Island (GIDI) where the coral community lies on a reef facing the ocean swells, strong swell conditions in late January/February were observed to deposit floating macroalgae and coarse sediments on many corals. In particular, this led to partial mortality of individuals of several species of *Acropora* causing coral cover to decline around 15-20%.

Dredging related mortality

The only site where mortality was related to dredging was at SUPB. Corals affected at that site were within a few hundred metres of intense dredging where propeller wash from constant positioning of a large trailer hopper suction dredge deposited substantial amounts of sediment directly on corals (Stoddart & Anstee, this volume). Blakeway (this volume) provides a discussion of the relationship between mortality and coral morphology at that site.

At sites between 500m and 1km from dredging and spoil disposal sites, coral communities were subject to turbidity levels elevated well above background for weeks at a time, or in very intense events of a few days (Stoddart & Anstee, this volume). No effects of this turbidity were apparent in cover estimates at these sites (HOLD, DPAN, ELI1, ELI2, ELI3, CONI) and at the East Lewis sites, several transects showed an increasing trend in cover.

Statistics of coral monitoring

The use of repeated estimates of coral cover to monitor coral mortality resulting from anthropogenic impacts or natural events is common. However, many reported studies provide little detail on the relationship between the statistical power of their methodology and the question they addressed. With considerable interest in the worldwide decline of coral reefs (Pandolfi et al 2003) much of the methodology used within research institutions or non government organisations (e.g. www.reefcheck.org) targets broad indicators which can be assessed rapidly over large areas of reef. The need for more precise repeatable estimates to detect small changes in coral communities has been recognised as requiring a different approach to monitoring (Brown et al 2004). Typically, the early detection of a decline in coral abundance (i.e. the start of mortality) associated with anthropogenic impacts is more likely to be successful using the latter methods rather than the former.

The requirement for this study was to provide a test design to detect a 10% effect size with a level of power appropriate to safeguard corals against an 'unacceptable level of mortality'. The latter was defined to be a 30% decline on which dredging near the effected site was to cease (EPA 2003a&b). In that regard, the post hoc estimates of power as varying between 0.35 and 0.95 suggest that the current design effectively met EPA requirements.

The positive correlation of power and mean coral cover appears to stem from 2 factors. Firstly, a 10% change in communities with sparse coral will be a smaller effect size than for more dense cover. Secondly, the variance of estimates between sampling events was relatively constant for many sites, resulting in the mean:variance ratio (an indicator of power) rising with mean coral cover. Where there is a requirement for monitoring programs to yield power above 50% for sparse coral communities, the only option may be to increase the number of transects surveyed.

Underwood and Chapman (2003) point out that in addition to the underlying distribution of the parameter being measured, power will also be a function of sampling error. A large variety of factors may influence apparent coral cover as a result of method error between setting foot in the water on each trip and results entering the database. One which played a large part here was the differences in depth and exposure to swell between the 19 sites. As well as having high coral cover, sites which returned high estimates of power were generally protected from strong wave action allowing divers to place and record transects accurately on most trips.

Our assessment of the importance of the sampling intensity of images for improving the precision of estimates of benthic cover agrees with the results of Harvey et al (2000) where coral cover is sparse (<30%). Where coral cover is more abundant (>60%) sampling intensity is less important and even the 5-point per frame scoring of the rapid survey techniques (Abdo et al 2003) can yield data with adequate precision.

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