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Parks Victoria Technical Paper Series No. 54

Marine National Park and Sanctuary Monitoring Plan 2007-2012

Bernadette Power and Anthony Boxshall

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EXECUTIVE SUMMARY

Victoria's Marine National Parks and Marine Sanctuaries are established under the *National Parks Act (1975)* and were gazetted in November 2002. The main environmental aim within the legislation is to "…*maintain biodiversity and natural processes*…".

This report discusses the monitoring program that is built around that aim. The report is designed to be broadly accessible but it is necessarily scientific, technical and detailed. Technical jargon has been kept to a minimum but where it is necessary definitions have been provided.

The report has a number of basic sections. It begins with some background to the Victorian system of Marine Protected Areas (MPAs). We follow with a brief discussion of the environmental (also known as "natural") values of the parks and some of the threats to those values. This can be thought of as a conceptual model of the possible interactions between those threats and values in the MPAs.

We continue with a section on why it is important to monitor these environments and discuss some of the expectations of such a monitoring program.

The remainder (and bulk) of the report is a more detailed discussion of the components that currently make up the program, as well as changes proposed or postulated.

Victoria's Marine National Parks and Marine Sanctuaries encompass many different ecosystems made up of a diverse range of organisms living in a diverse range of habitats. These areas vary physically from soft, sandy bottoms to rocky reefs, shallow waters to depths of 110 metres, and from protected bays to high energy open coast environments with strong currents and large waves. The different habitats, communities and physical environments dictate a range of different monitoring strategies. The marine monitoring proposed in this report is designed to effectively monitor different components of the marine environment. The report covers the following habitats areas and monitoring techniques:

1. Subtidal reef monitoring – SCUBA diver-based surveys (known as the SRMP – or Subtidal Reef Monitoring Program)
2. Intertidal reef monitoring
3. Deep water video monitoring
4. Baited-video fish surveys
5. Soft sediment monitoring
6. Monitoring iconic taxa including seagrass, mangrove/saltmarsh communities as well as penguins and marine mammals
7. The water column

Not all of these ecosystems currently has a monitoring program, however the aim of this report is to indicate why and how such programs could (or should) be implemented in the future if the need arises. At any one time, it is not possible (or necessary) for a management agency to deliver all possible monitoring programs. The decision on which ones do go ahead is based on management priorities especially in relation to threats and need for understanding the ecosystems involved.

This report concentrates mainly on the SRMP as it is a long-running and detailed program with 14 locations sampled in some cases since 1998. In 2006, the SRMP was reviewed (see Keough et al 2007) for statistical power, capacity to detect changes across functional types and species, and temporal and spatial design in the monitoring program. The report highlighted a number of important innovations as well as the need to link monitoring programs explicitly to specific management objectives. This review, combined with the greatly increased habitat information available through the recently completed and extensive habitat mapping program done by Parks Victoria, has led to the development of this

monitoring plan. Detailed changes to the SRMP are proposed in this report including alterations to the physical design of the method, the amount and types of parameters and the spatial and temporal arrangement of the monitoring. All are designed to focus the program on management aims while increasing the statistical capacity of the program to detect changes that will be important for management decisions. The great majority of this report is based around the changes to the SRMP for Marine National Parks and Marine Sanctuaries, with detailed discussion of the rationale and the format of the future program.

There is a short discussion of the approach to future Intertidal monitoring. This is a program aimed at managing impacts from visitors on the animals and algae that live on intertidal rock platforms. Monitoring is currently done at nine locations; those were deemed to be the platforms with the highest visitation. The program is entering its 4^{th} year at most parks and as such it is timely to review the results, design and statistical power. This is planned for the period following the 2008 surveys.

Following the large and successful marine habitat mapping programs over recent years, far more detail is known about the deeper water areas of the parks (those beyond the safe diving limits for surveying). It is essential that future monitoring programs include ecological assessments of these locations. To assist the process, this report reviews some of the methods that are used to survey such areas and the animals and algae living there. The review includes towed video, remotely operated video, baited fish video surveys as well as drop video techniques. Some recommendations are made for future decision-makers to review.

Monitoring options for soft sediment habitats, the water column and iconic animals and plants are also discussed and the report finishes with a proposed monitoring schedule for the years 2007 to 2017 encompassing all tractable habitats as well as management needs.

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VICTORIA'S MARINE NATIONAL PARKS AND SANCTUARIES

Victoria's Marine National Parks and Marine Sanctuaries are established under the *National Parks Act (1975)* and were gazetted in November 2002 (see Figure 1 for a map of the system). The sites for the National Parks and Sanctuaries were chosen to be representative of the diversity of Victoria's marine environment (Environmental Conservation Council 2000) and the 24 parks are spread across Victoria's five Marine Bioregions (see Figure 2). The Marine National Parks and Sanctuaries are highly protected areas where no fishing, extractive or damaging activities are allowed but to which access is unrestricted. Recreation, tourism, education and research are encouraged. They are classified Category II (MNP) and III (MS) under the IUCN classification (IUCN 1994).

Figure 1 Map of the Victorian system of Marine National Parks and Marine Sanctuaries.

The vision for Victoria's Marine National Parks and Marine Sanctuaries system, outlined in Parks Victoria's management strategy, is "*to preserve the diversity of our marine environment, its flora and its fauna, its natural beauty, and the diversity of activities that will be found there*" (Parks Victoria 2003). The main environmental aim within the legislation that defines the parks is to "*…maintain biodiversity and natural process…*". The management strategy further affirms that the system "*has been established primarily to protect and conserve representative examples of biodiversity, ecological processes and natural values.*"

Figure 2 Victorian marine bioregions

In its broadest sense *biodiversity* encompasses the genetic variation within species, the number of species and their relative abundances, the composition of communities at species level and other taxonomic levels, the diversity of ecosystems and the processes that drive them (Lindenmayer and Burgman 2005). *Natural processes*, referred to in the legislation and vision statement, and encompassed within the definition of biodiversity, comprise the complex dynamics operating in ecosystems including processes such as primary production, nutrient and waste assimilation, recruitment and predator-prey and competitive interactions. Research suggests that by removing human impacts from marine systems (primarily fishing activities) these natural processes can be protected and in some cases restored (Castilla 1999, Pauly et al 2002). The "natural values" are defined as the parts of the environment valued by people and we consider this to be a proxy for biodiversity and natural processes for the purposes of this report. However, "beauty" is a more abstract notion that may need to be measured and monitored via methods available in social sciences.

CONCEPTUAL MODEL OF MARINE PROTECTED AREAS - VALUES AND THREATS

A very useful and commonly employed approach when considering any protected area is to conceptualise the important environmental attributes (or values) and consider what threats to them might exist (e.g., see Coastal Zone CRC 2006; Turner et al 2004). Parks Victoria uses this method for management and evaluation and we have followed this method for consideration of the monitoring needs. Figure 4 shows some of the environmental values, and threats to the values, in a generic Victorian MPA. This is not a complete conceptual model as the links between threats and values are not shown however it does consider almost all of the threats and values generated by a state-wide marine risk assessment process covering all MPAs (see Carey *et al* 2006, Carey *et al* 2007a & b).

Simple diagrams like Figure 4 are useful tool for visually representing the mixture of environmental values and possible threats to a generic Victorian marine protected area.

Park managers can use such a tool to target monitoring of environmental values to those perceived to be most at threat. This representation also provides a simple explanation of why some threats are directly managed to protect the natural values in the park.

Fundamentally, environmental management is about managing uncertainty. A challenge for park managers is to fill knowledge gaps about the distribution and functioning of natural values, including ecosystem processes (Chapter 3 - Parks Victoria 2007a). Figure 3 shows a summary of a basic conceptual tool used by Parks Victoria to guide park management and hence monitoring needs: the Parks Victoria Environment Management Framework (known as the “EMF”). Consistently implementing the EMF relies on the availability of objective, comparable and reliable information that improves the understanding of what natural values occur in parks, where they are located, the risks to them and the type and level of management intervention required for their long-term viability (Parks Victoria 2007a). Much of this basic approach is consistent with the frameworks outlined in Hockings et al (2000). Proposed monitoring in Marine National Parks and Marine Sanctuaries conforms to this structure.

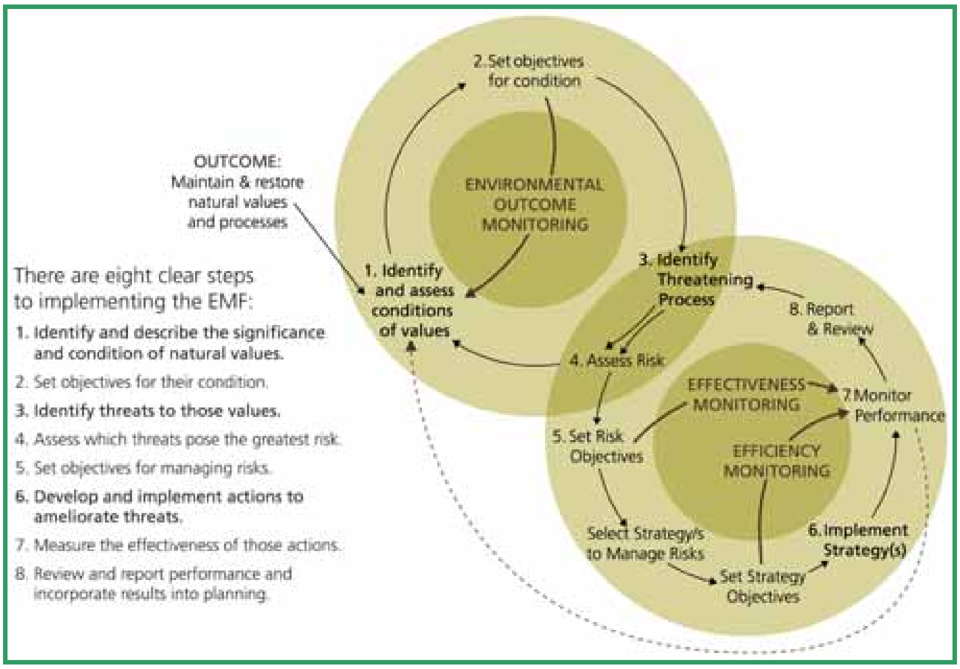

Figure 3 Parks Victoria Environment Management Framework. Image taken from State of the Parks Report (Parks Victoria 2007a)

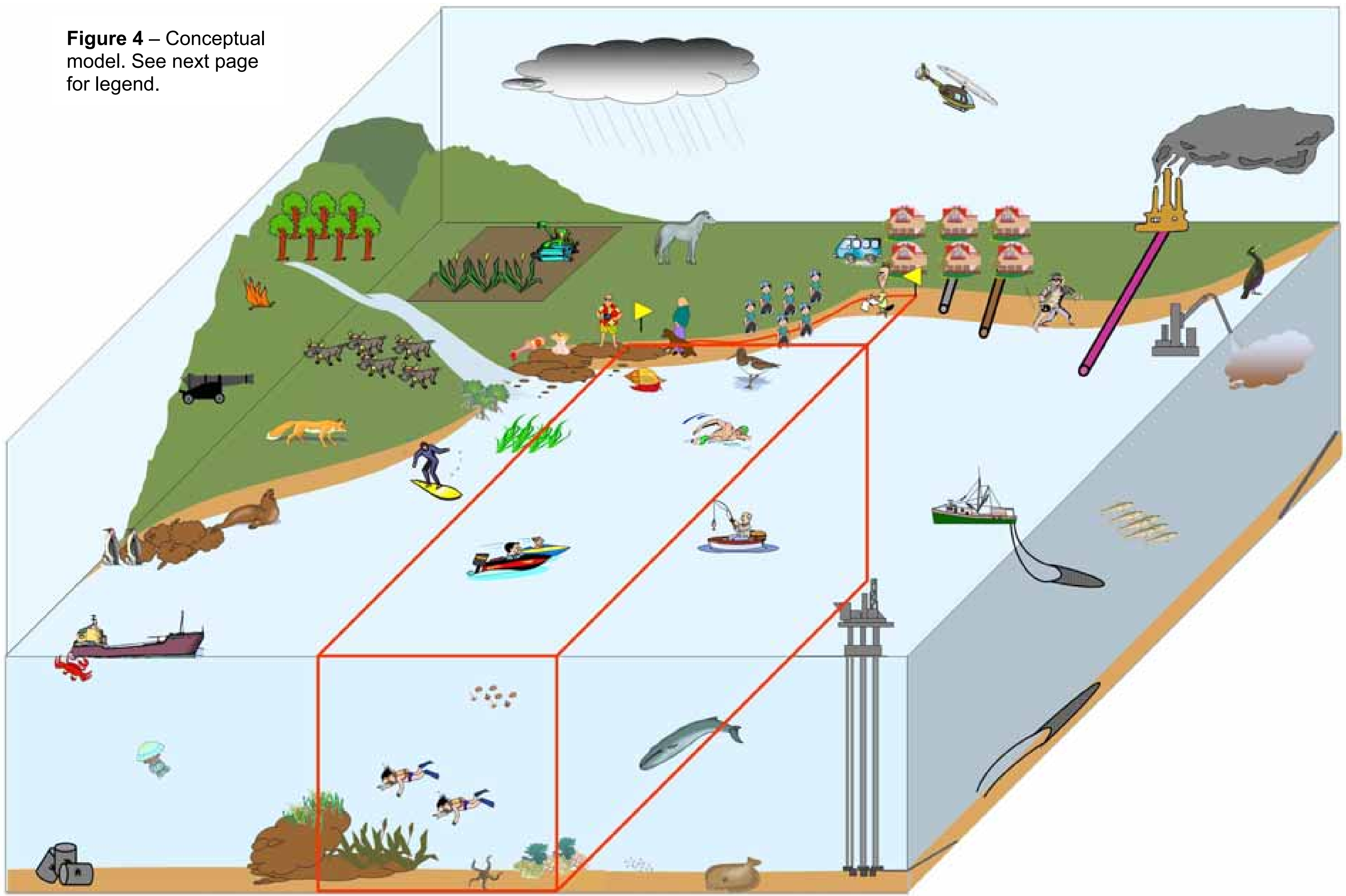

Figure 4 – Conceptual model. See next page for legend.

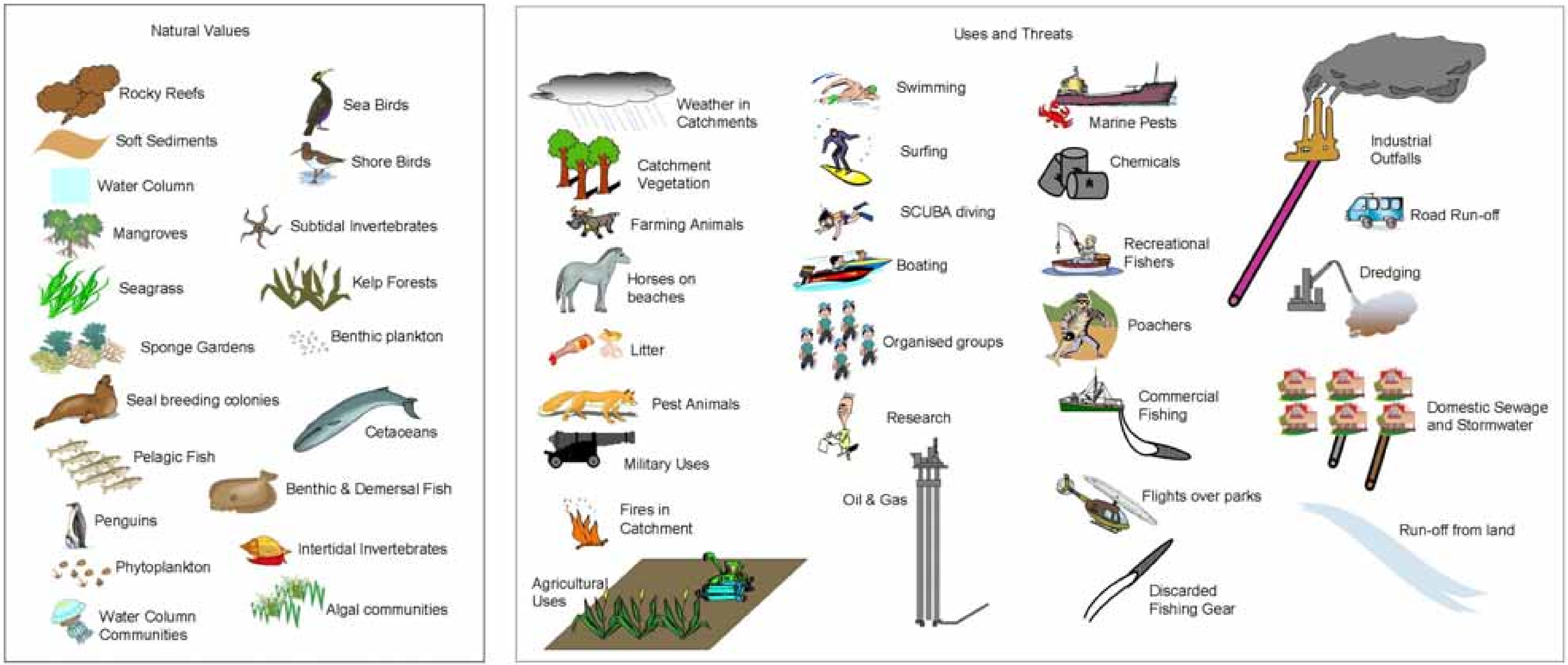

Figure 4 A conceptual model for a generic Victorian MPA. This information forms the basis for a conceptual ecological model. This image lists the environmental values, uses and threats in a generic Victorian Marine National Park and Marine Sanctuary. It is not a real park, but one that uses information from all parks drawn from Carey et al (2007a & b). Any final ecological model requires causal links to be drawn between values and threats where interactions exist as well as some indication of the strength and direction of the interaction. Park managers can use a model like this to target monitoring to those values most at risk from specific threats.

SECTION 1 - BASICS OF THE MONITORING PROGRAM

Why monitor the Marine Protected Areas?

Parks Victoria intends that environmental research and monitoring in the Victorian Marine National Parks and Sanctuaries should provide information on the status of natural values and threatening processes, determine the magnitude of trends through time, extend existing knowledge and develop an understanding of marine biodiversity within and outside the system. Monitoring should demonstrate a clear alignment with management objectives and be scientifically credible for their intended purpose (Parks Victoria 2003, Parks Victoria 2007b). Both broad-scale, long term monitoring programs and specific localized scientific research are referred to in the management strategy (Parks Victoria 2003); the monitoring strategy outlined in this document falls within the parameters of the broad-scale, long-term programs.

All monitoring at Parks Victoria is being integrated over the next 2-3 years into a broad program, known as “Signs of Healthy Parks” (SHP). This program is an integrated and adaptive monitoring program that aims to inform Parks Victoria of its management effectiveness by learning to read the signs of the environmental health of parks at a local and state level. It will be developed across all habitat types and multiple geographic locations and is proposed to roll-out in the near future (Parks Victoria 2007a). The proposed monitoring in this report links to that development.

The monitoring strategy outlined in this document aims to detect changes in biodiversity and natural processes inside marine parks relative to areas outside the parks known as “reference areas”. From a scientific perspective, these changes, if detected, would indicate that there has been a measurable effect of the presence of an MPA in a particular location. This can be used by Parks Victoria as a proxy for a more direct assessment of the ecological performance of the park and our effectiveness as managers. Any changes seen may be due to increases or decreases in species abundance and/or biodiversity inside the park relative to reference areas outside. In the situation where no difference is detected between the MPA and the reference areas it is important that we can distinguish between the two possible reasons for this. One possible reason is that there is no important difference and the other is that despite a real and important difference occurring, the sampling program did not have the power to detect this difference.

This approach may also allow for the detection of common changes in both MPAs and reference areas over time, such as would be expected in response to external local threats or regional (e.g., the “sea change” phenomenon) and global threatening processes (e.g., climate change). These programs are not expressly designed to detect changes due to these threats and hence the monitoring data may not show any causality between the impacts and these threats. However, if able to detect these types of large-scale changes, data from these programs could be used to target more specific monitoring or adaptive management programs based around those threats.

Expectations of MPA Monitoring: What will it tell us?

Protecting Marine National Parks and Sanctuaries from fishing, extractive processes and damaging activities has removed many of the localised negative impacts of human activity on biodiversity and natural process. It is expected that this will assist in maintaining (or enhancing) biodiversity and natural process in these areas. The primary goal of the monitoring program, therefore, is to quantitatively assess the effect MPAs have on maintaining biodiversity and natural processes.

This primary goal is achieved through ongoing, structured monitoring of environmental indicators inside the MPAs and at matched reference locations outside. We cannot measure all aspects of biodiversity, so instead we measure indicators that are a subset of all possible things we could measure (see Pomoroy et al 2004 for some indictors used in MPAs). The indicators (or parameters) chosen include things like number of species, abundance of organisms and size of organisms. Due to its broad definition we cannot hope to measure all biodiversity, hence we can only ever measure these parameters for a small subgroup of all species in a park and the species to be measured are determined by practical and ecological criteria (Keough and Mapstone 1995a). Analysis for differences in these parameters between inside and outside the MPAs allows us to test what effect the MPAs have on biodiversity (or at least the proxy of biodiversity that we have chosen). By monitoring species from different trophic levels (i.e., levels in the food chain) from high level carnivores to primary producers such as algae, changes in ecological processes due to the Marine National Parks and Sanctuaries may be detected in the form of changes in the relative numbers of these groups (Castilla 1999). Two fundamental questions can be asked of the data collected in this monitoring: 1) is there an important difference between inside and outside the MPA (known as a change that can be measured "spatially") and, 2) after data has been collected over a period of time, is that difference becoming larger, smaller or staying the same over time (known as a difference that can be measured "temporally")? These answers should tell us whether each MPA is achieving its intended aims of maintaining (or enhancing) biodiversity and natural processes.

Spatial Changes

It is important to measure exactly the same things and in the same way inside and outside the parks. The park and reference sites are used to allow an assessment of changes in the park compared with what is occurring in the rest of the coastal waters. Aside from the issue of inadequate statistical power (discussed later in the report), from a biodiversity perspective, there are three possible basic outcomes: 1) biodiversity parameters could increase in the park compared to the outside reference site, 2) biodiversity parameters could decrease in the park or 3) the parameters being measured could show that there is no difference inside and outside the park. These possible outcomes are shown in a hypothetical graph in Figure 5)

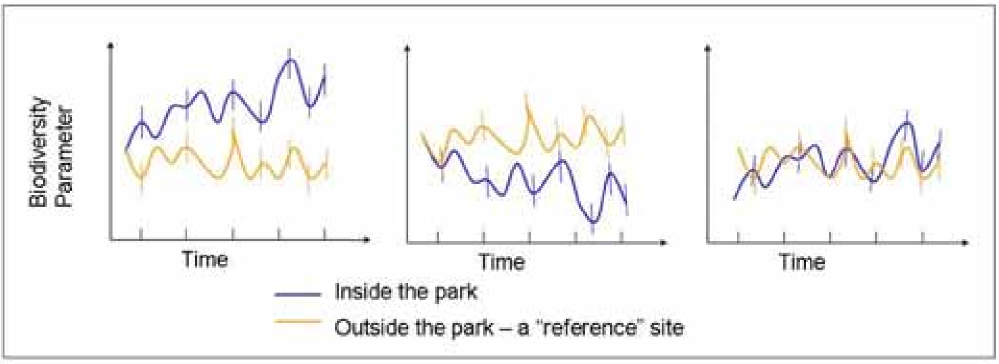

Figure 5 Three hypothetical graphs showing the 3 different outcomes described above in a biodiversity parameter over time measured inside and outside the park boundaries. The short vertical lines indicate the error (or variability) around each individual survey. This graph is for illustrative purposes only and does not represent actual data.

It is important to understand, however, that although some indicators of biodiversity might increase, others might decrease. Both are natural and to be expected and this is where the overall aim of "…*maintaining natural processes*.." is important. It is difficult to define a precise and easy-to-use measure which estimates the success (or otherwise) of maintaining natural processes with current scientific knowledge and technologies. Estimates that are measurable include recruitment, trophic interactions, inter- and intra-species competition, predation (see Pomoroy et al 2004 for some examples). It is, however, possible to explain how biodiversity and natural processes might interact with a simple example using fish (see Figure 6).

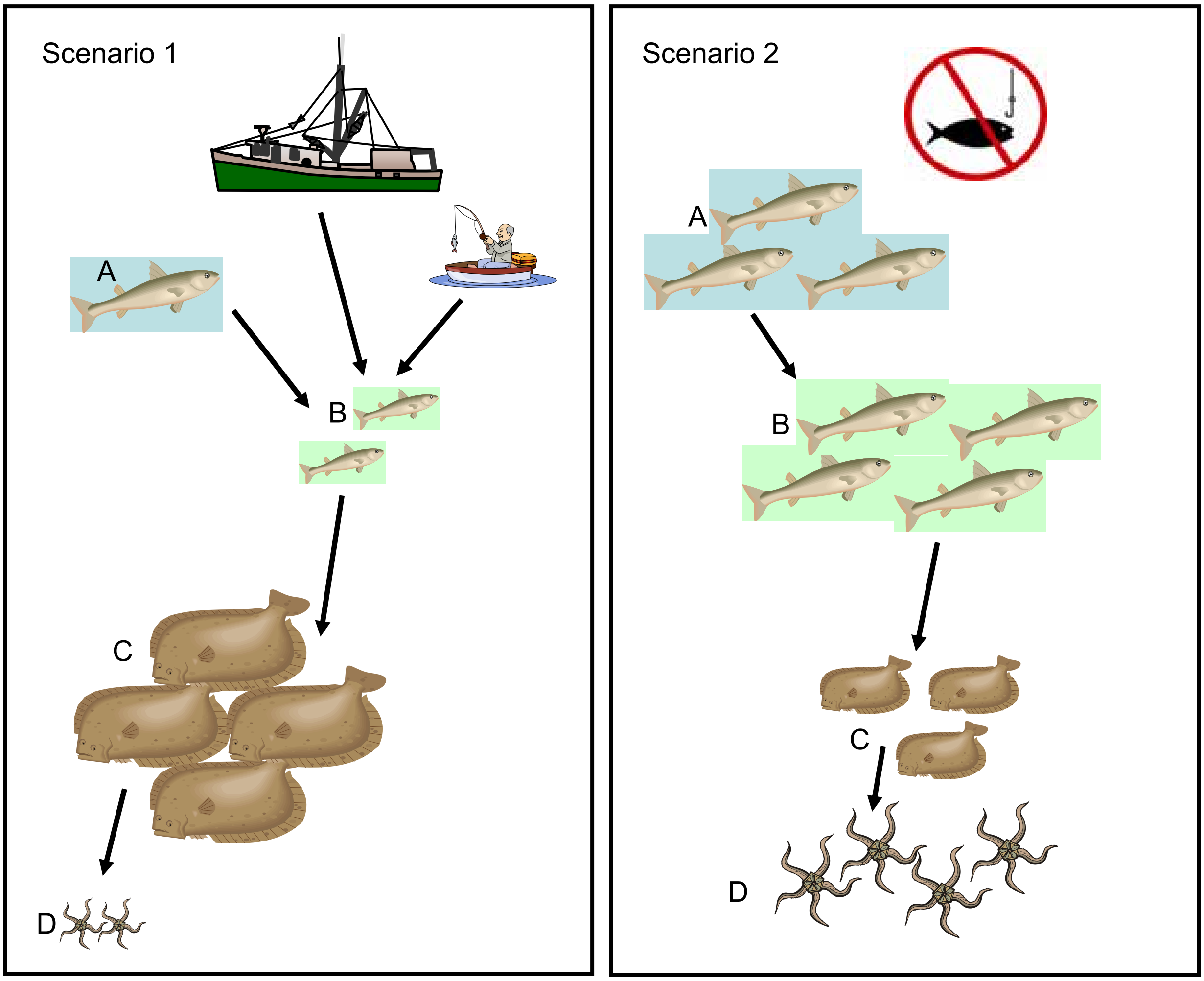

Figure 6 – Two management scenarios that produce different abundances of fish and invertebrates when monitored, but where animals in both cases increase in abundance or size of animals are seen in both cases. In scenario 1, fish B is targeted by commercial and recreational fishers thus reducing numbers and size of fish. Fish A eats B, but as there is less prey, its numbers decrease too. Fish C is eaten by B, but increases in size and number without an active predator having a flow-on impact on seastar D, its favourite prey item. In scenario 2, without fishing, the numbers and size of fish B increase as do those of fish A, but fish C has a decrease in numbers and size due the there being more of its predator around. However seastar D has an increase in numbers and size. Scenario 2 has less direct human impact so despite some fish numbers and sizes being less, it could be thought to be a more "natural" process.

Temporal Changes

In some cases data exists from the period before the parks were declared and for these parks it is possible to assert that any differences detected between inside and outside that park after declaration are the result of protection afforded by the MPA. The example of changes in fish and invertebrate size illustrates how such a difference might result from protection in an MPA (see Figure 6). Where no data from before declaration exists we cannot unequivocally attribute differences between inside and outside to the effect of the MPA, but clear inferences can be made if there is a consistent trend over time (Downes et al. 2002).

Other Outcomes

It is expected that monitoring of MPAs will also enable an identification of threatening processes and their magnitude (Barrett and Buxton 2002, Parks Victoria 2003). Insofar as such threats cause changes which are considered important to biodiversity and natural processes within the MPAs and/or reference areas, their effects should be detected over time. However, as the magnitude, timing and location of threats is unpredictable planning a monitoring system that can rigorously detect and quantify all of their effects and establish causal links is not logistically and economically viable. From a management perspective, where specific threats exist of a known intensity and extent, it may be appropriate to develop targeted monitoring program or an adaptive experimental management response to the threat.

Parameters (indicators) to be measured

This program is an integrated and adaptive monitoring program that aims to inform Parks Victoria of its management effectiveness by learning to read the signs of the environmental health of parks at a local and state level. It will be developed across all habitat types and multiple geographic locations and is proposed to roll-out in the near future (Parks Victoria 2007a). The program is likely to involve three types of indicators: environmental, threat and efficiency indicators (see Figure 3). The marine monitoring program primarily is concerned with the first two of these as the third is related to management activities and is beyond the scope of this report. The third type of indicator is discussed in the latest State of the Parks Report (Parks Victoria 2007a). Many environmental indicators exist for marine systems (e.g., Pomoroy et al 2004); however those considered in this report include measurements of selected species extent, (e.g., area covered by sea grass), species number (diversity), number of individuals of selected species, size of individuals of selected species, rates of recruitment of new individuals into a population or interactions between selected species in marine communities. These indicators will be primarily used for comparison with reference areas to measure the performance of the Marine Protected Area in meeting its objectives. They also have the potential to be used as response indicators to specific threats such as introduced marine pests, impacts from outfalls or catchment-derived pollutants. Further details of the specific parameters measured are discussed in the sections of this report detailing each individual monitoring approach.

SECTION 2 - COMPONENTS OF A VICTORIAN MPA MONITORING PROGRAM

Background

Victoria's Marine National Parks and Marine Sanctuaries encompass many different ecosystems made up of a large range of organisms living in a diverse range of habitats. These areas vary physically from soft, sandy bottoms to rocky reefs, shallow waters to depths of up to 110 metres, and from protected bays to high energy open coast environments with strong currents and large waves. These different habitats, communities and physical environments dictate the need for a range of different monitoring strategies. The proposed marine monitoring program is discussed in the following seven sections designed to effectively assess different components of the marine environment:

1. Subtidal reef monitoring – SCUBA diver-based surveys (known as the SRMP – or Subtidal Reef Monitoring Program)
2. Intertidal reef monitoring
3. Deep water video monitoring
4. Baited-video fish surveys
5. Soft sediment monitoring
6. Monitoring iconic taxa including seagrass, mangrove/saltmarsh communities as well as penguins and marine mammals
7. The water column

Not all of these ecosystems currently has a monitoring program, however the aim of this report is to indicate why and how such programs could (or should) be implemented in the future if the need arises. At any one time, it is not possible (or necessary) for a management agency to deliver all possible programs. Those that are delivered are done so based on management priorities especially in relation to threats, and the need to understand the ecosystems involved.

2.1 Subtidal Reef Monitoring – Diver based surveys (SRMP)

Historically known as the SRMP – the Subtidal Reef Monitoring Program, this monitoring has been in operation since before the MPAs were declared in 2002. Originally created by the then Department of Natural Resource and Environment (DNRE) in 1998 and expanded by Parks Victoria in 2003 (Edmunds & Hart 2003), monitoring began in four locations and has been progressively rolled out to incorporate all MPAs with accessible reef (currently 14 locations). The program involves diver-mediated visual census of fish, mobile invertebrates and algae associated with shallow rocky subtidal reef. A standard operating procedure is available (Edmunds & Hart 2003) and many reports exist in the Parks Victoria Technical Series detailing monitoring to date at 13 parks (and 1 site outside a park) across Victoria (examples of some of the latest reports are Gilmour et al 2006, Crozier et al 2007, Gilmour et al 2007, Edmunds et al 2007 & Williams et al 2007). The current locations are the Cape Howe, Point Hicks, Wilsons Promontory, Bunurong, Port Phillip Heads and Point Addis Marine National Parks, the Beware Reef, Ricketts Point, Jawbone, Point Cooke, Eagle Rock, Marengo Reefs and Merri Marine Sanctuaries and a site to the south of Phillip Island.

The SRMP monitors a specific suite of fish associated with reefs in shallow waters, i.e., in reefs in less than 10m depth (Edmunds & Hart 2003). It is not designed to assess cryptic or non-reef associated shallow water fish nor is it designed to assess the suite of species found in deeper water. It has been shown to target one subset of all available fish species (Harvey et al 2001). To effectively monitor these fish, different methods are required (Section 6 & 7).

Following an independent expert review (Keough et al 2007) and a subsequent internal review of the data collected since monitoring began in 1998[1], the monitoring program has now been redesigned building on the existing SRMP (see Section 4). The redesign process considered the recommendations of the independent review, and implemented them where possible, and included further analysis of existing data for individual MPAs to assess how the monitoring program could be made more powerful and cost effective. The redesign also took into account the extensive new information available from a recent large recent marine habitat mapping program (see Ball et al 2006, Ball & Blake 2007 a & b, and Holmes et al 2007a, b, c, d & e). New individual monitoring plans have been developed for each MPA where the SRMP is applied. The rationale for the new design and the plans for each MPA are presented in Section 4 in this report.

2.2 The Intertidal Reef Monitoring Program

To assist with the management of intertidal reefs, Parks Victoria established a long-term intertidal reef monitoring program in 2002-03 with the explicit aim of tracking changes due to human use of the platforms. The uses targeted by the monitoring were trampling and fossicking, which occur to a greater extent in high visitation parks. Intertidal platforms in Victoria have been the site of much study into the impacts from trampling on the diversity and status of intertidal organisms (e.g., Keough & Quinn 1998).

The program began in late 2002 with sampling at a number of sites in 2002-2003. The initial method used prior to 2003 was reviewed by independent experts and considered not fit for the proposed purpose. A new method was developed in consultation with the same independent experts and implemented by contractors in the summer of 2003-2004. Essentially, the method involves monitoring the invertebrates and macro-algae that live in the intertidal zone. Reefs within nine targeted MPAs as well as those in nine matched references areas outside the MPAs were monitored (see Hart and Edmunds 2005 for full details of the monitoring design).

This program will continue in its current form until a major review of the data and results after the summer of 2007-2008. See Section 5 in this report for more details.

2.3 Deep Water Video Monitoring

A program of video monitoring for deep reef communities will be developed for the purposes of monitoring MPAs in waters beyond safe diving limits (> 10 m). Parks Victoria does not currently have a program of ongoing monitoring for these areas; the well-established, diver-mediated visual survey methods are not possible at these depths and in the past other monitoring options have presented a suite of logistical and economic challenges. Nevertheless, these deep areas comprise a large proportion of Victoria's MPAs and it is important that a monitoring program be implemented for them. With the information gained through the recent habitat mapping program see Ball et al 2006, Ball & Blake 2007 a & b,

[1] Some data was collected inside and outside of what are now MPA's in the years leading up to declaration of the MPAs in 2002.

and Holmes et al 2007a, b, c, d & e), it is now possible to target this deep water video monitoring far more tightly than was previously possible. The availability of information from the habitat mapping program, combined with the fact that technology and methods for remote video monitoring of deep water habitats have become both more effective and cheaper to use, mean that remote monitoring of deep water habitats is now a viable option. It is therefore proposed that remote video be used to monitor deeper water habitats beyond safe diver survey limits. A review of deep water video monitoring has been completed as the first stage in implementing the method in deep reef areas in Victorian MPAs (see Section 6 in this report).

2.4 Baited Video Fish Monitoring (BRUVS)

There is substantial evidence in the scientific literature (Harvey et al 2001 and 2004) that only a subset of the fish species in an area is monitored using diver-mediated visual surveys (such as the SRMP) or other active monitoring techniques (such as the methods proposed for the Deep Reef Video Monitoring - above). Baited video monitoring (known as BRUVS) can be used to specifically target some of the fish species not effectively monitored by the other methods. BRUVS could be used to monitor fish in areas inaccessible to the SRMP and other methods (such as very complex high relief reef in deep water) and soft sediment areas where the SRMP and other methods do not apply. A review of baited video monitoring methods has been completed as the first stage of implementing the method in Victorian MPAs. See Section 7 for more detail.

2.5 Soft Sediment Monitoring

Soft sediment habitats in Victoria's Marine Protected Areas encompass areas where the sea floor is silt, sand or mud with no obvious epibenthic biota (that is, flora and fauna that lives on the top of the sediment). The habitats in some parks, such as those in the Yaringa, French Island, Churchill Island, 90 Mile Beach and Corner Inlet Marine National Parks are virtually completely made up of soft sediments. However there are areas of soft sediments habitats and communities in every MPA in Victoria.

Parks Victoria does not currently have a program of ongoing monitoring for these areas but has recently had a large historical dataset expanded and re-analysed Parry et al (in press). See Section 8 for more detail.

2.6 Monitoring Iconic Taxa

For the purposes of this report, we consider iconic taxa to be seagrass, saltmarsh, mangroves, seabirds, cetaceans (whales and dolphins), turtles and pinnepeds (seals). There is currently no structured program to assess or monitor ecological parameters for iconic taxa in Victorian MPAs. See Section 9 for more details.

2.7 The Water Column

The three dimensional nature of the sea results in the water column being an important habitat for many organisms, both for in transit or those who use it as a location for a permanent home. Some organisms (both flora and fauna) make their permanent home in the water column (e.g., sea jellies, salps, many fish, and much plankton). Others live on and near hard surfaces but use the water column for transport and food (and other resources like oxygen). Parks Victoria does not currently monitoring the water column as a habitat. See Section 10 for more details.

SECTION 3 - DETAILS OF THE MONITORING PROGRAMS

As previously noted, the bulk of this report deals with changes to the subtidal reef monitoring program (the SRMP) following a large recent review (Keough et al 2007). Other habitats are covered in the report but not in as much detail as subtidal reefs.

SUBTIDAL REEF MONITORING (SRMP)

Shallow subtidal reefs are considered important habitats to monitor ((Edgar and Barrett 1997, Edgar and Barrett 1999). These reefs are known for their high biological complexity, species diversity and productivity and in addition they have significant economic value through commercial and recreational fishing (outside of MPAs), diving and other tourism activities. It is also considered that as many reef-associated species are "attached" to the site they may be useful indicators of the performance of MPAs. This is in contrast to most open water and soft-bottom fishes, for example, which are highly mobile and may pass though a site before MPA management practices can have an effect (Stewart et al. 2006, Edmunds & Hart 2004).

The existing monitoring program was reviewed by independent experts from the Zoology Department, University of Melbourne (Keough et al 2007). The authors examined data from Bunurong and Wilson's Promontory Marine National Parks for which a long time series existed dating back to before these areas were declared National Parks. The review estimated the statistical ability of the existing program to detect meaningful biological changes and evaluated the effects of a range of different design options.

3.1 Current SRMP protocol

The current SRMP is based on the visual census methods of Edgar-Barrett (Edgar and Barrett 1997, Edgar and Barrett 1999). These procedures are detailed in the Parks Victoria Standard Operating Procedure, Biological Monitoring of Intertidal Reefs (Edmunds et al. 2005).

The language used when describing components of a monitoring program vary among authors and to avoid confusion we will define some of the terms used for this program. A site is an independent replicate area from which one or more subsamples may be taken. It should be spatially distinct from other sites and although the notion of what constitutes spatial independence is a contentious one (Downes et al. 2002) we use the guideline given in (Keough and Mapstone 1995a) that the buffer zone among replicate sites be at least as large as the sites themselves. For the single contiguous quadrats currently used in this program this would mean ensuring there was a least 220 m between transect lines for parallel sites and at least 200 m between transect lines that were end to end. A quadrat is a spatially distinct area within a site from which a subsample is taken, where measurements are made from multiple quadrats within a site they are pooled or averaged before analysis so they serve to better estimate the value of the variable at a site but don't add to the replication of the design. A transect is a linear strip (usually a measuring tape) laid out to allow spatial positioning of quadrats or other sampling tools at a site.

At most monitoring locations in Victoria, the 5 m depth contour is considered optimum for visual monitoring because dive times are not limited by decompression schedules. Many of the reefs in the Marine National Parks and Marine Sanctuaries are only present in shallower or deeper water. The actual depths used for site vary between 3 and 12 m depth.

Each site is located using differential GPS and marked with a buoy. A 100 m numbered and weighted transect line is run along the appropriate depth contour either side of the central marker. The resulting 200 m of line is divided into four contiguous 50 m sections of transect

(T1 to T4). Where possible the orientation of transect is the same for each survey, with T1 toward the north or east (i.e. anticlockwise along the open coast). However, depending upon the size, shape and orientation of available reef this can vary. The actual position of the current transects is maintained as a set of spatial layers in the Parks Victoria data system along with detailed site notes explaining the physical characteristics of each site.

For each transect line, four different census methods are used, involving census of the:

(1) abundance and size structure of large fishes,

(2) abundance of cryptic fishes and benthic invertebrates,

(3) percent cover of macroalgae and

(4) density of string-kelp Macrocystis plants.

The specifics of these methods are as follows (see Edmunds et al. 2005 for more details):

1) Mobile Fishes and Cephalopods: The densities of mobile large fishes and cephalopods are estimated visually by a diver within each of four contiguous 10 m x 50 m sections located along the transect line. The diver does this by swimming up one side of each 50 m transect section and then back along the other. The diver records the number and estimated size-class of fish.

2) Invertebrates and Cryptic Fishes: Cryptic fishes and large non-sessile invertebrates (e.g. large molluscs, echinoderms and crustaceans) are counted within each of four contiguous 1 m x 50 m sections located along the transect line. The maximum length of abalone and the carapace length and sex of rock lobsters are also measured whenever possible.

3) Macroalgae: The area covered by macroalgal species is quantified by placing a 0.25 m^2 quadrat at 10 m intervals along the transect line and determining the percent cover of the all identifiable plant and macroalgae within the quadrat. These yield twenty 25 m^2 quadrats along each 200 metre transect line at each site.

4) Macrocystis: In addition to macroalgal cover, the density of *Macrocystis angustifolia* plants is estimated by divers counting all observable plants within 20 * 10 m^2 contiguous quadrats along the transect line.

3.2 Power in monitoring program design

Why Statistical Power Matters

Because a monitoring program can only sample (count or measure a small sub-set of the organisms) from inside and outside an MPA, statistical tests are conducted on the data collected to establish the likelihood that any differences in the samples from inside and outside the MPA reflect real differences in population parameters, rather than just sampling error[2]. These statistical significance tests allow us to make objective judgements as to when we are confident (usually 95% confident) that we have detected a real (statistically significant) difference between inside and outside. These tests allow us to be confident that we are not inferring a difference between inside and outside when no such difference exists (known as a type I error).

There is, however, another type of error that can be made when statistically assessing the results of a monitoring program; we can conclude that no difference exists between inside and outside a park when in fact one does exist (known as a type II error). This type of error can occur if our sample size is too small or the populations we are sampling from are too variable. In order to protect against this type of error we need to assess the statistical power of our sampling program to detect a difference between sample populations, should such a difference exist. Type II errors have important implications for management of Marine Parks and Sanctuaries and it is important that monitoring programs are transparent about the likelihood of such errors occurring. Statistical power is equal to 1- the probability of making a type II error. Thus, achieving high power in a monitoring design means having a low, long-run likelihood of concluding that there has been no important effect, when in fact there actually has been an important effect but we did not detect it.

To determine the power of a monitoring design we need to have an explicit plan for how the results will be statistically tested (i.e., a statistical model). We also need to define what we consider to be an important difference (the effect size), what we consider an acceptable chance of a type II error (β) (typically set at 0.2) and what we consider an acceptable chance of a type I error (α) (typically set at 0.05). Finally, we need an estimate of the variation in the parameters of interest in the populations we plan to sample. From this information we can estimate how many samples (n) are needed for the monitoring program to have an acceptable likelihood of detecting a meaningful difference should such a difference exist.

In the independent review of the SRMP by experts from the Zoology Department, University of Melbourne (Keough et al 2007), the authors examined data from Bunurong and Wilson's Promontory Marine National Parks for which a long time series existed dating back to before these areas were declared National Parks. The review estimated the statistical power of the monitoring designs to detect a range of effect sizes from 20% to 200%, evaluated the effects of reduced sampling effort, and of modified sampling design to reduce background noise, determined the most cost effective design that included the optimal combination of spatial and temporal sampling to maximise the power of the SRMP, considered the relationship

[2] See chapters 1 and 2 in (Keough and Quinn 2002) for a discussion of sampling error and statistical hypothesis testing.

between existing parameters to determine whether any were redundant and examined the potential of alternative parameters to those currently measured.

Two sampling designs were considered by the reviewers, the first was a snap shot design in which samples are taken inside and outside of an MPA at a selected time after park declaration and the difference between the two sites assessed by ANOVA (*Analysis of Variance*), the other was a BACI (*Before - After - Control – Impact)* design in which MPAs are assessed at around the time of declaration and then again at a specified time after. They found that the existing monitoring program had low power to detect changes of < 100% in species abundance for most species but acceptable power to detect changes of 200%.

3.3 Statistical model for the Subtidal Reef Monitoring Program

It is important in a monitoring program that the design, aims and any proposed statistical models be explicitly defined. This section makes such information explicit and is divided into models and approaches for Marine National Parks and Marine Sanctuaries.

Marine National Parks

Based on the recent statistical review of the SRMP (Keough et al 2007), the reviewed SRMP is designed to have two questions asked of monitoring data collected from MPAs. The first question is whether, at a particular instant in time (known as a snap shot), there is a difference between samples taken inside and outside the MPA for a given variable (for example, numbers of fish, invertebrate, alga or a measure of marine community). The statistical test used is a one way ANOVA (*Analysis of Variance*). The null hypothesis is that there is no significant difference between samples taken inside and outside the park and the alternative hypothesis that there is a significant difference.

The second question is whether there is a change over time comparing the samples taken inside the MPA relative to the reference area. One of the analyses to be used is a “BACI” (*Before - After - Control – Impact)* analysis. Originally designed for environmental impact assessment studies, a BACI design refers to a test involving data from before and after an “impact”. In this case the “impact” is not really an impact, rather the declaration of the MPAs. One statistical test for this question is an mBACI analysis (which means: *modified Before - After - Control – Impact.* A second test used for a longer time series of data is a regression analysis to compare the trajectories of the change in through time in samples taken inside versus outside the park. In both of these analyses the term of interest is the interaction with time. A significant interaction indicates that the change over time is different inside and outside the MPA. The null hypothesis for these analyses is that there is no significant difference in change with time between areas inside and outside of the MPA. The alternative hypothesis is that changes inside the MPA follow a significantly different trajectory or change in a different way from outside the MPA. Where no data exists from before the MPAs were declared, a BACI (*Before - After - Control - Impact*) analysis is not strictly possible but a similar analysis can be done comparing data from a time nearer to declaration and another later time. In all cases sub-samples within sites are pooled before analysis and each site forms a replicate. The data will be averaged over two consecutive years before analysis.

The review by (Keough et al 2007) looks at the power of the data from Bunurong and Wilsons Promontory to detect a difference between inside and outside the park using a snapshot ANOVA and an mBACI design. The authors found that overall the power was similar or slightly less for the mBACI design compared to the snapshot design. The power analyses done for Point Addis, Point Hicks, Cape Howe and Port Phillip Heads considered the power of the data for a snap shot-style ANOVA design. We assumed that, given the similarity of power for the two models at Bunurong and Wilsons Promontory, there would not be a large difference in the power for these models at other sites. Furthermore, for all of these sites except Port Phillip Heads, there was no data from before declaration rendering the mBACI less meaningful than the ANOVA design for these parks. Neither the external review nor the work done for this report considered the power of a regression based trend over time analyses.

Marine Sanctuaries

The statistical model for the analysis of data collected from the Marine Sanctuaries is a BACIP (known as a Paired BACI). This is also an ANOVA-based analysis that has as its core the concept of two matched sites, one from inside the MPA and one from outside. The variable analysed in the BACIP is the difference (for a given parameter) between the inside site and the outside site and the replication comes from repeated samples of the same sites over time. Keough et al (2007) recommended this model for the Marine Sanctuaries as these MPAs don't have adequate space to establish multiple sampling sites inside them, and it is these sites that provide replication (and power) for the snap shot and mBACI models discussed above.

As was the case for information collected from within the Marine National Parks, two possible statistical questions can be asked of the data collected from Marine Sanctuaries under the new SRMP. The first is whether a change has occurred, between two selected time blocks, in the difference between inside and outside the MPA. The statistical test for this is a one-way ANOVA with repeated sampling dates as replicates and the time blocks (close to declaration and some time after) as the treatment (a statistical term for the thing to which the statistical test is applied). The second question that can be asked is whether there was a significant trend over time in the size of the difference between the two sites. This would be tested using regression analysis where a significant slope would indicate an effect of the MPA.

As no data exists for the Marine Sanctuaries from before MPA declaration, the analyses can only assess if the MPA and the reference site are diverging, converging or staying the same relative to each other.

3.4 Estimating effect size

The question regarding what constitutes a meaningful difference between an MPA and a reference area is a difficult one. For example, very small differences may not be biologically meaningful and detecting them may require an unjustifiable use of resources. We need to know that our monitoring program has the power to detect changes of the scale that are relevant and ecologically important. One way is to look at what has been detected in other MPA systems around Australia or the world, and hence is considered to be likely to occur as a result of park declaration.

Effect sizes have been set for monitoring of marine reserves in the Channel Islands National Park in California, USA (Davis 2002) where *a priori* goals of detecting 40% changes in mean values were declared. Keough and Quinn (2002) provide an example for calculating effect size for change in abalone density after declaration of "no-take" zones for abalone at Wilsons Promontory and Bunurong. The authors use fishing records to estimate that 50% of standing stock was removed each year before the "no-take" zone was declared. On this basis they set a 50% change in abundance as an important effect size that they should detect as the result of the removal by fishing. Depending upon how effectively animals of collectable size are replaced by recruitment from smaller size classes or larvae, and how many migrate in or out of the MPA, the effect of this removal over time may well be greater or less than 50 %.

Halpern (2002) reviewed and synthesized the findings of 89 empirical studies of marine reserves to assess marine reserve effectiveness. Looking at fish and invertebrates but not algae he found that, on average, creating a reserve appears to double density, almost triple biomass and increase organism size and diversity by 20 to 30% relative to non-protected areas. He cautions that these values have high variance and cannot be used to predict the response of particular organisms in particular reserves but they do provide some indication of the size of the changes that may be expected. These figures are similar to estimates of around 100% for density and 20% for size of individuals made by Keough (cited in Barrett and Buxton 2002) in the context of meaningful effect sizes for marine parks monitoring.

Based on these guidelines from the literature (see Keough et al 2007 for more information on effect sizes) and the need to set meaningful targets that can be realistically achieved given the nature of the data collected in the SRMP, this monitoring program will aim to detect effect sizes of a 100% change in abundance, 20% change in diversity and 20% change in size of individuals for a reasonable number of species within each of the fish, macro invertebrates and macro algae groups (see Section 4 for more detail on the parameters (or variables) to be measured).

3.5 Power of the revised SRMP

The SRMP outlined below aims to detect effect sizes of 100% for species abundance, for a reasonable number of the more common species. It also aims to be able to detect 20% changes in fish size and 20% for biodiversity measured as total number of species with power of 0.80 and an alpha level of 0.05. Detection of effect sizes of < 100% change in abundance may be achievable for a few of the common species. It should be noted that estimates of power are by their nature only approximate. They are based on preliminary estimates of variance, which will never be exactly the same as the variance of the actual data to be analysed (Keough in (Barrett and Buxton 2002).

SECTION 4 - CHANGES TO THE SUBTIDAL REEF MONITORING PROGRAM

4.1 Options for increasing power

Keough et al. (2007) found that the existing program had low power to detect changes of < 100% in species abundance for most species but acceptable power to detect changes of 200%. In summary, they examined 3 possibilities to increase power of the current program:

1. They suggest sampling less intensively, freeing up resources to allow for more sites to be sampled. They noted that 2 x 50 m non-contiguous quadrats rather than one 200m long quadrat didn't result in much loss of power.
2. They suggest averaging data across years to reduce variation due to short-term environmental fluctuations; this averaging was shown to increase in power.
3. Finally, they propose averaging data spatially to remove small-scale spatial variation; such averaging also led to an increase in power.

The authors found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if multiple non contiguous quadrats, rather than one large quadrat, were used to subsample each site. These have the effect of achieving greater averaging of the spatial variation within sites. Changing from 1 x 200 m quadrat to 2 x 200 m quadrats resulted in a substantial increase in power (10-20 %, Figures 21 and 22. Keough et al 2007). The authors also found that there was only a small loss of power (5%) when only two non-contiguous 50 m sections of the original 1 x 200 m quadrat were used at each site and that 4 non-contiguous 50m quadrats yielded considerably better power than one 200 m quadrat (Keough et. al 2007 Figures 23 and 24).

They found that by averaging data across years and averaging data spatially (to remove small-scale spatial variation) they achieved good power to detect moderate changes in most of the taxa examined.

4.2 Data redundancies

Keough et al (2007) examined correlations among current variables and suggested that using these correlations was not going to be useful to reduce the number of variables measured. They also found that none of the measured variables or groups of variables was a good predictor of broader "biodiversity", meaning that multiple components would need to be sampled if a broader picture of "biodiversity" was required.

4.3 Potential for using alternative variables

Keough et al (2007) looked at the potential for using alternative variables to those currently contained in the program. One example was rates of recruitment. Essentially due to constraints in the method, where recruitment rates are currently measured these variables do not have good power to detect large changes of 200%, therefore using recruitment variables would not be useful unless changes were made to the sampling process for these variables.

Examination of selected species suggested that there was some potential for detecting important changes in the size of abalone and the sex ratio of fishes that are sequential hermaphrodites.

4.4 New designs for use in the SRMP

Based on the recommendations of the review (Keough et al 2007) and on further internal analysis of existing data for other MPAs, some new design features have been proposed for the SRMP. The SRMP has been designed with individual plans for all the Marine Nationals Parks and Sanctuaries currently monitored via the program. The changes will result in a monitoring program with greater power to detect important differences between inside and outside and will also free up resources to expand the park monitoring strategy, allowing more components of park biodiversity to be sampled through additional monitoring methods.

As recommended in the review, in general the modified SRMP will reduce temporal variation by averaging across years and, where possible, reduce spatial variation by averaging across non contiguous sub samples within sites. In some cases, where sample size has been low, the number of sites within and outside the MPA will be increased to increase power. MNPs will be sampled less frequently but in consecutive years to allow temporal averaging of the data.

The power analyses presented in this report for individual parks focus on the snapshot ANOVA model described above. We did not analyse the data for the power of the second statistical model suggested (various BACI or trends over time) but the results of Keough et al (2007) suggest that power for these other models will be similar to or slightly less than for the snapshot model examined here. We also did not examine the power of the design for fish size and biodiversity parameters, thus decisions about these parameters will be based on the findings of the review by (Keough et al 2007).

4.5 Parameters to be measured

The types of parameters (variables) that are chosen for monitoring in MPAs include the following: common species that dominate community structure, endangered species, endemic species, exploited species, invasive or introduced species and charismatic species (Barrett and Buxton 2002, Davis 2002). Edgar and Barrett (1999) also selected heavily exploited species and those commonly occurring but not exploited.

In reality, the variable nature of the data coming from the existing SRMP and the prioritization of resources within future monitoring programs, mean that the most common species and/or those for which we have most power to detect important changes are predominantly the ones that will be measured. The species to be targeted by the monitoring program, and what aspects of these species should be measured (i.e. abundance, size, sex etc.), will be discussed at the end of the Section 4 after power analyses and monitoring programs for individual parks are presented.

As a potential strategy for making the monitoring process more resource efficient, (Keough et al 2007) considered whether it would be possible to use some of the currently measured variables as surrogates for others. They also examined whether measuring alternative variables would prove effective. They did not identify any effective surrogate variables or alternative variables that were viable for use at this time (see the section for each park for a more detailed discussion of their findings). Therefore, the monitoring program will continue to measure the suite of fish and invertebrates that are measured in the current SRMP. However, (Keough et al 2007) did observe that the algal program involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. Additional time/cost savings could be achieved by only sampling abundances of major, habitat-forming algae. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed as part of the tender process for the new SRMP. This saving will be weighed up against the potential loss of biodiversity information through not monitoring the rarer algae. A decision will then be made between the following three options: 1) only monitor abundant, habitat forming groups, 2) measure abundance of major habitat forming groups but record presence/absence data for less common species, 3) continue to monitor all macro algal groups.

4.6 Subtidal Reef Monitoring in Marine Sanctuaries

There are 11 Marine Sanctuaries in the Victorian MPA system ranging in size from 12 ha (Marengo Reefs MS) to 290 ha (Pt Cooke MS; see Figure 1 for map of all the parks). Eight Marine Sanctuaries have appropriate shallow subtidal reef habitat to warrant inclusion in the SRMP: Merri, Marengo Reefs, Eagle Rock, Pt Danger, Barwon Bluff, Pt Cooke, Jawbone, and Ricketts Point Marine Sanctuaries.

In 2003, the SRMP was expanded to include extra parks, including six Marine Sanctuaries: Point Cooke, Jawbone, Ricketts Point, Merri, Marengo Reefs and Eagle Rock Marine Sanctuaries. In 2004, Beware Reef and the Arches Marine Sanctuaries were added to the SRMP. Results and discussion of the program can be found in a number of Parks Victoria Technical Series reports (see Hart et al 2003, 2004 & 2005, Edmunds et al 2005 and Crozier et al 2007).

Statistical Model

The statistical model for the analysis of data collected from the Marine Sanctuaries is a BACIP (Paired BACI). The power for this model comes largely from the number of times the two sites (inside and outside) are sampled. As these parks do not have adequate space to establish multiple sampling sites on reef, (Keough et al 2007) recommended this model for use in Marine Sanctuaries. Multiple sampling sites provide replication (and statistical power) for the snap shot and MBACI models discussed previously for the MNPs. The BACIP design relies on repeated samples over time and hence is suited to the situation where only one site can be established inside the MPA.

Two statistical questions can be asked of the data collected from Marine Sanctuaries under the new SRMP:

4. Has a change occurred, between two selected time blocks, in the difference between inside and outside the MPA? The statistical test for this is a one-way ANOVA with repeated sampling dates as replicates and the time blocks (close to declaration and some time after) as the treatment.
5. Has there been a significant trend over time in the size of the difference between the two sites? This would be tested using regression analysis where a significant slope would indicate an effect of the MPA.

No data exists for the Marine Sanctuaries from before MPA declaration so the analyses can only assess if the parameters in the MPA and the reference site are diverging, converging or staying the same relative to each other.

New Design

As the power for the BACIP design comes from repeated samples over time and the longest time series of data for the sanctuaries is currently three years, it was not possible to test the power of data collected under this design to detect changes in data inside and outside the Marine Sanctuaries. The new design for sanctuaries is based on recommendations made in the review (Keough et al 2007) and the findings of power analyses done for the Marine National Parks (reported below for each park).

Temporal

Monitoring should be done annually and, where possible, individual sanctuaries should be monitored in the same season each year to reduce variation due to seasonal changes. A proposed long-term monitoring schedule, showing the proposed monitoring times for the Marine Sanctuaries is presented in Section 11.

Spatial

Monitoring should continue to be done at existing sites inside and outside each Marine Sanctuary. Consideration should be given to using the double transect method, and counting four non-contiguous subsections within the two transects (for an explanation of this method, see Figure 7). It is likely that space will be limiting for this change but ,where possible, the double-transect method should be considered. The justification for such a change is that there is not a lot of data continuity to be lost for the Sanctuaries by discontinuing the monitoring of two of the subsections in the current transects and power gains similar to those large gains shown for the Marine National Parks (Keough et al 2007 and results for Point Hicks Marine National Park presented in this document)could be expected.

Parameters to Measure

Parameters monitored should be the same as for the Marine National Parks (see Parameters to measure in the SRMP later in Section 4).

Table 1 Summary of new Subtidal Reef Monitoring Strategy for use in Marine Sanctuaries.

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
1 transect inside and 1 transect outside the park boundaries at depths of 4m to 12m depending on the Marine Sanctuary.	No change	In most cases, there is no extra space for more transects in these parks given the small size. Also, the power of the BACIP comes from the temporal replication and additional spatial replication will not greatly increase the power of the BACIP design.
Within Site Spatial Design		
Run a 100 m transect line either side of a marker buoy at a permanent location creating a 200 m transect. Divide transect into 4 * 50 m subsections, and sample contiguous subsections equating to one large quadrat.	The decision is Marine Sanctuary dependent. Where there is limited space, make no change. Where space for extra transects is not an issue, consider using the double transect method. Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 x 50 m sections. Run a second transect positioned to best fit within available reef and best sample the spatial variation of the site. It should be 50 m from the existing transect at its closest point and the virtual site size, as indicated on Figure 7, should be no more than 200 metres on any side. Sample all four contiguous quadrats on the original transect line and two non-contiguous quadrats on the new transect (see Figure 7 and the description in the associated text).	(Keough et al 2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate rather than a single transect were used at each site, however, space and data continuity are good reasons to consider not changing this design, especially as the BACIP style monitoring requires re-sampling fixed sites through time. If the double transect method is used, it should be considered an adaptive pilot. After the first two monitoring times the power gained by adding the two new non-contiguous transects to the design should be assessed. A cost/benefit decision should be made between dropping two quadrats from the original transect lines (approximately a 5 % percent loss in power was calculated for this in Keough et al 2007) versus keeping the two new quadrats. Running out a 2nd transect would take some additional time and effort compared to sampling all quadrats along a single line. And this cost can be weighed up against the gains in power.

Variables sampled		
All species of fish, macro invertebrates and algae that can be identified and measured are.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The algal program involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements are made for all species of fish and it is unlikely that meaningful changes in the size of rarer species are detectable with good power. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost of this practice are obtained. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.
Temporal design		
Sample annually, in the same season where possible	No change	As the statistical power of the BACIP analysis comes from the temporal replication, annual monitoring is important. Monitoring in the same season each year may also help to reduce between year variability within the MPA and reference areas and so increase the power of the design.

4.7 Subtidal Reef Monitoring in Wilsons Promontory MNP

Wilsons Promontory Marine National Park is the largest of Victoria's 13 marine national parks, and lies within Victoria's southernmost marine waters. Its marine flora and fauna is abundant and diverse, and is known to include more than 65 species believed to be at their eastern or western distributional limits (Plummer et al. 2003). It is the only Victorian marine protected area in the Flinders bioregion, and therefore makes an important contribution to the statewide system and the National Representative System of Marine Protected Areas(Parks Victoria 2006b).

There are 10 SRMP monitoring sites within the Marine National Park and 10 reference sites at depths of between 8 m and 10 m (see Edmunds et al 2007 for the latest reports). These sites have been monitored either once or twice per year from 1999 up until declaration in 2002 and twice (in 2004 and 2005) post declaration of the Marine National Park.

The monitoring at this location has been done according to the standard SRMP protocol (described in Edmunds and Hart 2003). For each time and site, one sub-sample was taken consisting of four contiguous 50 x10 (mobile fish and cephalopods) or 50 x 1 metre (cryptic fish and non-sessile invertebrates) quadrats. Macroalgal abundance was estimated from 2025 metre quadrats placed at 10 metre intervals along the same transect line according to the current protocol described above.

The new design for the SRMP at Wilsons Promontory is described below and summarised in Table 2.

Statistical Hypothesis & Analyses

As described previously for the monitoring program in all Marine National Parks, there are two questions that can be asked of the data collected from Wilsons Promontory. The first is whether there is a difference between sites inside and outside the MPA (tested with a one-way ANOVA comparing inside and outside park). The second is whether there is a difference in the change over time between data collected at sites inside and outside the park. This can be tested statistically with a BACI design using data from before declaration and some time after. It would test whether changes over time inside the park are significantly different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA since before declaration can also be tested for significant difference using regression analysis.

New Design

Spatial design

Site layout

At Willsons Promontory, the new design will not involve changes in the site placement or number of sites, and all sites will be retained in their current locations. Increasing the number of sites can increase the power of a monitoring design but this design has good replication already with 10 sites inside and outside. (Keough et al 2007) looked at the effect of moving to a design of 2, non-contiguous, 50 x 10 m quadrats within sites to reduce resource use. Using this approach saved resources to increase the number of sites in the total survey (one additional site for every 3 sites surveyed). However, they found that, for Wilsons Promontory and Bunurong, this did not achieve a significant increase in power. Given the value of the long time series of data for this location, dating back to before the MPA was declared, and the fact that little if any power would be gained by trading quadrat size for extra sites, this sort of reallocation of resources is not advised.

Design for sub-sampling within sites

(Keough et al 2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate rather than contiguous quadrats were used at each site. However, the fact that data exists for this site for before declaration of the MPA, and BACI style monitoring requires re-sampling fixed sites through time, there are only 2 options for redesigning the sub-sampling within the sites. First, retain the original spatial design, or second reduce effort/cost and accept some small loss of power (5%) by sampling only two of the original 4 contiguous quadrats at each site. Much of the time/cost involved in sampling is in deployment of the boat at the site and establishment of the transect line, thus the power retained by keeping all four quadrats would be traded for only a small saving of resources and does not seem expedient.

Temporal Design

Keough et al (2007) found that averaging data for each site over two years should reduce temporal variation and thus increase power by 10-20%. This finding is strongly persuasive because it suggests that a substantial increase in the power of the program to detect important differences can be achieved by adopting a cost effective regime of two consecutive yearly samples repeated over a longer monitoring cycle. We have chosen a cycle of 5 years as cost-effective and one that will enable meaningful changes to be detected within a suitable management time frame (see Section 11 for the full cycle over a 10 year period). The ecological appropriateness of this monitoring time scale is supported by evidence in the literature that the effects of marine parks are seen over long (15 to 40 year) time scales (Russ and Alcala 2004). Furthermore, initial monitoring has consisted of fairly intense (annual or more) surveys over the 4 years before and 3 years after declaration and this is the timeframe over which short-term effects of MPAs might be expected. The resources freed up by reducing monitoring frequency will enable improved SRMP monitoring of other MPAs and implementation of other components of the Marine National Parks and Sanctuaries Monitoring Program to monitor other habitats and components of biodiversity

Where possible, monitoring at this MPA should be done in the same season each year, as an extensive survey of shallow reefs in south-eastern Australia (Underwood et al. 1991) showed considerable variation within sites with season. As greater power to detect MPA effects is achieved by reducing the variation within MPAs and reference areas, restricting sampling to one season should reduce variation between years within MPAs and reference areas and so improve power.

The SRMP schedule, showing the proposed monitoring design and times for Wilsons Promontory is shown below in Table 2 (see also Section 11 for the long-term plan).

Table 2. Summary of new Subtidal Reef Monitoring Strategy for Wilsons Promontory Marine National Park

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
10 sites inside and 10 sites outside. Site depths between 8 m and 10 m. 4 contiguous subunits make up one large quadrat at each site.	No Change	This site already has good replication with 10 sites inside and out. (Keough et al 2007) found that moving to 2 non-contiguous 50 x 10 m quadrats and increasing the number of sites achieved no significant increase in power. There is good pre-declaration data for this MPA, which new sites would not have. Any subunits dropped would represent a loss of data with valuable continuity to pre-declaration for no overall power advantage.
Within Site Spatial Design		
Run a 100 m transect line either side of a marker buoy at a permanent location creating a 200 m transect. Divide transect into 4 * 50 m subsections, and sample contiguous subsections equating to one large quadrat.	No change	(Keough et al 2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate rather than a single quadrat were used at each site, however, valuable before data exists for this site, and BACI style monitoring requires re-sampling fixed sites through time. The options for the spatial design were to retain the original or reduce effort/cost for a loss of power (5%) by sampling only two of the original 4 contiguous subsections at each site. Much of the time/cost involved in sampling is in deployment of the boat at the site and establishment of the transect line, so the power gained by using the full quadrat would be traded for only a small saving of resources and does not seem expedient.
Variables sampled		
All species of fish, macro invertebrates and algae that can be identified and measured are.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The algal monitoring involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements of fish are made for all species and it is unlikely that meaningful changes, with good power, in the size of rarer species will be detectable. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost have been calculated. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.

Temporal design		
Sample annually or biannually	Survey in 2 consecutive years and average data over the two years for each site. Where possible, monitoring should take place in the same season each time.	Averaging data for each site over two consecutive years should reduce temporal variation and thus increase power by 10-20% (Keough et al 2007). Monitoring in the same season each year may also help to reduce between year variability within the MPA and reference areas and so increase the power of the design.
Sample every year where possible	Sample each site once in each five- year monitoring cycle.	A five-year cycle achieves a reduction in effort/cost for monitoring of this park, freeing up resources for monitoring other parks and other habitats. With the park being monitored once in each five-year cycle (with a maximum of 9 years between monitoring), changes will be picked up within an appropriate management time frame.

4.8 Subtidal Reef Monitoring in Bunurong MNP

Bunurong Marine National Park is an important component of Victoria's outstanding parks and reserves system, forming the largest continuous marine protected area within the Central Victorian Marine Bioregion. The shallow subtidal reefs of the Bunurong coast are different from elsewhere in Victoria, supporting species-rich red and brown algae communities. They provide habitat for a diverse range of marine flora and fauna, including a number species at the limits of their distributions (Parks Victoria 2006a). Over 300 different subtidal reef species have been observed during the monitoring program in, and around, Bunurong Marine National Park and the algal community is one of the most diverse of any found in Victorian Marine National Parks (Stewart et al. 2006). The Bunurong coast provides a spectacular natural setting for a variety of water-based activities, including diving, snorkelling, surfing and boating.

Monitoring began in 1999 and this park has been monitored annually or biannually up to and including 2006 but excluding 2005. There are currently 12 sites at this location that are monitored on an ongoing basis, 6 inside and 6 outside the Marine National Park, eight of these were established before the first sampling period in 1999 and the other four were established before the second survey in 2000 (see Stewart et al 2007 for the latest report). Two of the sites inside the park, Site Numbers 3205 and 3012 (54 m apart), are too close to each other to constitute independent sites using the criteria in this report. Parallel sites such as these two would need to be at least 220 m apart to be considered independent for the purposes of this monitoring program.

Sampling for Bunurong MNP is currently done according to the protocol outlined in Edmunds and Hart (2003). For each time and site, one sub-sample is taken consisting of four contiguous 50 x 10 (mobile fish and cephalopods) or 50 x 1 metre (cryptic fish and non-sessile invertebrates) quadrat sections. Macroalgal abundance is estimated from 20, 25 metre quadrats placed at 10 metre intervals along the same transect line. The new design for the SRMP at Bunurong MNP is described below and summarised in Table 3.

Statistical Hypothesis & Analyses

The SRMP for this park is designed so that two statistical questions can be asked of the data collected. The first is whether, at a snapshot in time, there is a difference between parameters inside and outside the MPA. The statistical model for this test is a one-way ANOVA comparing inside and outside park. Sub-samples (or quadrats) within sites are pooled before analysis and sites are replicates. The data will be averaged over two consecutive years before analysis. The second question that can be asked of the data is whether there is a difference in the change over time between sites inside and sites outside the park. This can be tested statistically using a BACI design, which would take data from two times, one before declaration and one at some time after. It would test whether changes over time inside the park have been significantly different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA can also be tested using regression analysis.

New Design

Spatial design

Site layout

This site has good replication with 6 sites inside and 6 outside, but two of the sites inside the park are too close to each other to constitute independent sites by the criteria defined in this report. As it is not valid to include sites # 3205 and # 3012 as replicates in the statistical tests for effect of MPAs, there are 3 options:

1. to discontinue the monitoring at one of these sites,

2. continue to monitor both sites but not include the data in the analysis for effect of MPA or
3. pool data from # 3005 and # 3212 before analysis.

The first option seems a waste of resources and the second is a questionable strategy, as the data from that averaged replicate would be derived from a greater spatial area than all the others in the analysis. The new design will thus retain the current spatial layout but discontinue monitoring at site # 3212 and establish a new site on suitable reef at a distance of greater than 200 m (220 if the transects from the two sites run parallel) from any other site. We propose discontinuing # 3212, as # 3205 has been monitored since the first survey in 1999 while monitoring at # 3212 did not begin until the second survey in 2000.

As noted previously in Section 3, increasing the number of sites can increase the power of a design and (Keough et al 2007) looked at the effect of moving to 2, non-contiguous 50 x 10 m quadrats within sites to reduce resource use, and using these saved resources to increase the number of sites (one additional site for every 3 sites surveyed). They found that for, Wilsons Promontory and Bunurong MNPs, this did not achieve a significant increase in power. Given the value of the long time series of data dating back to before the MPA was declared and the fact that little if any power would be gained by trading quadrat size for extra sites, we decided that apart from correcting the problem with the non-independence of sites # 3212 and # 3205, no other changes should be made to the site lay out.

Within site subsample design

(Keough et al 2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate rather than contiguous quadrats were used at each site. However, the fact that good before data exists for this site, and BACI style monitoring requires re-sampling fixed sites through time, the only options for the within site spatial design are to retain the original spatial design or reduce effort/cost and accept some small loss of power (approximately 5%) by sampling only two of the original 4 contiguous quadrats at each site. Much of the time/cost involved in sampling is in deployment of the boat at the site and establishment of the transect line, thus the power retained by keeping all four quadrats would be traded for only a small saving of resources and does not seem expedient.

Temporal Design

As noted previously in Section 3, a substantial increase in the power of the program to detect important differences can be achieved by adopting a cost effective regime of two consecutive yearly surveys repeated over a longer monitoring cycle. We have chosen a cycle of 5 years as cost-effective and one that will enable meaningful changes to be detected within a suitable management time frame (see Section 11 for the full cycle over a 10 year period). The ecological appropriateness of this monitoring time scale is supported by evidence in the literature that the effects of marine parks are seen over long (15 to 40 year) time scales (Russ and Alcala 2004). The resources freed up by reducing monitoring frequency will enable improved monitoring of other MPAs and implementation of other programs within Marine National Parks and Sanctuaries to monitor other habitats and components of biodiversity.

Where possible, monitoring at Bunurong MNP should be done in the same season each year, to reduce variability in the data (see section # above). The recent Parks Victoria Technical Series for Bunurong MNP (Stewart et al. 2007) includes an extensive consideration of the climatic conditions that limit the success of attempts to do SRMP surveys at Bunurong. The authors conclude that the most suitable seasons for monitoring at this site are January/February (summer) and June (winter). It is proposed that one of these two time periods be chosen for monitoring at Bunurong and then, if possible, monitoring should always take place in that season. The SRMP schedule, showing the proposed monitoring design and times for Bunurong MNP is shown below in Table 3 (see also Section 11 for the long-term plan).

Table 3. Summary of new Subtidal Reef Monitoring Strategy for Bunurong Marine National Park

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
6 sites inside and 6 sites outside. Site depths between 4 m and 7 m with 4 contiguous subunits making up the one large quadrat, which is sampled at each site. Sites # 3012 and # 3005 are too close to be analysed as independent replicates, separated at their nearest point by only 52 m.	Retain the current spatial layout but drop site # 3212 and establish a new site on suitable reef at a distance of greater than 200 m from any other site.	Increasing the replication can improve power if replication is low; however this site has good replication with 6 sites inside and 6 out. There is good pre-declaration data for this MPA, which any new sites would not have. Any subunits dropped to allow resources for extra sites would represent a loss of data with valuable continuity to pre-declaration for no overall power advantage. It is not valid to include sites # 3205 and # 3012 as replicates in the statistical tests for effect of MPA because they do not fit our criteria for independent replicates. The options are to discontinue the monitoring at one of these sites, continue to monitor the site but not include the data in the analysis for effect of MPA, or average data from # 3005 and # 3212 before analysis. The former seems a waste of resources and the latter is a questionable strategy as the data from that averaged replicate would be derived from a greater spatial area than all the others in the analysis.
Within Site Spatial Design		
Run a 100 m transect line either side of a marker buoy at a permanent location creating a 200 m transect. Divide transect into 4 x 50 m subsections, and sample contiguous subsections equating to one large quadrat.	No change	(Keough et al 2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate non contiguous quadrats rather than a single quadrat were used at each site. However, valuable pre-declaration data exists for this site, and BACI style monitoring requires re-sampling fixed sites through time. The options for the spatial design were to retain the original or reduce effort/cost for a loss of power (approximately 5%) by sampling only two of the original 4 contiguous subsections at each site. Much of the time/cost involved in sampling is in deployment of the boat at the site and establishment of the transect line, so the power gained by using the full quadrat would be traded for only a small saving of resources and does not seem expedient.
Variables sampled		
Every species of algae, fish and macro invertebrate able to be identified and counted.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The algal monitoring involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently

		monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements of fish are made for all species and it is unlikely that meaningful changes, with good power, in the size of rarer species will be detectable. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost can be calculated. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.
Temporal design		
Sample annually or biannually	Survey in 2 consecutive years and average data over the two years for each site. Monitoring should always be done in the same season.	Averaging data for each site over two years should reduce temporal variation and thus increase power by 10-20%. Monitoring in the same season each year may also help to reduce between year variability within MPA and reference areas and so increase the power of the design
Sample each year	Sample each site once in each five- year monitoring cycle	A five-year cycle achieves a reduction in effort/cost for monitoring of this park, freeing up resources for monitoring other parks. With the park being monitored once in each five-year cycle (with a maximum of 9 years between monitoring), changes will be picked up within an appropriate management time frame.

4.9 Subtidal Reef Monitoring in Pt Addis MNP

Point Addis Marine National Park makes an important contribution to the statewide system of Marine National Parks and Sanctuaries. Positioned within the Central Victorian Bioregion, it contains sandy beaches and offshore reefs that extend from shallow to intermediate waters and provides habitat for an abundance of reef associated marine life.

Monitoring for the SRMP began in 1999 and this park was monitored annually in summer / autumn from 2004, up to and including 2006 (see Crozier et al 2007 for the latest report). There are currently 4 sites at this location that are monitored on an ongoing basis, two inside and two outside the Marine National Park.

The site is monitored using the standard SRMP protocol detailed in (Edmunds and Hart 2003). For each time and site, one sub-sample is taken consisting of four contiguous 50 x 10 (mobile fish and cephalopods) or 50 x 1 metre (cryptic fish and non-sessile invertebrates) quadrat sections. Macroalgal abundance is estimated from 20, 25 metre quadrats placed at 10 metre intervals along the same transect line. The new design for the SRMP at Bunurong is described below and summarised in Table 7.

Statistical Model

Like the other parks above, the SRMP for this park is designed so that two statistical questions can be asked of the data collected. The first is whether, at a snapshot in time, there is a difference between parameters inside and outside the MPA. The statistical model for this test is a one-way ANOVA comparing inside and outside park. Sub-samples (or quadrats) within sites are pooled before analysis and sites are replicates. The data will be averaged over two consecutive years before analysis. The second question that can be asked of the data is whether there is a difference in the change over time between inside and outside the park. Although no data exists for Pt Addis MNP from before declaration, comparisons can still be made between changes inside and outside the MPA as time since declaration increases. This can be tested statistically using a BACI design, which would use data from two times, one close to declaration and one further away in time. It would test whether changes over time inside the park have been different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA can also be tested for significant difference using regression analysis.

Power Analyses

In order to plan a new program that is scientifically rigorous and sufficiently powerful to detect important changes in the Marine National Park at Point Addis we examined potential power of several monitoring designs. These power analyses followed on from analyses for other parks in Keough et al (2007). The analyses presented and discussed below examine the power of the first statistical model, the snapshot ANOVA analysis, with two different sample sizes.

Methods

The data used for these analyses were collected during surveys one and two of Point Addis Marine National Park and nearby reference sites. The sampling design consisted of two sites inside the park (sites # 3905 and # 3906) and two sites outside the park (sites # 3907 and # 3908). At each site the fish and mobile invertebrates and algae were surveyed according to the standard procedure outlined above and detailed in (Edmunds *et al*. 2005). Survey 1 was done between December 2003 and January 2004 and Survey 2 was done between February and April 2005.

Average abundances over the two sampling times at each site were calculated for each species of fish and mobile invertebrate and for algae, the average percent cover per transect was calculated. The averages were used in the power analyses. The analyses considered the relationship between sample size and power for a one-way ANOVA testing for a difference in abundance of species between inside and outside the park- known as a snapshot comparison. The standard deviations used for the analyses are those estimated from the existing data. Were the number of sites inside and outside the park to be increased in a new design, the variances within the park and outside the park would change so, as with all *a priori* power calculations, these are only an estimate of the potential power of the design.

All data for fish and invertebrates were appraised and power analyses done for species that had an average abundance of greater than zero inside and outside the park. This criterion meant that all but the very rare species were analysed. Species absent at one location were always very low in abundance at the other location. For algae, analyses were done on all species with an overall average abundance of >1 percent. This criterion resulted in the inclusion of all species for which there was > 1 % either inside or outside the park, i.e. all but the very rare species. The effect size was estimated relative to the overall mean abundance of the species in surveys one and two. For example, if the overall mean abundance was 7 individuals per quadrat, then a 100% effect size would be a difference of 7 individuals per quadrat between inside and outside the park.

Results

Fish

Power was tested for seventeen species of fish and none of these had good power (≥ 0.8) to detect a 100 % difference between inside and outside of the park with a sample size of n=2. There were four species of fish for which the analyses predicted good power (≥ 0.8) to detect a 100 % difference in the mean abundance inside and outside the park with a sample size of n=4. These were: *Parma victoriae* (scalyfin), *Cheilodactylus nigripes* (magpie perch) *Odax cyanomelas* (herring cale) and *Meuschenia hippocrepis* (horse shoe leatherjacket). For another three species, *Pictilabrus laticlavius, Notolabrus fucicola* (purple wrasse) and *Notolabrus tetricus* (blue throated wrasse), there was good power to detect an effect size of between 100 % and 150 % with a sample size of n=4. Effect sizes detectable with an n=4 design for the other ten species were greater than 150%. Details of power and detectable effect sizes for different sample sizes can be seen in Table 4.

Table 4. Power and effect size for mobile fish data from Point Addis averaged over survey one and survey two. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. * indicates species for which there was ≥ 0.8 power to detect a 100% change in average overall abundance with n= 4. + indicates species for which an effect size of between 100% and 150% could be detected with power of 0.8. P = power, n= number of replicates inside and outside. ADM is absolute difference in the means inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information.

Species Name	Average overall abundance	P for 100% change, n=2	P for 100% change, n=4	Effect Size (ADM) P≥0.8 and n=2	Effect Size (ADM) P≥0.8 and n=4	Effect Size (% of overall abundance) P≥0.8, n=2	Effect Size (% of overall abundance) P≥0.8, n=4
Trachinops caudimaculatus	9.50	0.1	0.1	NC	32.0	NC	336.8
Upeneichthys vlaminghii	0.38	0.1	0.1	NC	1.4	NC	373.3
Pempheris multiradiata	0.38	0.1	0.1	NC	1.4	NC	373.3

Enoplosus armatus	1.13	0.1	0.3	NC	2.2	NC	195.6
*Parma victoriae**	7.50	0.3	0.9	16.6	5.4	221.3	72.0
Aplodactylus arctidens	0.38	0.1	0.1	NC	1.4	NC	373.3
*Cheilodactylus nigripes**	1.25	0.5	1.0	1.9	0.8	152.0	64.0
Dactylophora nigricans	0.25	0.1	0.1	1.9	0.8	760.0	320.0
Dotalabrus aurantiacus	0.38	0.1	0.1	NC	1.4	0.0	373.3
Pictilabrus laticlavius+	3.13	0.2	0.7	18.4	3.6	588.8	115.2
Notolabrus fucicola+	1.13	0.2	0.6	4.2	1.4	373.3	124.4
Notolabrus tetricus+	20.88	0.1	0.5	NC	30.0	NC	143.7
*Odax cyanomelas**	3.25	0.2	0.9	14.8	3.0	455.4	92.3
Meuschenia flavolineata	3.25	0.1	0.4	NC	5.0	NC	153.8
Meuschenia freycineti	1.25	0.1	0.1	NC	4.3	NC	344.0
*Meuschenia hippocrepis**	3.38	0.6	1.0	4.2	1.4	124.4	41.5
Acanthaluteres vittiger	0.63	0.1	0.2	NC	1.4	NC	224.0

Invertebrates

Power was tested for fifteen invertebrate species. None of these had good power to detect a 100% difference between inside and outside the park with a sample size of n=2. There were three species for which the analyses predicted good power (>80%) to detect a 100% difference in the mean abundance inside and outside the park with a sample size of n=4. These were: the crinoid *Cenolia tricoptera*, the sea star *Nectaria ocellata* and the dog whelk *Dicathais orbita*. For one species of sea star *Nectaria macrobranchia* , there was good power to detect a change 162 % and for the predatory red foot snail *Pleuroploca australasia* there was good power to detect a change of 166%. For the remaining species analysed, a sample size of n=4 gave good power to detect larger differences (greater than 180%) in the abundance between inside and outside the MPA. Details of power and detectable effect sizes for different sample sizes for the invertebrates can be seen in Table 5.

Table 5. Power and effect size for abundance of macro invertebrates averaged over survey one and survey two, Point Addis. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. * indicates species for which there was ≥ 0.8 power to detect a 100% change in average overall abundance with n= 4. + indicates species for which an effect size of between 100% and 150% could be detected with power of 0.8. P = power, n= number of replicates inside and outside. ADM is absolute difference in the means inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information

Species Name	Average overall abundance	P for 100% change, n=2	P for 100% change, n=4	Effect Size (ADM) P≥0.8 and n=2	Effect Size (ADM) P≥0.8 and n=4	%Effect Size P≥0.8, n=2	%Effect Size P≥0.8, n=4
*Cenolia trichoptera**	1.63	0.4	1.0	3.7	0.7	228	43
Cenolia tasmaniae	1.25	0.1	0.1	NC	4.0	NC	320
Fromia polypora	0.25	0.1	0.1	NC	0.8	NC	320
Nectria ocellata	2.13	0.1	0.2	NC	5.6	NC	264
Pentagonaster dubeni	0.88	NC	0.2	NC	2.6	NC	297
Tosia australis	0.50	NC	0.1	NC	1.7	NC	338
*Nepanthia troughtoni**	2.13	0.4	1.0	4.6	1.4	214	68
Nectria macrobranchia	4.63	0.1	0.4	NC	7.5	NC	162
Holopneustes porosissimus	0.38	NC	0.1	NC	1.4	NC	385
Haliotis laevigata	0.63	0.1	0.1	NC	2.9	NC	468
Haliotis rubra	23.50	0.1	0.2	NC	65.5	NC	279
Pleuroploca australasia	0.50	0.1	0.4	NC	0.8	NC	167
Turbo undulatus	21.38	NC	0.1	NC	97.0	NC	454
*Dicathais orbita**	2.13	0.5	1	4.0	0.7	190	35
Astralium tentoriformis	0.38	0.1	0.3	NC	0.7	NC	199

Algae

Power analyses were done for twenty three species of algae. There was good power to detect a 100% change in abundance with a sample size of n=2 for only one species, the green alga *Caulerpa flexis* var. Muelleri. When the sample size was increased to n=4, there was good power to detect a 100% change in the overall mean percent cover for another seven species and the analyses predicted that there would be good power to detect an effect size of between 100 and 150% for a further six species. For the remaining nine species analysed, a sample size of n=4 had good power for very large effect sizes (>150% of overall mean). Details of power and detectable effect sizes for different sample sizes for the algae can be seen in Table 6.

Table 6. Power and effect size for percent cover of algae averaged over survey one and survey two, Point Addis. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. * indicates species for which there was ≥ 0.8 power to detect a 100% change in average overall abundance with n= 4. + indicates species for which an effect size of between 100% and 150% could be detected with power of 0.8. P = power, n= number of replicates inside and outside. ADM is absolute difference in the means inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information.

Species Name	Average overall abundance	P for 100% change, n=2	P for 100% change, n=4	Effect Size (ADM) P≥0.8, n=2	Effect Size (ADM) P≥0.8, n=4	%Effect Size, P≥0.8, n=2	%Effect Size, (P≥0.8, n=4
Acrocarpia paniculata+	6.8	0.1	0.5	NC	9.6	NC	141
Cystophora monilifera	1.0	0.1	0.4	NC	1.7	NC	173
Cystophora moniliformis	1.6	0.1	0.2	NC	4.3	NC	264
*Ecklonia radiata**	30.3	0.2	0.9	NC	22.9	NC	76
*Halopteris spp**	0.8	0.6	1.0	NC	0.5	NC	59
*Seirococcus axillaris**	15.7	0.3	0.8	NC	15.1	NC	96
Zonaria turneriana+	1.1	0.2	0.6	NC	1.4	NC	133
*Sargassum spp**	1.5	0.3	1.0	3.6	1.0	251	70
*Caulerpa flexilis var. muelleri**	2.3	0.8	1.0	2.4	0.6	105	27
*Ballia callitricha**	5.4	0.3	0.9	NC	4.7	NC	86
Melanthalia obtusata	1.1	0.1	0.2	NC	3.1	NC	295
Phacelocarpus peperocarpos+	3.0	0.2	0.6	NC	3.8	NC	129
Plocamium angustum	3.2	0.1	0.3	NC	7.1	NC	222
Plocamium dilatatum+	2.6	0.2	0.6	NC	3.5	NC	132
Other thallose red alga+	3.1	0.1	0.5	NC	4.7	NC	150
Pterocladia lucida	1.4	NC	0.2	NC	NC	NC	286
Plocamium preissianum	1.6	NC	0.2	NC	4.7	NC	297
Gelidium asperum	1.6	NC	0.2	NC	4.7	NC	292
Lobospira bicuspidata	1.7	0.1	0.4	NC	3.2	NC	182
Chlanidophora microphylla	1.0	NC	0.1	NC	5.1	NC	504
*Haliptilon roseum**	4.4	0.3	1.0	NC	2.9	NC	66
Areschougia congesta	2.5	NC	0.2	NC	7.4	NC	295
Encrusting corallines+	15.2	0.2	0.6	NC	19.0	NC	125

New Design

The design for the new SRMP at Point Addis is based on the analyses reported above and the findings and recommendations of the review by Keough et al (2007).

Temporal

Keough et al (2007) found that power to test for a change over time post-declaration improved by 10-20% when data pooled over two surveys was used rather than data from a single survey. In response to this finding and the need to optimise our power to cost ratio, Pt Addis will be monitored on a 5 year cycle with two consecutive annual surveys occurring back to back, i.e., the surveys will occur in the first year of one five year cycle and the last year of the next five year cycle. This design will allow data collected in the two consecutive years to be averaged, reducing temporal variation over the short (two year time scale). The SRMP schedule, showing the proposed monitoring design and times for Pt Addis MNP is shown below in Table 7 (see also Section 11 for the long-term plan).

Spatial

Number of sites

The monitoring design for Point Addis currently has two sites inside and two sites outside the MNP. With the current number of sites, the power to detect an important effect size (100% difference in abundance) was low (<.80) for all except one of fifty five species for which tests were done. When sample size was increased to four sites inside and four sites outside the power to detect an important effect size increased considerably yielding power of > 0.80 for a 100% effect size for fourteen species including four fish, three invertebrates and seven algae. For another eight species of fish and algae there was good power to detect an effect size of between 100 and 150 %.

These power analyses suggest that a sample size of four is the minimum acceptable allowing us to detect an important effect size for 14/55 of the more common species. If further gains of up to 10 to 20 % increase in power are achieved by the improved averaging of spatial variation within sites (see section # below), then the effect size detectable with good power for species with an effect size less than 150% may be reduced to close to 100 %, further increasing the number of species with good power to detect relevant effect sizes.

On the basis of these analyses, two new monitoring sites will be established inside Point Addis Marine National Park and two reference sites established outside the MPA. Sites should be positioned to ensure that all sites are a minimum of 220 m from each other at their closest point if they are parallel or 200 m at their closest point if they are end to end. If suitable reef can be found, then one new reference site should be added on each side of the MPA. If suitable reef can be found, the two new sites within the MPA should be in the north east region of the park, where no sites currently exist. Recent habitat mapping of the park (Ball & Blake 2007 and Holmes *et al* 2007a) show that extensive areas of suitable reef exist directly north of Point Addis itself and also between Addiscot Beach and Bells Beach.

Within Site Spatial Design

Keough et al (2007) found that the influence of small scale spatial variation within a site on the variation between sites would be reduced if replicate quadrats were used at each site rather than one large quadrat. These more widely spaced quadrats better sample and average the spatial variation within each site. For Bunurong and Wilsons Promontory MNP, 4 non-contiguous 50m quadrats yielded considerably better power than one 200 m quadrat (Keough et al 2007 - Figures 23 and 24). Given the lack of physical similarity between these sites and other differences in exposure and habitats, this suggests to us that 4 non-contiguous 50 m quadrats may also yield better power than one contiguous 200 m quadrat at other Victorian locations.

As no data exists from before the declaration of a park at Point Addis and new sites are being established, there is potential to develop a design that reduces the influence of small scale variation within sites on variation between sites by sampling more of the site with non-contiguous quadrats. An extra transect, allowing sampling of two extra, non-contiguous quadrats, will be added at each of the four sites. These will be permanent, as are the existing quadrats, and will be positioned using a similar process to that used for the existing quadrats. The protocol for positioning and laying out these double transect sites is described below.

Double transect site layout

At each site the new transect should be positioned to best fit within available reef and best sample the spatial variation of the site. It should be a minimum of 50 m from the existing transect at their closest points and also positioned so that the maximum size of the virtual site boundary (indicated by the dashed rectangles on Figure 7) is 200 m on any side. Figure 7 illustrates several examples of how the position of the new transect may be tailored to fit different spatial constraints at individual sites. The direction of the transect relative to the GPS locator should be specified as a compass bearing from the locator for future positioning and, for the purposes of identifying the subsections, T5 should be the closest to T1. The position of the new transect should also be marked on the transect map for the park that is kept as a GIS spatial layer by Parks Victoria.

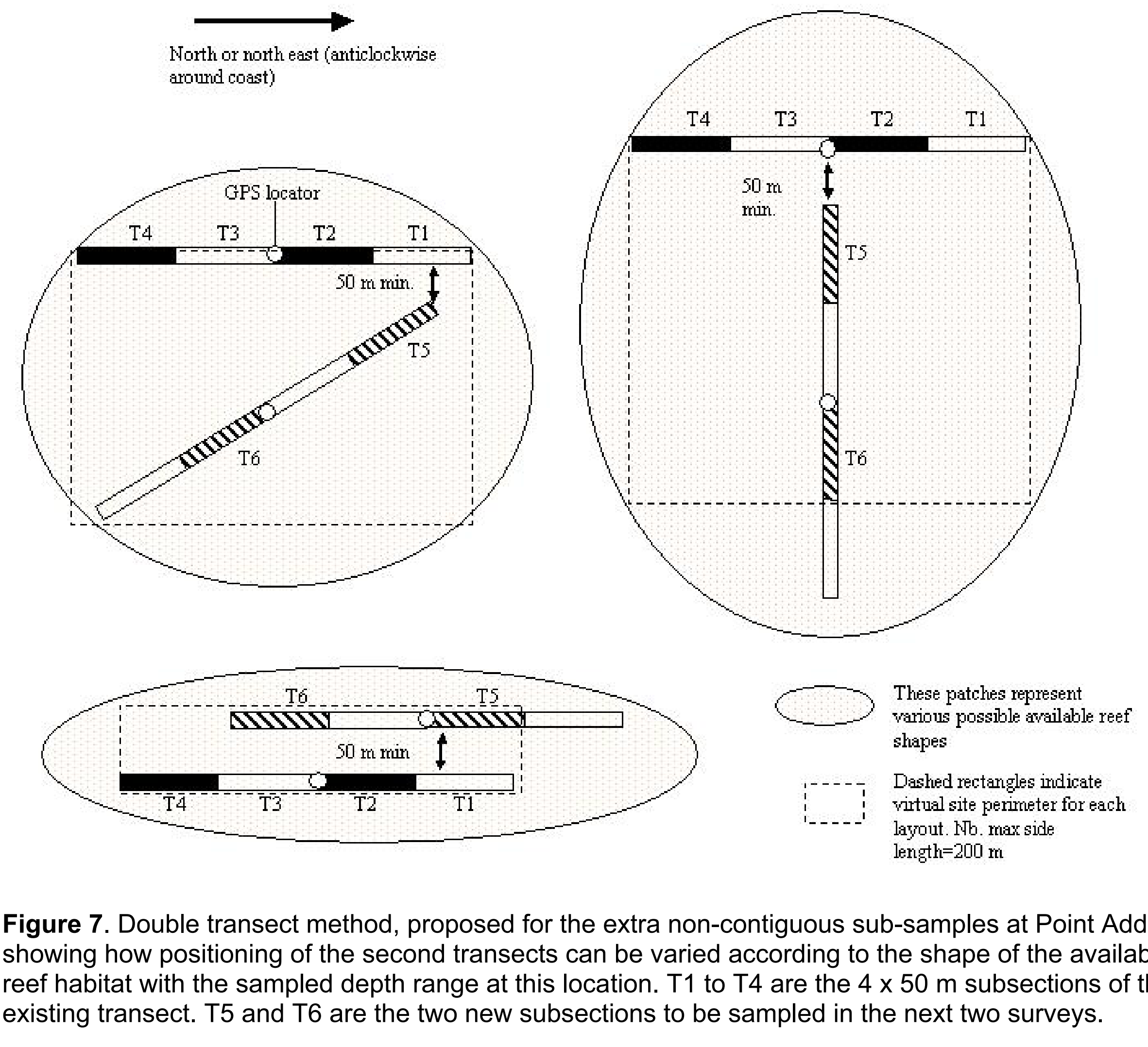

Figure 7. Double transect method, proposed for the extra non-contiguous sub-samples at Point Addis, showing how positioning of the second transects can be varied according to the shape of the available reef habitat with the sampled depth range at this location. T1 to T4 are the 4 x 50 m subsections of the existing transect. T5 and T6 are the two new subsections to be sampled in the next two surveys.

Sampling the double transects at Pt Addis

T5 and T6 should be sampled in an identical way to T1-T4 in the next two consecutive surveys. The data form these two surveys can then be analysed for power to detect an important differences between data collected inside and outside the park. It will then be possible to compare the power of a sampling program using T1-T4 (1 x 200 m quadrat) with a sampling program using T2, T4, T5 and T6 (4 x 50 m quadrats). The extra time (and cost) involved in laying out the second transect at each site will also be known, as will the time that could be saved by only sampling 2 of the original 4 sections. The cost and benefit of four smaller quadrats versus one large one can then be effectively weighed up. If it is judged that the extra sub-samples don't add enough extra power to justify their cost then the second transect can be dropped from the design and the continuity of data collection from the original transects will be unaffected. The next two consecutive surveys will be treated as adaptive monitoring surveys and will incorporate ongoing monitoring with a pilot study examining the scale of patchiness of the reef environment at Point Addis MNP and the power versus cost relationship of both potential sampling designs. If the power gained from the T2 - T6 option outweighs any extra expense then T1 and T3 should be dropped from future surveys while T5 and T6 on the second transect are continued.

Table 7. Summary of new Subtidal Reef Monitoring Strategy for Point Addis Marine National Park

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
Site depths between 3 m and 8 m with 2 sites inside and 2 sites outside. Four contiguous subunits make up the one large 200 m quadrat that is sampled at each site.	Increase the number of sites by 4, placing 2 new sites inside the park and 2 new sites outside. Locate one new reference site to the north of the MPA and one to the south. Locate the new inside park sites toward the north east section of the park where there are currently no sites and extensive suitable reef appears to exist.	Power calculations were done for a snapshot design using time-averaged data from surveys one and two. This indicated that for this design, a sample size of n=2 gave low power to detect important changes in abundances of all but one species and little improvement was achieved with n=3. However when sample size was increased to 4 the design had good power to detect important effect sizes for 4 of the more abundant species of fish, 3 species of invertebrates and 8 species of algae. Mapping of shallow sites at Point Addis shows that there are substantial sections of shallow subtidal reef on which sites inside the park could be established.
Within Site Spatial Design		
Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 x 50 m sections, sample all four sections.	Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 x 50 m sections. Run a second transect positioned to best fit within available reef and best sample the spatial variation of the site. It should be 50 m from the existing transect at its closest point and the virtual site size as indicated on Figure 7 should be no more than 200 metres on any side. Sample all four contiguous quadrats on the original transect line and two non-contiguous quadrats on the new transect (see Figure 7 and the description in the associate text).	Keough et. al. (in press) found that the influence of small scale spatial variation within a site on the variation between sites could be reduced if replicate rather than contiguous quadrats were used at each site. After the first two monitoring times the power gained by adding the two new non-contiguous transects to the design should be assessed and the program treated as adaptive. A cost/benefit decision should be made between dropping two quadrats from the original transect lines (approximately a 5 % percent loss in power was calculated for this in Keough et al 2007) versus keeping the two new quadrats. Running out a 2^{nd} transect would take some additional time and effort compared to sampling all quadrats along a single line. And this cost can be weighed up against the gains in power.

Variables sampled		
Every species of algae, fish and macro invertebrate able to be identified and counted.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The current algal monitoring involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements of fish are made for all species and it is unlikely that meaningful changes, with good power, in the size of rarer species will be detectable. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost can be calculated. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.
Temporal design		
Sample annually or biannually	Survey in 2 consecutive years and average data over the two years for each site. Monitoring should always be done in the same season.	Averaging data for each site over two years should reduce temporal variation and thus_increase power by 10-20%. Monitoring in the same season each year may also help to reduce between year variability within MPA and reference areas and so increase the power of the design
Sample each year	Sample each site once in each five-year monitoring cycle	A five-year cycle achieves a reduction in effort/cost for monitoring of this park, freeing up resources for monitoring other parks. With the park being monitored once in each five-year cycle (with a maximum of 9 years between monitoring), changes will be picked up within an appropriate management time frame.

4.10 Subtidal Reef Monitoring in Pt Hicks MNP

Point Hicks Marine National Park makes an important contribution to the statewide system of Marine National Parks and Sanctuaries. Located in the Twofold Shelf Bioregion, Point Hicks is considered to be the first land on the south-eastern Australian mainland sighted by Europeans, in 1770. The park has granite reefs with a rich and colourful array of marine life as well as stretches of sandy shores abutting the sand dunes and native heathland vegetation of the adjoining Croajingolong National Park (Parks Victoria 2006c).

The current sampling design consists of four sites inside the park (sites #'s 3204, 3221, 3222 and 3206) and four sites outside the park (sites #'s 3207, 3216, 3217, 3218) (see Williams et al 2007 for the latest report). Two of the sites inside the park, # 3216 and # 3207, are too close to each other (approximately 138 m apart) to constitute independent sites by the criteria defined in this report (see Section 11). Sites should be at least 220 m apart if they are parallel and 200 m if they are end to end. The first survey at this location was conducted 2001, before declaration of the MPA, but only 2 of the current sites inside the park (# 3204 and # 3206) and one outside (# 3207) were sampled. The extra sites were established before the second survey in March 2004 and all current sites were sampled again during a third survey in February 2006. The data used here in these power analyses were collected during surveys two and three of Point Hicks Marine National Park and nearby reference sites. Data from the first survey were not used as they only came from a subset of the current sites.

They surveys were conducted according to the standard SRMP protocol detailed in the SRMP protocol (Edmunds and Hart 2003). At each site the fish and mobile invertebrates were surveyed in one large quadrat placed along a transect line (200 x 10 metre for mobile fish and 200 x 1 metre for invertebrates and cryptic fishes). Algae were surveyed by placing a 0.25 m^2 quadrat every 10 metres along the 200 m transect at each site and scoring the presence or absence of each species under each of 50 regularly spaced points within the quadrat.

Statistical Model

As with the other MPAs, the new SRMP for Pt Hicks MNP is designed so that two statistical questions can be asked of the data collected. The first is whether, at a snapshot in time, there is a difference between inside and outside the MPA. The statistical model for this test is a one-way ANOVA comparing inside and outside park. Sub-samples (or quadrats) within sites are pooled before analysis and sites are replicates. The data will be averaged over two consecutive years before analysis. The second question that can be asked of the data is whether there is a difference in the change over time between inside and outside the park. Although little data exists for this MPA from before declaration, comparisons can still be made between changes inside and outside the MPA as time since declaration increases. This can be tested statistically using a BACI type design which would use data from two times, one close to declaration and later in time. This would test whether changes over time inside the park have been different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA can also be tested using regression analysis.

Power Analyses

As stated above, an issue with the current spatial design at Point Hicks is that two of the sites inside the park, # 3216 and # 3207, are too close to each other (approximately 138 m apart) to constitute independent sites by the criteria defined in this report. For the data be suitable for use in the statistical models described above, the sites need to be spatially independent.

In order to plan a new program that is scientifically rigorous and sufficiently powerful to detect important changes due to the Marine National Park at Point Hicks, the potential power of several monitoring designs was examined. First we examined the power of the existing spatial design ignoring the spatial non-independence; these analyses examined scenarios with the existing sample size of 4 sites and also with 5 and 6 sites inside and outside the park. As all of these scenarios had low power, and as we also needed to address the issue of the non-independence of sites # 3216 and # 3207, we then analysed and compared the power of two potential new designs (referred to as models 1 and 2 below).

The information presented and discussed below examines the power for the parameters to be analysed with a snapshot ANOVA approach. The standard deviations used for the analyses are those estimated from the time-averaged (over surveys 2 and 3) existing data. If the number of sites is increased in a new design or if new sites or transects are added the variances inside and outside the park may change substantially so, as with all *a priori* power calculations, these are only an estimate of the power of potential designs (Keough and Quinn 2002), p167).

Existing spatial design

Methods

Average abundances over the two sampling times at each site were calculated for each species of fish and macro invertebrate. For algae, the average percent cover per 0.25 m^2 quadrat was calculated.

All data for fish, algae and macro invertebrates were appraised and power analyses done for species for which there was an average abundance of greater than zero inside and outside the park. The decision was made to analyse data for the less abundant species of algae at Pt Hicks MNP (only those with an overall abundance of >1 % were analysed for the Point Addis MNP data) because there were so few species (7) for which the overall abundance was > 1. The effect size was estimated relative to the overall (from both inside and outside the park) mean abundance of the species for surveys one and two. For example, if the overall mean abundance was 7 individuals per quadrat, then a 100% effect size would be a difference of 7 individuals per quadrat between inside and outside the park.

The power analyses assessed what the predicted power was for the existing design (n=4) and for a design of n=5 and n=6 sites, both inside and outside the park. It also assessed what sample size would be needed to detect a 100% effect size with good power, i.e. ≥ 0.80, and what effect size could be detected with good power for a sample size of n= 4 and n=6.

Results

Fish

Power analyses were done for 28 species of fish. With a sample size of n=4, there was good power (≥ 0.80) to pick up a 100 % change in overall abundance for 3 species of fish, the blue-throated wrasse *Notolabrus tetricus*, purple wrasse *Notolabrus fucicola* and the rock cale *Crinodus lophodon*. There was also good power to detect an effect size of 107% for senator wrasse *Pictilabrus laticlavius* and 112 % for herring cale *Odax cyanomelas*. Increasing the sample size to 5 inside and 5 outside gave good power to detect a 100% effect size for the senator wrasse and herring cale but did not provide enough extra power to detect changes of 100% in the abundance of any other fish species. Increasing the sample size to 6 inside and 6 outside did not increase the number of species for which there was good power for ≤ 100% change in abundance but predicted good power to detect a change of ≤ 150% for an additional 2 species, the banded morwong *Cheilodactylus spectabilis* and sea sweep *Scorpis aequipinnis*. Thus these analyses predicted that with a sample size of 4 there was good power to pick up important effect sizes in 3 (5 if two marginal cases are counted) species of fish, the extra two sites inside and outside the park (i.e. a sample size of 6) don't make a great deal of difference to the number of species for which there is good power to detect a significant change. These additional sites only give the two already marginal cases better power. Details of power and detectable effect sizes for fish species for different sample sizes can be seen in Table 8.

Table 8. Power and effect size for fish abundance averaged over survey one and survey two for Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information.

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6	n for P=0.8 and 100% effect size	%Effect Size, P≥0.8, n=4	%Effect Size, P≥0.8, n=5	%Effect Size, P≥0.8, n=6
Notolabrus tetricus	22.44	1.00	1.00	1.00	3.00	41	NC	31
Crinodus lophodon	7.13	0.88	0.96	1.00	4.00	89	NC	67
Notolabrus fucicola	57.88	0.95	0.99	1.00	4.00	78	NC	58
Pictilabrus laticlavius	1.63	0.75	0.88	0.94	5.00	107	NC	79
Odax cyanomelas	12.81	0.71	0.85	0.93	5.00	112	NC	82
Cheilodactylus spectabilis	10.00	0.38	0.50	0.60	9.00	172	NC	127
Scorpis aequipinnis	1.69	0.33	0.44	0.53	10.00	190	NC	138
Meuschenia freycineti	0.75	0.25	0.34	0.41	13.00	223	NC	162
Achoerodus viridis	1.06	0.20	0.26	0.31	18.00	264	NC	191
Atypichthys strigatus	38.44	0.18	0.23	0.29	18.00	291	NC	202
Acanthaluteres vittiger	0.31	0.15	0.19	0.22	NC	NC	NC	NC
Tetractenos glaber	0.44	0.14	0.17	0.20	NC	NC	NC	NC
Heterodontus portusjacksoni	4.31	0.13	0.17	0.20	NC	NC	NC	NC
Dinolestes lewini	6.06	0.13	0.17	0.20	NC	NC	NC	NC
Pentaceropsis recurvirostris	0.19	0.13	0.16	0.20	NC	NC	NC	NC
Pseudolabrus luculentus	0.19	0.13	0.16	0.20	NC	NC	NC	NC
Kyphosus sydneyanus	1.75	0.12	0.15	0.18	NC	NC	NC	NC
Ophthalmolepis lineolata	1.19	0.11	0.13	0.15	NC	NC	NC	NC
Girella zebra	8.13	0.10	0.12	0.14	NC	NC	NC	NC
Upeneichthys vlaminghii	0.31	0.10	0.12	0.14	NC	NC	NC	NC
Parma microlepis	0.81	0.09	0.10	0.12	NC	NC	NC	NC
Pseudocaranx dentex	1.50	0.09	0.10	0.12	NC	NC	NC	NC
Scorpis lineolata	16.19	0.08	0.09	0.11	NC	NC	NC	NC
Unidentified fish	0.25	0.08	0.09	0.10	NC	NC	NC	NC
Aplodactylus arctidens	0.31	0.07	0.08	0.09	NC	NC	NC	NC
Diodon nichthemerus	0.31	0.07	0.08	0.09	NC	NC	NC	NC
Arripis georgiana	1.88	0.07	0.08	0.09	NC	NC	NC	NC
Trachurus novaezelandiae	190.6	0.07	0.08	0.09	NC	NC	NC	NC

Macro invertebrates

Analyses were done for 18 species of invertebrates. There were no species for which there was good power to detect a 100% change in abundance with a sample size of either n= 4 or n= 5. With n=6 there was one taxa, the predatory gastropod *Dicathais orbita*, in which there was good power to detect a 100% change. With a sample size of n=4 there was one species for which there was good power to detect an effect size of ≤ 150%, with a sample size of n=5 there were two species and with a sample size of n=6 there were 3 for which there was good power to detect an effect size of ≤ 150%. Details for macro invertebrates of power and detectable effect sizes for different sample sizes can be seen in Table 9.

Table 9. Power and effect size for macro invertebrate abundance, averaged over survey one and survey two, Point Hicks. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information.

Species Name	Average overall abund-ance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=5	n for P=0.8 and 100% effect size	%Effect Size, P≥0.8, n=4	%Effect Size, P≥0.8, n=5	%Effect Size, P≥0.8, n=6
Dicathais orbita	4.3	0.58	0.73	0.84	6	132	109	96
Haliotis rubra	47.6	0.47	0.60	0.71	7	151	127	112
Plagusia chabrus	10.4	0.34	0.47	0.58	9	190	152	130
Cabestana spengleri	62.6	0.25	0.33	0.40	14	221	187	165
Coscinasterias muricata	4.3	0.23	0.30	0.36	16	NC	198	175
Scutus antipodes	4.3	0.17	0.22	0.28	20	NC	239	208
Jasus verreauxi	1.1	0.17	0.22	0.27	20	NC	NC	NC
Heliocidaris erythrogramma	24.8	0.15	0.20	0.24	22	NC	NC	NC
Turbo undulatus	9.4	0.14	0.18	0.22	26	NC	NC	NC
Tosia australis	0.4	0.13	0.17	0.20	31	NC	NC	NC
Phasianella ventricosa	0.4	0.11	0.14	0.17	36	NC	NC	NC
Pagurid unidentified	0.6	0.10	0.12	0.14	50	NC	NC	NC
Patiriella calcar	342.1	0.10	0.13	0.15	39	NC	NC	NC
Ranella australasia	0.1	0.10	0.11	0.13	59	NC	NC	NC
Strigopagurus strigimanus	0.4	0.09	0.10	0.12	61	NC	NC	NC
Cenolia trichoptera	682.6	0.08	0.09	0.11	68	NC	NC	NC
Phasianotrochus eximius	1.1	0.08	0.09	0.10	78	NC	NC	NC
Centrostephanus rodgersii	10.4	0.07	0.08	0.09	101	NC	NC	NC

Algae

Analyses were done for 20 species of algae. With a sample size of 4 (inside and outside), no species showed good (≥ 0.80) power to detect a 100 % change in percent cover. With a sample size of 5 (inside and outside) there was good power to detect a 100% change in abundance for one species, *Phyllospora comosa*. When the sample size was increased to 6 inside and 6 out, there was marginally good power (≥.77) to detect a 100% change in abundance for a further 3 algae: the encrusting corallines, *Zonaria turneriana* and *Sargassum verruculosum*. There was also good power to detect an effect size of between 100 and 150% for a further 6 species of algae. Thus 6 sites (inside and outside) showed good power (≥0.77) for a total of 4 species with a detectable effect size of ≤ 100% and a total of 10 species with a detectable effect size of ≤150%. Relatively large increases in the sample size were required to achieve substantial improvement in the number of species able to be detected with good power and a 100% effect size; a sample size of ten gave good power to detect a 100% effect size for only 4 additional species. Details of the power and detectable effect sizes for different sample sizes for sampling algae are in Table 10.

Table 10. Power and effect size for percent cover of algae averaged over survey one and survey two, Point Hicks. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size supplies adequate information.

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6	n for P=0.8 and 100% effect size	%Effect Size, P≥0.8, n=6
Phyllospora comosa	19.37	0.68	0.82	0.92	5	86
Encrusting corallines	1.83	0.53	0.67	0.78	6	103
Zonaria turneriana	0.09	0.53	0.70	0.77	6	106
Sargassum verruculosum	0.23	0.53	0.67	0.77	6	105
Dictyopteris muelleri	0.47	0.41	0.56	0.68	7	117
Haliptilon roseum	0.08	0.37	0.49	0.60	9	136
Cordylecladia furcellata	0.22	0.31	0.43	0.54	9	138
Rhodymenia linearis	1.88	0.30	0.41	0.50	10	143
Cystophora moniliformis	1.19	0.29	0.40	0.49	11	146
Ecklonia radiata	12.76	0.29	0.39	0.48	11	147
Ulva spp	0.68	0.23	0.30	0.37	15	176
Other thallose red alga	0.36	0.19	0.26	0.33	16	189
Acrotylus australis	0.67	0.17	0.23	0.29	19	203
Sargassum vestitum	0.24	0.18	0.23	0.28	22	203
Carpomitra costata	0.05	0.14	0.18	0.19	29	271
Plocamium angustum	0.03	0.12	0.15	0.17	27	280
Sinkoraena tasmanica	0.63	0.11	0.14	0.16	35	293
Plocamium dilatatum	0.04	0.09	0.11	0.13	59	344
Rhodymenia australis	0.14	0.09	0.11	0.12	54	367
Nizymenia australis	0.02	0.07	0.09	0.10	83	459

Considering two alternative spatial designs

As mentioned above, there is an issue with the current design at Point Hicks as sites # 3216 and # 3207 are too close together to constitute independent sites by our criteria. Also, the power of the design with 4 sites inside and 4 sites outside (analysed above) is quite low. We compared two options to address the non independence and low power of the design. The first option we modelled is to combine the non-independent sites, treating them as subsamples within sites, and establish new sites that also consist of two non-contiguous transects. The extra transect at each site would allow better sampling of the within site variance (similar to that discussed for Bunurong MNP). The power of this design could be considered for different values of n to determine what level of replication is needed to result in good power for a reasonable number of species. The second option modelled is to drop one of the non-independent sites and establish new sites, consisting of single contiguous quadrats as used in the current design, to increase the n to a level that gives good power for a reasonable number of species.

We assessed the power for all species for which there was a presence inside and outside the MPA. Because the two different models below used data from different combinations of sites, some different species feature in each analysis. The power of the two models for each of the fish, invertebrates and algae was compared for the 12 species or groups with the highest power. The power of the current design is low with n=4 sites inside and outside. In the light of this fact, and taking into account the sample size indicated by power analysis for Point Addis MNP, these power analyses for models 1 and 2 only considered sample sizes of ≥4.

Model 1

To test the power for option 1 we combined data, averaged over two consecutive surveys, from existing sites (listed below and see Williams et al 2007 for details of the sites). This was not an ideal test of the design option for two transects per site, but is the best test available at this site.

- 3206 and 3222 averaged as one site (inside)
- 3204 and 3221 averaged as one site (inside)
- 3216 and 3207 averaged as one site (outside)
- 3217 used as second outside site as there are no close sites to average.

Model 2

To test the power of option 2 we used data from the following sites:

- 3206 (inside)
- 2304 (inside)
- 3216 (outside)
- 3217 (outside)

Results from the models

Model 1

For invertebrates this model had high power for 3 species to detect a change of 100% abundance with n=4, 4 species with n=5 and 4 species with n=6. The average power over the 12 species with highest power was 0.45 (n=4), 0.51 (n=5) and 0.55 (n=6). See Table 11 for results for all taxa analysed.

For fish this model had high power for 9 species at n=4, n=5 and n=6. The average power for the 12 species with the highest power was 0.76 (n=4) 0.83 (n=5) and 0.86 (n=6). See Table 12 for results for all taxa analysed.

For algae this model had high power for 2 species at n=4 and 3 species at n=5 and n=6. The average power for the 12 species with the highest power was 0.41 (n=4) 0.48 (n=5) and 0.54 (n=6). See Table 13 for results for all taxa analysed.

Table 11. Results from Model 1: the power for the abundance of macro invertebrates from data averaged over survey one and survey two at Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 1 uses combined data from # 3206 and # 3222 averaged as one site (inside), # 3204 and # 3221 averaged as one site (inside), # 3216 and # 3207 averaged as one site (outside) and # 3217 as the other outside site.

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Dicathais orbita	4.63	1.00	1.00	1.00
Coscinasterias muricata	4.53	1.00	1.00	1.00
Jasus verreauxi	1.03	0.98	1.00	1.00
Haliotis rubra	42.66	0.75	0.91	0.97
Plagusia chabrus	9.81	0.56	0.74	0.85
Cabestana spengleri	68.84	0.24	0.32	0.39
Pagurid unidentified	1.03	0.20	0.26	0.32
Tosia australis	0.38	0.17	0.21	0.26
Scutus antipodes	5.06	0.15	0.20	0.25
Heliocidaris erythrogramma	22.25	0.14	0.19	0.23
Centrostephanus rodgersii	1.84	0.13	0.17	0.21
Turbo undulatus	8.56	0.10	0.13	0.15
Patiriella calcar	246.53	0.10	0.12	0.15
Strigopagurus strigimanus	0.31	0.10	0.12	0.14
Phasianella ventricosa	0.34	0.09	0.10	0.12
Cenolia trichoptera	1062.78	0.08	0.10	0.11
Phasianotrochus eximius	0.66	0.07	0.08	0.09

Table 12. Results from Model 1: the power for the abundance of fish from data averaged over survey one and survey two at Point Hicks MNP.. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 1 uses combined data from # 3206 and # 3222 averaged as one site (inside), # 3204 and # 3221 averaged as one site (inside), # 3216 and # 3207 averaged as one site (outside) and # 3217 as the other outside site.

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Notolabrus tetricus	22.25	1.00	1.00	1.00
Cheilodactylus spectabilis	8.06	1.00	1.00	1.00
Achoerodus viridis	1.00	1.00	1.00	1.00
Odax cyanomelas	12.44	0.97	1.00	1.00
Dinolestes lewini	3.63	0.96	1.00	1.00
Crinodus lophodon	6.50	0.90	0.97	0.99
Notolabrus fucicola	52.50	0.89	0.97	0.99
Pictilabrus laticlavius	1.56	0.84	0.95	0.99
Atypichthys strigatus	41.63	0.80	0.93	0.98

Acanthaluteres vittiger	0.19	0.31	0.43	0.54
Scorpis aequipinnis	1.69	0.27	0.37	0.47
Meuschenia freycineti	0.88	0.22	0.29	0.37
Tetractenos glaber	0.44	0.21	0.27	0.32
Pseudocaranx dentex	1.50	0.17	0.23	0.29
Girella zebra	8.13	0.16	0.20	0.24
Ophthalmolepis lineolata	1.31	0.15	0.19	0.22
Scorpis lineolata	16.50	0.15	0.19	0.24
Kyphosus sydneyanus	1.75	0.13	0.17	0.21
Upeneichthys vlaminghii	0.50	0.13	0.17	0.20
Pentaceropsis recurvirostris	0.19	0.12	0.15	0.18
Heterodontus portusjacksoni	5.50	0.11	0.14	0.17
Arripis georgiana	1.88	0.09	0.11	0.14
Trachurus novaezelandiae	195.94	0.08	0.12	0.15

Table 13. Results from Model 1: the power for the abundance of algae from data averaged over survey one and survey two at Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 1 uses combined data from # 3206 and # 3222 averaged as one site (inside), # 3204 and # 3221 averaged as one site (inside), # 3216 and # 3207 averaged as one site (outside) and # 3217 as the other outside site.

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Encrusting corallines	3.01	1.00	1.00	1.00
Rhodymenia linearis	3.69	1.00	1.00	1.00
Phyllospora comosa	40.74	0.75	0.90	0.97
Other thallose red alga	0.66	0.37	0.51	0.63
Haliptilon roseum	0.15	0.29	0.38	0.46
Sargassum vestitum	0.49	0.27	0.37	0.46
Galaxaura marginata	0.22	0.27	0.35	0.42
Cystophora moniliformis	2.38	0.26	0.36	0.45
Zonaria turneriana	0.18	0.19	0.26	0.33
Ecklonia radiata	24.89	0.16	0.21	0.26
Rhodymenia australis	0.29	0.16	0.21	0.26
Acrotylus australis	1.34	0.15	0.20	0.24
Ulva spp	1.36	0.14	0.18	0.22
Carpomitra costata	0.11	0.13	0.16	0.19
Halopteris spp	0.55	0.13	0.16	0.20
Plocamium angustum	0.04	0.12	0.15	0.18
Sargassum verruculosum	0.46	0.11	0.14	0.17
Sinkoraena tasmanica	1.36	0.10	0.12	0.14
Nizymenia australis	0.04	0.10	0.12	0.14
Dictyopteris muelleri	0.94	0.10	0.12	0.14
Cordylecladia furcellata	0.44	0.08	0.10	0.12

Model 2

For invertebrates, this model had high power for 4 species with n=4 and 5 species at n=5 and n=6. The average power for the 12 species with the highest power was 0.46 (n=4), 0.50 (n=5), and 0.52 (n=6). See Table 14 for results for all taxa analysed.

For fish this model had good power to detect a 100% change in the abundance in the overall mean for 5 species at n= 4 and 6 species at n=5 and n=6. The average power of the 12 species with the highest power was 0.58 (n=4) 0.64 (n=5) and 0.68 (n=6). See Table 15 for results for all taxa analysed.

For algae this model had one taxonomic group for which there was good power to detect a 100% change in abundance with n=4 and n=5 and two species with n=6. The average power of the 12 species with the highest power was 0.25 (n=4), 0.30 (n=5) and 0.35 (n=6). See Table 16 for results for all taxa analysed.

Table 14. Results from Model 2: the power for the abundance of macro invertebrates from data averaged over survey one and survey two at Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 2 uses combined data from # 3206 (inside), # 3204 (inside), # 3216 (outside) and # 3217 (outside).

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Plagusia chabrus	9.38	1.00	1.00	1.00
Jasus verreauxi	1.75	1.00	1.00	1.00
Haliotis rubra	33.00	0.94	0.99	1.00
Dicathais orbita	3.63	0.93	0.98	0.99
Patiriella calcar	208.87	0.75	0.89	0.96
Coscinasterias muricata	2.88	0.20	0.27	0.33
Cabestana spengleri	63.38	0.16	0.20	0.24
Phasianella ventricosa	0.38	0.12	0.15	0.17
Turbo undulatus	12.00	0.11	0.14	0.17
Centrostephanus rodgersii	1.75	0.11	0.14	0.16
Phasianotrochus eximius	1.88	0.09	0.11	0.13
Heliocidaris erythrogramma	18.00	0.08	0.09	0.12
Cenolia trichoptera	1064.13	0.08	0.10	0.11

Table 15. Results from Model 2: the power for the abundance of fish from data averaged over survey one and survey two at Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 2 uses combined data from # 3206 (inside), # 3204 (inside), # 3216 (outside) and # 3217 (outside).

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Notolabrus tetricus	20.88	1.00	1.00	1.00
Notolabrus fucicola	44.63	0.99	1.00	1.00
Atypichthys strigatus	63.13	0.92	0.98	0.99
Crinodus lophodon	5.25	0.91	0.97	0.99
Pictilabrus laticlavius	1.75	0.83	0.93	0.97
Cheilodactylus spectabilis	8.00	0.74	0.88	0.94
Odax cyanomelas	10.75	0.47	0.60	0.70
Meuschenia freycineti	1.00	0.39	0.50	0.60
Tetractenos glaber	0.63	0.24	0.32	0.39
Kyphosus sydneyanus	2.88	0.16	0.21	0.26
Scorpis lineolata	27.75	0.14	0.12	0.14
Achoerodus viridis	0.25	0.14	0.17	0.20
Girella zebra	13.13	0.12	0.15	0.19
Dinolestes lewini	0.38	0.12	0.15	0.17
Heterodontus portusjacksoni	4.75	0.10	0.12	0.14

Table 16. Results from Model 2: the power for the abundance of fish from data averaged over survey one and survey two at Point Hicks MNP. Power analyses are for predicted power of a one way ANOVA with the single factor being location inside or outside of MPA. P = power, n= number of replicates inside and outside. Overall abundance is the average of the abundance inside and outside the MPA. NC indicates the value was not calculated because the value for the alternative sample size/s supplies adequate information. Model 2 uses combined data from # 3206 (inside), # 3204 (inside), # 3216 (outside) and # 3217 (outside).

Species Name	Average overall abundance	P for 100% change, n=4	P for 100% change, n=5	P for 100% change, n=6
Encrusting corallines	3.26	0.84	0.96	0.99
Phyllospora comosa	48.49	0.62	0.75	0.84
Rhodymenia linearis	4.36	0.33	0.44	0.54
Sargassum vestitum	0.60	0.17	0.22	0.26
Ecklonia radiata	24.10	0.16	0.21	0.26
Other thallose red alga	1.01	0.16	0.21	0.26
Ulva spp	2.20	0.14	0.17	0.20
Galaxaura marginata	0.28	0.13	0.16	0.19

Sinkoraena tasmanica	1.13	0.11	0.14	0.17
Rhodymenia australis	0.50	0.11	0.14	0.17
Sargassum verruculosum	0.80	0.11	0.13	0.16
Acrotylus australis	1.76	0.10	0.12	0.14
Cystophora moniliformis	3.63	0.10	0.12	0.14
Halopteris spp	0.99	0.10	0.12	0.14
Dictyopteris muelleri	0.28	0.09	0.11	0.13

Comparison of model 1 and model 2

For invertebrates, model 1 performed much the same as model 2. It resulted in a slight loss in average power (-1.28) over Model 2 for n=4 and small gains (2.84 and 5.28%) for n= 5 and n=6. Model 1 gave one less species of invertebrate (at n=3 and n=4 but not n=5) for which there was good power to detect a meaningful effect size.

For fish, using model 1 rather than model 2 resulted in substantial increases in power. The number or species for which there was good power to detect a 100% change increased from 5 species to 9 and the average power for the top 12 species increased from between 31% (n=4) and 26 % (n=6).

For algae, using model 1 rather than model 2 resulted in substantial increases in power. With model 1, there was high power to detect important effects for three species of algae for (for all n values tested) compared to 1 or 2 species for model 2. The algae showed the largest increase in any group for average power of top 12 species, with increases of between 54% and 64% with model 1 compared to model 2. See Table 17 for results for all functional groups.

Table 17. Results for the average power for the top 12 species (in each group - fish, algae and invertebrates) with highest power to detect an effect of 100% change in abundance. The % increase shows the change in average power for the top 12 species in model 1 over the average power of the top 12 species in Model 2, results are shown for power with n=4, n=5 and n=6.

	P model 1	P model 2	Difference	% increase
Fish				
n=4	0.76	0.58	0.18	31.03
n=5	0.83	0.64	0.19	29.69
n=6	0.86	0.68	0.18	26.47
Algae				
n=4	0.41	0.25	0.16	64.00
n=5	0.48	0.3	0.18	60.00
n=6	0.54	0.35	0.19	54.29
Invertebrates				
n=4	0.45	0.46	-0.01	-1.28
n=5	0.51	0.50	0.01	2.84
n=6	0.55	0.52	0.03	5.28

Discussion of the models

Model 1 performs considerably better for fish and algae than model 2 but makes little difference to the power to detect changes in invertebrate numbers. This presumably reflects a different scale of patchiness of invertebrate distribution.

Notably, both models give more power than the existing design. For any given area of interest, the variance, and therefore power calculated, will change depending upon sampling location and number of samples taken. Depending upon the spatial patchiness of the biota in the area and the spatial scale of the sampling, taking more samples may increase or decrease the variance estimated (note that this effect is distinct from the sample size put into the power analysis and determining the degrees of freedom of the statistical test, which was set at n=4 or above). In the situation examined here for Point Hicks MNP, dropping the number of samples from 4 inside and out to 2 inside and out decreased the variance and increased the power. This is effectively an artefact of only using two sites to estimate the variation inside and outside the park. Using two sites inside and outside for model 1 was unavoidable because the testing Model 1 required averaging the available sites leaving only two inside. To make this a legitimate test of the effect of the averaging (i.e., using non-contiguous quadrats to better sample within site variation), it was necessary to compare Model 1 to a design where the variance estimate came from the same number of sites inside and outside, thus two sites were used in model 2 as well.

New Design

The power of the existing design for Point Hicks to detect important differences in abundance between inside and outside the park is low and two of the existing sites are not spatially independent by our criteria. The following new design addresses these issues based on the power analyses presented above and the recommendations of the review in Keough et al (2007).

Temporal

Keough et al (2007) showed that power to test for a change over time post-declaration improved by 10-20% when data pooled over two surveys were used rather than data from a single survey. In response to this finding and the need to optimise the data returns for investment, Pt Hicks MNP will be monitored on a 5 year cycle with two consecutive annual surveys occurring back to back, i.e., the surveys will occur in the first year of one five year cycle and the last year of the next five year cycle. This design will allow data collected in the two consecutive years to be averaged, reducing temporal variation over the short (two year time scale). The power analyses for Point Hicks, presented above, are based on this temporal design. The SRMP schedule, showing the proposed monitoring design and times for Pt Hicks MNP is shown below in Table 18 (see also Section 11 for the long-term plan).

Spatial

Number of sites

The results of the power analysis for Pt Hicks MPA indicate that increasing the sample size beyond n=4 would not substantially increase the power for this model. We therefore propose that the new design for Point Hicks be based on 4 sites inside and outside. If available reef space is insufficient to allow establishment of 4 double transect sites inside the park then it would be preferable to use 3 double transect sites rather than 4 single transects.

The sites that are not spatially independent by the criteria we have set are Sites # 3216 and # 3207. We propose that site # 3207 be retained unchanged, as this site was established before declaration in 2001, and that monitoring at site # 3216 be discontinued. Other sites should remain in their current positions to maximise the continuity of the sampling program and a new site should be established. Benthic habitat mapping for Point Hicks (Blake & Ball 2007b, Holmes et al 2007b) suggests that most of the substrate in the MPA is sand and that reef is restricted to the area immediately around the point itself. Thus, there may not be much reef in the area adjacent to the park where sites # 3216 and # 3207 are currently established, and suitable reef for the new reference site may be difficult to find. Sites should be positioned so that they are at least 220 m apart from each other (200 m if transects of adjacent sites are end to end).

Within site spatial design

The power analyses suggest that a model using 2, non-contiguous transects at each site will be considerably more powerful than one using single transects. We therefore propose that each site be comprised of two non-contiguous transects (as described for Point Addis MNP and shown in Figure 7). Existing transects (other than # 3216) should remain in their current positions to maximize the continuity of the sampling program with the extra transects being added to these existing sites and the new site. The configuration of the new double transect sites should be as described in Figure 7. All sites should be at least 220 m from each other after addition of the new transects, or 200 m if transects of one site are end to end with transects of the adjacent site.

As detailed in the design for Point Addis MNP, only two, non-contiguous sections of the second transect should be sampled. If resources are insufficient, it would be preferable to consider only sampling 2 non contiguous sections of both the first and second transect rather than dropping the second transect or dropping sites. As per previous discussions of the findings from Keough et al (2007), the authors found that there was only a small loss of power (5%) when two non-contiguous 50 m sections, of the original 1 x 200 m quadrat were used and that 4 non-contiguous 50 m quadrats yielded considerably better power than one 200 m quadrat. If reef space is insufficient to allow the establishment of 4 double transect sites inside the park then it would be preferable to use 3 double transect sites rather than 4 single transects.

Table 18. Summary of new Subtidal Reef Monitoring Strategy for Point Hicks Marine National Park

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
Site depths between 4 m and 8 m, 4 sites inside and 4 sites outside. Four contiguous subunits make up the one large quadrat that is sampled at each site. 2 of the reference sites are not far enough apart to be spatially independent.	Retain 4 sites inside and outside but drop site # 3216 and establish a new reference site on a suitable area of reef at a distance of at least 200 m from any existing sites.	Power calculations were done for a snapshot design using time-averaged data from surveys one and two. This indicated that for the current design, a sample size of n=4 gave low power, overall, to detect important changes in abundances. There was no substantial increase in power with sample sizes greater than 4 so it was decided that 4 sites inside and outside should be used and other methods of increasing power should be employed. In the existing design, two of the reference sites are not spatially independent making it necessary to remove either site # 3216 or site # 3207 from the design. As site # 3207 was established before declaration of the Marine National Park it was judged preferable that # 3216 be dropped.
Within Site Spatial Design		
Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 * 50 m sections, sample all four sections.	Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 x 50 m sections. Run a second transect positioned to best fit within available reef and best sample the spatial variation of the site. The second transect should be 50 m from the existing transect at their closest points and the virtual site size as indicated on Figure 7 should be no more than 200 m on any side. Sample all four contiguous quadrats on the original transect line and two non-contiguous quadrats on the new transect (see Figure 7 and the description given under Within Site Spatial Design for Point Addis).	Keough et. al. (in press) found that the influence of small scale spatial variation within a site on the variation between sites could be reduced if replicate rather than contiguous quadrats were used at each site. The power analyses were done for this site in which data from two nearby sites were averaged in an imperfect test of the effect of using two non-contiguous transects at each site. Comparison of the results of this design (Model 1) with the results of a comparable design with single transects at each site (Model 2) indicated that a substantial increase in power would be gained if two transects were used at each site. After the first two monitoring times the power gained by adding the two new non-contiguous transects to the design should be assessed. A cost/benefit decision should be made between dropping two quadrats from the original transect lines (approximately a 5 % percent loss in power was calculated for this in Keough et al (2007) versus dropping the new transects or keeping all the quadrats in the new design.

Variables sampled		
Every species of algae, fish and macro invertebrate able to be identified and counted.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The algal monitoring involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements of fish are made for all species and it is unlikely that meaningful changes, with good power, in the size of rarer species will be detectable. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost can be calculated. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.
Temporal design		
Sample annually or biannually	Survey in 2 consecutive years and average data over the two years for each site. Monitoring should always be done in the same season.	Averaging data for each site over two years should reduce temporal variation and thus increase power by 10-20%. Monitoring in the same season each year may also help to reduce between year variability within MPA and reference areas and so increase the power of the design
Sample each year	Sample each site once in each five-year monitoring cycle	A five-year cycle achieves a reduction in effort/cost for monitoring of this park, freeing up resources for monitoring other parks. With the park being monitored once in each five-year cycle (with a maximum of 9 years between monitoring), changes will be picked up within an appropriate management time frame.

4.11 Subtidal Reef Monitoring in Cape Howe MNP

Cape Howe Marine National Park makes an important contribution to the statewide system of Marine National Parks and Sanctuaries. Located in the Twofold Shelf Bioregion, Cape Howe MNP is relatively unexplored from an ecological perspective, although this is changing (see Holmes et al 2007a). It contains sandy beaches, small intertidal platforms and subtidal rocky reefs, subtidal soft sediments and pelagic communities, all of which provide habitat for a range of species. Its location in Victoria is unique as the warmer eastern waters (from NSW) mix with cooler southern waters, creating an environment rich in nutrients and high in diversity. Many species in these waters reach their southern limits in Far East Gippsland (Parks Victoria 2006d).

Monitoring commenced at Cape Howe in 2001 when 3 sites inside (#'s 3213, 3214 and 3215) and 3 sites outside (#'s 3208, 3210 and 3212) what is now the Cape Howe MNP were surveyed (see Williams et al 2007 for details). In 2004, these sites were surveyed again and two additional sites were established, one inside (# 3220) and one outside (# 3219) the marine park, a third survey was done in February 2006. There are several issues with the existing monitoring design for this park. The four sites inside are very close together and two of the reference sites outside the park are very close together, creating problems of spatial non-independence. Also, the physical habitat and communities of the reference sites is very different form each other as well as from the sites inside the MPA.

The surveys were done according to the standard procedure (Edmunds and Hart 2003). At each site the fish and mobile invertebrates were surveyed in one large quadrat placed along a transect line (200 x 10 m for mobile fish and 200 x 1 m for invertebrates and cryptic fishes). Algae were surveyed by placing a 0.25 m^2 quadrat every 10 m along the 200 m transect at each site and scoring the presence or absence of each species under each of 50 regularly spaced points within the quadrat.

Statistical Model

Like other MPAs discussed, the new SRMP for Cape Howe MNP is designed to ask two statistical questions of the data collected. The first is whether, at a snapshot in time, there is a difference between parameters inside and outside the MPA. The statistical model for this test is a one-way ANOVA comparing inside and outside park. Sub-samples (or quadrats) within sites are pooled before analysis and sites are replicates. The data will be averaged over two consecutive years. The second question that can be asked of the data is whether there is a difference in the change over time between inside and outside the park. Although no data exists for this MPA from before declaration, comparisons can still be made between changes inside and outside the MPA as time since declaration increases (as discussed in Section 3). This can be tested statistically using a BACI type design which would use data from two times, one close to declaration and one further away in time; it would test whether changes over time inside the park have been significantly different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA can also be tested for using regression analysis.

New Design

Temporal

Keough et al (2007) found that averaging data for each site over two years should reduce temporal variation and thus increase power by 10-20%. This finding is strongly persuasive because it suggests that a substantial increase in the power of the program to detect important differences can be achieved by adopting a cost effective regime of two consecutive yearly samples repeated over a longer monitoring cycle. We have decided on a 5 year cycle as appropriate to enable meaningful changes to be detected within a suitable management time frame as well as achieve a good return for monitoring investment. The ecological aptness of this monitoring time scale is supported by evidence in the literature that the effects related to marine protected areas are seen over long (15 to 40 year) time scales (Russ and Alcala 2004). Furthermore, initial monitoring has consisted of fairly intense (annual or more) surveys over the year before and 3 to 4 years after declaration at Cape Howe MNP and this is the timeframe over which short- term effects might be expected to be monitored within the MPA. The resources freed up by reducing monitoring frequency will enable improved monitoring of other MPAs and implementation of programs to monitor other habitats and components of biodiversity.

Where possible the monitoring at Cape Howe MNP should be done in the same season each year. An extensive survey of shallow reefs in south-eastern Australia (Underwood et al. 1991) showed considerable variation within sites with season. As greater power to detect effects is achieved by reducing the variation within MPAs and reference areas, restricting sampling to one season should reduce variation between years within MPAs and reference areas and so improve power. The SRMP schedule, showing the proposed monitoring design and times for Cape Howe MNP is shown below in Table 20 (see also Section 11 for the long-term plan).

Spatial

Sites

Several sites within the MPA and two of the sites outside are closer than 220 m to each other. These sites, which are considered too close to be spatially independent according to our criteria are:

Inside the park

- Sites # 3214 and # 3220 (approximately 120 m apart)
- Sites # 2314 and # 3213 (approximately 190 m apart)
- Sites # 3214 and # 3215 (approximately 182 m apart)
- Sites # 3213 and # 3220 (these sites cross each other)

Outside the park

- Sites # 3212 and # 3219 (approximately 83 m apart)

Inside the park

Sites # 3220 and # 3213 cross each other. We considered whether it would be possible to combine the two transects into one site and monitor non-contiguous sections on each transect, however, because of the configuration of the two transects this was not possible (the non-contiguous sections would be less than 50 m apart. We decided, therefore, that as data from before declaration of the park exists for site # 3213 but not for # 3220, and the continuity of data is considered valuable, monitoring of site # 3220 should be discontinued.

Once # 3220 is discontinued the remaining problems inside the park lie with sites # 2314 and # 3213 (approximately 190 m apart) and sites # 3214 and # 3215 (approximately 182 m apart). These sites were all established before declaration of the park making them all equally valuable in terms of data continuity. The following two options were considered for these sites: a) discontinue monitoring of # 3214, which would ensure spatial independence for all sites or b) allow the spatial discrepancy of 30 m in the distance between # 3214 and # 3213 and discontinue monitoring just the last subsection of site # 3215 putting the remainder of this site more than 200 m from site # 3214. Option b) retains the continuity of monitoring at the sites but means that ongoing resources are put into collecting data for which there is no explicit statistical hypothesis. We decided to discontinue monitoring at site # 3214 (option a).

Outside the park

Sites # 3212 and # 3219 are also too close together (approximately 85 m) to be considered spatially independent. The two sites are, however, too far apart to be combined into a single site as the whole site would then cover a distance of over 280 m. It was decided that monitoring at # 3219 be discontinued as this site was established after declaration and # 3212 was established prior to declaration.

There is a further issue with the two of the other reference sites that are spatially independent which raises an interesting conundrum. In the earliest stages of the program, these two sites (# 3208 and # 3210) were deliberately chosen by a contractor because they were different from the sites inside the park and were sites of intrinsic interest and value beyond the park monitoring program. Site # 3208 is predominantly an urchin barren and all of # 3210 is an urchin barren. According to the contractors, they were chosen to provide a reference for "*assessing the direction of community changes within the marine protected area, and whether there is a succession toward urchin barrens within the park*" (Williams et al. 2007). The sites within the park are dominated by crayweed *Phyllospora comosa* although the report from the last monitoring survey noted large decreases in the cover of crayweed at sites within the park (Williams et al. 2007) suggesting one of two things. First that there is large natural variation of crayweed over time or second, that these sites may be becoming more like # 3208 and # 3210 over time. The remaining two reference sites outside the park (# 3212 and # 3219) are more similar to those inside the park and are dominated by *Phyllospora comosa.*

The difference between the two urchin barren reference sites and the two that are more like sites within the park results in very high variance among sites outside the park and thus low power for an ANOVA testing for a difference between inside and outside the park. The following example illustrates this point. The average standard deviation for abundance of species within the invertebrate group, when calculated for all four current reference sites and all four current inside MPA sites was 20.3 outside the park and 7.2 inside (data averaged over surveys 2 and 3). The standard deviation for the same group and times, when calculated for just # 3212 and # 3219 outside and # 3213 and # 3215 inside, was 10.2 inside (9.2 if # 3215 and # 3220 are retained) and 8.5 outside. The fact that dropping two sites from inside causes little change to the standard deviation, while dropping the two urchin barren reference sites from outside reduces the variance to less than half the original figure, suggests that # 3208 and # 3210 are considerably reducing the power of the design to detect a difference between inside and outside.

There is also a logical problem with using sites # 3208 and # 3210 as reference sites. By definition, reference site are chosen to be as closely representative as possible of the areas inside the MPA (or treatment area) when they are established. Thus, if the reference sites change in a different way from the impact sites, the change can be attributed to the "treatment" (or a causal link can be inferred) - in this case to a result of having an MPA in that location (Keough and Mapstone 1995b, Downes et al. 2002). If the reference sites are chosen because they are different from the treatment site then, if they change in a different way after the treatment, nothing can be concluded about the existence of the MPA. The sites were already ecologically very different and in this particular case were chosen for that difference. The trends over time at the urchin barren sites compared to the other sites may answer interesting questions about ecological changes associated with the development of urchin barrens but they don't assist with detecting changes due to the location of the MPA. In this light, it has been decided that annual monitoring should be discontinued at # 3208 and # 3210, however, in future years these sites may contain very important historical data relating to the development and spread of urchin barrens in eastern Victoria. This potential role should not be lost and consideration should be given to infrequent assessment of these sites by similar or other methods.

Power analyses for the current design at Cape Howe were not considered justified. The non-independence means that three of the current eight sites will be discontinued and two sites need to be discontinued from outside the park due to their lack of suitability as reference sites. This leaves only three sites in the new design from the original eight sites. Hence it is unclear whether analyses of the current data would be a much better predictor of the power of a new design, with five new sites, than the results obtained from analyses at Wilsons Promontory, Bunurong, Point Addis and Point Hicks MNPs.

However, a power analysis was done for one group at Cape Howe MNP, the invertebrates, to establish if the power at this site was actually similar to that at other parks. We used data from the current design, averaged over surveys one and two, with 4 sites inside and outside the park. The analysis was done for every species of invertebrate that had a presence both inside and outside the MPA. The analysis tested the power for an ANOVA snapshot design to detect a change in abundance of 100% with a sample size of n=4. Thirteen species of invertebrate were tested and there were no species for which the power to detect a 100% change in abundance was good (≥0.8) (see Table 19). The species for which the test had highest power to detect an effect size of 100% was the black lip abalone *Haliotis rubra* with a power of 0.47 or high power to detect a 150 % effect size. The next smallest effect size was for the unidentified pagurid for which there was good power to detect a change of 270 %. Unsurprisingly, comparing this power to that of Point Addis or Point Hicks MNPs with the same number of sites shows that power at Cape Howe MNP is inferior.

We have assumed that the lower power is due to the spatial confounding of two very different reference sites. If this problem is corrected, by implementing a spatial design with non-contiguous subsamples, similar power could be expected at this site as is predicted at Point Hicks and Point Addis MNPs.

Table 19. shows power to detect a 100% change in abundance for invertebrates at Cape Howe. This is calculated for the data monitored under the current design with 4 sites inside and 4 sites outside and averaged over surveys 2 and 3.

Species Name	Average overall abundance	Power for 100% change in overall mean, n=4	Effect size for P>.80, n=4
Heliocidaris erythrogramma	18.94	0.14	343.0
Centrostephanus rodgersii	330.94	0.13	362.6
Holopneustes inflatus	0.19	0.08	542.9
Haliotis rubra	45.63	0.47	151.5
Turbo undulatus	52.88	0.11	407.2
Dicathais orbita	4.69	0.10	457.8
Ranella australasia	0.38	0.15	314.9
Cabestana spengleri	6.81	0.08	595.4
Astralium tentoriformis	8.25	0.14	339.8
Plagusia chabrus	2.13	0.13	364.0
Strigopagurus strigimanus	0.44	0.14	349.5
Jasus verreauxi	0.13	0.10	476.2
Pagurid unidentified	0.31	0.18	270.6

Summary of new design

Discontinue monitoring at site #'s 3220, 3214, 3219, 3210 and 3208. However, it is paramount to acknowledge the importance of sites #'s 3210 and 3208 in understanding urchin barren development in eastern Victoria and consideration should be given to other monitoring whilst maintaining this useful historical data. Two new sites should be established inside Cape Howe MNP and three new reference sites should be established outside the park each consisting of two transects (see Figure 7). The remaining reference site # 3212, and the remaining two sites inside the park, # 3213 and # 3215 should have an extra transect added so that they each consist of 2 transects. All sites should be at least 220 m from each other. At each site, two non-contiguous sections of each 200 m transect should be monitored.

One remaining logistical issue is that, even with the use of recently acquired habitat mapping products, it may not be possible to find enough similar reef inside and outside the park for four independent MPA sites and four valid reference sites. If adequate suitable reef cannot be found then the only option may be to treat this site in the same way as the Marine Sanctuaries are treated (see earlier in Section 4), establishing one or two closely matched sites inside and outside and using a BACIP model. If this approach proves necessary, as many of the existing sites as possible should be retained, in order to maximise data continuity.

Table 20. Summary of new Subtidal Reef Monitoring Strategy for Cape Howe Marine National Park

Present Protocol	**Proposed Changes**	**Rationale**
General Spatial Design		
Site depths between 5 m and 10 m with, 4 sites inside and 4 sites outside the park. Four contiguous subunits make up the one large quadrat that is sampled at each site. Several sites inside the park and 2 of the reference sites are not far enough apart to be spatially independent. The other two reference sites are very different to sites within the park.	Discontinue monitoring at sites # 3220, # 3214, # 3219, # 3210 and # 3208. Two new sites should be established inside Cape Howe MPA and three new reference sites should be established outside the park. Each should consist of two transects as described for Point Addis Marine National Park. **Note** the potential importance of sites # 3210 & 3208 for future assessment of urchin barrens in eastern Victoria.	With the current layout, the non-independent sites do no constitute independent replicates and can't legitimately be used as such in the snapshot or BACI designs outlined for analysis of data collected in the monitoring program. No alternative could be seen apart from discontinuing the monitoring of the sites indicated. The power analyses at this (only limited analyses were done for Cape Howe) and other MPAs indicated that 4 sites inside and outside was the minimum number to achieve good power to detect meaningful changes in the abundance of a reasonable number of species
Within Site Spatial Design		
Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 * 50 m sections, sample all four sections.	Run a 100 m transect line either side of the buoy creating a 200 m transect, divided into 4 x 50 m sections. Run a second transect positioned to best fit within available reef and best sample the spatial variation of the site. The second transect should be 50 m from the existing transect at their closest points and the virtual site size (see Figure 7) should be no more than 200 m on any side. Sample two non-contiguous quadrats on the original transect line and two non-contiguous quadrats on the new transect (see Figure 7 and the associated description)	Keough et. al. (in press) found that the influence of small scale spatial variation within a site on the variation between sites could be reduced if replicate rather than contiguous quadrats were used at each site. Power analyses were also done for Point Hicks Marine National Park (see above), in which data from nearby sites were averaged in an imperfect test of the effect of using two non-contiguous transects at each site and results indicated that a substantial increase in power would be gained if two transects were used at each site. In the proposed spatial designs for Point Addis and Point Hicks MNPs we recommended monitoring all sub-sections of the original transects, as well as discontinuous sub-sections of the new transects, until cost/ benefit analysis was done of the new design. However, as sites at this location will be mostly new, there is little to be gained by monitoring all sub-sections of either transect.

Variables sampled		
Every species of algae, fish and macro invertebrate able to be identified and counted.	Will need to be determined after cost estimates of changes are obtained from monitoring contractors.	Examination of the potential for using surrogate or alternative variables failed to find any that could be used effectively at this time. The algal monitoring involves time-consuming assessment of a large number of taxa, most of which are too rare for analysis. The potential time/cost savings involved in only monitoring a subgroup of the algae currently monitored will be assessed. This saving will be weighed up against the loss of biodiversity information through not monitoring the rarer algae and other options for monitoring rare species of interest. Measurements of fish are made for all species and it is unlikely that meaningful changes, with good power, in the size of rarer species will be detectable. A cost benefit analysis will be done for fish measurement for rarer species when estimates of the cost can be calculated. Alternative methods for obtaining data on total number of species and abundance of rare species will be assessed.
Temporal design		
Sample annually or biannually	Survey in 2 consecutive years and average data over the two years for each site. Monitoring should always be done in the same season.	Averaging data for each site over two years should reduce temporal variation and thus increase power by 10-20%. Monitoring in the same season each year may also help to reduce between year variability within MPA and reference areas and so increase the power of the design
Sample each year	Sample each site once in each five-year monitoring cycle	A five-year cycle achieves a reduction in effort/cost for monitoring of this park, freeing up resources for monitoring other parks. With the park being monitored once in each five-year cycle (with a maximum of 9 years between monitoring), changes will be picked up within an appropriate management time frame.

4.12 Subtidal Reef Monitoring in Port Phillip Heads MNP

The Port Phillip Heads Marine National Park was declared along with all other current Marine National Parks under the *National Parks (Amendment) Act 2002*, on 16 November 2002. The boundaries of the Marine National Park incorporated five locations in areas previously protected in the Harold Holt Marine Reserve – Point Lonsdale, Point Nepean, The Annulus (Popes Eye), Mud Islands and Swan Bay as well as including one new location – Portsea Hole. Port Phillip Heads MNP is sometimes referred as 6 MPAs in one park. It spans 6 different locations and many different habitat types including deep water canyons, manmade rock structures, high energy outer coast beaches and platforms, calmer bights inside the bay and shallow seagrass beds in a benign embayment (to name a few). This diversity of habitats creates a challenge when planning a monitoring program to assess changes in environmental values for this park.

Monitoring commenced at Port Phillip heads in May 1998 with an original 15 sites (10 inside and 5 outside the final park boundaries). The site locations were based on the habitats within the old Harold Holt Marine Reserve boundaries and the original park boundaries proposed by the Environmental Conservation Council (ECC 1998, 2000). The initial sites locations were more balanced than the final design would indicate, as the declared park boundaries were different to initial proposals. In essence there are currently 4 regions being monitored under SRMP at Port Philip Heads MNP, each structural distinct and varying markedly in wave exposure. These regions are: Pope's Eye (and a reference site at South Channel Fort), the outer coast of Pt Lonsdale, the inner coast at Nepean Bay and the inner coast Lonsdale Bight. Currently there is no monitoring in Swan Bay, Portsea Hole or the outer coast of Pt Nepean.

Since 1998, there have been 12 surveys across a number of seasons. Following advice in Keough et al (2007), in an attempt to balance the representation of habitats inside and outside the park in the monitoring program, three new sites were established in survey 12 done in 2006. Two are in Lonsdale Bight, outside the park (#'s 2816 and 2817) and one on the open coast of Point Lonsdale, inside the park (# 2818). Monitoring at two existing sites in Lonsdale Bight, inside the park, was discontinued. Full details of the current design and data are available in Gilmour *et al* (2007).

The main issues with the existing monitoring design for this park have been discussed in Keough et al (2007) and include the lack of balance in the representation of habitats, the lack of replication inside and outside the park for certain habitats and resultant decrease in power to detect changes. Another issue exists with the proximity of some of the sites to each other, thus creating problems of spatial non-independence.

The surveys have all been done according to the standard procedure (Edmunds and Hart 2003). At each site the fish and mobile invertebrates were surveyed in one large quadrat placed along a transect line (200 x 10 m for mobile fish and 200 x 1 m for invertebrates and cryptic fishes). Algae were surveyed by placing a 0.25 m^2 quadrat every 10 m along the 200 m transect at each site and scoring the presence or absence of each species under each of 50 regularly spaced points within the quadrat.

In this report, we will not present a complete assessment of the monitoring at Port Phillip Heads, as a more detailed review of the program at this park is required. We will, however, create framework for the future and detail a number of possible scenarios that should be assessed prior to any further decision been made about SRMP monitoring at Port Phillip Heads MNP. The most important of these is that Port Phillip Heads MNP should be considered as a series of unconnected smaller MPAs with different flora and fauna and monitored accordingly.

Statistical Model

Like other MPAs discussed, the new SRMP for Port Phillip Heads MNP should be designed to ask two statistical questions of the data collected. The first is whether, at a snapshot in time, there is a difference between parameters inside and outside the MPA. The statistical model for this test is a one-way ANOVA comparing inside and outside park. Sub-samples (or quadrats) within sites are pooled before analysis and sites are replicates. The data will be averaged over two consecutive years. The second question that can be asked of the data is whether there is a difference in the change over time between inside and outside the park. This can be tested statistically using a BACI design with data from before declaration and some time after. It would test whether changes over time inside the park are significantly different to changes over the same time outside the park. Differences in trends over time inside and outside the MPA since before declaration can also be tested for significant difference using regression analysis.

Information for New Design

We have not done any power analyses for this site. The general approach to the final monitoring design should be finalised prior to further power analyses followed by a assessment of the plan with current data, where applicable. The long-term SRMP schedule showing the proposed times for Port Phillip Heads MNP is in Section 11. However, we have not provided a summary table for the proposed monitoring program in Port Phillip Heads Marine National Park. Below, we list a series of questions and issues to be addressed prior to further large scale monitoring in the park using the SRMP.

Temporal

Keough et al (2007) found that power to test for a change over time post-declaration improved by 10-20% when data pooled over two surveys was used rather than data from a single survey. In response to this finding and the need to optimise our power to cost ratio, Port Phillip Heads Marine National Park should be monitored on a 5 year cycle with two consecutive annual surveys occurring back to back, i.e., the surveys should occur in the first year of one five year cycle and the last year of the next five year cycle. This design will allow data collected in the two consecutive years to be averaged, reducing temporal variation over the short (two year time scale).

Spatial

The monitoring of each of the 4 defined regions of Port Phillip Heads MNP should be separately considered ensuring that the power, aims, parameters and timescales for each region are assessed individually..

For the foreseeable future, we propose that monitoring should be reviewed with a view to it continuing in some form in the 4 regions historically monitored: Pope's Eye, the outer coast of Pt Lonsdale, the inner coast at Nepean Bay and the inner coast Lonsdale Bight. Unless there is a specific threat to it, we do not propose expanding the SRMP to Portsea Hole. It is not appropriate to monitor Swan Bay using the SRMP methods and it should be considered as a part of any seagrass monitoring done in future. Finally, although the outer coast of Pt Nepean has appropriate habitats for the SRMP, it is a very difficult environment to monitor given the wave exposure. It should be considered for future monitoring, however logistics may make its inclusion problematic. Should it prove statistically powerful, maintaining some monitoring at the other 4 regions ensures a continuity of the oldest historical dataset in the SRMP.

Pope's Eye

We propose that the Pope's Eye region of the park and its reference site at South Channel Fort should be treated as a Sanctuary for the purposes of the future monitoring program (see plan for Marine Sanctuaries earlier in Section 4), mainly due to the restricted size for replicate transects. Also, both are self-contained and manmade sites with rock annuli and similar flora and fauna. One important threat noted for Pope's Eye (Carey et al 2007) relates to uncontrolled or over use from recreational pursuits. A program designed to assess this risk should involve annual surveys rather than a 5 yearly cycle.

Approach to other regions monitored via the SRMP in the park

We propose considering the remaining sites as being within three sections: Outer Lonsdale, Inner Lonsdale and Inner Nepean. This following presents a framework for a more detailed review of the data at these sites.

It is important to establish which sites are too close together (not spatially independent) according to the criteria defined in this report. There are different ways to effectively solve any issues of spatial non-independence. Based only on geographic location, our initial assessment has concluded that in the Inner Nepean region # 2808 should be discontinued due to it's proximity to #'s 2802 and 2803, however detailed further examination of the historical data may lead to a different conclusion.

In the Outer Lonsdale region # 2818 was, unfortunately, recently established too close to # 2813. This means that #2818 will need to be discontinued after a single survey in favour of #2813 because #2813 was one of the original sites and this will ensure maintenance of data continuity from before declaration of the park. A new site will need to be established in the Outer Lonsdale region should power analyses show that the Outer Lonsdale region could be effectively monitored by continuing with the SRMP. We are unsure that this region has enough space to cater for the replication needed for the SRMP. The challenge will be finding space for enough independent sites (probably 4) using double transects (see Figure 7 for details). It is possible that 3 sites would be adequate, however power analyses are essential to answer this question. A further assessment of the habitats and logistics of sampling at the outer coast at Pt Nepean may validate the creation of one single region for the outer coast in this park. Alternatively, the Outer Lonsdale region may need to be treated similarly to the Marine Sanctuaries for monitoring and analyses. In this case, we make no recommendation regarding the temporal scale of this monitoring. The existence of a long historical dataset suggests that monitoring less than annually is possible, but this must be weighed against the statistical power to detect changes over time.

For the remaining sites at Inner Lonsdale and Inner Nepean, we suggest that 2 models be assessed.

4. Treat each region as a separate monitoring area for analysis and examine power to assess how many sites would be needed to detect meaningful change for each region. From our initial assessment, this option will probably require the addition of new independent sites in Inner Nepean and this may be difficult to achieve spatially. We predict that the existing sample size for the Inner Lonsdale location should be sufficient.

Important questions to be answered regarding the data for Inner Nepean and Lonsdale are:

- Which model results in the higher power to detect meaningful changes?
- For model 2 what sample size is needed for each region to result in good power to detect meaningful changes?
- Is the power high enough with the current site layout (in both regions) or should double transects be used at these sites?

4.13 Parameters to measure in the SRMP

The new SRMP designs outlined above should have good power to detect meaningful differences in abundance for a subset of the species currently counted in the SRMP. The power analyses presented above and those done for Bunurong and Wilsons Promontory by (Keough et al 2007) show that many species are counted for which there is very little likelihood of detecting a meaningful effect. For example, in the 3 surveys at Point Addis MNP, 38 fish, 37 invertebrate and 107 algal taxa were counted as well as the fish length being measured. Even with a new, more powerful monitoring program in place it is likely that important changes in abundance will only be detectable for 7 fish, 5 invertebrate and 11 algae taxa (at the most). While it is important to monitor the taxa that have high power for the meaningful measure of MPA performance they provide, the situation leads us to ask whether it is meaningful to continue to count and measure all the species for which we have low and very low power to detect a meaningful effect. This section discusses this issue and we make a number of recommendations..

Abundance data collected for species with a low power to detect a meaningful change (known from here as “species with low power”) is also used in the calculation of diversity indices but power to detect meaningful changes (>20%) in biodiversity indices was consistently low at Bunurong and Wilsons Promontory MNPs (Keough et al 2007). If the power to detect meaningful changes in the biodiversity indices is also low in the new designs, as is predicted by the results from Bunurong and Wilsons Promontory MNP, then collection of abundance data on species with low power for use in biodiversity indices would not be justified[3].

Generally, but not always, species with low power are less common and rare taxa. It is possible that a streamlined system of scoring the presence / absence of the species with low power could be developed, thus giving estimates of total number of species, a measure of biodiversity for which there was good power at Bunurong and Wilsons Promontory MNP. It is also possible that separate monitoring methods to target rare species could be developed for species of particular management interest. Such monitoring might involve searching larger areas for particular species or using video to record large areas and then reviewing the footage in the laboratory for rare species. There may also be good reason to continue monitoring abundance of some species with low power where there is an expectation that very large changes in abundance are possible (e.g., introduced marine pests or previously harvested taxa that have no known major predators).

The ultimate decision about whether and how to measure the species with low power is dependent upon whether substantial savings in time/money would be made by not counting and/or measuring them. Given that the deployment of boat, divers and survey equipment for each method is a major proportion of the cost of a survey, the savings that would be made are not clear cut. Accurate estimates of time involved in counting low power species are therefore essential before decisions on what parameters to measure can be made. The most likely place for savings to be made seems to be in reducing the large number of less common algae currently counted.

[3] Power to detect changes in biodiversity indices of data from MPAs other than Bunurong and Wilsons Promontory MNP was not tested as a part of this study.

Fish size is also measured for all species counted in the current SRMP protocol. Power analyses of fish size measurements for protogynous hermaphrodite fish (i.e., fish that change from female to male) were done by (Keough et al 2007) for Bunurong and Wilsons Promontory MNP. They found good power to detect meaningful change in some of the common protogynous hermaphrodite fish species. On the basis of this, the continuation of size measurements for these common species,, such as blue- throated wrasse (*Notolarbrus tetricus*), senator wrasse (*Pictilabrus laticlavius*), barber perch (*Caesioperca rasor*) and herring cale (*Odax cyanomelas*) seems advisable. Size measurements for these fish, along with a record of sex, allow for a different measure of MPA performance and possibly a better reflection of ecological process than that given by other variables measured in the current SRMP design (see Keough et al (2007) for more information on this topic). The advisability of continuing to take size measurements on other fish species is less clear. An estimate of the potential cost savings associated with dropping the measurements of fish size for species with low power is necessary to decide if these measurements should continue.

Power analyses for the minimum detectable change in size of adult (in this case legal catch size) black lip abalone showed good power to detect changes at Bunurong and Wilsons Promontory MNP (Keough et al 2007). On the basis of this it would seem advisable to continue to measure size for this species.

(Keough et al 2007) considered whether there were alternative measures that would better measure ecological process than the ones made in the current SRMP. One measure they considered was average size of male protogynous hermaphrodite fish, as discussed above, and the other was recruitment processes. The power analyses showed detectable effect sizes for the fish and abalone species that were analysed to be high (> 200 % in most cases). Further investigation would need to done into alternative methods of sampling for recruit numbers before recruitment could be incorporated, as a useful parameter, into the SRMP (Keough et al 2007).

Summary of recommendations for parameters to be measured

The SRMP should continue to count and measure those taxa and processes highlighted as having a good statistical power to detect a change. In most parks, there were a number (generally less than 10) of species of fish, invertebrate and algal taxa each with high enough power for the monitoring to detect an ecologically meaningful change. They are listed in each park section in this report.

- After detailed estimates and logistical input are sought from potential contractors for the cost and time involved in changes to the SRMP, the following options should be considered: Reducing the species of algae counted to include only those for which there is good power to detect a meaningful change. Consideration could be given to using a large detectable change for this criterion (such as 200% change) or choose those algae with numbers above a particular abundance threshold defined for an ecologically important reason.
- Using alternative methods to assess the presence/absence of the less common (and hence less powerful) species to estimate total number of species.
- Reducing the length measurement of fish species to only the common (and possibly only the common protogynous hermaphrodite) species.
- Using different and targeted methods to measure rare species of management concern (e.g., rare indigenous species, introduced pest species or previously harvested species with low numbers). This could involve searching specific areas within the park for particular species of interest or the use of video to record such areas for later review. This approach may not necessarily result in a cost saving (except where it allowed monitoring of rare species to be discontinued within the SRMP protocol) but would guard against loss of important information that may result

from reducing the number of species counted within standard SRMP method. Most importantly, it would provide a more powerful measure of the abundance of these species than that provided by the current monitoring protocol.

SECTION 5 - INTERTIDAL REEF MONITORING PROGRAM

Current Program

Rocky intertidal reefs (also called rocky reefs or intertidal platforms) are generally found in Victoria on and near headlands with stretches of sandy beaches either side. Along with beaches, intertidal reefs are one of the most accessible components of the marine environment as they are the interface between the ocean and the land. As such they are valued as important habitats by people and tend to be visited more than other sections of the coast (see Carey et al 2007 a 7 b). As noted in a recent marine risk assessment process done in partnership between Parks Victoria and the University of Melbourne (Carey et al 2007 a & b), intertidal reefs are often subjected to human pressures like harvesting, fossicking, and trampling as well as pressures from pollution sources on land and in the sea.

To assist with the management of intertidal reefs, Parks Victoria established a long-term intertidal reef monitoring program in 2004 with the explicit aim of tracking changes due to human use of the platforms. The uses targeted by the monitoring are trampling and fossicking, which occur in greater amounts in high visitation parks. Intertidal platforms in Victoria have been the site of much study into the impacts from trampling on the diversity and status of intertidal organisms (e.g., Keough & Quinn 1998).

The program began in late 2002 with sampling at a number of sites in 2002-2003. The initial method used prior to 2003 was reviewed by independent experts and considered not fit for the proposed purpose. A new method was developed in consultation with the same independent experts and implemented by contractors in the summer of 2003-2004. Essentially, the method involves monitoring the invertebrates and macroalgae that live in the intertidal zone. Reefs within 9 targeted MPAs as well as those in 9 matched references areas outside the MPAs(see Hart and Edmunds 2005 for full details of the monitoring design).

Initially 14 fixed sites were established on intertidal reef habitats inside and outside the following marine protected areas in 2003-2004:

- Point Addis Marine National Park;
- Point Danger Marine Sanctuary;
- Barwon Heads Marine Sanctuary;
- Point Cooke Marine Sanctuary;
- Jawbone Marine Sanctuary;
- Ricketts Point Marine Sanctuary; and
- Mushroom Reef Marine Sanctuary.

In the summer of 2004/2005, 2 new sites were established and surveyed inside and outside the park boundaries:

- Bunurong Marine National Park; and
- Port Phillip Heads Marine National Park.

As per Hart and Edmunds (2005), the general method is to survey a single reef during a single low tide. Surveys target the predominant substratum type at each intertidal reef. After identifying the high and low shore regions, 2 baseline transects are run horizontal to the shore. Five fixed transects, each running from high to low shore, are positioned roughly equidistant to each other across the survey area. Invertebrate and algal abundances (and/or % cover) are surveyed from quadrats placed in five fixed sampling locations (2 x 2 m) that distribute sampling effort evenly along each transect and endeavour to cover any changes in substratum height across the reef.

The latest report on the results of the monitoring is in Gilmour and Edmunds (2007). This program will continue in its current form until a major review of the data and results after the summer of 2007-2008.

Proposed Review

Most parks have been sampled between 3 and 4 times since summer 2003-2004. There has been no formal review of the data collected in the program and the statistical power it has to detect changes in the biota, and hence inform management directions. Each of the 9 parks was chosen because it was known to be an area of high visitation. The associated reference sites are assumed to be less visited.

Once the next round of surveys is done (2007-2008), there will be enough data for an extended and detailed review of the statistical power to detect changes. The power analyses should use the published effect sizes noted from local platforms in the published literature, and a required power of 0.80. Minor refinements to the method or to parameters measured may be required following the review.

This program should continue in its current form until this review of the data and results is done after the summer of 2007-2008.

SECTION 6 - DEEP REEF MONITORING

6.1 Review of remote video options for deep reef monitoring

Introduction

The Monitoring Program for subtidal areas of Victoria's Marine National Parks and Sanctuaries uses diver-based underwater visual census techniques, however these types of diver-based surveys become more difficult and more costly at depths greater than about 10 metres (Edmunds and Hart 2003). Safe dive times decrease below this depth, along with the amount of work that can be achieved per dive, while deployment costs per dive remain constant or increase for deeper water. The deepest practical and safe depth for diver surveys is approximately 27 m in southern Victorian waters (Edmunds et al. 2006b), which means that habitats and communities below this depth cannot be monitored unless alternative methods are used.

There are good management reasons, however, to monitor deep-water marine habitats. Large areas of Victoria's MPAs are in deep water (Plumber et al 2003, Holmes et al. 2007a, to e) and shallow water species and communities are generally different from those at greater depth. Stronger wave action can lead to simpler communities in shallow waters, reflecting the suite of species that are tolerant to the high-energy conditions found there (Barrett and Buxton 2002). The dominance of algae declines with depth due to decreasing light, and a concomitant increase in the dominance of sessile invertebrates is usually seen on deeper substrates (Keough 1999). Existing data for deep reef assemblages in Victoria and elsewhere in the world suggests that these assemblages contain a unique combination of organisms (Barrett and Buxton 2002) and the biological and physical differences mean that deeper areas may also respond differently to threats. It is important therefore, that survey and monitoring programs for Victoria's MPAs include deep-water habitats, as well as more easily accessible, shallow areas.

There are also other issues with diver surveys. Diver bias and training (Barrett and Buxton 2002, Douchette et al. 2002) and avoidance or attraction of fish by divers can influence data collected in visual surveys (Willis et al. 2000, Barrett and Buxton 2002, Harvey et al. 2001). The cold waters of the temperate zones can limit dive times, even in shallow water, and cause increased diver fatigue and decreased accuracy and precision.

The primary purpose of this review is to examine the potential of video techniques that could form a permanent part of a program monitoring deep habitats in Victorian MPAs. The review includes discussion of what techniques are available for use and how data collected via video might compare to, or augment, the current diver-based data that is collected on shallow subtidal reefs the MPAs.

Overview of video use for marine monitoring

The major advantage of video is that it can be used in places where depth, time or safety considerations preclude the use of divers. In these situations, videos attached to remotely operated vehicles (ROVs), towed by boat or directed at bait stations have been used. Video monitoring has also been done from submersible vehicles. Video can also be adapted to most of the techniques used in diver mediated visual monitoring programs (where divers visually identify and count biota as they swim) and offers a permanent record that can be revisited at any time. It can be used by divers with little biological training and be analysed by biologists in a laboratory, free from the negative and potentially confounding effects of narcosis, cold and fatigue which can be experienced by divers (see discussion by Sweatman in Barrett and Buxton 2002). Studies suggest that there may also be advantages like greater precision and accuracy from diver collected video data over diver visual census (see discussion in Cappo et al. 2002).

Neville Barrett discusses diver mediated visual census techniques currently in use in temperate Australia in (Barrett and Buxton 2002) and notes that cost is one important consideration. Underwater housing for standard video cameras is relatively inexpensive (approximately $7000 for a unit with lights). If size and distance estimates are needed, however, as they are where consistent sample areas or animal size measurements are required, then stereo video systems (Cappo et al 2002) or distance estimating lasers ((Edmunds et al. 2006b) are required, all of which add considerably to the cost.

Video performs best in two dimensional situations; on highly structured reefs with caves and overhangs a lot of information can be missed by video. The presence of large upright organisms such as kelp or large arborescent sponges can also obscure other, smaller or flatter organisms from view in video footage (B. Power pers. obs.). This may make video surveys less suitable for use in communities with a high density of large arborescent species, such as dense algal beds, and better suited to surveying communities of more discrete individuals such as fish or lobster, or encrusting sessile communities. Other limitations with video include poor performance in high contrast situations where dark areas cannot be seen and the loss of colour with depth (Barrett and Buxton 2002) which can be a problem when colour is important for taxonomic identification. Barrett and Buxton (2002) also note that analysis of video footage in the laboratory is very time consuming so that savings made on time and expertise in the field may be lost as costs for biologist’s time in the laboratory are met. Costs, advantages and difficulties of video deployment differ according to the particular application and technique. Some of these techniques, and studies comparing them, are discussed in more detail below.

Underwater video techniques

Stereo Video for fish surveys

The use of ‘swimmable’ stereo vision systems (stereo video systems that can be held by a diver swimming along a transect) are discussed in (Cappo et al. 2002). Although that study focuses on use of stereo vision systems as an alternative to visual diver surveys in shallow water, the findings of the study have implications for the potential of stereo video use via remote deployment in deep water as well. Cappo et al (2002) observe that stereo video systems can now allow for very accurate assessment of fish size (up to 0.05% of measurements done on a boat deck of the same fish) and that customized software is available to provide these measurements from video footage. The authors note that increased precision of stereo vision fish measurements can substantially increase the power of surveys to detect differences in fish length between samples. (Harvey et al. 2001) found that stereo video detected a 15 % difference in mean length of blue cod with a power of 0.9 with 63% fewer samples than needed when measurements were made by experienced divers. Stereo video can also potentially measure very large animals (e.g., whale sharks) and very small fish (such as new recruits) and can also measure rugosity and other parameters of the underlying physical habitat (Douchette et al. 2002). Single video cameras don’t provide accurate measurements of fish size unless the subject fish is swimming in a precise plane, measurements from stationary single video cameras have a very high error rate 13.62 $\pm$ 1.41 mm compared to 0.05% of target lengths with stereo video (Harvey et al. 2002).

Image analysis and modelling can allow fish weight and volume to be accurately and precisely measured from stereo video images and there is potential for automated species recognition from video footage using computer vision and artificial neural networks (Storbeck and Daan 2001). Another positive characteristic of stereo video is that transect boundaries (and which individuals are inside and outside those boundaries) can be accurately estimated from stereo video whereas there is potential for considerable error when divers estimate unmarked boundaries to survey areas. Harvey (1999 cited in (Cappo et al. 2002) demonstrated that these errors could be up to 82% underestimates and 194% overestimates of survey areas and hence of the subsequent density measurements).

Remote Operated Vehicle with Video

Video surveys have been conducted using remote operated vehicles (ROVs), especially in deeper waters on continental shelves (Auster et al. 1998; Auster et al. 2003) and in very deep seas (Lauerman 1998, Fijikura et al. 1999, Scoltwedel and Vopel 2001), where diving is not possible. ROVs have also been used for video surveys and environmental monitoring in shallow water habitats (Greene and Alevizon 1989, Kasprzak and Wilson 1994), and for coral reef surveys (Williams et al. 1999, Williams and Mahon 2004).

ROVs with mounted video were used in Port Phillip Bay in Victoria in the Trial Dredge Deep Reef Impact Assessment (Edmunds et al. 2006a); The Deep Reef Existing Conditions Report for the Port of Melbourne Authority Channel Deepening SEES (Edmunds et al. 2006) ; in surveys at The Twelve Apostles Marine National Park and The Arches Marine Sanctuary for Parks Victoria (Edmunds et al. in prep) and also at Wilson's Promontory National Park (Edmunds et al. 2006b). In all of these studies the system used consisted of a small Seabotix LBV150 S^2 ROV that was deployed from a boat at the site of interest. Other brands and types of ROVs are also available. A high resolution digital video camera was mounted to a rotating chassis to provide inspection of horizontal, sloped and vertical surfaces. Two parallel lasers were mounted to the chassis, to provide a means of scaling the field of view and capturing the appropriate frame size. A sonar tracking system was used to determine the position of the ROV underwater relative to the vessel. A differential GPS, electronic compass, pitch and roll sensors, and scanning sonar were also used to aid in navigation and supply spatial positioning data for the frames. The vessel-mounted instruments and the tracking data fed data into a mapping and logging software program, and this software calculated the absolute position of the ROV and plotted the ROV track on bathymetry charts. Real time video footage could be viewed on an onboard monitor.

The Port Phillip Heads study (Edmunds et al. 2006a) surveyed sessile organisms on primarily high relief, deep water, rock walls and the survey method involved taking video transects of the substratum along depth contours (27 - 57 m). At The Twelve Apostles MNP and The Arches MS, the terrain consisted of 1-3 m high ridges and gullies over a slight gradient and hence the transects were more haphazard. Non-overlapping consecutive frames (0.5 x 0.7 m) were grabbed from this footage and percentage cover estimated for each species of sessile organism using software to assess onscreen points. Physical habitat features such as slope were also noted. An 8 MP resolution digital still camera in an Ikelite housing was also mounted on an ROV for some deployments; set to fire automatically at a given time intervals it was used to obtain high quality still images of biota for taxonomic records and identification purposes as well as possible other management uses (e.g., for use in communications or educational products).

The authors noted that the amount of data that could be collected over any set period of time was limited by the nature of the Port Phillip Heads environment (which includes strong tides, rough conditions, a shallow central reef and heavy shipping traffic). The principal strategy for dealing with these limitations was to have a field team ready to mobilize whenever a slack water period that had suitable tidal and sea conditions occurred (Edmunds et al. 2006). As a further illustration of the logistical difficulties encountered using ROVs in severe weather and sea conditions, the AIMS survey at Lord Howe Island (Speare et al. 2004) discussed below, carried two ROVs, similar to those used in the Victorian studies (Sea Botix LVBs), on the vessel to examine selected areas of the benthos in detail but were never actually deployed due to unsuitable sea conditions.

Advantages of the ROV system used for the 4 Victorian studies include the portable lightweight nature of the ROV unit (a trait shared across a number of brands), which could be lifted and deployed from the vessel by hand, and the manoeuvrability of the ROV in the water. In the hands of a skilled pilot the camera can be trained on selected sample areas, habitats or organisms with a high degree of control and areas of consistent size sampled. Large amounts of data can be collected over relatively short time periods when sea conditions allow use. Given that the tidal flows in Port Phillip Heads are relatively extreme and shipping traffic place further limitations on deployment, it is likely that other Victorian sites would probably allow greater windows of time for deployment. Disadvantages include the cost of the ROV and onboard equipment, the cost of chartering a suitable vessel and the requirements for skilled people to pilot the ROV and operate the onboard equipment. The method may not be suitable for use in areas dominated by large arborescent organisms (such as substrates with high density of large thallose algae) or for certain substrate types such highly structured and complex reef where entanglement of cables and loss of video information due to contrast can be problem.

The ability to use trained biologists and specialists such as taxonomists to analyse video footage in a laboratory environment is a major advantage but definitive taxonomic identification of less common species often requires collection of samples. Such collection would have to be done using dredge or remote operated claws in water beyond safe diving depths. (Speare et al. 2004) note that one of the ROV's carried on the vessel for the Lord Howe Island Survey was equipped with a manipulator arm for the purpose of recovering specimens to support identification and equipment such at this would greatly enhance the taxonomic robustness of surveys or monitoring work done using ROV and video.

A recent study (Lam et al. 2006) evaluated the use of video transects obtained from divers or (ROV) and a point intercept transect method (where divers swimming along a transect visually assessing the substrate under predetermined points) for monitoring subtropical coral communities. Unfortunately the comparisons confound the effect of increased sampling within sites and the effect of the different data collection methods. Power analysis of results from each method showed that the minimum detectable change in coral percentage cover had very low mean values (0.39 - 1.65% for the ROV dataset, 0.66% for the diver mediated video dataset, and much higher values (12.11%) for the diver mediated visual transect dataset. These differences may have been explained, however, by the fact that for both ROV and SCUBA video methods, a total of 5000 points were examined per transect while for the diver mediated visual transect method, only 40 points were analysed per transect. The authors suggest that the video survey methods have higher power to detect temporal changes in coral communities than the diver mediated point intercept transect method. This may be the case, but the increased power may well come from the time efficiency of video compared to the diver mediated transect method (which was 30 min per five transects per site versus 1.5 h of diving per site in this study). These efficiency gains could allow for more sampling within transects, larger transects or more transects.

Towed Video

Towed Video has been used in a joint venture between Parks Victoria and the Coastal Zone CRC (major contributions from the University of Western Australia and Fugro Survey Pty Ltd) as part of a state of the art surveying and mapping project across 5 of Victoria's national parks spanning 5 bioregions (Holmes et al. 2007 a to e). In the course of this project (and other associated partnerships), 20 000 hectares of near-shore benthic habitat were surveyed with hydroacoustic equipment and more than 350 kilometres of underwater towed video footage was collected. The imagery was analysed for substrate and biotic benthos characteristics and the resulting data were used for habitat modelling and mapping.

The authors note that there were problems with poor visibility making identification of biota and substrate impossible for a percentage of the video frames and also that there was a problem with the varying height of the camera above the substrate which made some frames unusable due to being too distant or too close for identification. Although the system was precisely geo-referenced by using an USBL (ultra-short baseline GPS system), the varying distance of the camera from the substrate meant that the data collected was not suitable for density estimates as there was not a constant frame size. Density estimates were not necessary for the mapping and modelling purposes of that project, but accurate and precise density estimates would be necessary for ongoing MPA monitoring. This issue would need to be resolved in any tow-video system used for such monitoring.

CSIRO Marine Research has resolved the problem of maintaining the camera unit at a constant height above the sea floor. CSIRO has been using and developing towed video systems to survey deep waters since the 1990's (Barker et al. 2000) and have developed a series of towed platforms for video. Early systems were developed to survey 20 - 200 m depths and these have been refined to operate down to much greater depths (>1500 m), with variable undersea terrain, rough open ocean conditions and strong currents. These systems are explained in detail in Barker et al. (2000). The critical advantage of the system from the perspective of surveying Victorian Marine Protected areas, is its ability to maintain the camera unit at a height of 1 to 2 metres off the seabed while it is towed along a transect at a speed of approximately 1 m/sec over variable terrain. This is achieved by having the camera unit trail horizontally behind a weight, which de-couples the wave-induced vessel movement from the camera. The camera unit is positively buoyant and is held at the desired distance above the seafloor by trailing a given length of chain.

The CSIRO camera unit has two digital DVCAM format video cameras; individually housed and separated by 250 mm. Lighting is supplied by 2 x 250 Watt incandescent lights. Data is transmitted to and from the ship electronically and sensors are incorporated into the system to measure depth, pitch and roll angles, water temperature, housing temperature and height above bottom (altitude). At the time the review was written (2000) real-time video could not be displayed on the vessel but note that the ROV system operated by AME in Victorian waters (described above), and the AIMS system (described below), do allow for this. The use of two cameras extends the available recording time from 40 to 80 minutes when used consecutively or the two cameras can record simultaneously for stereo video.

The authors note that the system is expensive due to the length of conducting cable, the cost of accurate deep-rated sensors and the need specialised deep water materials. They also note that the system has been judged to be cost effective against ocean-depth rated ROVs or submersibles. A towed video system for the depths encountered in Victoria's Marine Protected areas (<110 m) would be less expensive. Future developments planned for the system by CSIRO include: replacing the tow-cable with a combination fibre-optics and wire conductor to enable real-time video display on the vessel; and modifications to decrease risk to the camera unit over irregular topographies through break away weak-links and thrusters which can react to forward looking sensors, to help avoid obstructions.

The Australian institute of Marine Science conducted towed video studies in intermediate and deep waters in Ningaloo Marine Park (Rees 2004) and also at Lord Howe Island (Speare et al. 2004). The study at Ningaloo Marine Park had the principal aim of describing the general nature and spatial patterns of macrobenthic communities below normal diving depths (>30 metres) within the Marine Park. In this study the survey was done at three zones adjacent to existing State Marine Sanctuary zones from the seaward edge of Ningaloo Reef. The spatial layout of the survey was designed to test the spatial variation of benthic communities along a north-south gradient at the 10-100km scales. At each zone, fifteen transects at each of five depth zones were surveyed using a towed video camera system. Transects were spaced approximately 1.5 km apart within each depth zone and were approximately 300m long.

Equipment consisted of a Cunard Series 50 subsea video camera connected to 300m of load-bearing coaxial tow cable. This was attached to a purpose built wing that provided negative lift and maintained a stable angle through the water while towing. The camera and wing were deployed over the stern of the boat supported by a lightweight A-frame. During filming an observer watched the footage on the boat above and recorded seabed substrate and relative abundance of dominant organisms. The video stream was recorded to miniDV tape with real time ship position, from GPS, added to the tape. Whilst recording, speed was maintained at 1-1.5 knots. The actual position of the camera was not determined and boat position may have differed from camera position by up to 200m (a problem overcome in the Victorian mapping program see Holmes et al. (2007 a - e)).

Analysis of the video data from the Ningaloo study was broad and descriptive rather than quantitative. Results included discussion of intra site trends ~10km scale, and inter-site trends ~100km scale and images of typical community and habitat types. The authors observe that although the video sampling provided an initial insight into the complexity and range of biodiversity in Ningaloo Marine Park, it remained important to obtain sample specimens to characterise the uniqueness and extent of seabed biodiversity in the region. It was also noted that ocean conditions meant that it was not possible to keep the camera in 'an ideal position for filming'. This was noted again in report on the Lord Howe Island study (Speare et al. 2004) where rough conditions caused boat movement which was translated into camera movement and resolution and focus problems. Deployment in rough sea conditions was an issue in the Ningaloo study (Rees et al. 2004) where authors note that the vessel initially intended for use was a 7m research vessel, but it proved unable to operate effectively in the sea conditions and the survey was completed using a larger charter vessel (size not given).

A towed video system was also used by Australian Marine Ecology in reef surveys done for Parks Victoria at the Twelve Apostles Marine National Park (Edmunds et al. in prep) and discussed above in relation to the ROV technology used. The relatively simple and inexpensive towed video unit consisted of an underwater video camera mounted obliquely in a 1m tow fish (a hollow cylinder with vertical stabilising fins at one end and variable weights). It could be towed stably behind a vessel at speeds up to 4 knots. Depth was controlled by the length of tow rope and speed of vessel. GPS vessel position was logged by a computer.

Drop Video

Drop video footage collected in Wilson's Promontory Marine National Park was used as a part of the Deep Reef Existing Conditions Report for the Port of Melbourne Authority Channel Deepening SEES (Edmunds et al. 2006). This method involved lowering a low-resolution video camera to the sea floor. The camera, which was connected via cable to video recorder on board the vessel, was held stationary to record an image of a 25 cm quadrat. It was lifted and lowered multiple times to provide a series of images. This method was only suitable for surveys of flat reefs and also may not allow for very accurate positioning of frames along transect lines. The system does allow for consistently sized quadrats to be sampled with the camera at a set distance from the substrate but would only be suitable for sampling sessile and semi-sessile benthic organisms, as it is highly unlikely that mobile organisms such as fish would remain under the descending camera frame.

Submersible with video

Staffed submersibles have been used for assessing fish species abundance, habitat associations and assemblage types by (Gibbons et al. 2002). These authors note that submersibles have been used in other studies to describe the physical habitat and environment of benthic communities (Sibuet et al. 1998), to determine habitat use by nekton (Parker R 1986, Felley J and M 1995, Felley and Vecchione 1995) and in data collection for fish stock assessment ((Parker R 1986, Giguère M and S 1994, Giguère and Brulotte 1994). Submersibles however, are very expensive and often difficult to access compared to other methods of deploying underwater video (Cappo et al. 2004, Cappo M 2004). They are considered very unlikely to be used in any ongoing MPA monitoring program and are more likely to be used as "ships of opportunity" when in Victoria and access can be gained.

6.2 Video monitoring options for Victoria's Marine Parks and Sanctuaries

Fish Surveys

Remote video could be used for one off descriptive surveys or ongoing quantitative monitoring of fish in deep waters in Victorias MPAs. Given the accuracy and precision of stereo video systems for surveying fish transects (Cappo et al. 2002) a promising possibility for monitoring fish communities in water beyond diving depths is the use of a stereo video mounted to an ROV or towed on apparatus behind a boat. If stereo-vision video systems prove to be too expensive or impractical to deploy, single camera systems could be considered, provided a method of standardizing the field of view was available. Fish size data should not be collected in this case as fish size can't be reliably estimated from video footage. A video monitoring system should be used quantitatively to survey fixed transects, in a design similar to that used for the SRMP, or at multiple fixed points at sites inside and outside MPAs. At the very least, equipment for logging camera position on the vessel, such as that used in Edmunds et al. (2006b) or the USBL used in Holmes et al. (2007 a - e), should be used to allow accurate positioning of the camera over permanent transects or points. Powerful designs could be used as multiple transects within sites and multiple sites could be surveyed in less time than divers can conduct visual surveys in shallower waters. The potential for use of software to automate some of the laboratory analysis should also be investigated. This system would have the added advantages of providing detailed habitat information in addition to fish data, providing a permanent record that can be revisited and reanalysed if required and providing material that can be used by management agencies for illustrative, descriptive and public education purposes.

Potential problems with the use of remote video for fish monitoring may include the cost of sophisticated stereo video systems such as those described in (Cappo et al. 2002). There may also be issues with instability and lack of power and manoeuvrability of small ROV's (such as those used in (Edmunds et al. 2006b) when equipped with a large (> 1 m wide)) stereo camera system. The ongoing issue of negative or positive response of fish to the survey equipment remains a problem to be solved or at least assessed and made explicit. There may be logistical problems with towed systems when attempting to position the camera accurately enough to repeatedly survey a permanent transects or points in difficult sea conditions. The problems encountered by Speare et al. (2004) in 'keeping the camera in an ideal position') are an example although the systems used by CSIRO (discussed above) seem to have resolved these problems to some extent.

Benthic biota surveys

Remote video surveys of benthic assemblages (made up of organisms such as bottom dwelling mobile invertebrates e.g., abalone and lobster; sessile invertebrates e.g., sponges, bryozoans and hydroids; and algae) in deep waters could also be possible. As with the fish surveys, these could be one off descriptive surveys or form part of a systematic, ongoing monitoring program. It may be possible to collect benthic data and fish data simultaneously or, if this is not possible, at least collect benthic and fish data in the same boat deployment. Benthic data could be collected using less expensive single video or with stereo video (which could allow for better measurement of 3 D organisms and assessment of sample area size). The video system could be mounted on an ROV or towed. ROV Technology such as that described in (Edmunds et al. 2006b) or towed video such as used by (Rees et al. 2004) and (Barker et al. 2000) could be used to obtain video surveys of permanent transects from which images could be grabbed for quantitative analyses. The existing literature suggests that it may be difficult to develop a towed video system that allows sufficient control over camera position to accurately survey permanent transects and that finer scale control over camera position is more likely to be achieved with ROV. Provided video cameras are able to be accurately positioned then high resolution video, coupled with software allowing accurate assessment of frame size, and should allow for robust quantitative analysis of such images. Some benthic assemblages, such as those with a high abundance of large thallose and arborescent organisms and some substrates, such as highly structured reef with caves and overhangs, may be the most difficult to survey with these remote video methods. A remote controlled collection arm or dredge, for one off collection of samples of benthic organisms for ID purposes, would greatly enhance the taxonomic robustness of such monitoring. The arguments for the use of video as a compliment to visual diver surveys are also persuasive and the potential of these techniques in shallow waters could also be considered.

SECTION 7 - BAITED VIDEO FISH MONITORING

Review of baited video techniques for fish surveys

Baited video techniques have been used to count juvenile fish (Ellis and DeMartini 1995), to identify the scavengers of prawn trawl discards (Hill and Wassenberg 2000), to measure abundance of abyssal scavengers (Priede and Merrett 1996) and other deep-water species (Gledhill et al., 1996; Yau et al., 2001) and to measure the performance of marine protected areas (Willis and Babcock 2000, Willis et al. 2000). Unlike towed video and ROV video systems, Baited Remote Underwater Video Stations (or BRUVS) can be used without problems in complex topographies. There are many types of BRUVS used by different groups and researchers (e.g., Harvey and Mladenov 2000). however in this review we will only review a few to show the potential of the system. There are slight differences between systems, however from a management perspective they result in similar information, potential uses and limitations.

In the study by Rees et al. (2004), BRUVS were deployed at the same time as video tows were done in each of 3 study zones in Ningaloo MP. BRUVS were deployed with the aim of providing information on fish community composition in relation to habitat. There were 9 deployments at one zone, 6 at another and 3 at the third. Each BRUVS (three were used in the study) consisted of a camera housing made from PVC pipe with transparent acrylic front and rear sections. The housing was held within a galvanised roll-bar frame. Bait arms made from 20 mm plastic conduit held crushed pilchards in mesh bait containers and 6kg galvanised ballast weights were attached before deployment. BRUVS were deployed with 8mm, polyethylene, ropes and surface floats and flags were used to mark the position of the BRUVs . They were retrieved using a lobster pot hauler. SonyTM MiniDV Handicams (relatively simple and inexpensive cameras) with wide-angle lenses were placed in the underwater camera housings. The BRUVS were deployed to provide 83 minutes of film and were set 1km-2km apart along depth contours to ensure independence of each replicate unit,(Cappo et al. 2004).

Video footage from the BRUVS revealed 454 individuals of 52 species from 25 families over all zones. Depth, habitat class (3 habitat classes were defined from video footage: coarse sand, fine sand and 'megabenthos') and fish species abundance were analysed using multivariate techniques to examine relationships among sites, environmental variables and fish species. These analyses were used to describe the major groupings in the BRUVS data, measure the strength of fish-habitat associations, and identify indicator species defining these patterns. These multivariate analysis using classification and regression trees produced three major groupings and indicator fish species were identified for these three groups. No formal statistical tests for difference between these multivariate groups or for significance of correlations between indicator species and groups were conducted. There was also no consideration of the power of this method to detect differences in fish species or abundance over space or time.

A recent study in the Great Barrier Reef Marine Park (Cappo et al. 2004) compared the results of fish surveys using BRUVS (very similar in design to those used by Rees above) with those from surveys using a prawn trawler. The aim was to evaluate the relative performance, biases and selectivity of the video technique. Replicated comparisons of trawling and BRUV deployments were made during the day and night over different substrate types. For each fish species, the sum of the maximum number of fish sighted on BRUVS at each time was compared with the number of fish caught in trawls. The study found that the two techniques recorded significantly different components of the fish assemblage in the area studied (but note that despite sampling different components of the fauna, both trawl and BRUVS indicated similar patterns of groupings of fish in multivariate analysis). Trawls caught mainly small (<300 mm), sedentary or cryptic, demersal species while BRUVS recorded more larger, mobile species from a much wider size range of families, including large elasmobranchs, more fusiform pelagic species and eels. Species accumulation curves were similar for both techniques, but about 11 extra species were found in the trawls. Thirty-eight small mobile species in 21 families were common to both techniques, but most of these showed differences in relative abundance. Trawls recorded higher species richness at night and baited videos recorded higher species richness during the day. The authors also note that the BRUVS are most limited by water clarity and that modelling of bait plumes is needed to allow estimates of fish density from video sightings.

This study by Cappo et al (2004) suggests that BRUVS may have a particular role in studies of larger, rarer fish of special conservation interest and illustrates that different survey techniques can deliver very different data about the same fish assemblage. Another study (Willis et al. 2000) compared surveys of snapper *Pagrus auratus* (Sparidae) and blue cod *Parapercis colias* (Pinguipedidae) conducted using 3 methods: underwater visual census; experimental angling; and baited underwater video. These were done inside and outside the Cape Rodney-Okakari Point marine reserve in New Zealand. The authors found that angling and baited video provided estimates of almost 40 times greater density of fishable *P. auratus* within the marine reserve. Visual surveys, by contrast, only detected adult *P. auratus* at the reserve centre, where fish had been habituated to divers by hand-feeding. While size measurements for *P. auratus* were consistent between angling and video, mean size was significantly smaller using visual census methods. This study suggests that standard methods across all species are not always appropriate for ongoing monitoring programs, and that different survey methods should be considered according to the biology and behaviour of the species of interest. It is possible that even within practical depths for diving, alternatives to diver mediated visual census may be indicated for some fish species within the Victorian system of MPAs.

At present there is no established method for estimating fish density from stationary (baited or unbaited) video techniques. The major problems with making such density estimates include separating repeat visitors from new arrivals into the camera's field of view, estimating sampling area, and the fact that the number of fish visiting a station may reflect the other food opportunities in the area rather than fish density (Cappo 2004).

Baited Remote Underwater Video options for Victoria's Marine National Parks and Sanctuaries

Use of BRUVS in deep water to obtain fish data should be seriously considered and trials of the technique have already begun in Cape Howe MNP in a project linking habitat mapping data to fish assemblages (a project with Cordelia Moore and Euan Harvey from the University of Western Australia based on the techniques used in Cappo et al 2003 and Harvey et al 2004). BRUVS units are relatively inexpensive to build and can be deployed repeatedly in different positions. BRUVS stations should remain in place for a short fixed period (generally 1 hour), so deployment can be done during SRMP or other survey trips thus reducing deployment costs. Data from these baited video stations should be compared to that obtained from stereo video surveys to accesses whether ongoing use of both techniques would be beneficial. Results from (Cappo et al. 2004) suggest that, like diver-based visual surveys, each method allows for monitoring of a different suite of fish species. If preliminary data suggests that this is the case in Victorian waters, then ongoing periodic deployment of BRUVS should be used in conjunction with a video monitoring program to monitor fish in deep waters inside and outside MPAs with the appropriate development of associated statistical models aimed at addressing the management aims of the park.

SECTION 8 - SOFT SEDIMENT MONITORING

Current Program

Soft sediment habitats in Victoria's Marine Protected Areas encompass areas where the sea floor is silt, sand or mud with no obvious epibenthic biota. The habitats in some parks, such as Yaringa, French Island, Churchill Island, 90 Mile Beach and Corner Inlet MNPs, are virtually completely made up of soft sediments. However there are areas of soft sediments habitats and communities in every MPA in Victoria.

There is no current structured program to assess or monitor ecological parameters in soft sediment habitats in Victorian MPAs. Recently, Parks Victoria engaged the Marine and Freshwater Resources Institute at Queenscliff to use existing datasets and unprocessed survey samples to give an assessment of known conditions across the parks system. Parry et al (in press) combined available survey data and processed some unprocessed historic samples to gather basic inventory data for parks. The report includes some basic exploratory data analysis for many areas that are now MPAs. This report will be used as a basis to plan any future offshore soft sediment monitoring program.

Future Program

Should a future soft sediment monitoring program be developed it will most likely monitor infauna only. Fish associated with soft sediment areas would not initially be monitored as part of a soft sediment monitoring program. Seagrass habitats are considered under Iconic Taxa (see Section 10).

The data collated in Heislers & Parry 2007 should be used to guide the aims of any future program as well as the methods, target species and the spatial distribution of the program. An ongoing soft sediment monitoring program will need to be explicit about its aims and the link to any possible management responses. However, note that an aim may be to collect inventory and build a solid baseline of soft sediment infauna in the MPAs (that builds on current data. In this case, Parry et al (in press) and the references sited within will be useful in planning the design and spatial extent of the program.

SECTION 9 – MONITORING ICONIC TAXA

For the purposes of this report, we consider iconic taxa to be seagrass, saltmarsh, mangroves, seabirds, cetaceans (whales and dolphins), turtles and pinnepeds (seals). There is currently no structured program to assess or monitor ecological parameters for iconic taxa in Victorian MPAs.

At times, monitoring programs target iconic taxa (animals or plants) for specific management need. An example is the mapping and monitoring of saltmarsh and mangrove communities in Westernport, Victoria (e.g., Saintilan & Rogers 2001 and Rogers et al 2005 for information on the global network of SET – Surface Elevation Tables – some of which are in Westernport) or the seagrass communities within Port Phillip Bay and Westernport (e.g., Blake & Ball 2001a & b). Parks Victoria is often a partner in these programs and is likely to continue to maintain involvement at that level, aiming to add value to proposed, or on-going monitoring, programs to ensure they cover the parks or appropriate adjacent areas. Subsequently, Parks Victoria may use the data produced and analyse it in a way relevant to management aims. It is unlikely that Parks Victoria will initiate a broad scale on-going program of monitoring saltmarsh, mangrove or seagrass communities.

Other examples of monitoring iconic taxa are one-off or ongoing surveys for penguins, turtles, whales or seals. Such surveys do occur from time to time in parks however they are largely initiated as a part of a broader research project (e.g., Arnould et al 2004, Kirkwood et al 2005 or a separate program (e.g., Channel Deepening Project: Mustoe & Waugh 2006). Parks Victoria is unlikely to initiate such programs but should ensure it has access to the data and understands the extent of available information or alternatively, act as a research partner.

SECTION 10 – THE WATER COLUMN

The water column is a habitat. The 3-dimensional nature of the sea results in the water column being an important habitat for many organisms, either in transit or as the location for a permanent home. Some organisms (both flora and fauna) make their permanent home in the water column (e.g., sea jellies, salps, many fish, many plankton). Others live on and near hard surfaces but use the water column for transport and food (and other resources like oxygen).

Parks Victoria does not currently monitoring the water column as a habitat. In the future, if it did there are likely two options. The first involves monitoring the quality of the water in the water column. As an on-going program, that is most likely to be done in partnership with other organisations whose primary role is to assess water quality (e.g., EPA Victoria). However, there might be occasions where water quality is monitored as a part of the response to an incident (e.g., a pollutant spill on land nearby or in the water).

The second option involves monitoring the "condition" of the ecosystem or community that makes up the water column. This could involve monitoring the quantity and extent of pelagic organisms or their interactions. Although this could be seen conceptually as a water column version of the SRMP, it is unlikely that the "condition" of the water column communities will become a major management concern for Parks Victoria in the foreseeable future.

Section 11 - Table 21 - Proposed monitoring schedule for Marine National Parks and Marine Sanctuaries 2007-2012

Key			
SRMP		Soft Sediment/ Seagrass	
Deep Reefs		Intertidal	
Inventory (e.g., BURVS)			

	Sites	Times	Pre Declaration					November 2007	Post Declaration (Past)				Monitoring review	Post Declaration (Future)										First 10 year cycle	
		Times	1998	1999	2000	2001	2002		2003	2004	2005	2006		2007	2008	2009	2010	2011	2012	2013	2014	2015	2016		2017
Two Fold Shelf	Cape Howe					1				1		1													
	Point Hicks					1				1		1													
	Beware Reef									1		1													
	Ninety Mile Beach																								
	Corner Inlet																								
	Wilsons Promontory			1	2	2	2			1	1														
	Bunurong			1	2	2	1		1	1		1													
										1	1	1													
	Phillip Island			1	1	2	1		1		1														
	Churchhill Island																								
	French Island																								
	Yaringa																								
	Mushroom Reef								1	1	1	1													
	Port Phillip Heads		2	2	2 (ran into 2001)	1	1		1	2		1													
										1	1	1													
	Popes Eye																								
Port Phillip Bays Marine Sanctuaries	Ricketts Point								1	1	1														
	Jawbone								1	1	1														
	Point Cooke								1	1	1														
	The 3 PPB MS								1	1		1													
	Barwon Bluff								1	1	1	1													
	Point Danger									1	1	1													
	Point Addis									1	1	1													
										1	1	1													
	Eagle Rock								1		2														
	Marengo									1	2														
	Merri									1	1	1													
	Twelve Appostles																								
	The Arches																								
	Discovery Bay																								
	Inventory (especially initially via BURVS)																								

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Parks Victoria is responsible for managing the Victorian protected area network, which ranges from wilderness areas to metropolitan parks and includes both marine and terrestrial components.

Our role is to protect the natural and cultural values of the parks and other assets we manage, while providing a great range of outdoor opportunities for all Victorians and visitors.

A broad range of environmental research and monitoring activities supported by Parks Victoria provides information to enhance park management decisions. This Technical Series highlights some of the environmental research and monitoring activities done within Victoria's protected area network.

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