Australian Government
Great Barrier Reef
Marine Park Authority

Queensland
Government

# Supplementary Report to the Final Report of the Coral Reef Expert Group:

## S5. Statistical power of existing AIMS Long-Term Reef Monitoring Programs

Thompson, A., Menendez, P.

Australian Institute of Marine Science

The Great Barrier Reef Marine Park Authority acknowledges the continuing sea country management and custodianship of the Great Barrier Reef by Aboriginal and Torres Strait Islander Traditional Owners whose rich cultures, heritage values, enduring connections and shared efforts protect the Reef for future generations.

Published by the Great Barrier Reef Marine Park Authority

ISBN 9780648589297

**A catalogue record for this publication is available from the National Library of Australia**

**This publication should be cited as:**

Thompson, A., and Menendez, P. 2020, *Supplementary report to the final report of the coral reef expert group: S5. Statistical power of existing AIMS Long-Term Reef Monitoring Programs*, Great Barrier Reef Marine Park Authority, Townsville.

Front cover image: Cape Ferguson AIMS research vessel in Cape Tribulation © Commonwealth of Australia (GBRMPA), photographer: Katherine Martin.

**DISCLAIMER**

While reasonable effort has been made to ensure that the contents of this publication are factually correct, the Commonwealth of Australia, represented by the Great Barrier Reef Marine Park Authority, does not accept responsibility for the accuracy or completeness of the contents, and shall not be liable for any loss or damage that may be occasioned directly or indirectly through the use of, or reliance on, the contents of this publication. The views and opinions in this publication are those of the authors and do not necessarily reflect those of the Australian Government or the Minister for the Environment.

Australian Government
Great Barrier Reef
Marine Park Authority

Great Barrier Reef Marine Park Authority
280 Flinders Street Townsville | PO Box 1379 Townsville QLD 4810
Phone: (07) 4750 0700
Fax: 07 4772 6093
Email: info@gbrmpa.gov.au
www.gbrmpa.gov.au

# Contents

# 1.0 Executive Summary

This report presents estimates of power to detect changes in the rate of coral cover recovery and species richness of herbivorous fishes. Estimates are based on the variability in existing time-series derived from the Australian Institute of Marine Science (AIMS) long-term reef monitoring programs. The objective is to provide a basis for monitoring program design considerations under the Reef 2050 Integrated Monitoring and Reporting Program (RIMReP).

Collectively, the AIMS programs provide the only time series of sufficient spatial and temporal coverage to allow estimation of variability at the scales necessary for considering design options at the scale of the Great Barrier Reef (the Reef).

Power analyses are specific to the hypothesis being tested by the underlying models. To compare across the multiple spatial scales within the AIMS monitoring designs required the use of a standard model across all reefs. As a result, the power estimates reported should be considered as conservative compared to the power that would be realised should more flexible models be applied to investigate specific questions of sub sets of the data.

Using the methods, and within reef replication of three sites used by the representative areas and long-term monitoring programs, there was reasonable power (>0.8) to detect trends in coral cover within a ‘region’ of 1 per cent p.a. over a five-year period. Detecting this level of change was reliant on annual sampling of 4-5 reefs within the region, where regions are defined as areas of similar location across the shelf with reefs separated by tens rather than hundreds of kilometres.

In general, the power to detect changes in the species richness of fishes was low compared to the power to detect changes in trend of coral cover. Annual sampling of 3-4 reefs over a ten-year period was required to ensure reasonable power to detect a change in richness of 3 species of herbivorous fish. This lower power is to be expected given the added variability in fish census data as a result of the mobility of fishes. From a design recommendation perspective, we have chosen to provide recommendations that aim to provide a sampling design for future monitoring that would provide high power to detect an approximate halving, over a five-year period, in the mean rate at which coral cover is recovering.

- Return to the annual sampling frequency originally intended for the long-term monitoring of coral communities: This will reduce the period over which changes can be detected, reduce the magnitude of changes than can be detected and improve the attribution of changes to specific pressures.
- Increase the number of reefs per “cluster” to at least 4/5. Where a cluster should encompass reefs in broadly similar environmental settings so that exposure to pressures are likely to be similar. This will ensure that regional trends are accurately estimated and facilitate the spatial delineation of where pressures are negatively impacting coral communities.
- Maintain within reef precision of estimates of coral cover to a least that currently applied by continued use of fixed, marked, sampling sites and an adequate intensity of sampling within those locations.

## 2.0 FIGURES

## 3.0 TABLES

# 4.0 Introduction

A core objective of long-term monitoring of the coral and fish communities is the detection of changes in key biological indicators that can be assessed in terms of the pressures being imposed on the system. The size of the Great Barrier Reef (the Reef) and the heterogeneity of communities, at all spatial scales, when coupled with variable exposure to pressures, limits the inference space about any particular observation of ecosystem condition. Sampling effort in existing monitoring programs has been spatially constrained due primarily to logistical and monetary constraints. This has limited the information available for some areas and habitats, and questions the representativeness of trends observed at the necessarily few monitoring locations.

The purpose of this report is to understand the historical variability observed in two high-end measurements of ecological condition, hard coral cover (HC) and species-richness of herbivorous reef fishes (HR), and how this variability influences the power to detect changes over time. This knowledge can be used as a guide to developing appropriate sampling designs for future monitoring in the Reef.

The analysis presented focuses on the three most extensive long-term coral reef monitoring programs on the Reef, the Long-Term Monitoring Program (LTMP – run since 1992), the Representative Areas Program (RAP – run since 2005), and the Marine Monitoring Program (MMP – run since 2005), each undertaken by the Australian Institute of Marine Science (AIMS). All three programs share similar sampling designs that include: replication of sampling within individual reefs to account for fine scale spatial heterogeneity of communities; clusters of reefs within tens of kilometres to allow generalisation of trends at individual reefs to larger spatial scales of within or between these clusters. Given the size of the Reef there is a necessary trade-off between the intensity of sampling undertaken at a particular site, that will influence the precision with which an indicator is measured, and the number of sites visited, that will allow greater certainty about any observed trends in a given indicator. Collectively the AIMS programs provide the only time-series of sufficient spatial and temporal coverage to allow estimation of variability at the scales necessary for considering design options at the scale of the Reef.

Importantly, statistical power relates specifically to the underlying model used to estimate the linear trend in indicators of interest and the resulting measurement variance estimates. Power calculations rely on statistical tests that aim to differentiate between two statistical hypotheses: the null hypothesis, $H_0$, that an effect of interest did not occur, and the alternative, $H_A$, that the effect of interest took place. The incorrect acceptance of either hypothesis leads to error in the interpretation of the test results. A Type I error occurs if $H_0$ is rejected, when it is, in fact, true (“false positive”). That is, a change in the indicator is identified when no change occurred. Framing this in the context of ecological management, a Type I error could result in the unnecessary use of resources directed to understanding or mitigating potential drivers of the observed change when, in fact, no action was required. While this might lead to a waste of resources, there is no ecological cost as no change in the indicator had occurred. In contrast, a Type II error occurs when $H_0$ is accepted as no change was detected, despite a change having occurred (“false negative”). In such cases, a need for action is not identified, giving no basis for consideration of appropriate management actions, and so, allowing pressures influencing the ecosystem to continue.

The probability of Type I error is defined by the significance of level α, typically set to 0.05. The probability of Type II error is defined by a parameter β, that it is not controlled for, rather, varies in response to the magnitude of the effect of interest and measurement variance in the indicator across the sampling design. The power of a test is defined as 1−β and can be understood as the probability of obtaining a significant result under the null hypothesis, $H_O$. A typical level of power aspired to in experimental designs is at least 80 per cent (Zar 1984).

Large data sets, such as those accumulated by AIMS monitoring, provide for a multitude of possible hypotheses. Here we focus on two questions critical to the long-term maintenance of the system. For corals, a key indicator of resilience is that cover increases during periods free from acute disturbances. It is important that a monitoring program has the ability to identify situations where recovery is not occurring, so that potential pressures can be identified and management options pursued. Here we assess the power to detect trends in coral cover during periods free from the influence of acute disturbances that can be interpreted as the ability to detect changes in the rate of coral recovery.

The indicator chosen for reef fish was species-richness of herbivores. The ability to confidently identify species of reef fishes, as compared to the ambiguity associated with field identification of corals, make the fish data more suited to the detection of a reduction in species-level diversity. In addition, the maintenance of herbivore diversity is seen as critical for the maintenance of coral reefs, as compensatory feeding produces indirect, though positive, effects on corals (Burkepile & Hay 2008). On the Reef, species-richness of herbivorous fish communities has been shown to be positively associated with their functional diversity and redundancy (Cheal et al. 2013).

# 5.0 Methods

## 5.1 Sampling design of existing programs

The LTMP sampling design clusters reefs into six "sectors" that define latitudinal swaths of the Reef (Figure 1). Within sectors sampled reefs are spread among "shelf positions" that describe broad differences in environmental conditions of water quality and exposure to swell across the continental shelf (Table 1). Shelf positions are categorised as:

- Inner – reefs periodically exposed turbidity arising from resuspended coastal sediments. These reefs lie within 20km of the mainland coast or major continental island groups.
- Mid – reefs separated from the Inner by the "shipping channel" which is an area largely devoid of platform reef development landward of the main Reef complex, and do not lie along the offshore margin of the Reef.
- Outer – reefs along the offshore margin of the Reef.

The RAP samples reefs primarily in the mid-shelf though some outer-shelf reefs are included (Table 1). The RAP design includes five clusters of reefs that align broadly with the Innisfail (includes southern Cairns), Townsville, Pompey (includes 2 western Swains reefs), Swains and Capricorn Bunker regions (Figure 1, Table 1). Ten reefs are sampled by both the LTMP and RAP.

MMP samples inner-shelf reefs only. Reefs are clustered in to four natural resource management regions areas that align with the Cairns and Innisfail, Townsville, Whitsunday and Capricorn Bunker sectors (Figure 1, Table 1).

The LTMP time-series from 1992-2004 included annual sampling of each reef. The RAP began in 2005 and, from this time on, sampling of both RAP and LTMP occurred in alternate years on a biennial cycle (Table 2). Sampling periodicity at the MMP sites has varied from annual sampling in 2005, 2006 to a mixture of annual and biennial sampling through to present.

All programs use permanently marked transects as the base sampling unit although the depth, length, and within reef replication of transects varies between those used by the LTMP and RAP, and those used by the MMP (Table 2). In each program, estimates of the composition of benthic communities were derived from the identification of organisms on digital photographs taken along the permanently marked transects. The method followed closely the Standard Operation Procedure Number 10 of the AIMS Long-Term Monitoring Program (Jonker *et al.* 2008). In short, digital photographs were taken at set intervals (Table 2) along each transect. Estimates of benthic cover were derived from the proportion of points along transects identified as the category of interest.

**Table 1: Sampling design of the LTMP and MMP.**

| Project | Sector / Region or Subregion | Latitudinal range | Reefs sampled | | |
|---|---|---|---|---|---|
| | | | Inner | Mid | Outer |
| LTMP | Cooktown/Lizard | 14.52 S - 14.92S | 2 | 3 | 3 |
| | Cairns | 16.04S – 16.92S | 3 | 4 | 3 |
| | Townsville | 18.26S – 19.19S | 3 | 3 | 3 |
| | Whitsunday | 19.66S – 20.18S | 3 | 3 | 3 |
| | Swains | 21.47S – 22.02S | | 5 | 2 |
| | Capricorn/Bunker | 23.25S – 23.88S | | | 4 |
| RAP | Cairns | 16.04S – 16.80S | | 3 | 2 |
| | Innisfail | 16.84S – 17.81S | | 7 | |
| | Townsville | 18.42S – 18.73S | | 9 | 3 |
| | Pompey | 20.88S 21.05S | | 10 | |
| | Swains | 21.11S – 22.00S | | 11 | 3 |
| | Capricorn/Bunker | 23.17S – 23.88S | | | 8 |
| MMP | Wet Tropics | 16.29S – 18.01S | 12 | | |
| | Burdekin | 18.57S – 19.15 | 6 | | |
| | Mackay Whitsunday | 20.10S -20.47S | 7 | | |
| | Fitzroy | 23.09S – 23.34S | 6 | | |

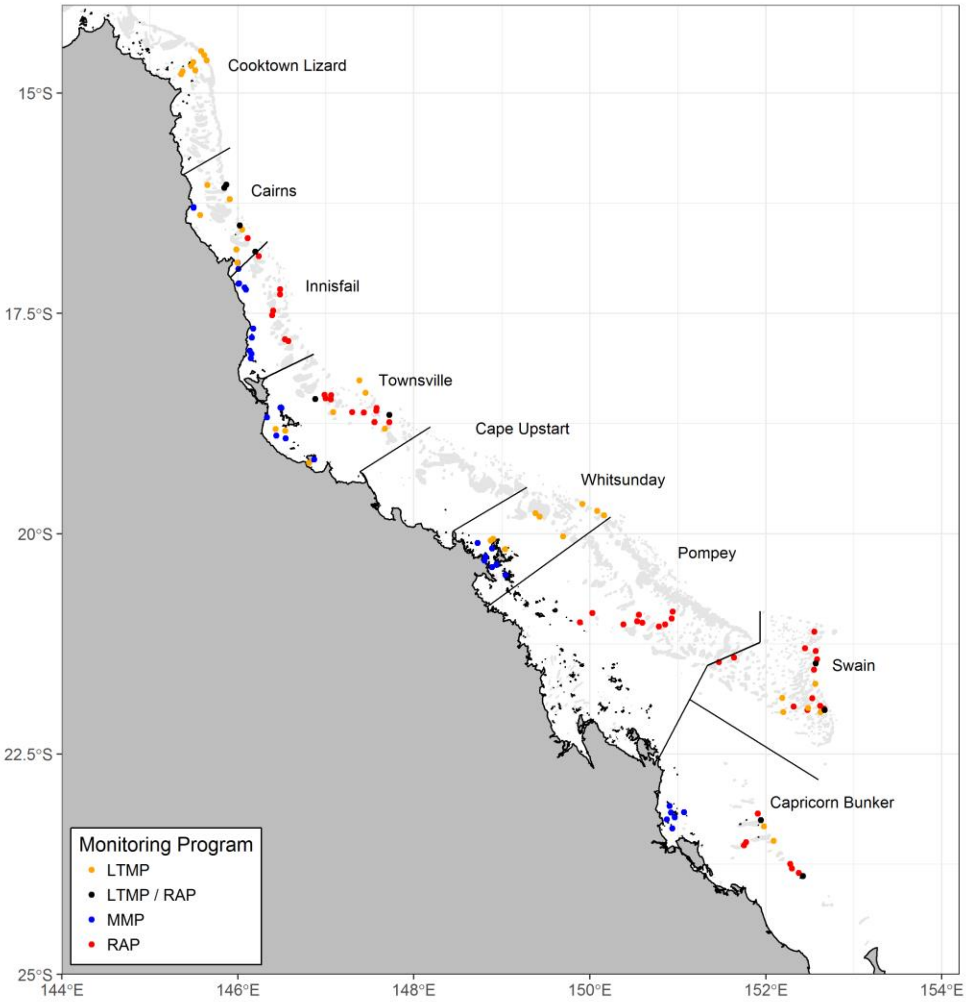

**Figure 1. Location of reefs sampled by the AIMS monitoring programs.**

For consideration of relative precision between programs the sampling intensity used by Reef Check Australia was also considered. Reef Check Australia estimates coral cover using an in-situ point intercept technique with cover estimated under points separated by 50cm along a transect line laid along the substrate (Table 2).

The LTMP and RAP additionally estimate fish abundances along the benthic transects. Larger fish species are counted within 5 m wide belts with smaller damsel fish species counted within a 1 m wide belt. For this report the richness (number of distinct species) of herbivorous fishes was used as the indicator variable.

**Table 2. Survey methods and within reef sub-sampling design for the MMP and LTMP**

| | MMP | LTMP & RAP | Reef Check |
|---|---|---|---|
| Spatial design | | | |
| Transect length | 20m | 50m | 20m |
| Transects per site | 5 | 5 | 4 |
| Sites per reef | 2 | 3 | Typically 1, up to 3 |
| Depths per site | 2m and 5m | ~6m | variable |
| Transects marker interval | 10 m | 10 m | Unmarked, rely on maps |
| Point intercept method for benthic classification | | | |
| Image interval along transects | 0.5m | 1m | |
| Images sampled per transect | 32 | 40 | |
| Points identified per site / depth combination | 800 | 1000 | 160 |
| Visual census for fish | | | |
| Transect width Damsel fish | n/a | 1 m | |
| Transect width Other reef fish | n/a | 5 m | |

## 5.2 Sampling error

The primary focus of monitoring is to identify change in an indicator of interest. The ability to confidently ascribe change relies on minimising unexplained variability in observations. Variability in the observations within a time-series occurs partly as a result of differences in the indicator at a range of spatial and temporal scales as well as sampling imprecision. The power to detect a change in the indicator is reduced by the combination of sampling error and real differences in the trend of the indicator that are not captured by the model applied.

For coral cover, sampling error will include a combination of: random variability in the intersection of sampled points across the benthic community; differences in the percentage cover of the benthos beneath the transect line as a result of variability in the location of the line; observer error or bias in identification of the benthos below selected sampling points. For reef fish species richness, the movement of fishes across transect boundaries, along with observer bias, are likely the primary factors influencing sampling error.

It is not the aim of this report to tease apart sampling error as the combined errors are implicitly accounted for in the error term of the models for which we assess power. That said, sampling intensity is a consideration relevant to future sampling design options as it will impact on the precision of coral cover estimates for individual surveys. Coral cover can be estimated as the proportion of points from a given survey that are classified as coral, as opposed to anything else. As such, it is possible to assume that coral cover is distributed according to a binomial distribution B($n, p$) where a given point in a survey of sample size $n$ is classified as coral as opposed to anything else according to probability $p$. Under this scenario, to illustrate the influence of the sample size $n$ on the precision of 95 per cent confidence intervals for $p$, confidence intervals were calculated for coral cover estimates of 5, 10, 25 and 50 percent and for sampling intensities ranging from 0 to 3500 points. Confidence intervals were computed based on Normal approximations as follows:

$$CI = \hat{p} \pm 1.96 * \sqrt{\frac{\hat{p}\ (1-\hat{p})}{n}}$$

## 5.3 Power analysis

While sampling error can be reduced by the implementation of an adequate sampling protocol, real variability at various spatial and temporal scales is more appropriately accounted for by partitioning variability on the basis of the sampling design.

The sites of both the MMP and LTMP monitoring programs investigated in this report have suffered over time from a range of acute disturbances. Disturbances can drastically reduce coral cover, adding substantial variability to estimates of hard coral cover over time. As the focus of the power analysis for coral cover is the detection of an increase in coral cover (recovery) during periods free from disturbance, it was essential to first account for the variability associated with these disturbance events.

In order to utilise the entire time series of observations from each reef to estimate power, our approach was to correct the hard coral measurements at time points where a known disturbance had occurred. This was possible as obvious reductions in coral cover as a result of acute events are identified by each of the monitoring programs. The correction applied was to replace hard coral cover estimates recorded after a disturbance with the value observed prior to the disturbance. Subsequent observations were adjusted to maintain the incremental changes so applied.

Although simple, this correction has proved to be an effective method for adjusting hard coral cover as it preserves the general trend in coral cover increase during periods of recovery without introducing larger variability. Ideally, more flexible models that focus specifically on coral cover growth would be used to account for the effects of disturbances. However, when power is compared between different sampling designs and locations that would require that the growth models for different reefs under study would be fairly similar, the predicted values might increase the variability. An easy correction, like the one presented here, has the advantage of being easy to implement in any program, and it is comparable across programs. No correction was applied to the fish richness data as we were not confident that observed changes in species richness between pre and post disturbance observations reliably indicated a response to disturbance.

The power calculations presented in this report relate to the power to detect a trend in hard coral cover recovery or species richness of herbivorous fish at the spatial scales of reefs or clusters of reefs within localised areas of the Reef defined by a combination of latitude and position on the continental shelf. The choice of scales is informed by the underlying sampling design of the existing monitoring programs. The sampling design was, however, developed based on the understanding that coral reef communities on the Reef show clear variability both across the shelf and with latitude. Here, the calculations were performed via simulation using hierarchical models (Gelman and Hill, 2007) as follows:

1. Population parameters were estimated based on existing data.

   Hierarchical linear models with random effects for log transformed indicator variables were used to estimate variability in hard coral cover at the different study scales, namely site, reef and region. The models used in this study can be summarized as:

   $y = b_0 + tb_1 + Zu + \epsilon$ , with $u \sim N(0, \Sigma_u)$ and $\epsilon \sim N(0, \Sigma_\epsilon)$

   Where $b_0$ is the intercept, $b_1$ is the time slope and *u* are the random effects estimates, $Z$ corresponding to sites, when studying individual reefs, or reefs and sites nested within reefs when looking at regions.

2. A sample of *B* = 1600 data sets were simulated using estimated population parameters.

   Using the variance model estimates $\hat{\Sigma}_u$ and $\hat{\Sigma}_\epsilon$, data were generated with an added time trend with known slope (change over time) using model estimated intercepts $\hat{b}_0$. Data were simulated for a range of trends over 5-and-10-year periods for both annual and biennial sampling; specifically changes in hard coral cover of 1 per cent, 2 per cent, 3 per cent per annum, and changes in species richness of herbivorous fish of 0.1, 0.3 and 0.5 species per annum. Noting that the mean rate of cover increase within the time-series was ~2 per cent, as such, power to detect a change of 1 per cent per annum would allow both the detection of recovery at lower than average rates, but also the reduction of recovery at the majority of reefs. Higher rates of change will allow detection of recovery but reduce the capacity to detect a slow down to only regions exhibiting historically above mean levels of recovery.

3. Model trend parameters were estimated for each simulated data set based on the full sampling design of the original data, and subsets of diminishing numbers of sites within reefs and reefs within regions.

   Each of the simulated data sets had the same variability as the original data, allowing investigation of the experimental design power under different scenarios, such as differing number of sites per reef or reefs per region.

   For the selected experimental design fit:

   $y = b_0^* + tb_1^* + Zu^* + \epsilon^*$ , with $u^* \sim N(0, \Sigma_{u^*})$ and $\epsilon^* \sim N(0, \Sigma_{\epsilon^*})$

Where $b_0^*$ is the intercept, $b_1^*$ is time slope and $u^*$are the random effects corresponding to sites when studying an individual reef's power, or reefs and sites within reefs when looking at regions.

4. Hypothesis tests were carried out for each model trend estimate (slope) in the simulation sequence.

For each of the simulated data sets under a given design the fitted model estimates were used to test for changes on the slope parameter estimates for a rejection level $\alpha$ = Probability (Type I error) = 0.05 for $H_0 : b_1^* = 0$ versus $H_A : b_1^* \neq 0$.

5. Power computed.

With *R* being the number of times that the null hypothesis of no change ($H_0$) was rejected out of the *B* simulations for a given experimental design. Power is then calculated as the proportion of times that the null hypothesis was rejected:

$$\text{Power} = \text{Probability (Reject } H_0 \mid H_0 \text{ is false)} = R/B$$

# 6.0 Results

## 6.1 Sampling intensity and precision of coral cover estimates

Theoretical improvement in the precision of hard coral cover estimates based on random samples drawn from a binomial distribution where hard coral covers ranges between 5 per cent and 50 per cent are displayed in Figure 2. This figure shows the theoretical improvement in the precision of hard coral cover estimates based on inclusion of multiple sites sampled at the intensity used by the LTMP and RAP, the MMP, and Reef Check Australia. The distributions presented in Figure 2 give some guidance as to the confidence one should have in observed differences in cover between any two observations. The 95 per cent confidence intervals are at a maximum for a 50 per cent estimated coral cover and decline to 0 at covers of 0 or 100 per cent. For instance, at an estimated coral cover of 50 per cent and at site level, increasing sampling effort from the 160 points identified by Reef Check to the 800 points sampled by the MMP improves precision more than 2 fold (Table 3). However, there is a more moderate improvement in precision with an increase from the 800 points sampled by the MMP and the 1000 points sampled by the LTMP and RAP (Table 3). The value of additional sites in each program is to increase the sampling intensity at the reef-level and gain improved precision in the reef-level estimate of coral cover, expressed in Table 3. From this table, it is clear that as the number of sites increases, the corresponding 95 per cent confidence intervals reduce in size.

**Table 3. Improvement in precision of coral cover estimates with increasing sampling intensity.** Values represent the span between upper and lower normal approximations of 95 per cent confidence intervals for coral cover estimated at 50 per cent. Additional sites represent multiplicative increases in points sampled at a single site. Four sites are included for MMP as this is the sampling intensity at the reef level where two depths are surveyed.

| Program (points per site) | 1 site | 2 sites | 3 sites | 4 sites |
|---|---|---|---|---|
| Reef Check (160) | 15.49 | 10.96 | 8.95 | |
| MMP (800) | 6.93 | 4.90 | | 3.46 |
| LTMP and RAP (1000) | 6.20 | 4.28 | 3.58 | |

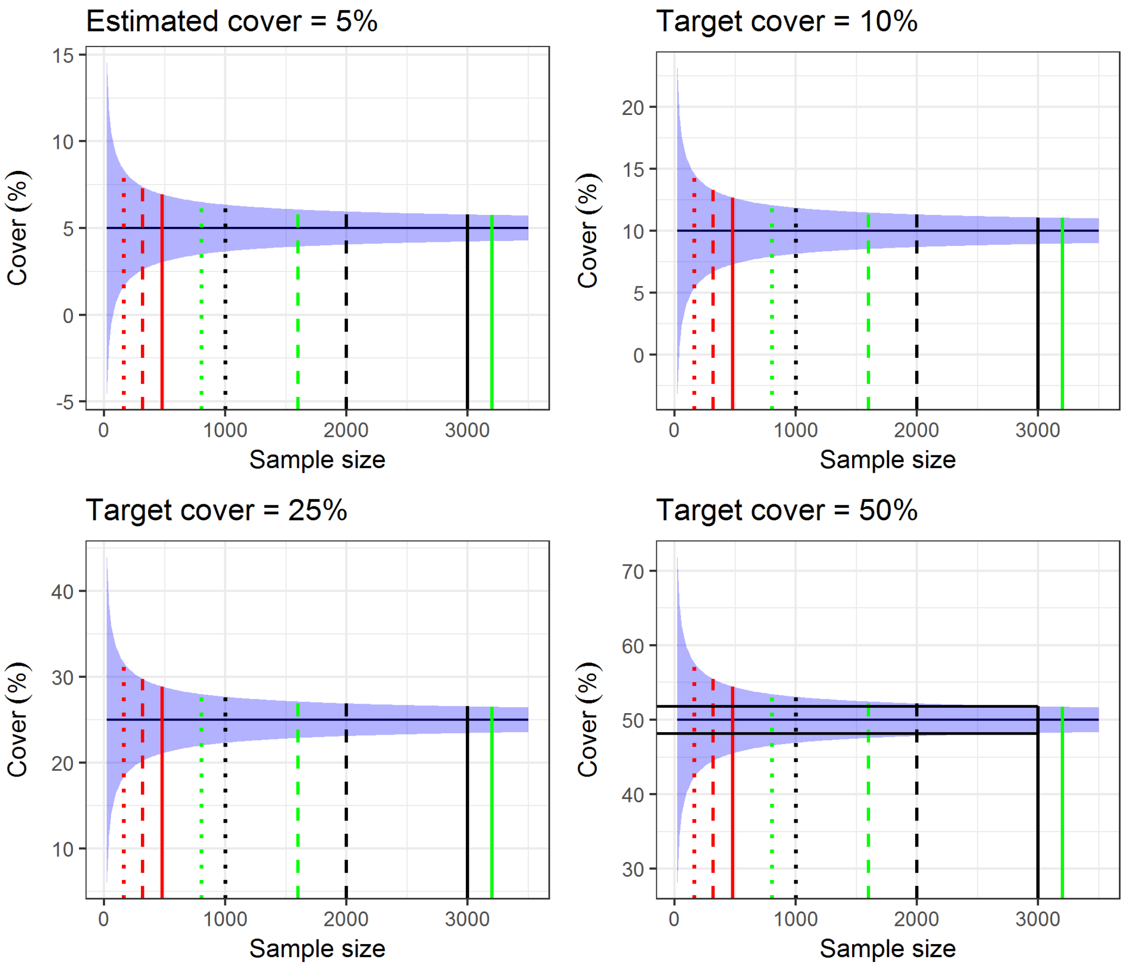

**Figure 2. Theoretical influence of sampling intensity in confidence intervals about mean coral cover.** Reference lines indicate sampling intensity used by Reef Check (red), MMP (green) and LTMP/RAP (black) for one (dotted), two (dashed) and solid for three (Reef Check, LTMP/RAP) or four (MMP) sites within a reef.

## 6.2 Power to detect changes within a reef

Comparing power between the MMP, LTMP and RAP at a sampling rate of 2 sites demonstrates the similarity of results between the two sampling designs, with little, if any, improvement in power realised by the more intensive sampling undertaken be the LTMP of 1000 cf. 800 points per site (Figure 3). Improvements in power were logically achieved as the magnitude of the trend in coral cover increased.

For each program there was substantial variability among reefs in the power to detect recovery trends of a given magnitude and this is likely to have overwhelmed any effect of the sampling intensity. The reefs demonstrating the least power to detect linear recovery trends in the LTMP and RAP were the outer-shelf reefs in the Cooktown Lizard and Capricorn Bunker sectors (Figure 4, Figure A1). The two reefs with least power in the MMP were Franklands East and Franklands West at 5 m depth (Figure A1, A2). Site level trajectories of the disturbance-corrected coral cover, used to estimate power, at Franklands East and Wreck Island are provided (Figure 5) showing large variability in trends over time at site level. Those figures illustrate important points relevant to the interpretation of the power estimates that are raised in the discussion.

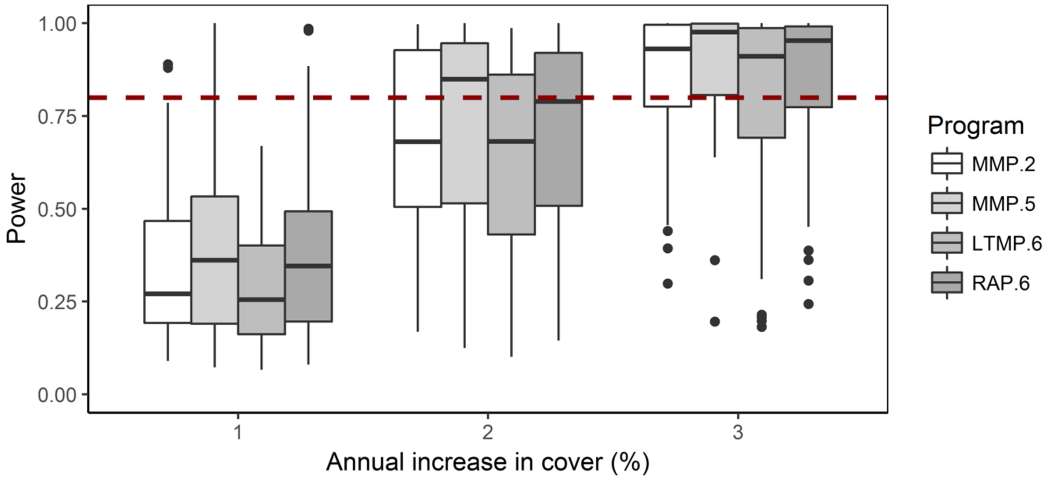

**Figure 3. Power to detect changes in hard coral cover over a 5 year period based on annual sampling. Boxplots represent the distribution of estimates from all reefs sampled by the MMP at 2m or 5m depths, the LTMP and RAP programs based on within reef replication of 2 sites. Columns represent power estimates to detect trends ranging from 1 to 3 per cent per annum. A horizontal reference line at a power of 0.8 is provided.**

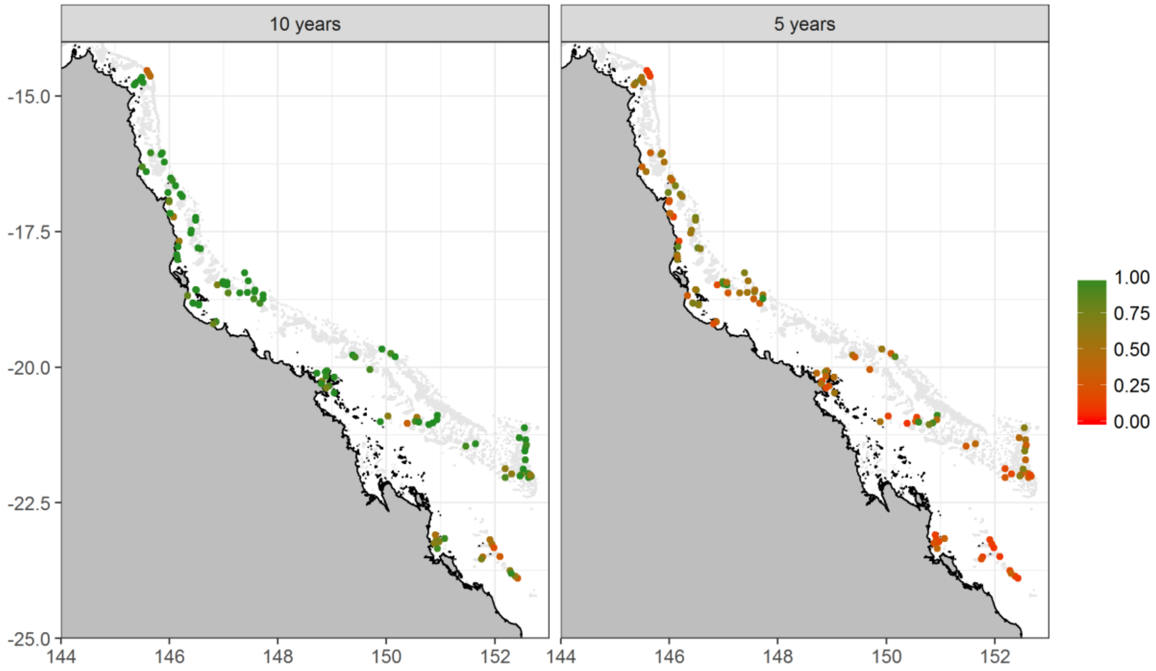

**Figure 4. Spatial distribution of power to detect a change in coral cover. Power estimates are for a 1 per cent per annum change in trend based on annual sampling over a 10 year (left) or 5 year (right) period. For MMP reefs estimates represent the mean of power estimated for 2m and 5m sites. Estimates are based on the full sampling design of 2 site per depth for MMP reefs and 3 sites at LTMP and RAP reefs.**

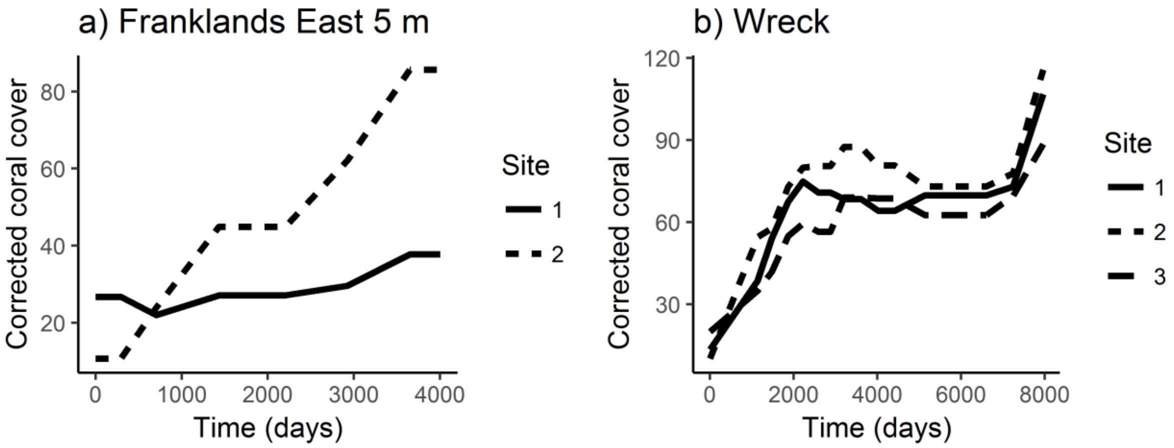

**Figure 5. Examples of time-series with low power to detect change at reef level. a) Differing rate of change in coral cover between sites, and b) no linear rate of change in coral cover over the time-series.**

At the level of individual reefs, the power to detect changes in coral cover at a given rate increases with: the number of sites sampled, the duration of the time series, and the frequency of sampling (Figure 6). For example, the median power to detect a change in trend of 1 per cent per annum at the lowest sampling intensity considered, of one site per reef over a 5 year period is minimal, although improved when annual rather than biennial sampling is undertaken (Figure 6a). Power to detect changes of 1 per cent per annum improve markedly with both the sampling of additional sites (Figures 5b, c), or when a longer, 10 year time-series is considered (Figures 6d-f).

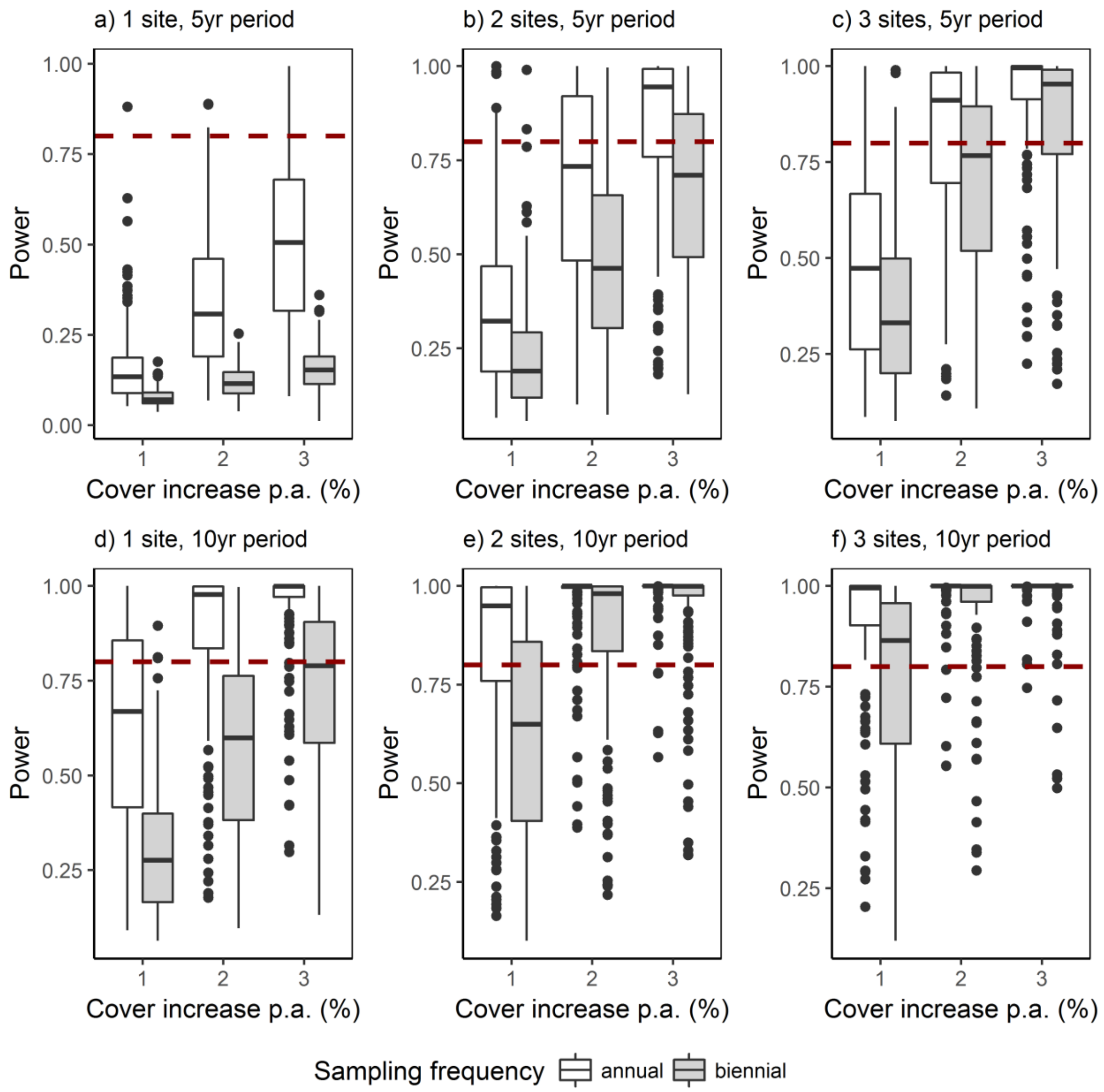

**Figure 6. Power to detect trends in hard coral cover at the scale of sector and shelf combinations. Sampling designs options include variable number of sites sampled per reef (displayed on the x axis of the plots), variable number of reefs sampled in the sector-shelf combination (defined by boxplot shading), and either biennial (top row) or annual (bottom row) sampling. Only observations from the mid-shelf are included. Data are derived from the LTMP and RAP programs.**

In addition to the number of sites sampled within a reef, the number of reefs sampled within a “region” of the Reef has a clear influence on power to detect changes of a given magnitude (Figure 6, see also Figures A2, A3 for power to detect trends of 2 per cent and 3 per cent per annum). The number of reefs required to achieve a desired level of power does, however, vary among regions being lowest in the Capricorn Bunker Region (CB) and highest in the Townsville Region (TO) (Figure 7).

Taking the Pompey Region (PO) as an example, achieving a power of 0.8 to detect a change in coral cover trend of 1 per cent per annum over a 5 year period requires the annual sampling of three sites at each of 5 reefs (Figure 7). The number of reefs required to achieve the same level of power over a 10 year period declines to 1 or two depending on the annual or biennial frequency of monitoring (Figure 7).

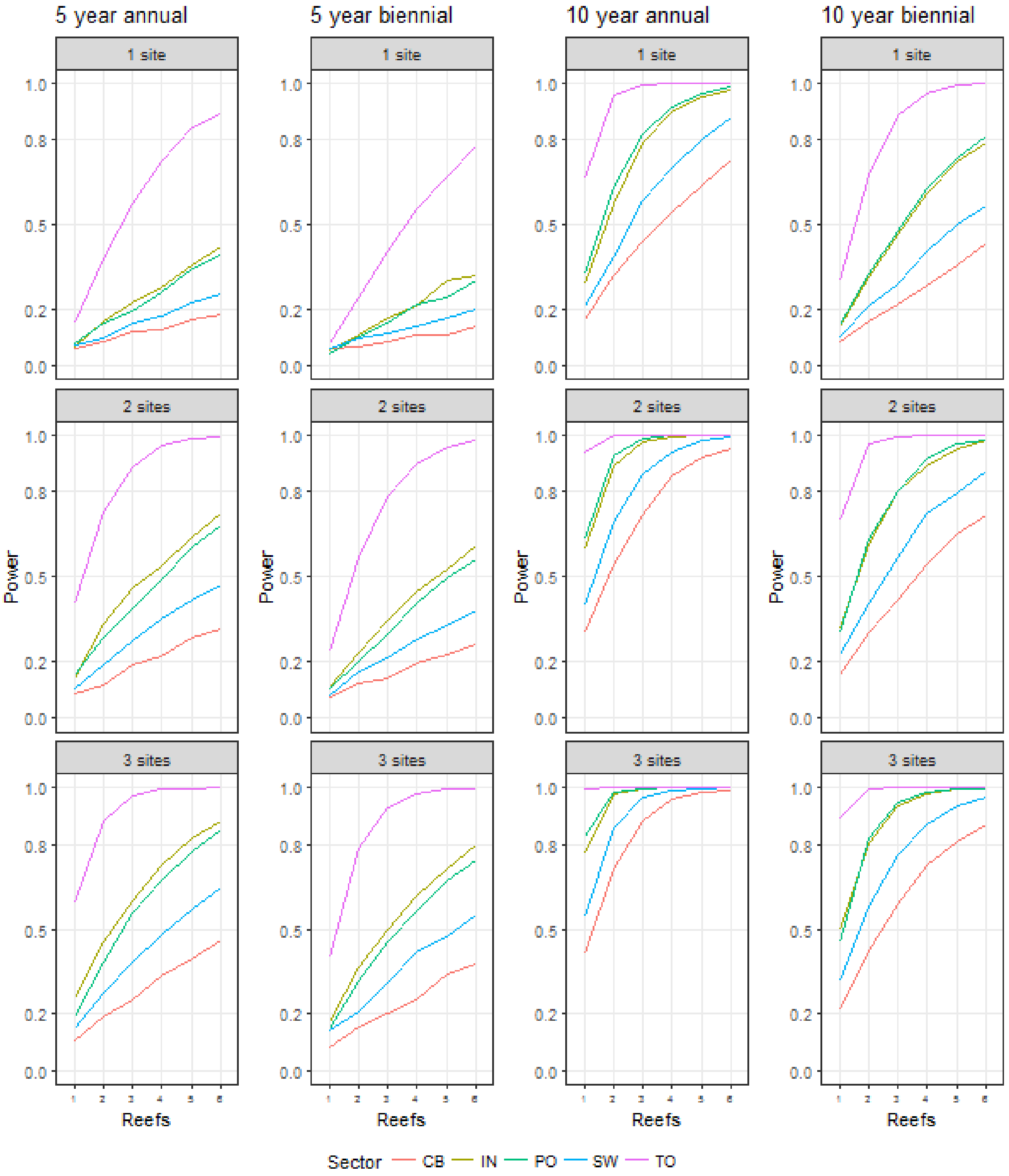

**Figure 7. Influence of the number of reef per region on power to detect changes in hard coral cover trends of 1 per cent p.a. over a period of 5 and 10 years. Estimates based on mid-shelf reefs monitored by RAP in each sector, indicated by colour. The exception is the Capricorn Bunker Sector where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.**

At the level of individual reefs, there is relatively low power to detect changes in the species richness of herbivorous fish lower than at a rate of 0.5 species per annum (five species over a 10 year period). It is only at a sampling intensity of three sites per reef, sampled annually over a 10 year period that changes of this magnitude were detectable with greater than a power of 0.6 at the majority of reefs (Figure 8).

Within sectors monitored by the RAP program three to five reefs, sampling at the intensity of three sites on an annual basis should allow the detection of a change in species richness of 3 species over a 10 year period, the exception being the Capricorn Bunker Sector (CB) where there was limited power to detect changes in species richness, irrespective of the sampling intensities investigated (Figure 9).

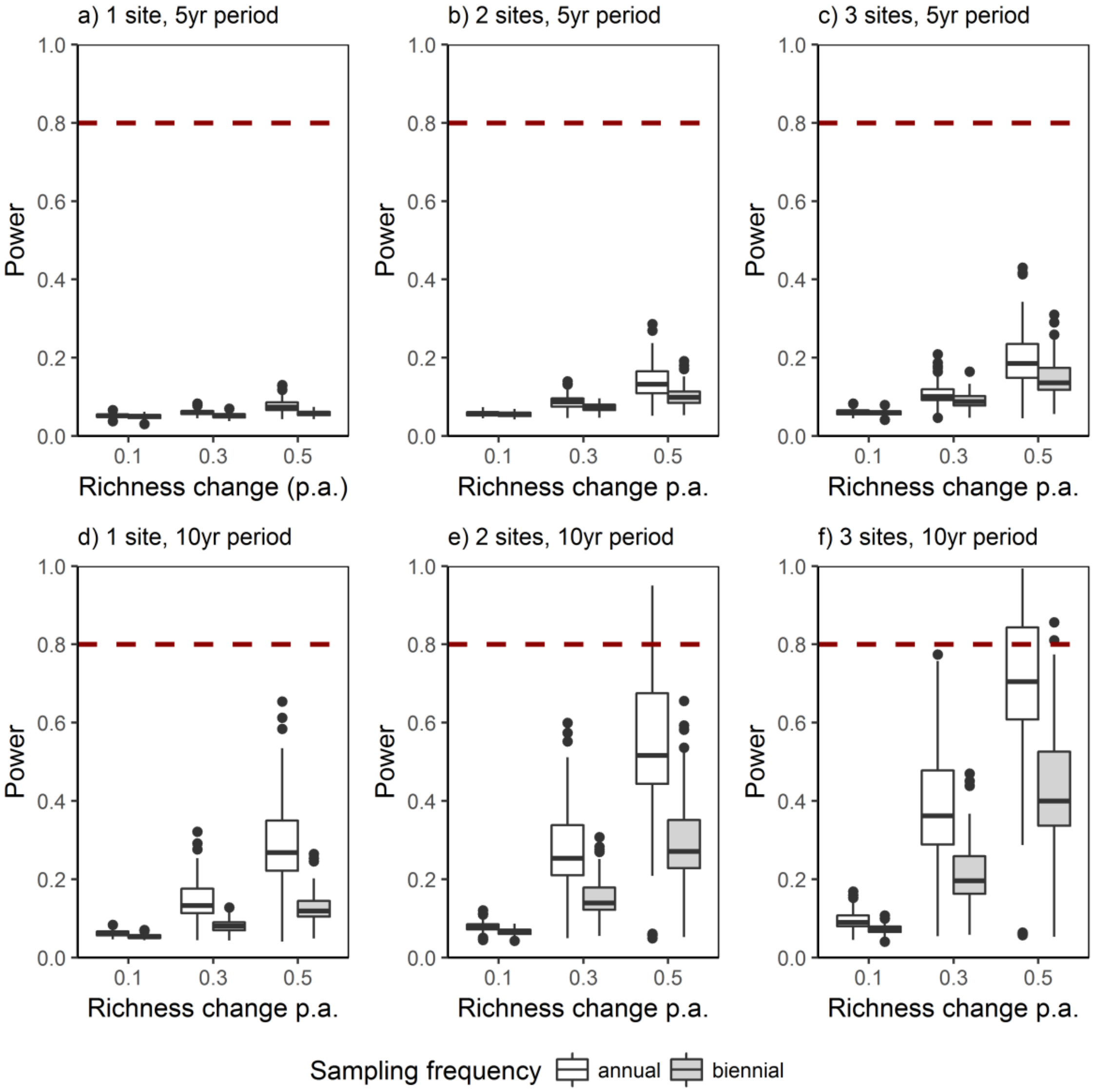

**Figure 8. Power to detect changes in the richness of herbivorous reef fish over a 5 and 10 year period based on biennial sampling (top row) or annual sampling (second row). Boxplots represent the distribution of estimates from all reefs sampled by LTMP and RAP programs. Within each plot the estimates based on sampling 1, 2 or 3 sites per reef are displayed. Columns represent power estimates to detect trends ranging from 0.1 to 0.5 species per annum. A horizontal reference line at a power of 0.8 is provided.**

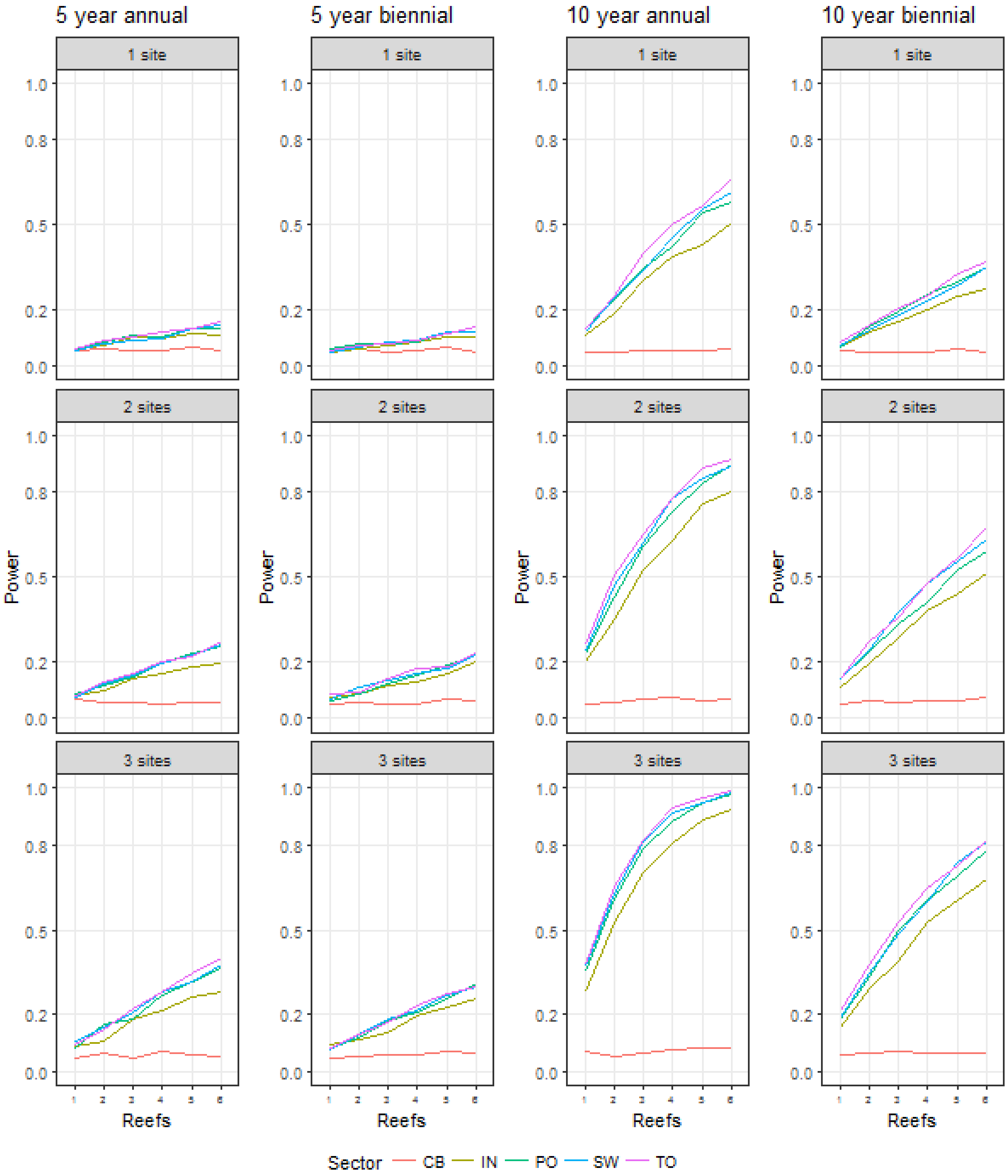

**Figure 9. Influence of number of reefs on power to detect a change in fish species richness at a rate of 0.3 species per annum (plus or minus three species over 5 and 10 years). Estimates based on to mid-shelf reefs monitored by RAP in sectors, indicated by colour. The exception is the Capricorn Bunker Sector where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.**

## 7.0 Conclusions

This report investigates the power of the three AIMS-run monitoring programs, namely LTMP, MMP and RAP, to detect changes in the recovery rate of coral cover and the species richness of herbivorous fishes. The primary focus was to understand possible deficiencies or redundancies in the programs' sampling designs that could guide future design options. The results presented here need to be interpreted with due consideration of two important aspects of the analyses performed: that they are specific to the model applied, and that they are dependent on the historical variability within and between the sites sampled (Button et al. 2013). While there is no guarantee that future variability will be the same as that previously observed in this report, we have not attempted to guess at how variability may change and the effect that such changes would have on the power reported.

The fitting of generic hierarchical linear models to each time-series was necessary to allow the estimation of variability at the various spatial scales of the Reef. The underlying assumption was that there was a consistent trend throughout the time-series. For coral cover we applied a simple correction to account for losses of coral cover that were attributed to major disturbance events. The reason for that correction was to separate the volatility in coral cover resulting from large disturbances from our capacity to estimate recovery rates before and after the disturbance event took place. While the correction applied helps to remove the influences of disturbance events, the correction itself introduced biases resulting from non-linearity in the rate of coral cover increase across the range of covers observed at some reefs that will have resulted in underestimates of power. Modelling the rate of increase in coral cover as a log-linear response is reasonable at low through to moderately high covers, however, at high cover space becomes limiting and as such the rate of cover increase diminishes. For our models a slowing in coral cover increase at high cover will have resulted in additional variance and so reduced power estimates for reefs where space limitation occurred. Secondly, where cover was severely reduced, the correction of the resulting low coral cover to a much higher cover comes with an expectation of proportional increase at the corrected value: again this would have had the effect of reducing power estimated for those reefs. The low power estimates for many of the Tabulate *Acropora* dominated reefs of the Capricorn Bunker and outer-shelf Cooktown Lizard sectors should be considered in light of the likely biases introduced as a result of both the correction for disturbance events and high covers attained at these reefs; the case of Wreck Reef (Figure 6b) is a clear example. The coral cover time-series at Wreck Reef tracks coral cover from initially very low levels through a period of increasing rate of recovery until recovery slowed at ~60-70 per cent, at this point coral cover was reduced to very low levels and the process repeated. The result was a clearly non-linear time-series for corrected coral cover (Figure 6b) contributing to the very low power estimate for that reef as the corrected time-series was not log-linear across its range. This lack of fit of the underlying model partly explains the low sector level power estimated for the Capricorn Bunker Sector. In practice, Gompertz growth equations have been fitted to coral cover time-series and these, more appropriate models, have allowed the demonstration of reduced rates of cover increase in both the Capricorn Bunker and Cooktown Lizard Sectors (Osborne et al. 2017). Low power may also reflect variable trends in coral cover among sites within a reef, the low power estimated for Franklands East (Figure 3a) is an example. This again highlights the limitation of power analysis applied across such a range of reefs and scales that did not allow the more nuanced fitting. The reason for that being the reliance of power analysis on the hypothesis testing of single parameters that aim at explaining changes over time and which might be too restrictive in many cases.

Despite the limitations imposed by the necessarily linear models underlying the power analyses, there was reasonable power to detect a 1 per cent per annum change in coral cover recovery over a five year period at the scale of sector and shelf combinations. This level of change was reliant, however, on the annual sampling of 4 to 5 reefs, a level of replication that is higher than the current 3 reefs sampled biennially by the LTMP. The choice of a 1 per cent per annum change as a desirable level was made as it represents a halving or doubling of the ~2 per cent per annum mean recovery rate for coral cover over the existing monitoring time series. A reduced sampling design aimed at detection of larger changes in recovery rate would risk missing important downturns of recovery and so severely limit the utility of the program for the early identification of areas where coral resilience was compromised. Similarly, the focus on a five-year period recognised not only the need for early detection to maximise the potential for management intervention, but also recognised the often short periods between disturbance events within which estimates of recovery rate can be derived. In addition to the clear improvement of power we observed with annual rather than biennial sampling, the ability to confidently ascribe causation to losses of coral cover, and so tease out the influences of multiple disturbances, is greatly improved with annual visitation.

A preliminary investigation of power to detect changes in hard coral cover trends of the MMP using a broadly similar approach (linear regression models instead of hierarchical models) was applied by Kuhnert et al. (2014)) to estimate power to detect changes in trends at individual reefs. That study reported some very low estimates of power as power was calculated in terms of proportional changes in coral cover rather than absolute changes in coral cover considered herein. The estimation of power based on proportional changes is quite biased across the range of coral cover observed on reefs. As an extreme example, if coral cover was initially low, say 2 per cent cover, then increased over a period to 4 per cent cover, that small increase represents a proportional change of 100 per cent, understandably there would be negligible power to detect such a small change in coral cover. Further, Kuhnert et al. (2014) did not account for the real, and often large, changes in coral cover resulting in from exposure to known disturbances. Certainly, the higher power reported in this study was critically reliant on the identification of disturbance events that allowed the influence of the resulting large changes in cover to be removed from the variance estimates that together with the effect size under consideration dictate the power to detect changes in trends.

Deciding on an appropriate indicator for reef fish was problematic as unlike coral cover there was no single summary of the fish community that clearly represented the “condition” of the system. Although Emslie et al. (2104) clearly demonstrate a reduction in richness of the main herbivore families, Scaridae, Acanthuridae, in response to a loss of habitat complexity, as can occur as a result of disturbance, cyclones in particular. However, the relationship reported by Emslie et al. (2014) varied spatially and was not evident for other disturbance types. This inconsistent response to disturbance precluded the correction of the species richness time-series for any response to disturbances meaning that any real changes will have been added to the “noise” in the time-series and contributed to low power. Further, census of fish are naturally variable because of the mobility of fish across transect boundaries meaning that the sites are not as ‘fixed’ as they were for corals. In addition, as abundance declines, the probability of observation will also decline, meaning that observed species richness change may exceed actual local losses. Given these limitations, it is not surprising that the power to detect changes in species richness of herbivorous fish was limited compared to that demonstrated for coral cover recovery. There is no evidence of redundancy in the sampling design for reef fish.

Implicit in the level of power demonstrated in this report is the precision in coral cover estimates attained by the sampling methods. Any reduction in precision will logically result in a reduction in the power to detect changes of a given magnitude. The current methods used by the AIMS programs are based on the sampling of five points per image from images spread at 1m (LTMP and RAP) or 50cm (MMP) intervals along the fixed transects. In part, the selection of five points per frame is a technological legacy, balancing the diminishing information content of additional points per frame, due to point’s spatial dependency, and the time-cost of capturing and viewing multiple frames. Costs were high in the early years of the LTMP when images were captured from video footage and computing power considerably lower than that of today (Ryan 2004). Even with these constraints, Ryan (2004) suggested that efficiency of sampling was increased by maximising the separation of points along transects. That is, for the same number of sampled points, precision in cover estimates improves by considering more images with fewer points per image. A further influence on precision of estimates through time is that of real differences in the sample population from one sample to the next resulting from imprecise placement of the sampling unit. In practice, the position of the transect line will vary from one survey to the next, potentially adding placement error to the underlying estimated sampling error. It is to minimise this additional source of error that permanently marked transects are used by the LTMP, MMP and RAP monitoring programs. From a sampling design perspective, the accurate relocation of the transect lines minimise the additional uncertainty introduced between samples that would occur if transects were randomly deployed. For instance, Ryan & Heyward (2003) demonstrated substantial improvement in sampling precision when marked transects are used, rather than random deployments in a fixed area. We reiterate this consideration of precision here only to acknowledge that as technologies improve, alternate sampling methods may be adopted and we must ensure that they deliver precision of estimates at least equal to those currently achieved, if relevant change in key indicators is to be detected.

Finally, any given experimental design should not only have a sound design which will allows us to make statistical inferences, but it should also be driven by a very strong ecological rationale. For the programs assessed in this report, broad patterns that defined both coral and fish communities were implicitly considered in the underlying design and influenced the focus on clustering of reefs within regions or locations across the shelf. Ninio et al. (2000) described coral community composition in terms of a spatio-temporal mosaic recognising that communities in close proximity to each other are both selected for by their co-location along environmental gradients of light, temperature and water quality as well as similar historic exposures to large scale disturbances. These features mean that the current clustering of sites likely improved the power to detect consistent changes within regions of the Reef, though it potentially limits the validity of interpolating those changes out to areas of reef that are more distant. The indication that power will be improved with additional reefs may provide the opportunity to increase the geographical coverage of the program. However, the likelihood that communities will respond in a consistent way to any given pressure will almost certainly decline with the distance between those reefs as pressure intensity and community compositions vary (Ninio et al. 2000).

## 7.1 Recommendations

Based on the above analyses and considerations, here we provide a number of recommendations for a revised sampling design for an improved Reef monitoring program. These recommendations are aimed at promoting the timely detection of spatial-temporal fluctuations in coral cover as basic information against which management actions may be considered. We do not explicitly provide recommendations for the monitoring of species richness of herbivorous fish, in the understanding that improved ability to detect changes in reef fish communities will naturally flow from any uptake of the below recommendations for increased sampling:

- Return to annual sampling frequency. This will reduce the period over which changes can be detected, reduce the magnitude of changes than can be detected and improve the attribution of changes to specific pressures.
- Increase the number of reefs per “cluster” to at least 4/5. A cluster should encompass reefs in broadly similar environmental settings so that exposure to manageable pressures are likely to be similar. This will ensure that regional trends are accurately estimated and it will facilitate the spatial delineation of where pressures are negatively impacting coral communities.
- Maintain within reef precision of estimates of coral cover to a least that currently applied by continued use of fixed marked sampling location and adequate intensity of sampling within those locations.

## 8.1 References

Burkepile DE, Hay ME (2008) Herbivore species richness and feeding complementarity affect community structure and function on a coral reef. Proceedings of the National Academy of Sciences USA 105:16201–16206.

Button KS, Ioannidis JP, Mokrysz C, Nosek BA, Flint J, Robinson ES, Munafò MR (2013) Confidence and precision increase with high statistical power. Nature Reviews Neuroscience 14(8):585.

Cheal AJ, Emslie ME, McNeil MA, Miller I, Sweatman H (2013) Spatial variation in the functional characteristics of herbivorous fish communities and the resilience of coral reefs. Ecological Applications 23(1): 174-188.

Emslie MJ, Cheal AJ, Johns KA (2014) Retention of habitat complexity minimises disassembly of reef fish communities following disturbance: a large-scale natural experiment. PLoS ONE 9(8): e105384. Doi: 10.1371/journal.pone.0105384

Gelman A, Hill J (2007) *Data analysis using regression and multilevel/hierarchical models*. Cambridge University Press, Cambridge.

Jonker M, Johns K, Osborne K (2008) Surveys of benthic reef communities using underwater digital photography and counts of juvenile corals. Long-Term Monitoring of the Great Barrier Reef, Standard Operational Procedure Number 10, Australian Institute of Marine Science, Townsville

Kuhnert PM, Liu Y, Henderson B, Dambacher J, Lawrence E, Kroon F (2014) Review of the Marine Monitoring Program (MMP), Final Report for the Great Barrier Reef Marine Park Authority, CSIRO, Australia.

Ninio R, Meekan M, Done T, Sweatman H (2000) Temporal patterns in coral assemblages on the Great Barrier Reef from local to large spatial scales. Marine Ecology Progress Series 194: 65-74.

Osborne K, Thompson AA, Cheal AJ, Emslie MJ, Johns KA, Jonker MJ, Logan M, Miller IR, Sweatman H. (2017) Delayed coral recovery in a warming ocean. Global Change Biology 23(9):3869-81.

Ryan DJ, Heyward A (2003) Improving precision of longitudinal ecological surveys using precisely defined observational units. Environmentrics 14: 283-293 DOI: 10.1002/env.586

Ryan DAJ (2004) Point sampling strategies for estimating coverage from benthic video transects. Environmetrics 15: 193-207 DOI: 10.1002/env.634

Zar J. H. (1984) *Biostatistical analysis*, second edition. Prentice-Hall, Inc., London.

# 9.0 Appendices

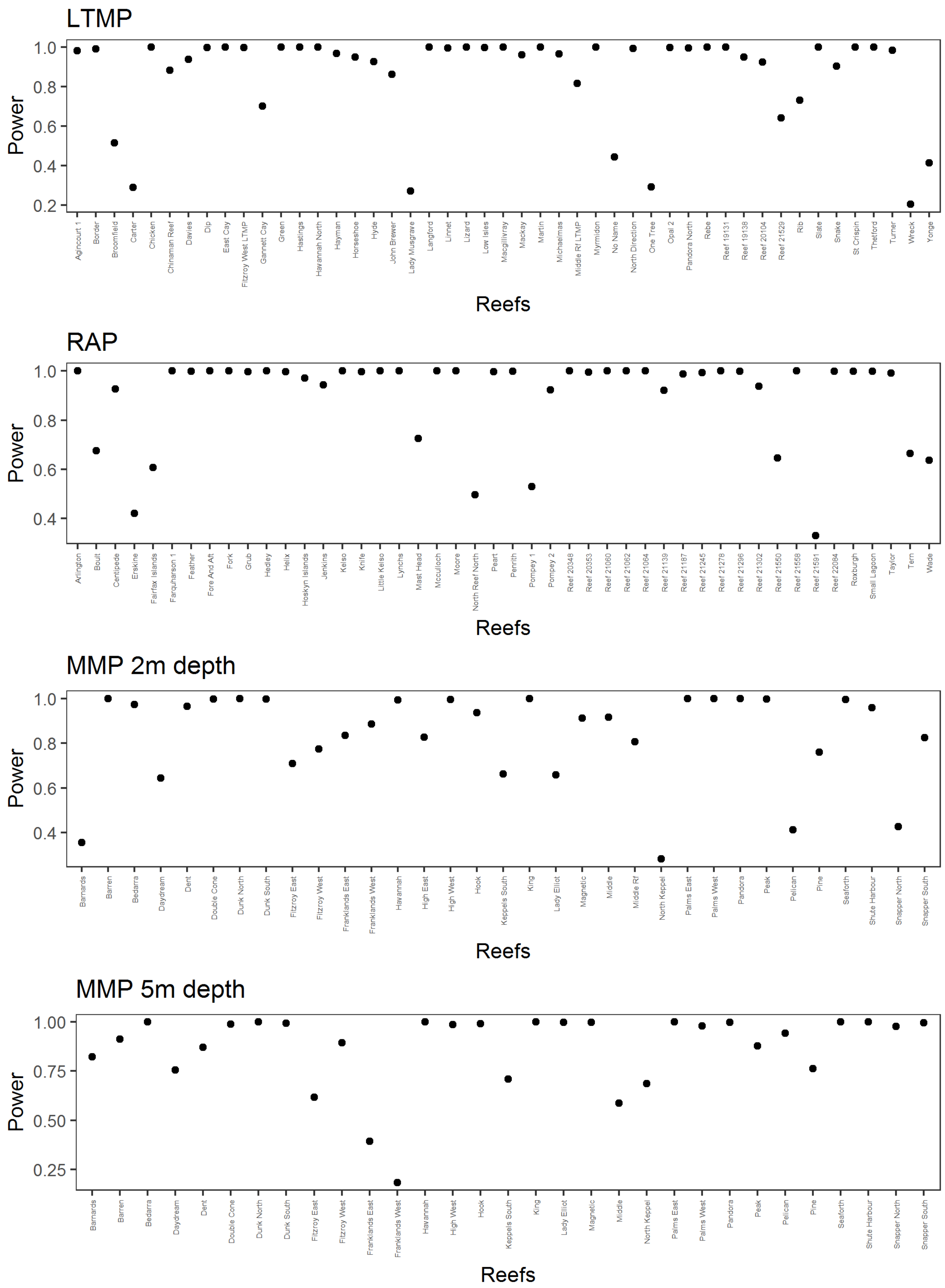

**Figure A.1. Power to detect trends of 1 per cent per annum change in hard coral cover over a 10-year period. For each program reefs are ordered along the y axis by mean coral cover. Power estimates are based on the full sample design of 3 sites ate each LTMP and RAP reef and 2 sites ate each depth for MMP.**

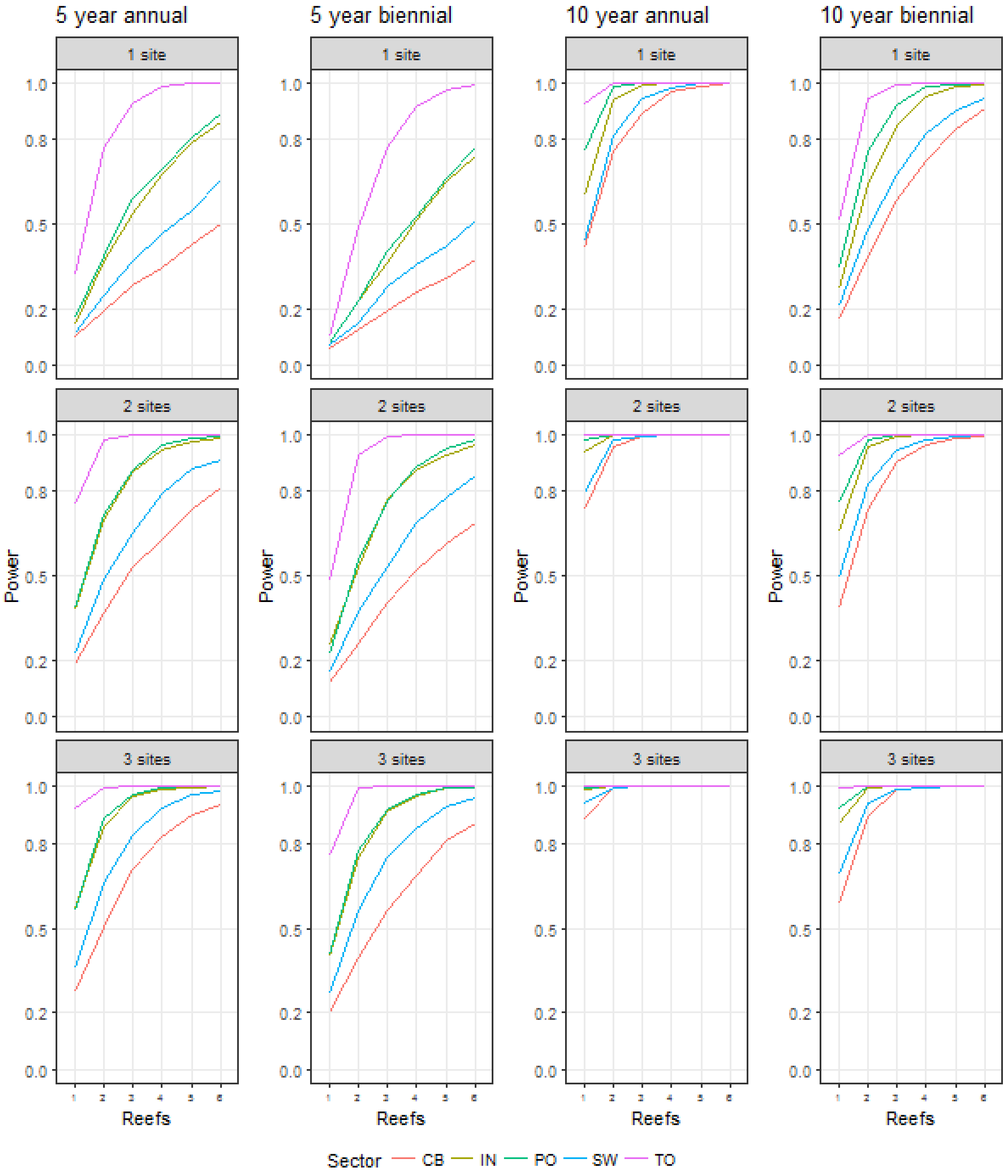

Figure A.2. Influence of number of reef on power to detect a trend of 2 per cent p.a. increase in hard coral cover. Estimates based on to mid-shelf reefs monitored by LTMP and RAP in each sector, indicated by colour. The exception was the Capricorn Bunker Sector (CB) where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.

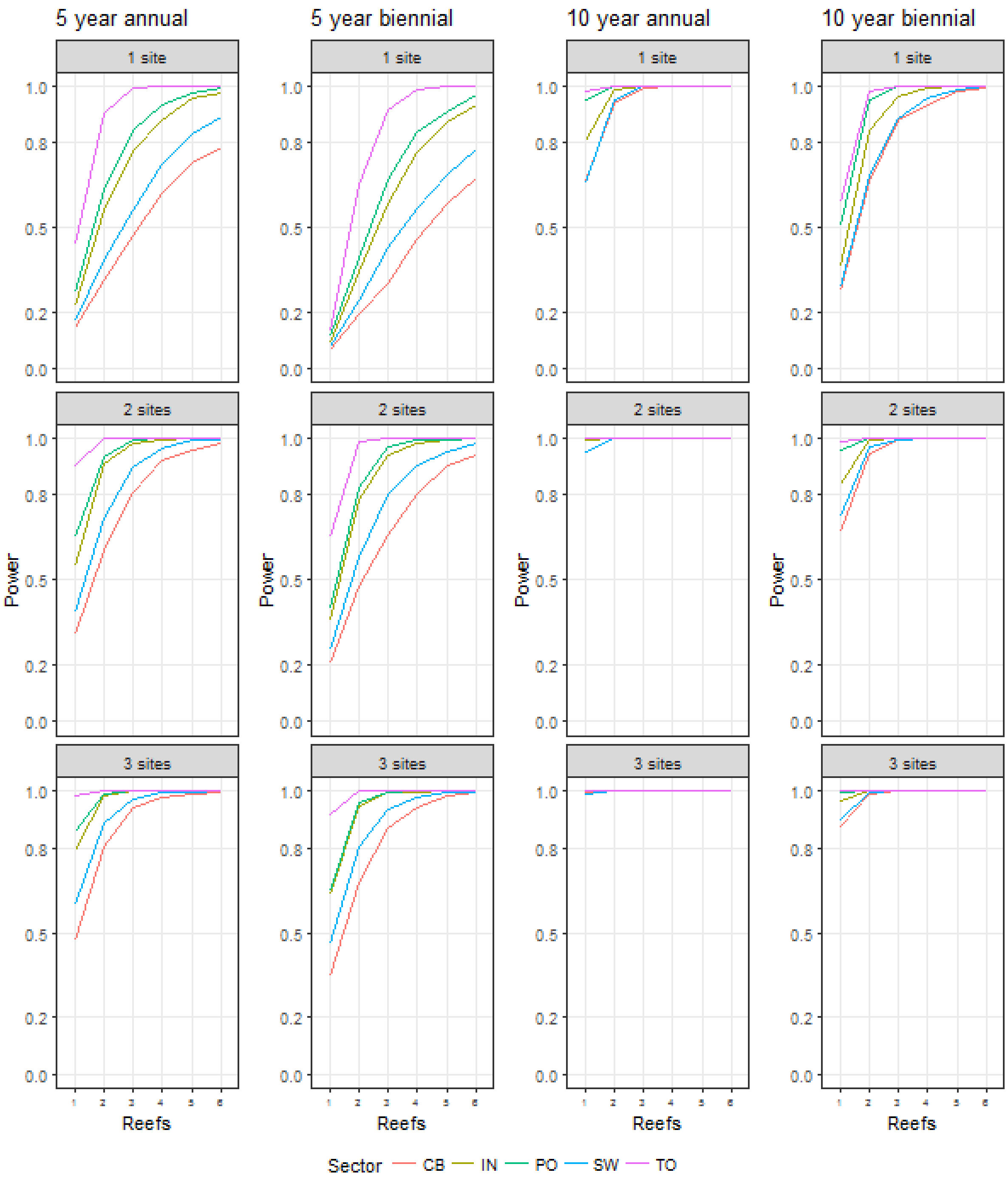

**Figure A.3. Influence of number of reef on power to detect a trend of 3 per cent p.a. increase in hard coral cover over a period of 5 and 10 years. Estimates based on to mid-shelf reefs monitored by LTMP and RAP in each sector, indicated by colour. The exception was the Capricorn Bunker Sector (CB) where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.**

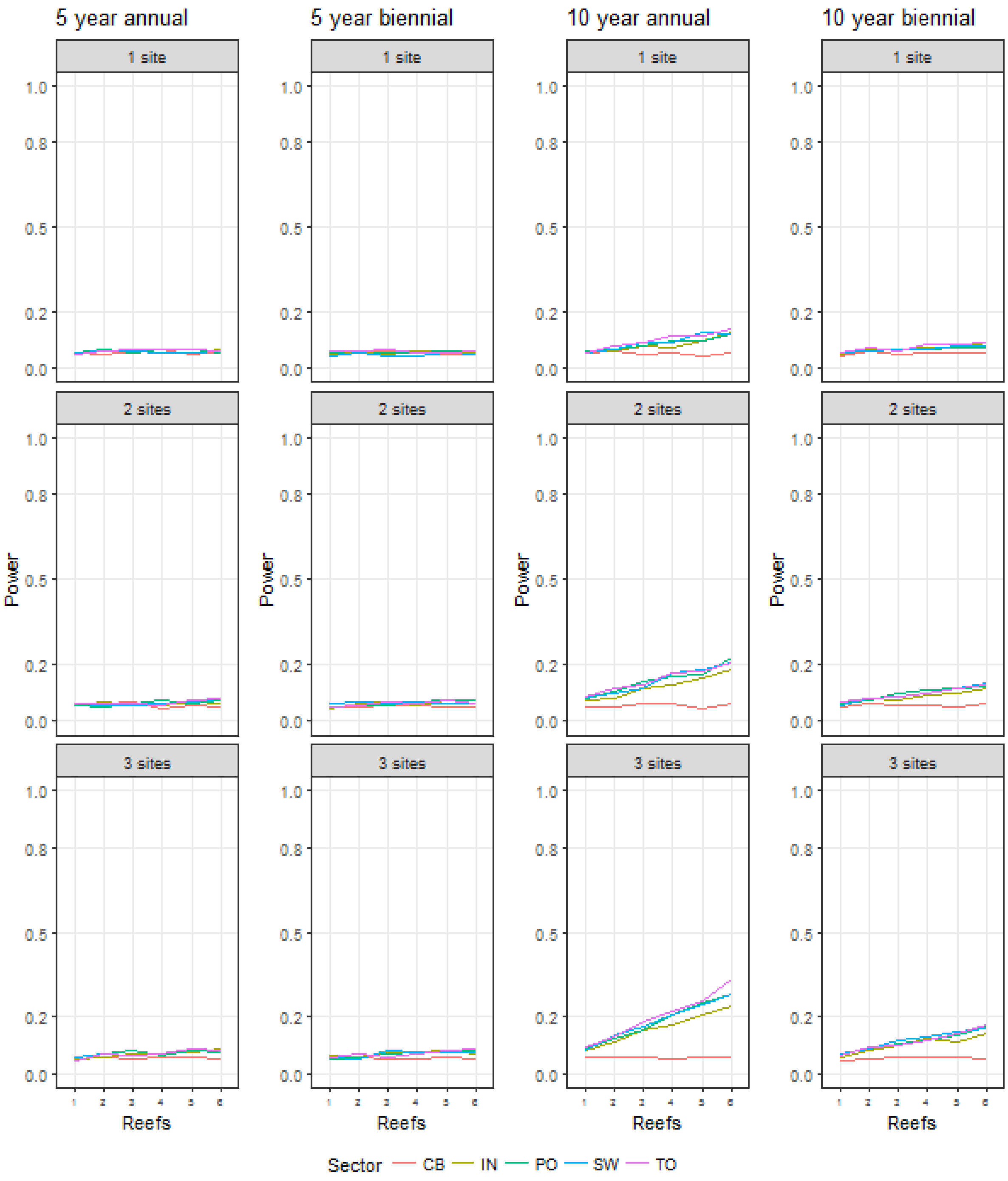

**Figure A.4. Influence of number of reefs on power to detect a change in fish species richness at a rate of 0.1 species per annum (plus or minus 1 species over 10 years). Estimates based on to mid-shelf reefs monitored by RAP in each sector, indicated by colour. The exception was the Capricorn Bunker Sector (CB) where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.**

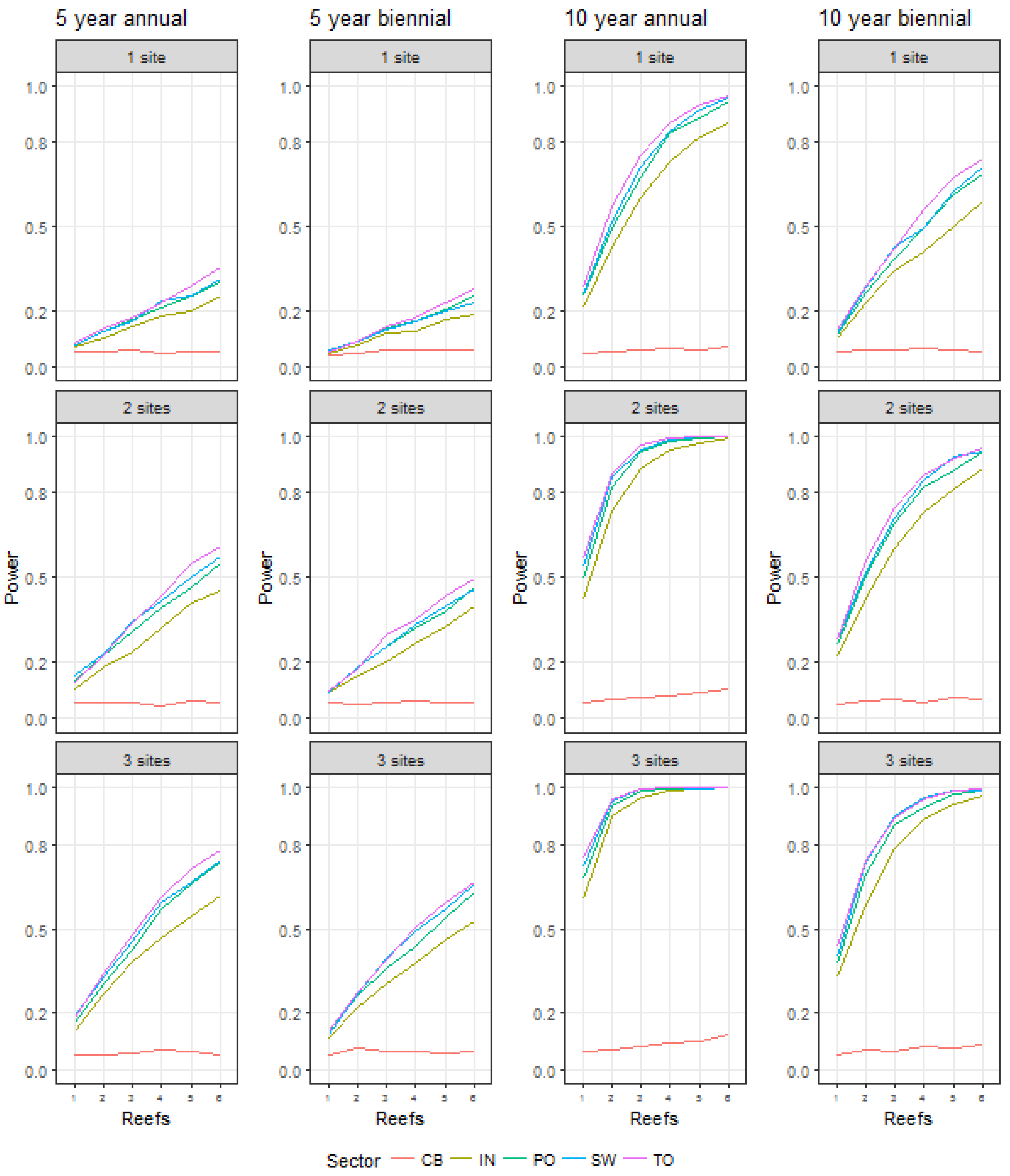

**Figure A.5. Influence of number of reefs on power to detect a change in fish species richness at a rate of 0.5 species per annum (plus or minus 5 species over 10 years). Estimates based on to mid-shelf reefs monitored by RAP in each sector, indicated by colour. The exception was the Capricorn Bunker Sector (CB) where outer-shelf reefs were used. Rows provide estimates based on monitoring of 1, 2 or 3 sites per reef. Columns group estimates by sampling period of 5 years or 10 years and frequency, annual or biennial.**