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Developing Integrated Performance Measures for Spatial Management of Marine Systems

David C Smith, Beth Fulton, Penny Johnson, Greg Jenkins, Neville Barrett and Colin Buxton

Final Report

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Developing Integrated Performance Measures for Spatial Management of Marine Systems

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OBJECTIVES

1. Through an analysis of monitoring data from existing marine system management regimes (including MPAs) and an identification of observational approaches that are available to be used, develop simple biophysical and management models of impact and response at various spatial scales.
2. Use these models to develop and evaluate measures to report performance for specified management objectives, particularly in respect of power to detect change.

NON-TECHNICAL SUMMARY

Outcomes Achieved

This study is, as far as the authors' are aware the first 'whole of system' approach to performance evaluation of spatial management. The results of this study should improve effectiveness and efficiency of spatial management, through development of better performance assessment methods. More importantly, is should lead to greater uptake of spatial management approaches to achieve ESD for marine resources and ecosystems.

An important outcome of the project has been that our model (Atlantis-SM) and what we have learned from its development and application, provides a "transportable" framework for developing performance measures. The basic approach, both the telescoping treatment of habitats in and around spatial closures and the management strategy evaluation framework for representing the estimation of indicators, can be applied to other systems via new implementations of the Atlantis framework.

However, even without going that far, it is possible to take the lessons learnt in this case to other systems. In particular it contributes to the policy debate around the implementation of EBM. It highlights the potential for monitoring for EBM performance to be far from simple. Monitoring schemes with small spatial coverage or infrequent temporal repetition (on the order of 3-5 years or more) had no power to rapidly detect changes at the broader system level. In addition, the finding that there is likely to be system specific reference points is particularly important, as the tendency within the literature has been to try and find generic rules and approaches for universal application and broad scale comparisons. The indication from this study that universal reference points (analogous to $B_{0.4}$ in fisheries) or directions, which do not take into account local specificity, may not be feasible has important consequences as is at odds with recent literature and suggests that adoption of such approaches could lead to a very misleading interpretation of management performance.

There is increasing interest in the spatial management of marine systems worldwide and it is seen as a crucial step towards implementation of ecosystem-based management. This has seen a growing focus on managing marine systems at various spatial scales and assessing the relative roles of different spatial components to large marine systems as a whole. However, the scientific basis for the spatial management of marine systems is limited, although spatial area management, as illustrated by Marine Protected Areas (MPAs), has received considerable attention as the ‘new’ tool to control over-exploitation of fish stocks. To properly evaluate the potential merit of spatial management, there is a distinct need for such approaches to have clearly stated objectives, meaningful indicators and effective monitoring of performance with respect to management objectives. Where performance assessment has been undertaken, it is usually focuses on examining the consequences in the immediate area of the spatial management zone (e.g. what accumulates in a closed area) rather than examining the system-wide effects and benefits. In addition, while an enormous number of candidate ecological indicators have been proposed in the literature, these are generally at the whole of system level rather than being spatially explicit.

In this study we reviewed the available information on monitoring for spatial management and associated performance measures, for programs both in Australia and overseas. Overseas monitoring programs reviewed were from the Philippines, the Caribbean, Indonesia, California, New Zealand, South Africa, Kenya, France and Ecuador. Australian programs reviewed were from Queensland, Tasmania, New South Wales, Victoria, and the Great Australian Bight. The majority of these programs were associated with spatial management of marine protected areas (MPAs). In addition the review considered monitoring for social and economic objectives of spatial management, and observational approaches for the spatial management of marine systems. A key outcome from the study into performance measurements of spatial management are the implications for monitoring designs.

The Atlantis modelling framework provided the basis for a model developed explicitly for this study; Atlantis-SM. It was calibrated using time series data from Victoria and Tasmania and was able to spatially simulate MPAs in the south east of Australia. It was developed to evaluate indicators at various spatial scales and how well they perform under a range of specifications and scenarios. We do not address whether or not there should be MPAs, rather the model is designed to develop an effective means to assess the performance of indicators of the system and the spatial management within it. The rationale for this focus is that no-take MPAs are likely to show the strongest contrast in the influence of human activity and so would contain the greatest potential differential and signal strength. If indicators are not effective in evaluating performance here they are unlikely to be useful in other forms of spatial management. To the authors' knowledge it represents the first such 'whole of system' study undertaken on appropriate indicators for assessing the performance of spatial management.

We applied a 3x4x4x4* (productivity x MPA size x sampling schemes x impact type, with the * indicating that one of the impacts (fisheries) was also considered at 3 levels) matrix of specifications and scenarios to assess indicator robustness (ie how well they perform under different conditions), giving 432 individual outputs. The other impacts considered included climate change, nutrients, and illegal, unreported and unregulated (IUU) fishing.

As we were not addressing a specific management objective we drew upon indicators commonly used to address a range of spatial management objectives. The indicators evaluated were drawn from previous studies on ecological indicators and from the results of a literature review undertaken as part of this study. We also chose indicators that could be feasibly calculated and tested in the Atlantis-SM model, which cover the majority of indicators that can be feasibly and repeatedly measured in reality. These indicators cover the primary indicators used to date to monitor MPAs, and the recommended set from past studies of ecological indicators of the effects of anthropogenic impacts (especially fishing).

Compared with previous studies, the indicators checked show that, in broad terms, overall indicator performance still holds. However it also highlighted that monitoring for EBM performance may be far from simple. While sampling schemes of low frequency or spatial coverage are acceptable for detecting change inside and outside closures (also needing a reasonable time series to enable causes of the signal to be evaluated) they have little power to detect signals at broader spatial scales. Monitoring schemes with small spatial coverage or infrequent temporal repetition (on the order of 3-5 years or more) had no power to rapidly detect changes in the system; while intensive sampling was confounded by natural system variation and shifts through time, unless carefully planned around stratified sampling schemes. Moreover, indicators, such as pelagic:demersal fish biomass, that have been found to be useful across different ecosystem types proved sensitive to scale. These indicators were informative in the immediate area of closures (as the data at this scale is within habitat patches and individual species ranges and so avoid species-scale mis-matches) and globally (because at such large scales the ratio integrates across many species effectively smoothing out any potential mis-matches). However, they do not work at intermediate spatial scales because these exceed the typical spatial range of activity of individual species, but are not yet at a point where they smoothly integrate across sufficient groups.

The ecology of the groups in the system also impacts the performance of individual indicators based on those groups. For example, signals for mobile species can be over-stated outside reserves, while signals for more sedentary species decay rapidly with distance from the closures.

Atlantis-SM also suggests that variation in community dynamics between regions can lead to locally specific indicator-attribute relationships; meaning that while indicator signals are representative of the attribute at a specific locale, they may not always be consistent site-to-site. For instance the relationship between the indicator "relative lobster biomass" and the attribute "diversity" was linear (with $R^2 > 0.92$), but in opposite directions (in one case there was a positive correlation and in one a negative) at sites less than 300km apart. This difference in direction of response is due to locally specific environmental drivers and community dynamics and has significant implications for monitoring and management, as it shows that an understanding of system dynamics at regional scales will be necessary to understand the signal obtained from indicators. This suggests that universal reference points (analogous to $B_{0.4}$ in fisheries) or directions, which do not take into account local specificity, may not be feasible. This finding is at odds with recent literature on indicators which not only recommends a definitive set of indicators across many systems and scales but also recommends the use of reference points that are intended to be consistent across systems. Instead, suites of indicators drawn from the main general classes of indicators noted above (e.g. relative biomass, biomass ratios, relative habitat cover) will need their associated reference points or directions adjusted to suit status and processes at the locations of interest (and potentially through time as the system changes). Crucially, this also means that a lack of a temporal dimension in monitoring cannot be completely compensated for by periodically applying very intensive surveys across broad spatial scales.

The results of the study indicated that fisheries dependent indicators should not be used alone unless there is absolutely no alternative (industry independent data is much preferred). Fishery independent surveys using commercial or research vessels (often using trawl or other extractive methods), are commonly used around the world. However in some spatial zoning arrangements, such as no-take MPAs, extractive sampling methods may be prohibited. It has been argued that by using a combination of (non-extractive) observational techniques (eg underwater visual census, video, BRUVS, acoustics, 'smart tags' etc) to target specific species or habitats, spatial monitoring surveys can provide information on the whole ecosystem. While this might be the case, the efficacy of many of these methods for the sustained observing required to monitor marine systems has still to be demonstrated.

Finally, these are strong ecological reasons why a suite of indicators will be needed to capture performance of spatial management. When moving to triple bottom line objectives, the inclusion of social and economic objectives is still more reason for use of a broad suite of indicators to cover all aspects of the system (and triple bottom line objectives).

Keywords:

Spatial management, indicators, performance measures, management strategy evaluation, ecosystem model

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Background

There is increasing worldwide interest in the spatial management of marine systems and it is seen as a crucial step towards implementation of ecosystem-based management (Douvere and Ehler 2008). This has seen a growing focus on managing marine systems at various spatial scales and assessing the relative roles of different spatial components to large marine systems as a whole. However, the scientific basis for the spatial management of marine systems and the potential contribution of a given area of marine-space (eg a Marine Protected Area, MPA) is limited (Smith et al 2004). Although spatial area management, as illustrated by MPAs, has received considerable attention as a 'new' tool to control over-exploitation of fish stocks (eg Pauly et al 2002), particularly where the tools are absent or failing[1]. For example, the potential benefits to fisheries are often listed to be: to increase the spawning biomass, to act as an insurance policy against fishery management errors, to protect critical habitats, to damp ecosystem wide fluctuations and to provide reference sites to be used in fishery resource assessments. However, while some studies have shown that the harvesting regimes for specific areas within a system can change biomass, density, size of organisms, quality of habitats and species diversity, the causes of the extent and nature of these changes at various spatial and temporal scales impedes the selection of performance indicators.

Ward et al (2000) states that there are "...very few examples where benefits to a fishery (as opposed to the closed area) have been well studied and documented". Similar conclusions are drawn in other reviews currently appearing (e.g. Halpern in press). Socio-economic impacts are even less well studied (Sanchirico 2000).

Clearly, spatial management, for example as expressed by MPAs, is here to stay. For example FRDC suggests that nations will set targets such as 20% of the coastal zone for high degrees of protection through MPAs (FRDC R&D Plan 2000-2005). The objectives of MPAs are usually to support the achievement of ESD for the regional ecosystem and for the various sectoral users of the ecosystem.

The importance of specific objectives, meaningful indicators, monitoring and evaluation for spatial management has been stressed (Day 2008; Gilliland and Laffoley 2008, Jenkins et al 2008, Appendix E). However, where performance assessment has been undertaken it is usually focuses on examining the consequences in the immediate area of the spatial management zone (eg what accumulates in a closed area with reference to nearby fished areas (Babcock et al 2010, McClanahan 2010)) rather than examining the system-wide effects at a broader scale and benefits at a broad scale where that was the aim of introducing the spatial management measures (eg protection of the breeding stock or maintaining ecosystem biodiversity). Whereas the use of other management tools includes the use of performance assessment and the triggering of management responses under different circumstances, spatial management has often been put in place with no direct means of assessing its performance (Pomeroy et al 2005) or specific management goals. There is typically no analogy to the system of periodic review seen in industries such as fisheries (where there is evaluation of the efficacy of management actions based on consideration of indicators such as catch or fishing effort levels in relation to stock condition) (Pomeroy et al 2005). In many instances key uncertainties remain about spatial management approaches and these are a

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[1] Clearly, here we are not discussing the fishery spatial management tools that are widely used for fishery specific objectives, for example protecting vulnerable life history stages such as nursery areas for juveniles and adult spawning aggregations.

significant constraint on decision making to achieve ecosystem based management objectives.

While ecological or ecosystem indicators have been proposed particularly in relation to managing the effects of fishing within an EBFM context (Fulton et al 2005a, Link 2005, Shin et al 2010), the focus has primarily been at the system-level. What we are interested in here is how well do such indicators or others perform at various spatial scales and can they be used to inform spatial management.

Appropriate tools are required to support the effective management of fisheries and biodiversity at diverse spatial scales. Ecosystem models are one tool that has seen increased use over the last decade as a means of capturing the dynamics of large marine systems, both at the local and regional scale. In the original project design it was anticipated that prototype models would initially be developed, mostly based on existing modelling frameworks, such as the EwE models (Ecopath with Ecosim software suite, Walters et al 1997, Christensen et al. 2000), or the 'In Vitro' model originally developed for the Australia's North West Shelf (NWSJEMS 1999). However, the Atlantis modelling framework (Fulton et al. 2004) was ultimately selected as:

1. a south east Australia implementation of this model already existed (which had been used to consider alternative management strategies in Commonwealth fisheries in southern Australia (Fulton et al. 2007));
2. the modelling platform is extremely flexible (and so could easily span a wide range of scenarios and strategies);
3. it was already set-up for consideration of indicator performance (as Fulton et al 2005 used it as the basis of research into ecosystem indicators for the effects of fishing). In addition, Atlantis is now widely regarded as the leading marine ecosystem model available for ecosystem-level evaluation of management strategies (Plaganyi 2007).

In this study we use the Atlantis modelling framework (Fulton et al. 2004) to evaluate indicators at various spatial scales and how well they perform under a range of specifications and scenarios. In this approach we use actual coastal no-take MPAs in south eastern Australia to provide spatial contrast. We do not address whether or not there should be MPAs, rather our approach is designed to develop an effective means to assess the performance of the system and the spatial management within it. The rationale for this focus is that no-take MPAs are likely to show the strongest contrast in the influence of human activity and so would contain the greatest potential differential and signal strength. If indicators are not effective in evaluating performance here they are unlikely to be useful in other forms of spatial management.

Need

The need to report on the ecologically sustainable use of marine systems, which have been 'zoned' at a variety of spatial and temporal scales, is gaining considerable support in Australia and world-wide. Spatial management is seen as a major role in ecosystem based management and yet it is argued that what is missing is a clear demonstration of how it can be implemented (Ehler 2008). What is needed are specific objectives, meaningful indicators, monitoring and evaluation for spatial management (Day 2008; Gilliland and Lafolley 2008, Jenkins et al 2008, Appendix E). An enormous number of candidate ecological indicators have been proposed in the literature (e.g. Rochet and Trenkel 2003, Trenkel and Rochet 2003, Fulton et al 2005, Link 2005, Rice and Rochet 2005, Rodionov and Overland 2005) but these are generally at the whole of system level rather than spatially explicit.

In addition, in response to a perceived failure of traditional fisheries management, there are increasingly frequent calls for widespread use of MPAs (primarily no-take zones) as fisheries management tools (eg Roberts and Hawkins 2000, Gell and Roberts 2002, Pauly et al. 2002, Worm et al. 2006). The veracity of such calls can be debated but closures are seen as an integral component of modern fisheries management.

MPAs and other spatial management arrangements are being introduced in most Australian management jurisdictions through the NRSMPA process and at the Commonwealth level, this is being implemented through Marine Bioregional Planning under Oceans Policy. In Australia, MPAs are primarily being introduced as a precautionary measure to protect biodiversity from a range of potential impacts, although flow-on fishery benefits are often claimed, and their reference role for informing fisheries management of EBM and ESD issues is evolving.

Outcome oriented assessment of performance has been rare and superficial for spatial management measures (Day 2008; Gilliland and Lafolley 2008), even for those that have had widespread use in fisheries for many years. There is also limited assessment of this for more recently introduced measures such as no-take MPAs (eg, Ward et al. 2001, Sainsbury and Sumalia 2003, Jenkins et al 2008 Appendix E).

In framing management objectives, many agencies have considered a relatively small spatial scale, associated with individual MPAs and adjacent areas. However, it is not yet clear under what circumstances specific areas within large systems contribute to the system as a whole, nor is it clear how large system behaviour in turn influences areas within it.

Without such performance assessment, managers and resource users may become locked into sub-optimal management arrangements, and if MPAs and other area management arrangements are not working as intended, then achieving goals such as Ecologically Sustainable Development may be unknowingly at risk. Consequently, even with objectives that are clearly defined and agreed by all stakeholders, the most challenging work still remains as how to evaluate performance.

This project does not address whether or not there should be MPAs rather it is designed to develop an effective means to assess the performance of spatial management using MPAs as the focus for evaluating the performance of a range of indicators under different specifications and scenarios.

Objectives

1. Through an analysis of monitoring data from existing marine system management regimes (including MPAs) and an identification of observational approaches that are available to be used, develop simple biophysical and management models of impact and response at various spatial scales.
2. Use these models to develop and evaluate measures to report performance for specified management objectives particularly in respect of power to detect change.

Methods

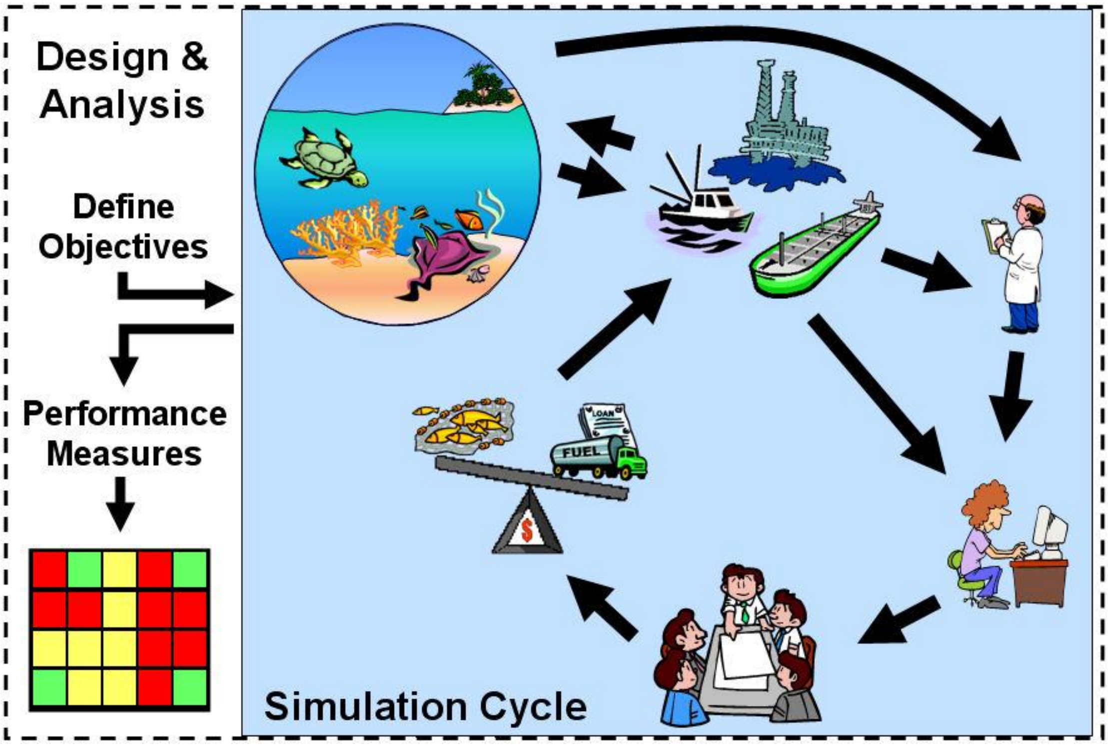
Atlantis Overview

Atlantis is an end-to-end ecosystem model that includes modules for each of the major steps in the adaptive management cycle (Figure 1) – biophysical, industry, monitoring, assessment, management and socio-economic aspects (Fulton et al 2004, Fulton et al 2007, Fulton et al

2010). Monitoring and assessment are key components of any adaptive management approach, including EBM. Atlantis has frequently been used to evaluate these two components of management, as management strategy evaluation (MSE) testing of indicators is an effective means of evaluating indicator performance under a range of circumstances (e.g. alternative levels of fishing, climate forcing or cumulative pressures). These evaluations compare indicators calculated from "sampled" data with "true values" of key variables (attributes) of interest from the Atlantis biophysical sub-model.

At the core of Atlantis is a deterministic biophysical operating model, coarsely spatially-resolved in three dimensions, which tracks nutrient (usually Nitrogen and Silica) flows through the main biological groups in the system. The primary ecological processes modelled are consumption, production, waste production, migration, predation, recruitment, habitat dependency, and mortality. The trophic resolution is typically at the functional group level. Invertebrates are typically represented as biomass pools, while vertebrates are represented using an explicit age-structured formulation. The physical environment is represented explicitly, via a set of polygons matched to the major geographical and bioregional features of the simulated marine system. Biological model components are replicated in each depth layer of each of these polygons. (Fulton et al in press).

a)



b)

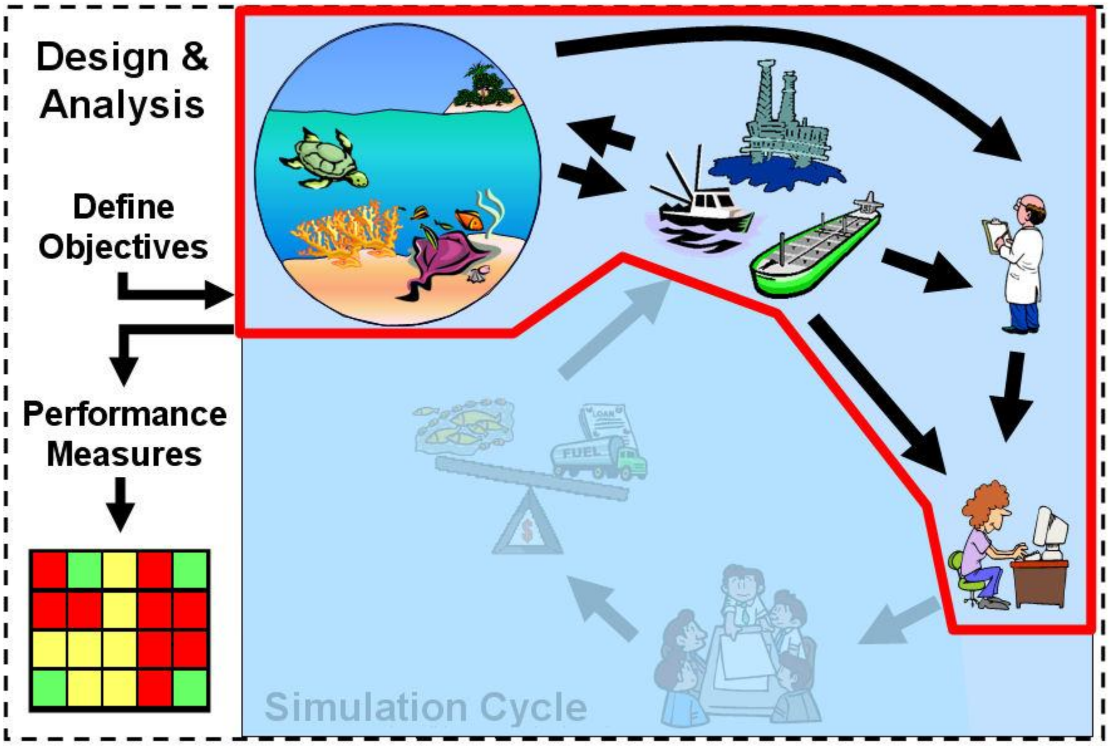


FIGURE 1. A) THE ATLANTIS SE MANAGEMENT STRATEGY EVALUATION MODELLING FRAMEWORK, B) THE VARIANT USED FOR THE CURRENT PROJECT

Atlantis Spatial Management (Atlantis-SM)

Atlantis-SM was developed specifically for this study (Johnson et al 2010, Appendix F). It is a variant on Atlantis SE (Fulton et al 2007) but with a greater focus on enabling resolution of inshore habitats at finer spatial scales. In addition, because here we are interested in assessing the performance of indicators, there is a stronger emphasis on the biophysical components of the system compared to the management and socio-economic components.

The Atlantis-SM domain consists of approximately 640,035 km^2 off the south east coast of Australia. The domain extends from the NSW/Victorian border to just west of Port Phillip bay, and around the eastern coast of Tasmania, including Bass Straight (Figure 2). Soft sediment habitats, including sand, mud and seagrass form the dominant component of the area’s inshore environment. Rocky reefs and kelp forests also play important roles in the ecology of the region. Offshore the model incorporates both shelf and open ocean environments. Ecologically, the area is highly diverse and contains a high proportion of endemic species (Coleman et al. 2007). The area supports significant commercial fisheries; abalone and rock lobster, squid, small pelagic, scallop, and Danish seine and trawl fisheries and significant coastal recreational fisheries (Smith and Smith 2001).

The spatial geometry is represented using a physical transport model based on a polygonal box geometry of 80 spatial boxes (see Figure 2) with up to 7 layers per box. Physical properties (e.g. depth, seabed type, porosity, salinity, and temperature) are defined for each box (and may change dynamically through time) and transport between boxes (or water column layers within a box) represents processes such as advection, diffusion, settling, mixing and re-suspension (Fulton et al. 2004). Currents, temperature and salinity were incorporated as per the Atlantis SE model study (Fulton et al 2007).

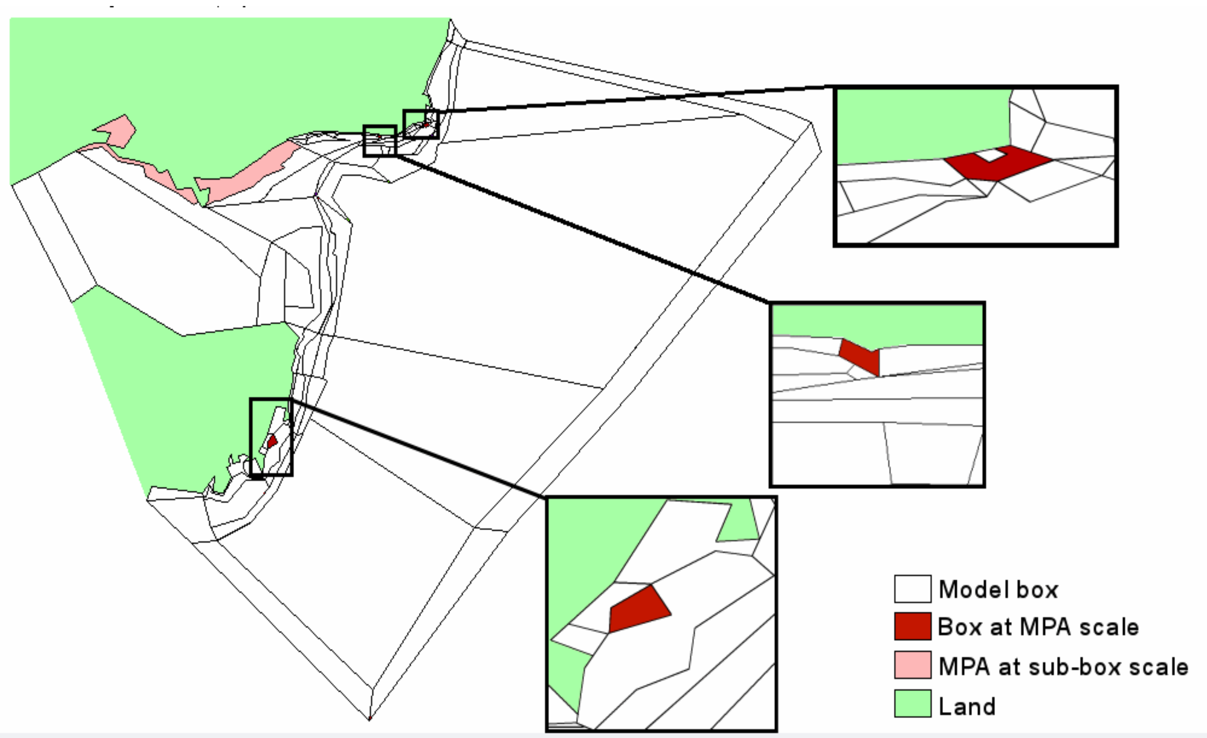


FIGURE 2. ATLANTIS-SM DOMAIN: VICTORIAN AND TASMANIAN COASTS SHOWING THE ATLANTIS-SM SPATIAL STRUCTURE AND MPAS OF INTEREST

A novel telescoping geometry was used to explicitly include the Point Hicks, Cape Howe and Maria Island Marine National Parks at their native resolutions and gradually build up to the larger boxes applicable along the rest of the shelf and offshore (Figure 2) (Johnson et al 2010, Appendix F). Other MPAs are dealt with at the sub-box scale, shown in Figure 2 in grey. These MPAs were not all treated explicitly, because their small size would have made explicit handling exceptionally computationally difficult. It was felt that treating the three larger MPAs explicitly and capturing other smaller MPAs using distributions within boxes would be the most effective means of representing the system. This still allows for inside and outside reserve responses in biomasses, animal size and ecological functioning to be captured with sufficient clarity to address the study questions.

The biological components of the model provide a representation of the entire foodweb; inshore and offshore, pelagic and demersal and from bacteria and phytoplankton up to top predators. The model includes 3 types of detritus, 3 types of primary producer, 21 invertebrate and 31 vertebrate groups, some represented at the species level and others as functional groups (Table 1 shows the complete list along with initial biomass for each group, further details of the life history of the animals can be found in Appendix C). The vertebrate, abalone, lobster, cephalopod and prawn groups are age-structured, but all other groups are handled as biomass pools. Those groups represented as functional groups were aggregate groups of species with similar size, diet, predators, habitat preferences, migratory patterns and life history strategy. In addition to these living biological groups, pools of ammonia, nitrate, silica, carrion, labile and refractory detritus are also represented dynamically.

Data for biological parameters such as initial abundance, seasonal distribution, fecundity and timing of reproduction, growth and habitat preference, were obtained from a variety of sources including: the databases of the Central Ageing Facility, Fisheries Victoria; unpublished data (Fisheries Victoria, TAFI), the Fishbase database (www.fishbase.org); re-parameterised from ecosystem models that encompassed the study domain (Fulton et al. 2007) and literature on the region (Kuiter 1993, Gomon et al. 1994, Edgar 1997, Taylor & Willis 1998, Edgar & Barrett 1999, Greely et al. 1999, Ewing et al. 2003, Edgar et al. 2004, Lyle et al. 2004, Barrett et al. 2007). As a single set of biological parameters is used across the model domain, fitting of the model must be done simultaneously across each group and spatial area. A detailed description of the biological components and parameters is given Johnson et al (2010, Appendix F)

Model calibration is presented by Johnson et al (2010, Appendix F). Time series trajectories of both biomass and abundance of reef fish and some invertebrate groups were constructed from available observational data (Gilmour et al. 2005, Barrett et al. 2007). These time series showed 10 year biomass trajectories for the reef species, the first 6 years in these data sets were used to calibrate the trajectories of the reef groups in our model. For the groups where no time series data were available (e.g. the off-shore pelagic groups), biological parameters were calculated simply to achieve a stable ecosystem within the range of biomass values reported for these groups in the literature. Care was taken during calibration not to take any parameter beyond the plausible range of values defined by the literature (where available) or expert advice from researchers active in the region.

TABLE 1. FUNCTIONAL GROUPS IN ATLANTIS-SM AND THEIR INITIAL BIOMASS VALUES

Model Component	**Group Composition**	**Initial Biomass (t)**
Large phytoplankton	Diatoms	4975869
Small phytoplankton	Picophytoplankton	4448839
Gelatinous zooplankton	Salps, coelentrates	95029941
Large zooplankton	Krill and chaetognaths	2930676
Mesozooplankton	Copepods	1901186
Small zooplankton	Heterotrophic flagellates	95108706
Carvivorous infauna	Polychaetes	5417350
Deposit feeders	Holothurians, echinoderms, burrowing bivalves	2.42E+08
Deep water filter feeders	Sponges, corals, crinoids, bivalves	500249
Shallow water filter feeders	Mussels, oysters, sponges, corals	64833
Urchins	Echinoidea	6914
Deep water megazoobenthos	Crustacea, asteroids, molluscs	1500159
Shallow water megazoobenthos	Stomatopods, octopus, seastar, gastropod, crustacea	103891
Meiobenthos	Meiobenthos	6299170
Macroalgae	Macroalgae	417926
Seagrass	Seagrass	21352
Squid	*Sepioteuthis australis*, *Notodarus gouldi*	94893
Shallow water herbivores	e.g. *Girella tricuspidata*, *Liza argentea*, *Dactylophora nigricans*	51114
Banded morwong	*Cheilodactylus spectabilis*	511
Shallow demersal fish	*Pagrus auratus*, Labridae, *Chelidonichthys kumu*, *Pterygotrigla*, *Sillaginoides punctata*, *Zeus faber*	200654
Planktivorous reef fish	eg. Atypichthys strigatus, *Enoplosus armatus*, *Trachinops caudimaculatus*	8020
Deep demersal fish	Oreosomatidae, Macrouridae, *Zenopsis*	68611
Zebra fish	*Girella zebra*	824
Silver sweep	*Scorpis lineolata*	284
Magpie perch	*Cheilodactylus nigripes*	584
Seahorses, pipefish, gobies	Syngnathidae, Gobiidae	11011
Herring cale	*Odax cyanomelas*	1823
Purple wrasse	*Notolabrus fucicola*	2978
Blue throat wrasse	*Notolabrus tetricus*	3817
Blue-eye trevalla, warehou	*Hyperoglyphe antarctica*, *Seriolella*	13919
Small pelagic fish	*Engraulis*, *Sardinops*, sprat	142795
Mackerels	*Trachurus declivis*, *Scomber australisicus*	34906
Shallow piscivores	e.g. Sphyraena novaehollandiae, *Arripis truttacea*, *Pomatomus*	281896
Migratory mesopelagics	Myctophidae	173976
Non-migratory mesopelagics	Sternoptychids, cyclothene	403340
Pink snapper	*Pagrus auratus*	30656
Tunas and billfish	*Thunnus*, *Makaira*, *Tetrapturus*, *Xiphias*	15158
Dogfish	Squalidae	476245
Demersal sharks	*Heterodontus portusjacksoni*, Scyliorhinidae, Orectolobidae	290432
Large Pelagic sharks	*Prionace glauca*, *Isurus oxyrunchus*, *Carcharodon carcharias*, *Carcharhinus*	106812
Dogshark	*Centrophorus*	184953
Skates and rays	Rajidae, Dasyatidae	17672
Baleen whales	*Megaptera novaeangliae*, *Balaenoptera*, *Eubalaena australis*	4423
Dolphins	Delphinidae	675
Orcas	*Orcinus orca*	1028
Seals	*Arctocephalus pusillus doriferus*, *Arctocephalus forsteri*	643
Abalone	*Haliotis*	5510
Prawns	*Haliporoides sibogae*	390124
Lobster	*Jasus*	21646
Seabirds	Albatross, shearwater, gulls, terns, gannets, penguins	261

The Sampling Sub-Model

The sampling model in Atlantis generates simulated data that can (in turn) be used to calculate indicators. This simulated data generated by the sampling model has components for both sector dependent and independent data types. This data is generated with realistic levels of measurement uncertainty – evaluated as bias and variance (see Tables D1 and D2 in Appendix D) that covers sampling and handling uncertainty. These simulated data are based on the outputs from the biophysical and exploitation sub-models, using a user-specified monitoring scheme. The monitoring scheme specifies the precision and spatiotemporal coverage of the data collection for each data type and source. A range of data collection methods are represented, including fisheries statistics, fishery independent surveys (eg., dive or fishing surveys), mammal and bird surveys, diet information, oceanographic surveys or moorings collecting physical and chemical properties (Fulton et al. 2004).

The boxes within Atlantis are homogenous so there is no point in repeatedly sampling the same box. Instead the process of running multiple transects or survey cruises in a geographic area in reality is represented using a combined error distribution (with error terms drawn from this distribution). Sampled individuals are sub-sampled to collect length, age, growth, consumption and diet information – with appropriate errors added with each additional data handling step (just as in reality) (Fulton et al. 2004).

After base sampling and indicator calculations, the results are pooled across user-defined zones, to ease comparison of results at scales relevant to the study in question. In this case local, regional, broad and global scales (see Figure 3).

The key benefit to simulation testing indicators is that the true state of the system is known. This means the performance of the indicators can be evaluated by comparing the indicator values (and their trends) against the true values (and trends) for the main system characteristics of interest. To this end attribute (system) values are also recorded directly from the operating model by the sampling model.

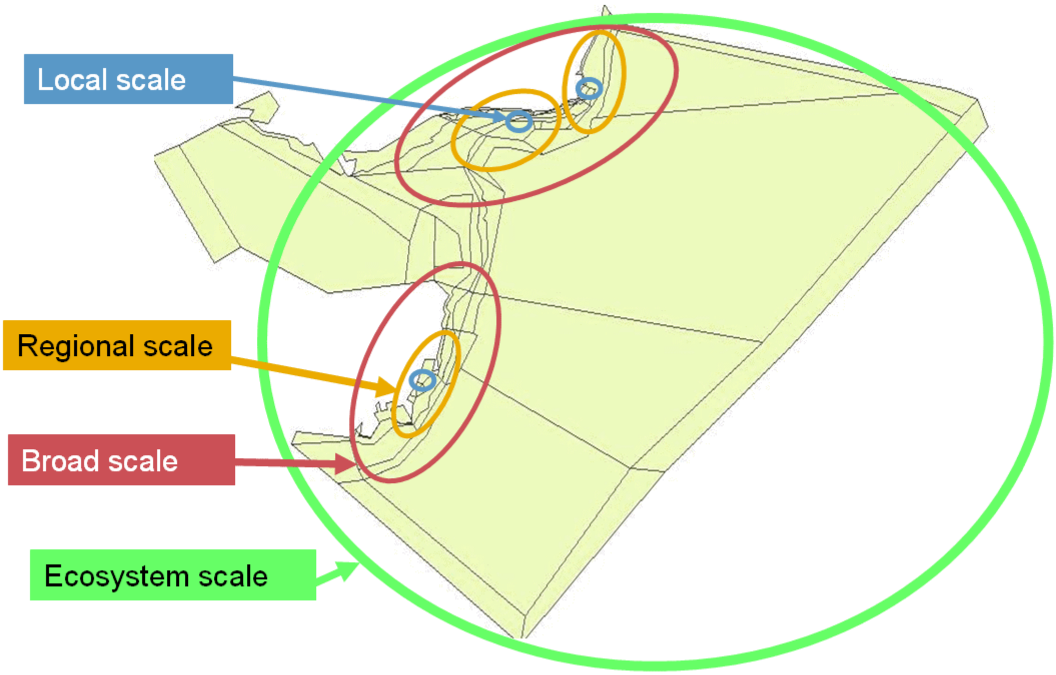


FIGURE 3. SPATIAL SCALE OF AGGREGATE SAMPLING ZONES USED IN THE ANALYSIS

Specifications and Scenarios

Given the complexity of ecosystem models, systematic 1-factor at a time sensitivity analyses are inappropriate (due to feedbacks in the system) and computationally infeasible. Consequently we applied a 3x4x4x4* matrix of specifications and scenarios to assess indicator robustness, where the dimensions of the matrix are productivity x MPA size x sampling schemes x impact type, with the * indicating that one of the impacts (fisheries) was also considered at 3 levels. This approach has previously been used quite effectively with system-level MSE (Little et al 2005, Fulton et al 2007).

As the size and complexity of the model precludes a traditional sensitivity analysis uncertainty in system state and response was represented using a bounded set of alternative parameterisations. During the calibration of the model (Johnson et al 2010, Appendix F) a small set of parameters were found to meet the fitting criteria equally well. These became the basis of one of the specification dimensions of this matrix of alternatives and were expanded to cover a broader set of potential productivity and vulnerability levels under the direction of experts in the system (including co-authors of this report, Jenkins and Barrett). The alternative specifications were referred to as the high productivity version, low productivity version and medium productivity version. The latter represents our best understanding of the real world. The high and low productivity models represented the outer levels of system productivity that were described in the literature, and as such were used as a type of 'confidence interval' for the ecosystem state. The 'high' level is defined as the system where each part of the system is at the upper bound of confidence levels from the literature and with the fastest rates of recovery; similarly the 'low' level is when all groups are at their lowest level from the literature with the slowest rates of recovery.

The sensitivity of indicators to different spatial scenarios was assessed by considering four MPA sizes:

1. current MPA size (1-5% of area)
2. large MPAs that covered approximately 20% of the shelf area
3. intermediate size (10%), and
4. no MPA protection at all.

We investigated four sampling strategies (Figure 4):

1. Intensive: This was the most intensive sampling regime with a monthly sampling frequency and a spatial range covering over half the boxes in the model domain.
2. Periodic Snapshots: This sampling regime had the same spatial coverage as the intensive regime, however the frequency of sampling was vastly reduced to only once every 5 years
3. Low: This sampling regime is the closest to the current level of sampling that is occurring to monitor the Victorian MPAs. The sampling occurs only inside and immediately outside each MPA, at a frequency of twice yearly.
4. Mixed: This sampling regime is a combination of the Periodic Snapshots and Low sampling strategies. The low sampling regime is run, with the higher spatial coverage sampling imposed over the top every 5 years.

Atlantis includes a detailed industry (or exploitation) component. This model deals not only with the impact of fishing but also with pollution, coastal development and broad-scale environmental (e.g. climate) change. All forms of fishing may be represented, including recreational fishing (which is based on the dynamically changing human population in the area).

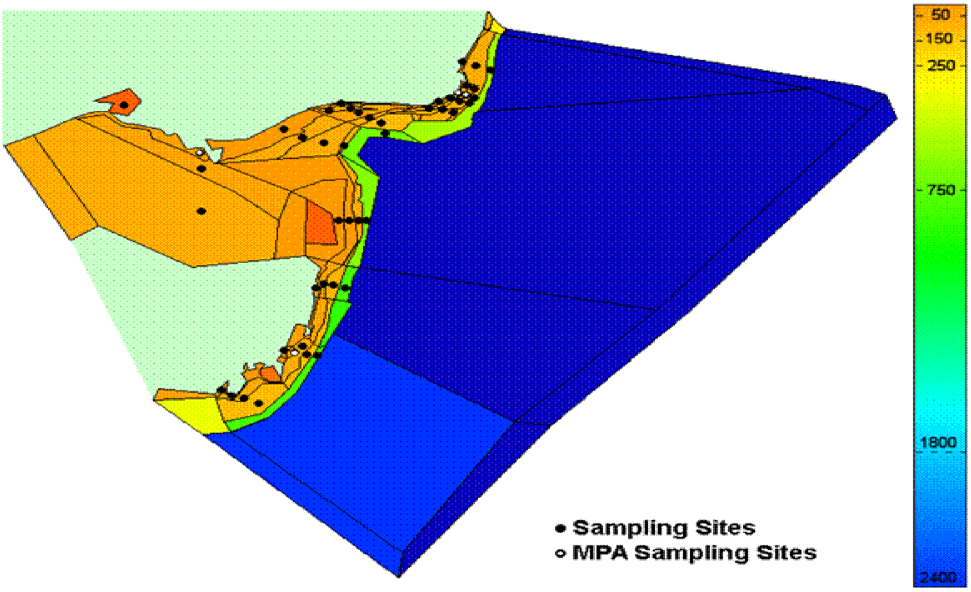


FIGURE 4. EXAMPLE OF INTENSIVE SAMPLING DESIGN

Fishing was imposed at about the current level, at 5 times this level and at half this level. A fishing mortality (F value) was estimated for each fished group by setting F to the proportion of the total population of each group that was taken as catch in 2004. This fishing pressure was imposed based on estimates of the 2004 rates of fishing by both federal and state fleets (Anon 2004, Smith & Waytes 2004). These F estimates were refined during the calibration process so as to achieve a stable model biomass (i.e. with no evidence of numerical instability) that matched the available biomass trajectories that have been observed in the ecosystem over the past 10 years (Gilmour et al. 2005, Barrett et al. 2007).

In addition to fishing, the following impacts were considered in order to test the robustness of indicators to:

– Illegal, unreported and unregulated (IUU) fishing in the MPAs - fishing pressure within the MPA reaches 30% of that applied outside the MPA

– Climate change - a temperature increase of 5ºC (this matches the upper level of increase anticipated under climate change and increasing strength of the East Australian Current). This impacts biota through temperature dependent physiology. Note the temperature increase is extreme over the time scale evaluated but chosen on purpose for signal detection.

– Nutrients - a point source input of nutrients was added close to the MPAs to represent outflow from hypothetical tourism ventures that could grow up around the park, as well as additional coastal development in the area. The nutrient scenarios were created by scaling the time series of inputs to the model domain - so the pattern of the nutrient supply from rivers and upwelling remained the same (both spatially and temporally), but the magnitude was increased/decreased to cover scenarios of gross productivity change (due to changed physical supply of nutrients).

– Gauntlet fishing - produced by increasing fishing pressure beyond background levels only on the margins of the spatial management zones. As fishing pressure was only represented via application of a fishing mortality rate, complex spatial effort allocations were not dynamically produced, Consequently, to reflect the behaviour observed in other real world systems after the introduction of spatial closures, where fishing pressure was concentrated on the boundary of spatial closures, fishing pressure along the margins of the closures in the model was scaled (>1) relative to the fishing pressure applied elsewhere in the system. Typically the fishing pressure along the boundary was at least twice that seen elsewhere, but could be as high as 5x higher.

Simulations were run over a 20 year period for each possible combination.

Selection of Indicators

Indicators are quantities of interest that reflect system attributes or a component of management performance. In this study we were not addressing a specific management objective so we drew upon indicators commonly used to address a range of spatial management objectives. The indicators evaluated were drawn from Fulton et al (2004, 2005, 2007) and Link (2005) and from the results of a literature review undertaken as part of this study (Jenkins et al 2009, Appendix E). We also chose indicators that could be feasibly calculated and tested in the Altantis-SM model, which cover the majority of indicators that can be feasibly and repeatedly measured in reality. These indicators cover the primary indicators used to date to monitor MPAs, and the recommended set from past studies of ecological indicators of the effects of anthropogenic impacts (especially fishing). The complete set of indicators tested are listed in Table 2 and include the following broad types:

relative biomass, structural, network, diversity, water quality and industry (where the indicator total value is the sum of catch multiplied by price across all groups).

TABLE 2. SUMMARY OF INDICATORS EVALUATED IN THIS STUDY

Type	Indicator
Relative biomass	
	fish and invertebrate species of interest
	habitat associated fish, demersal fish, medium, piscivore
	demersal shark, pelagic shark, small pelagics, zooplankton
	TL4+ biomass (Link 2005), proportion mature, habitat cover
Structural	
	average trophic level, high/low value biomass, large:small biomass
	plantivore:piscivore biomass, pelagic:demersal biomass
	sedentary:mobile fish biomass, infauna:epifauna biomass
Network	
	total consumption, total production
Diversity	
	Reyni-0.1, Reyni-10 (Kindt 2002)
Water quality	
	Chla, DIN
Industry	
	total catch, total value
	CPUE for abalone lobster fish shark
Aggregate	
	'State of nature'
	'State of socioeconomics'

While reasonable attention has been paid in fisheries and conservation science to ecological indicators, less focus has been on socioeconomic indicators. So for this study we drew on a range of social surveys that teased out how people value the marine environment and judge recreational and spatial management sites (Alder 1996, Bunce 1997, Goodridge et al 1997, Bunce and Gustavson 1998, McClanahan 1999, Badalamenti et al 2000, Bunce et al 2000, White et al 2000, Bunce and Pomeroy 2003, Wilkinson et al 2003, Lynch et al 2004, Moscardo and Ormsby 2004, Ormsby 1999, Alcala et al 2005). From these it was possible to identify a list of system features of social interest and simple social and economic indices (to begin addressing the triple bottom line).

Lastly these factors were combined to produce two aggregate indicators. The first is a "state of nature index" based on things listed in the surveys as aesthetically attractive or desirable (in their opinion or socially or culturally). This is based on the normalised sum of the relative levels of: dissolved inorganic nutrients (as water quality measure); biomasses of target species, sharks, whales (or other charismatics); proportional of the fish populations that are adult (as people like old, big fish); biomass ratios of demersal:pelagic fish and piscivorous:planktivorous fish (so we capture the preferred system structure that recreational fishers, tourists and divers say they want, which is large demersal or piscivorous fish not lots of little pelagic animals); and biomass (or cover) of epibenthos (as an index of habitat quality).

The second index we have referred to as a ‘state of the socioeconomics index’ indicator which combines system features that people want to target recreationally, commercially or visually (i.e. tourists and divers want to see them). This index is calculated as the normalised sum of the relative total catch; total effort; catch-per-unit-effort rates; inverse rate of illegal, unregulated or unrecorded fishing (i.e. 1/IUU_rate); and the average size of target fish. These are typically related more to commercial value of the system around the area of spatial management or to what the recreational fishers can catch (trophy fish often matching the most valuable commercial fish by the way). Note that in this instance, the IUU rates are specified in the scenario definition rather than being an emergent dynamic property of the simulations.

Measuring Performance

We are interested in looking at how the indicators work on a range of spatial scales (Figure 3).

The local scale is simply looking at the MPAs themselves and the areas immediately surrounding them. We would expect that these areas would show the strongest contrast and that indicators would most strongly detect a change in these areas. It is anticipated that indicators would detect some signal from the MPAs at the regional scale. This is a larger area than the local scale regions, but should potentially incorporate some similar bioregions as those covered by the MPAs.

The broader regional and ecosystem scales are also considered, to see whether we can detect any impact of the MPAs throughout a much larger area. If, for example, the MPAs are acting as a refuge for juvenile stages of pelagic fish, then it is plausible that we may see an increase in these fish in the outer boxes (in the red and green circles in Figure 3). We are looking at whether any of our tested indicators can detect a change at this scale.

Output from the sampling model was evaluated in three ways.

1. *Strength:* at each sampling step this was calculated as:

$$\text{strength} = \frac{\text{indicator\_at\_distance\_x}}{\text{indicator\_in\_MPA}}$$

   this captures the indicator values at a range of distances from the centre of the MPAs compared against the value inside the MPA to assess the signal content and strength at distance from the MPA.

2. *Fidelity:* at each sampling step this was calculated as:

$$\text{fidelity} = \frac{\text{indicator\_at\_distance\_x} / \text{indicator\_in\_MPA}}{\text{attribute\_at\_distance\_x} / \text{attribute\_in\_MPA}}$$

   the pattern of change of the indicator values was compared inside and outside the MPAs with the pattern for attributes to check for fidelity of the signal with distance from the MPA.

3. *Correlations:* attribute values were correlated with the sampled indicator values in order to check whether indicators captured the actual system properties of interest.

These analyses were repeated under the various scenarios to assess whether the performance of the indicators was robust to changes in management, environmental condition and breaches in protection of the MPAs. Results were classified across all scenarios and sampling schemes as follows:

Fidelity - ideally this is a value about 1 (so indicator faithfully tracks the attribute), but even a biased result (e.g. the indicator value is consistently half the attribute value etc) is acceptable so long as it is consistent under all circumstances. Consequently indicators were classified as robust if across all scenarios and sampling schemes it produced a flat line (so nothing deviated by more than +/- 20% from the average value); intermediate if the line was relatively flat (nothing deviated more than +/-50% from the average); poor if there were deviations of >+/-50%; and uninformative if the results showed large scale variation or no clear pattern at all.

Signal strength - ideally this shows a linear trend with distance from the spatial closure so that an effect of the spatial management can be detected. The classifications of signal strength were based on a linear trend analysis, with the robustness of the patterns assigned as follows: robust = 75% of all trends linear (and remaining 25% showing only plateaus with distance from the zone, no nonlinear behaviour detected); intermediate = 25-75% show clear linear trend and the rest show either plateaus with distance from the zone or a flat response in the immediate area of the spatial management zone and then a linear signal with distance from the closure (no non-monotonic behaviour detected); poor = if less than 25% show linear trends; uninformative = if non-monotonic results were found.

These qualitative classifications were completed using statistical and automated binning based on the criteria stated above and verified by the researchers.

Results/Discussion

Calibrating Atlantis-SM

Comparison with observations

A qualitative comparison of the modelled time series with survey data for individual species groups is presented in Table 3. Results are presented only for those locations where sufficient survey data were available for thorough comparison. Examples are given in Figure 5a of the calibration matches and matches to test data sets (which were not available for all species and areas). The final 3 years in both figure show test data that was not used in the calibration process, and demonstrate both a good fit (5a i) and a medium fit (5a ii) to the test data. Figure 5b shows examples for good, moderate and poor matches to the survey data where no test data was available. In most cases the objective of the calibration process was to fit the model to trends in the available data as well as possible across the whole domain. This did mean that while the fit for individual boxes could have in theory been improved beyond what is presented here, this could not have been achieved without significant degradation in the overall fit (i.e. the correct observed spatial distributions and the magnitude of population level biomass would have been too adversely effected)[2]. Moreover, as the fisheries

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[2] While spatial parameterisations, where different biological parameter values are used in different boxes, are technically possible in Atlantis, it was felt that the available amount of data did not warrant such an approach in this case. This is because it would have lead to a drastic overfitting of an already heavily parameterised model.

component in the model is static we did not expect to see exact matches on an annual basis at any rate. Rather, the focus was on whether the overall directional trend in the biomass of the single species groups was captured by the model.

Of the species considered, rock lobster and abalone are given greater prominence in ensuing sections because of their importance in this region.

TABLE 3. QUALITATIVE ASSESSMENT OF CALIBRATION MATCHES OF MODEL OUTPUT TO SURVEY DATA FOR SINGLE SPECIES GROUP BY REGION

Species	Central Vic	Port Philip Bay	Maria Island
lobster	good	NA	medium
abalone	NA	good	medium
band morwong	medium	NA	medium
silver sweep	poor	NA	NA
blue throat wrasse	good	medium	poor
purple wrasse	medium	good	good
herring cale	poor	good	medium
magpie perch	poor	good	NA
zebra fish	good	good	NA

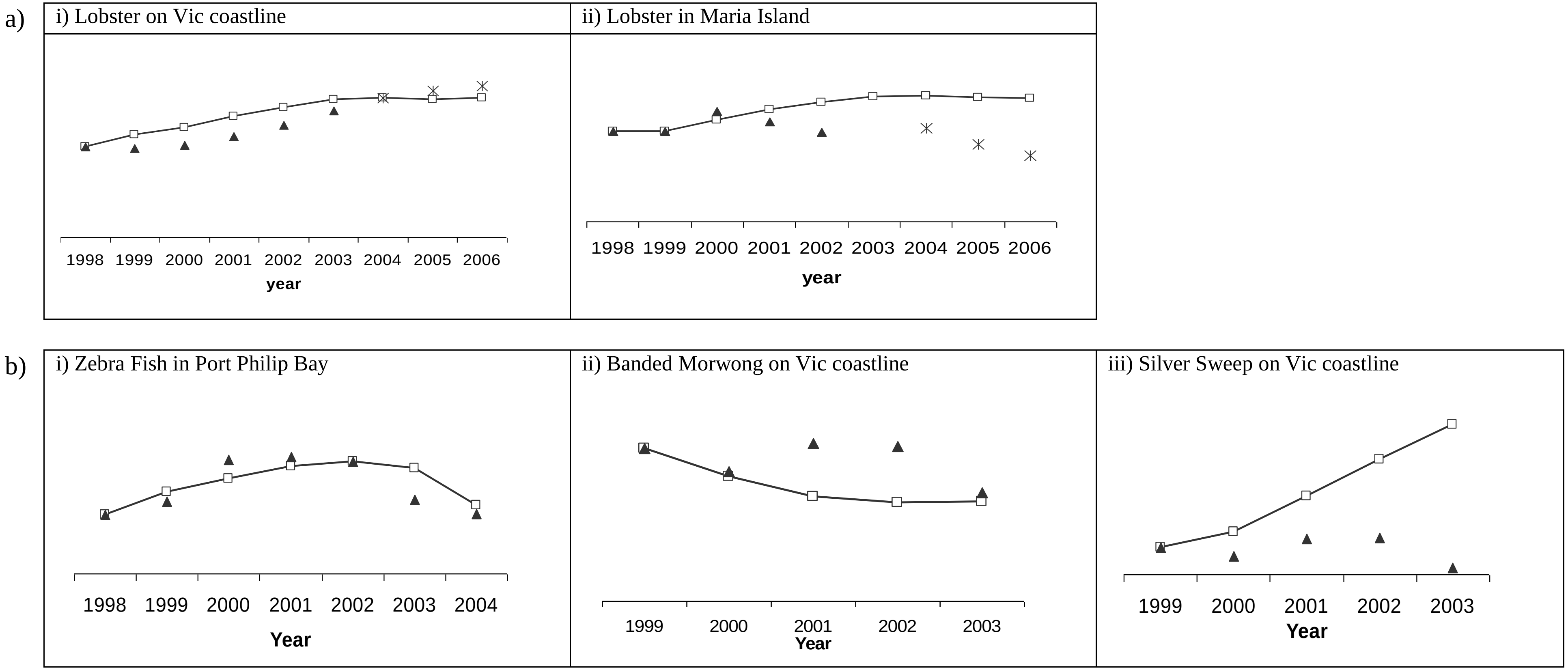


Figure 5 a. Examples of time series with test data sets included. i) lobster time series along the Central Victorian coastline, ii) lobster at Maria Island **b.** Comparison of survey data with model output for i) Zebra fish, ii) Banded morwong and iii) Silver sweep. Open squares are model output, black triangles are survey data and stars are test data.

In addition, in some cases the survey data was unreliable, in which case we did not try to match the model trajectories exactly. For example, it is likely that the observed trend in the abalone surveys in the Maria Island regions was, at least partially, a result of behavioural changes in abalone due to the increased size of lobsters in the area after marine park protection. the abalone may have become more cryptic and therefore less likely to be counted in diver surveys of the region (Barrett, pers. com.). This very fine scale change in behaviour is beyond the simple behavioural model used in Atlantis, although preliminary results from more refined representations does result in emergent behaviour such as crypsis, which leads to a concurrent fall in "observed" biomass (Fulton unpub).

Overall, of the 20 species/area combinations for which survey data were available 50% of matches were qualitatively assessed as good and 80% moderate-good. Generally matches were slightly better along the Victorian coastline than for Maria Island off the east coast of Tasmania. The reasons for this are unclear, though it is plausible that system drivers may differ between the sub-systems and that an important driver was not captured for the Tasmanian coast.

Comparison of spatial representation in Atlantis-SM and Atlantis-SE

The Atlantis-SM model domain is a sub-domain from a larger model, Atlantis-SE (Fulton et al 2007). The Atlantis-SE model covers most of southern Australia and whilst using polygons of various sizes to represent spatial structure and large geomorphic features, it did not use the dedicated telescoping of box sizes used here. Consequently it could not be used to assess the finer scale dynamics around specific spatial management zones required for this study.

The impact of telescoping can be assessed by comparing outputs from the two models. Figure 6 shows the distribution of squid in Atlantis-SM and the corresponding area of Atlantis-SE. Similarly, Figure 7 shows the distribution of shallow demersal fish in Atlantis-SM and Atlantis-SE. There is broad agreement between the two models of the overall distributions of both squid and shallow demersal fish along the south east of Australia. However, by using the telescoped modelling approach of Atlantis-SM, a more detailed description of the cross shelf distributions of both groups is obtained. It captures the heterogeneity of species at the spatial scale required for the assessment of indicators for use with spatial management. Figures 8a and 8b show the distribution of lobster in Atlantis-SM and Atlantis-SE respectively, with the Tasmanian east coast expanded to show the finer telescoping detail. Again there is broad agreement, however, the coastal portion of Atlantis-SM shows the variation from slope to coastal areas, whereas Atlantis-SE aggregates these areas, which leads to a loss of detail. However, by using the telescoped modelling approach of Atlantis-SM, a more detailed description of the cross shelf distributions of both groups is gained, with the subtle changes in density from the coast to the shelf much more apparent than in the Atlantis-SE model.

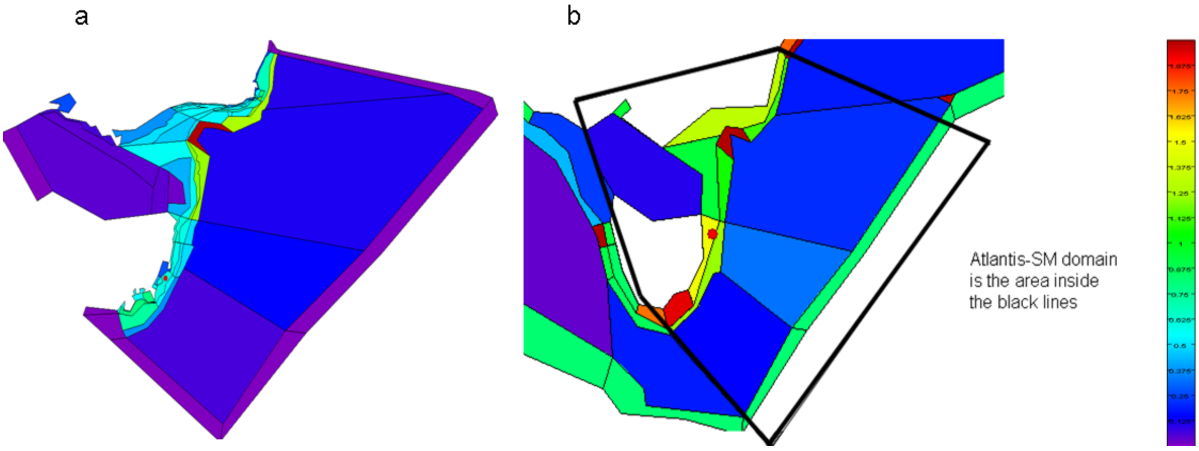


Figure 6. Spatial distribution of squid in a) the Atlantis-SM model, and b) the Atlantis-SE model. Colour scale is purple to red, with purple being the lowest density and red being the highest.

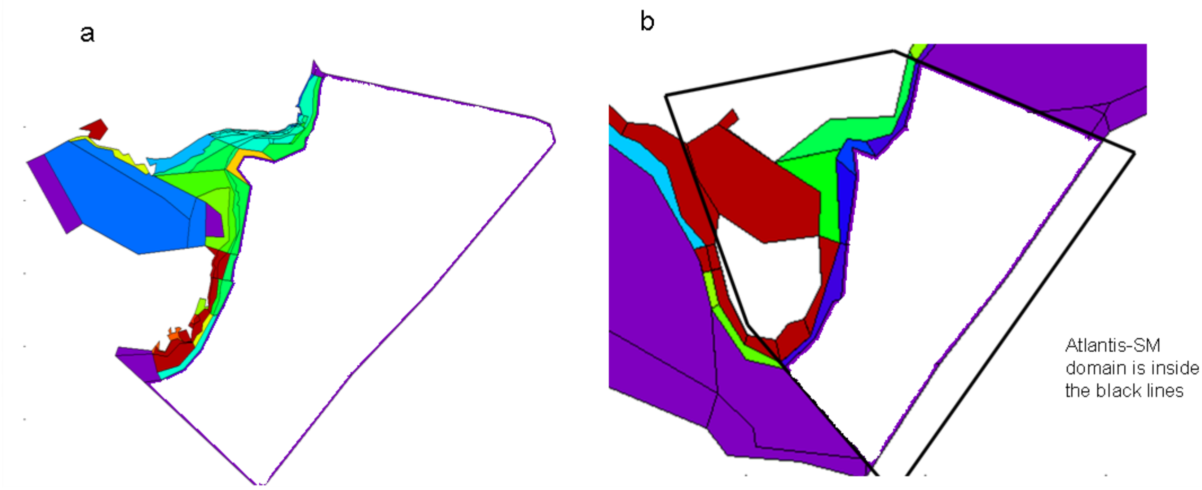


Figure 7. Spatial distribution of shallow demersal fish in a) the Atlantis-SM model, and b) the Atlantis-SE model. White areas area either land or areas where shallow demersal fish do not occur. Colour scale is purple to red, with purple being the lowest density and red being the highest.

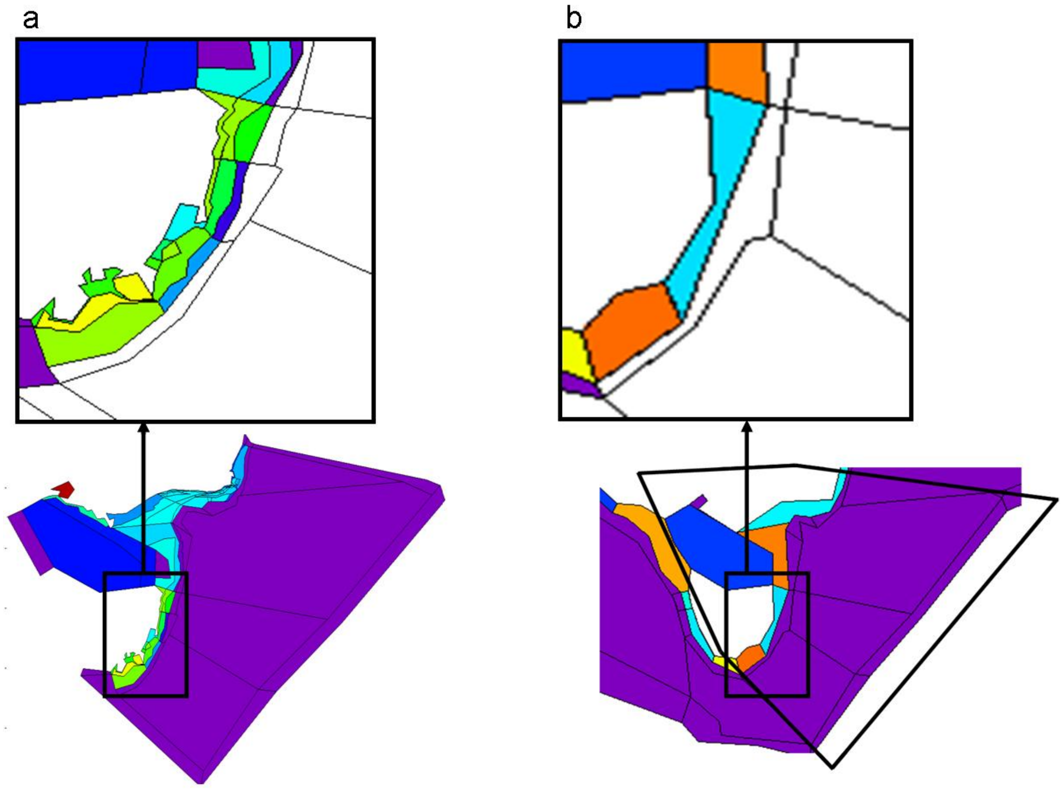


FIGURE 8. DISTRIBUTION OF LOBSTER IN A) ATLANTIS-SM, AND B) ATLANTIS-SE. THE EAST COAST OF TASMANIA IS EXPANDED TO SHOW FURTHER TELESCOPING DETAIL. COLOUR SCALE IS PURPLE TO RED, WITH PURPLE BEING LOWEST DENSITY AND RED BEING HIGHEST (AREAS WHERE LOBSTER DO NOT OCCUR HAVE BEEN MADE WHITE IN EXPANDED SQUARES).

The telescoping technique allows the construction of a model with non-uniform levels of spatial complexity across the modelled domain. By using polygons based on observed bioregional structures, the telescoped modelled domain represents the heterogeneity found along coastal regions (where geomorphological features and benthic habitats have a greater impact on species distribution), and also benefits from reduced computational demands in the ocean environment where species distributions are not so strongly constrained spatially. In this way we have been able to create a model of a large area without unnecessarily increasing the level of spatial complexity in the offshore regions, thus tailoring it according to both the data available, and the dynamics of each part of the system that are relevant.

Spatial resolution has been shown to strongly influence modelled ecosystem dynamics (Sharov 1996), including competition (Johnson & Seinen 2002), trophic structure, community composition and ecosystem complexity (Fulton et al. 2004). While an increase in spatial resolution can potentially provide more detail and accuracy when modelling an ecosystem, there are also drawbacks. Issues such as increased data requirements for parameterisation and calibration, increased computational power requirements and the increased complexity of model output, make producing and using highly spatially-resolved models difficult (Fulton et al. 2003).

The spatially structured telescoping approach described here allows large, diverse ecosystems to be modelled reasonably well, although some limitations in this study restricted the performance of the model. The static biological parameterisation across all regions, and the static fisheries parameterisation across time and space, meant that the dynamics of some biological groups were not captured at all scales. Where data were available to support spatially-resolved biological parameterisation, we found that model performance improved. Similarly, forcing a realistic fisheries time series also improved model performance in the boxes and for the groups for which it was applied. However, since this detail of data cannot be supplied for all groups in all boxes, the extra complexity cannot generally be justified and so was not used in the standard runs presented here. Despite these issues, the reasonable performance for most species demonstrates that the telescoping approach produced a comprehensive representation of processes across all scales relevant to regional management.

Model performance

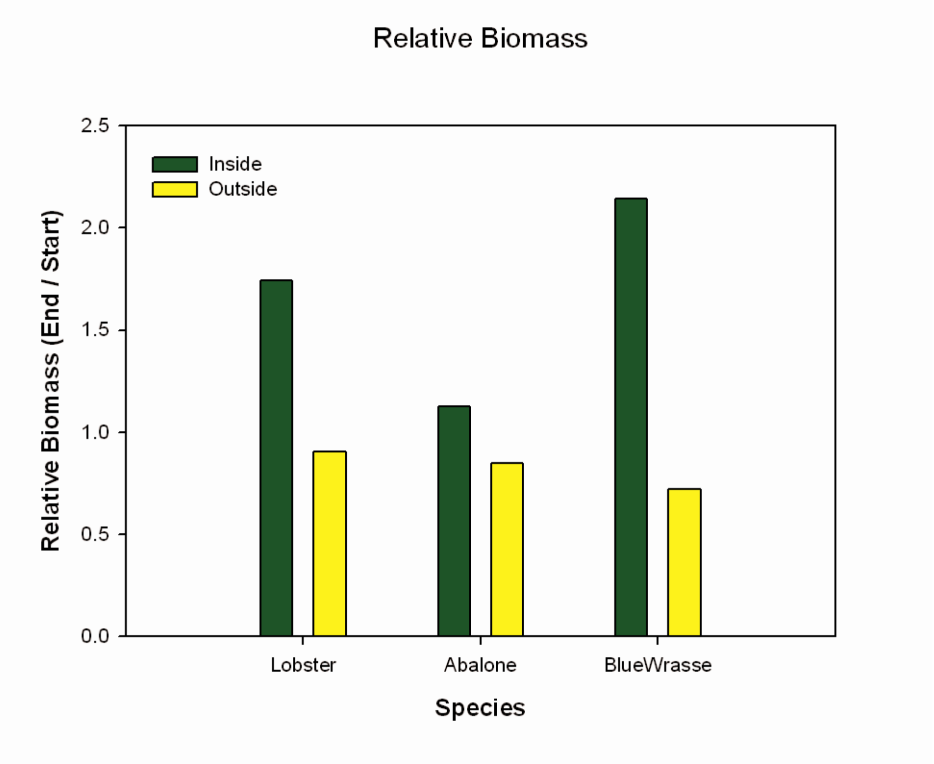
Prior to evaluating indicator performance it is important to consider overall model performance and behaviour. The calibration discussed above shows that the model can capture past dynamics reasonably well, but given the model's role as a testbed, the question remained about whether it could capture ecosystem dynamics resulting from spatial management at the scale we were interested in. To resolve this, we looked at outputs inside and outside the modelled reserves to verify the model was producing realistic results (and thus would be a fair testbed).

Relative biomass trends

The relative biomass for lobsters, abalone and blue throat wrasse at the end of the 20 year projections for Maria Island using the baseline run (medium productivity) are shown in Figure 9a. The abundance of lobsters and wrasse are considerably higher inside the closures whereas abalone, although higher, is considerably less so. Mimicked survey counts at Maria Island for these species for a range of modelled closures (from none to large) are generally consistent with observations (Figure 9b).

The trajectory of relative biomass for lobsters and abalone under different spatial arrangements (in this example at Maria Island under the low productivity model specification) are shown in Figure 10. As expected, there are different responses for each species based on whether there is a closure and what the size of that closure is. Both species show clear differences in abundance inside and outside the closure, but the impact of the closure appears to be more pronounced for lobsters. This is a good example of Atlantis-SM's capacity to capture complex differences in abundance at relatively fine spatial scales. Further, Figure 11 shows the relative abundance of lobster in each polygon at the end of the 20 year projection period with moderate productivity. The relative biomass trajectories for lobsters at two example locations, inside and outside the Point Hicks and Maria Island MPAs show how there can even be regional differences in the signals generated by the presence of a spatial closure. Figures 12, 13 and 14 show the regional differences for abalone, demersal sharks and blue throat wrasse, respectively.

a)



b)

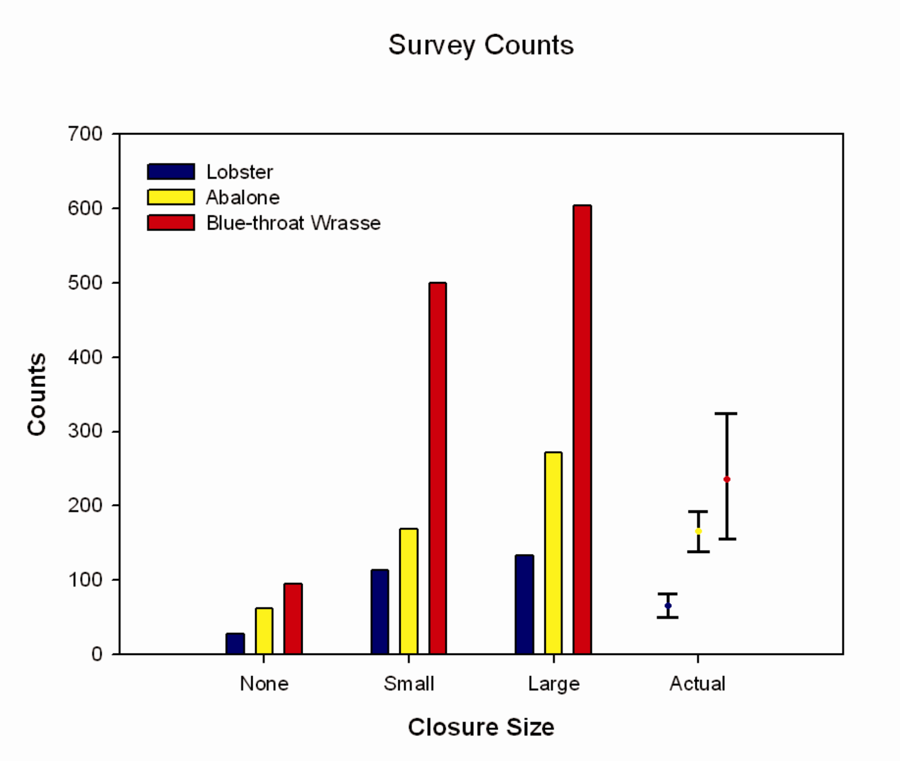
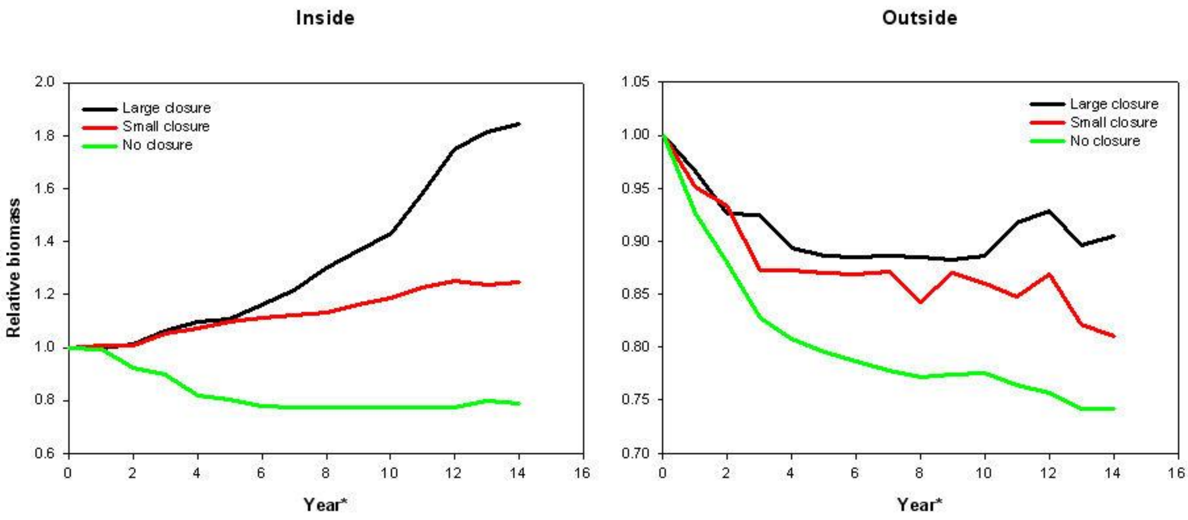
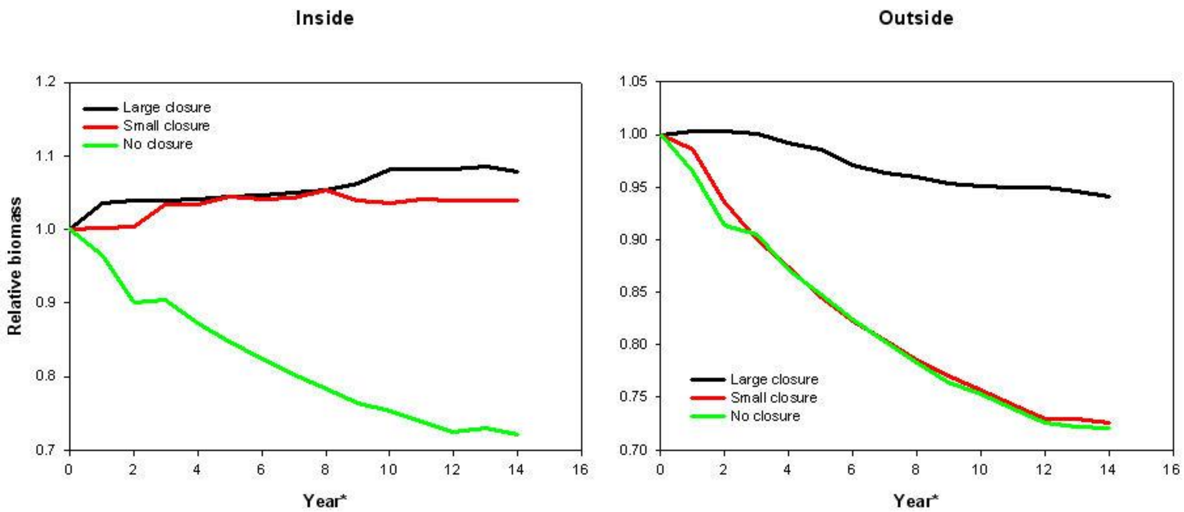


FIGURE 9. A) RELATIVE BIOMASS PLOT SHOWING BIOMASS BEHAVIOUR INSIDE-OUTSIDE CLOSURES, B) SURVEY COUNT MIMICKED TO COMPARE MODEL VALUES AT MARIA VS OBSERVATIONS



a) Lobster



b) Abalone

FIGURE 10. RELATIVE BIOMASS TRAJECTORIES – FOR MARIA ISLAND UNDER LOW PRODUCTIVITY, A) LOBSTER AND B) ABALONE. YEAR* MEANS THAT IT IS YEAR AFTER BURN IN PERIOD.

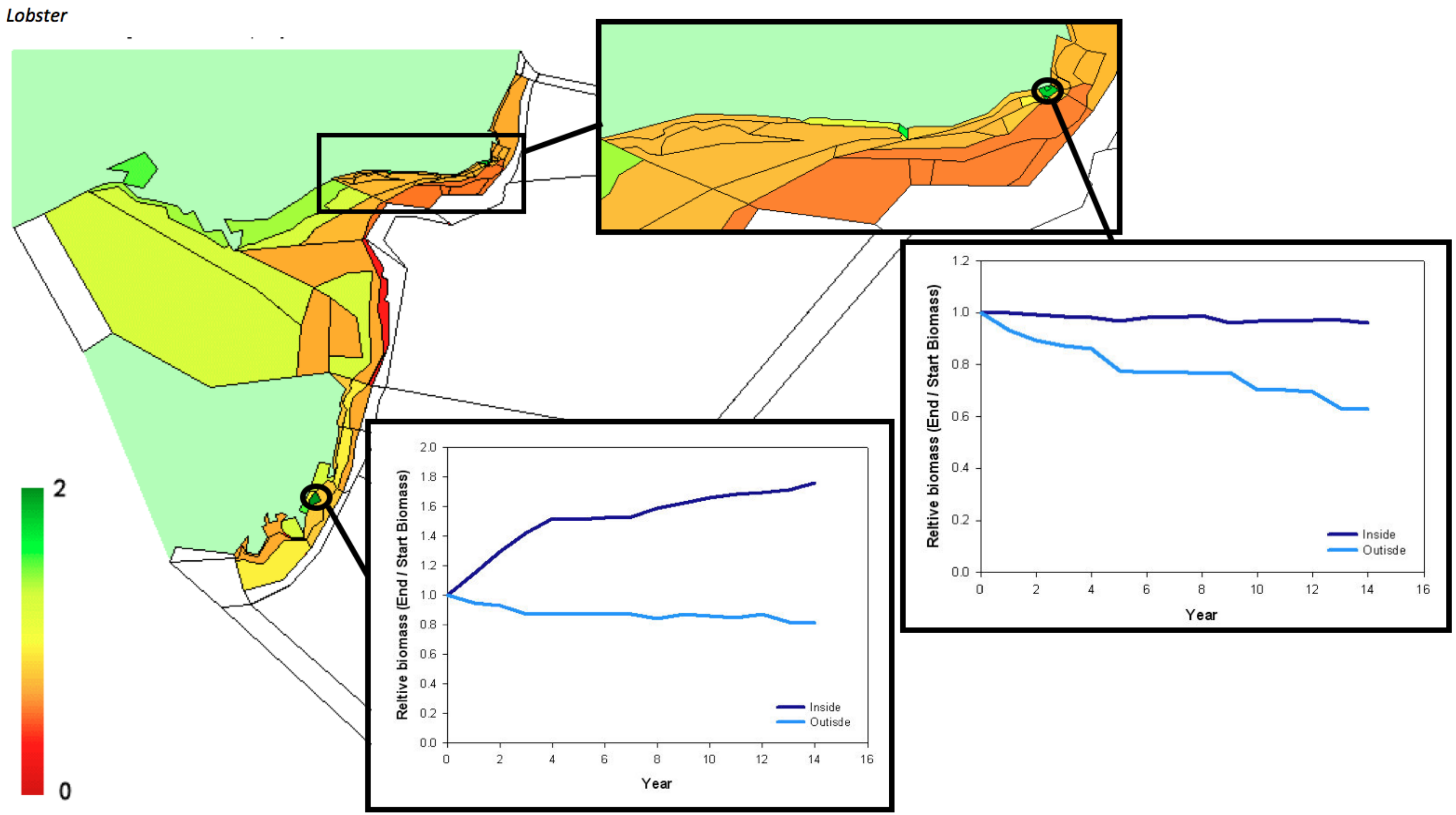


Figure 11. The relative abundance of lobster in each polygon at the end of the 20 year projection period with moderate productivity

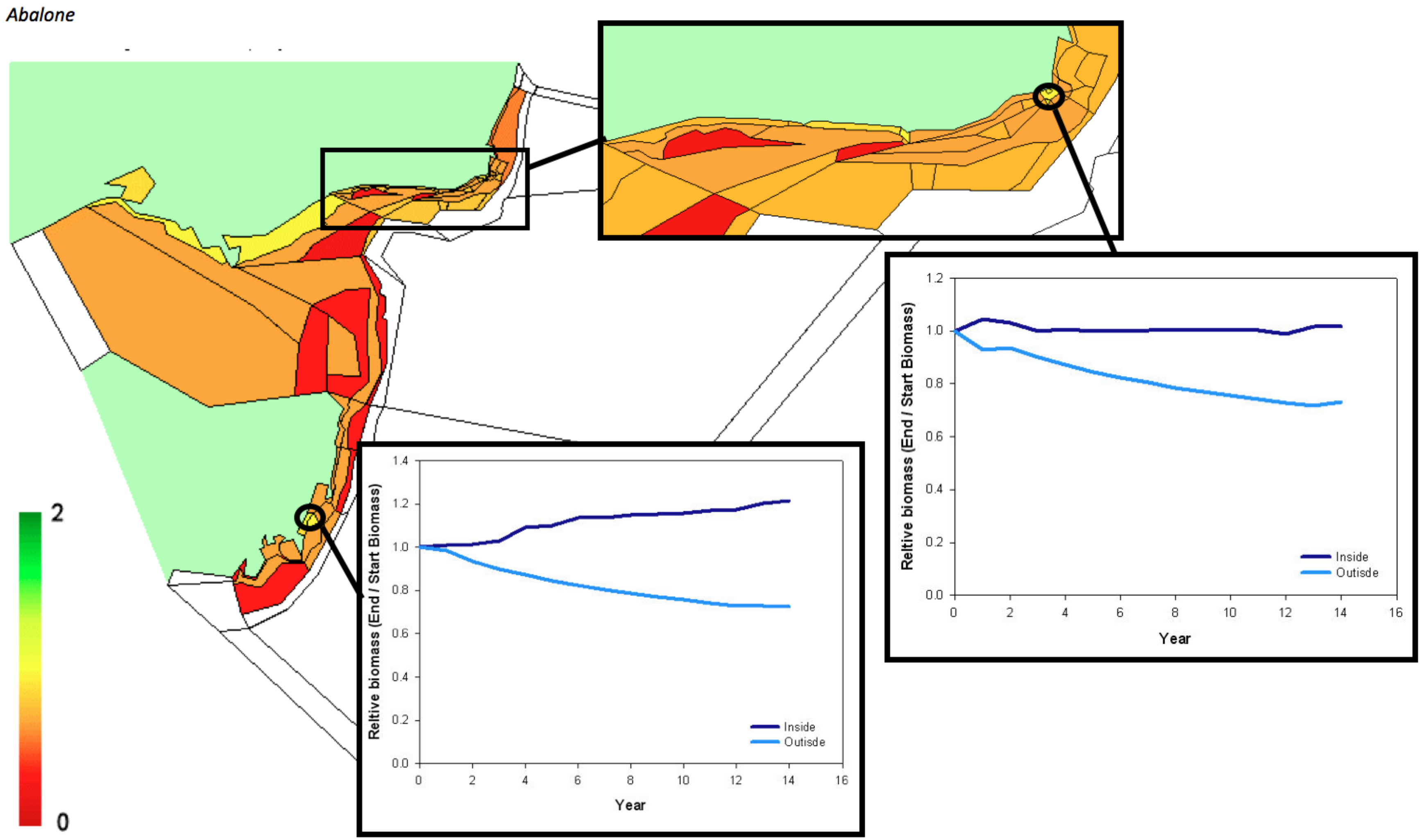


FIGURE 12. THE RELATIVE ABUNDANCE OF ABALONE IN EACH POLYGON AT THE END OF THE 20 YEAR PROJECTION PERIOD WITH MODERATE PRODUCTIVITY

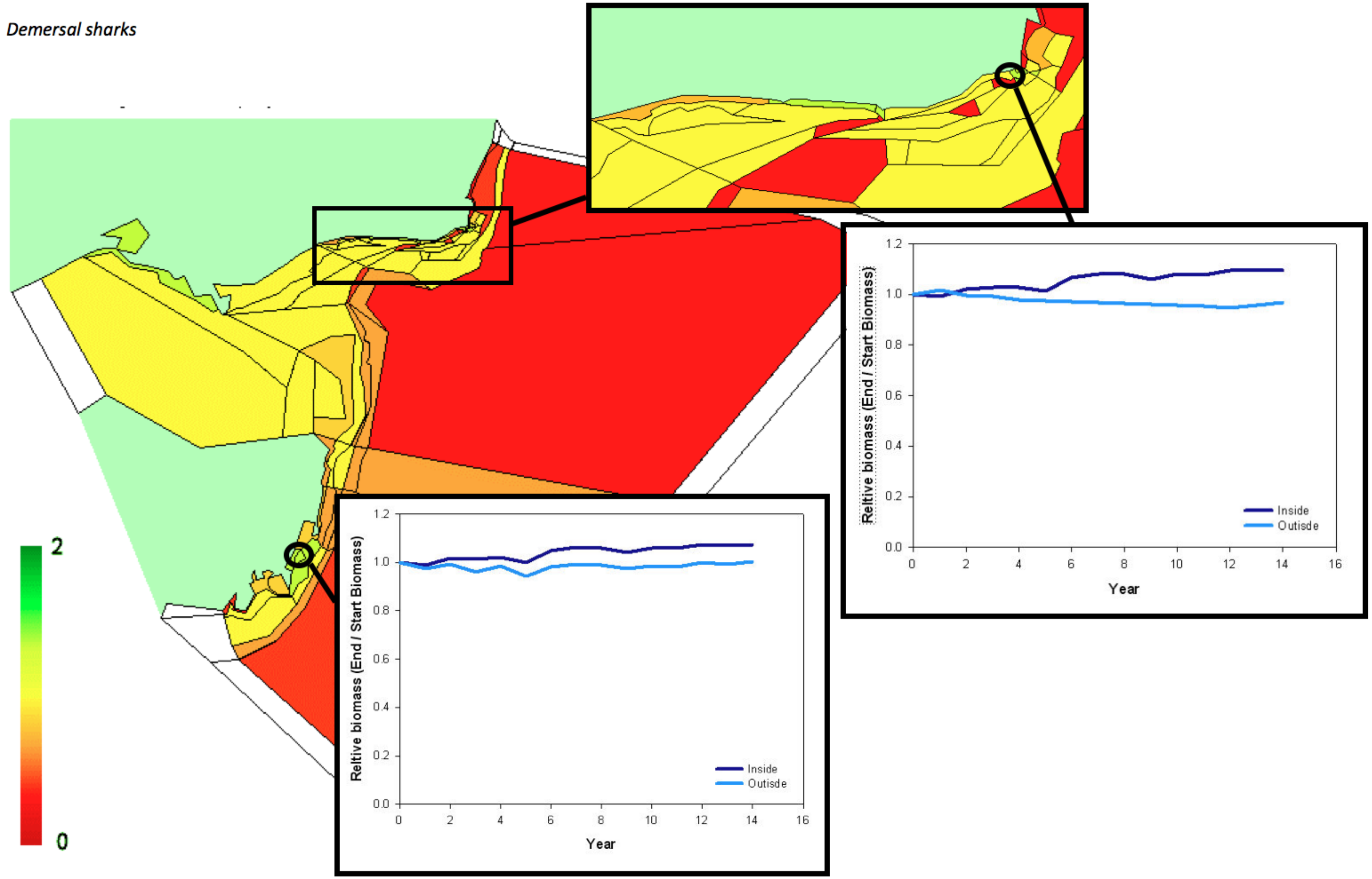


FIGURE 13. THE RELATIVE ABUNDANCE OF DEMERSAL SHARKS IN EACH POLYGON AT THE END OF THE 20 YEAR PROJECTION PERIOD WITH MODERATE PRODUCTIVITY

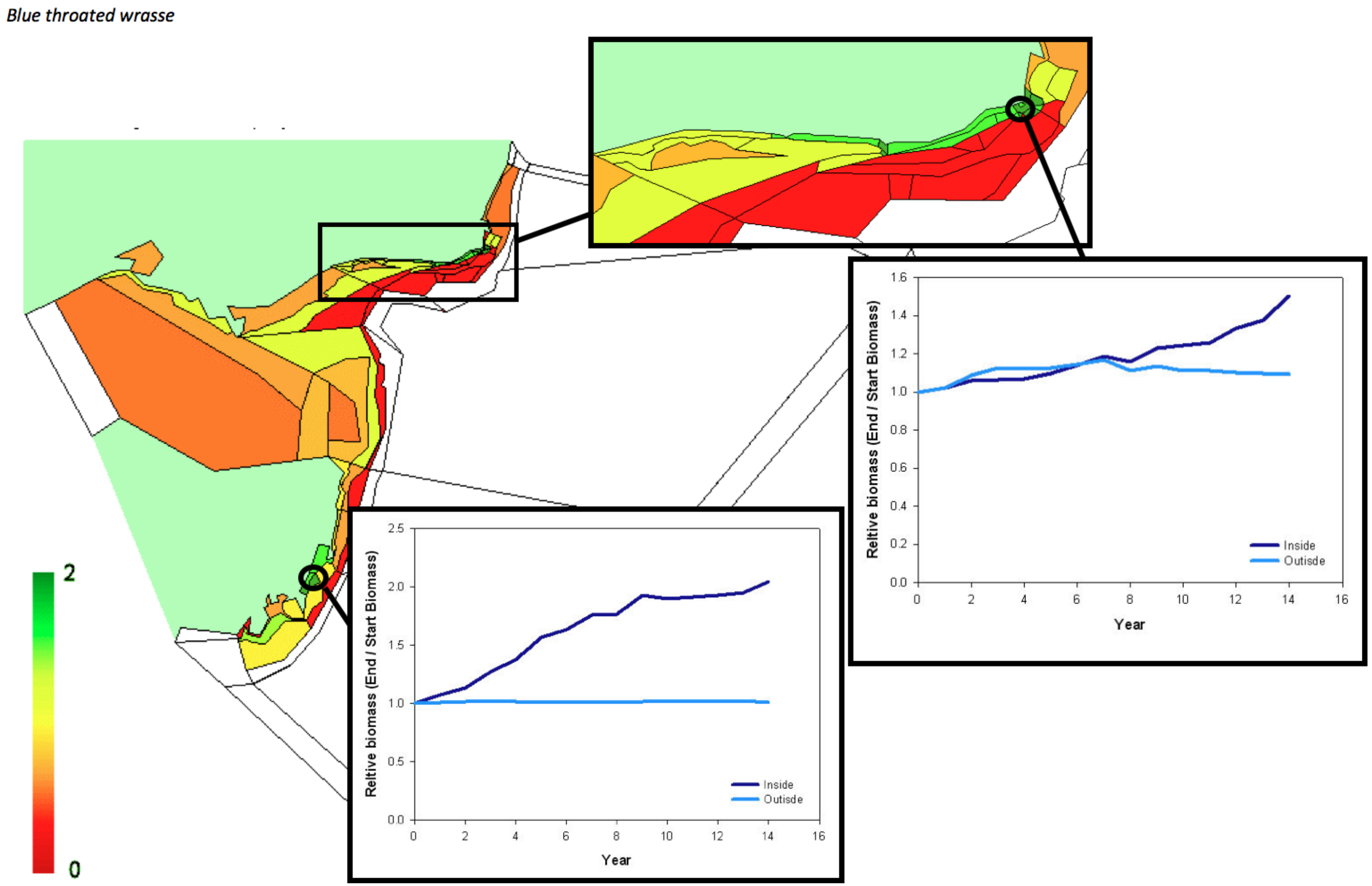


FIGURE 14. THE RELATIVE ABUNDANCE OF BLUE THROAT WRASSE IN EACH POLYGON AT THE END OF THE 20 YEAR PROJECTION PERIOD WITH MODERATE PRODUCTIVITY

However the biology of the animals is also important to this performance. In this context, the pronounced difference in the abundance of abalone outside a large closure compared to that for a small or no closure in Figure 10 does require some explanation. Given that the group is effectively sedentary in the model once settled, the only mechanisms for this result are (i) that there are altered forage and predation conditions outside the large closures in comparison with the small (i.e. indirect effects) or (ii) that there is a spill over of larvae from the population within the closure or (iii) a combination of the two. In this case, the second mechanism is a drastic overstatement of a real world mechanism, given in reality larval dispersal is likely on the order of hundreds of metres to tens of kilometres at most for abalone. Within Atlantis however, it is possible for a patch at the edge of a box to provide larvae to habitat in an adjacent box. This is not an issue in fine-scale boxes, but becomes more of an issue for larger boxes, where those larvae are then available for settlement across that entire box. This effect is exacerbated in the case of large closures, because all the smaller boxes lie within the closure and only larger boxes remain beyond the closure and so hyperdiffusion of larvae is possible. While aberrant at such large scales for abalone, the mechanism only becomes an issue for the largest spatial closure cases and is not an issue for the vast majority of groups. Thus it does not invalidate the general performance of the model across the system as a whole, and the model remains a good basis for assessing indicator performance.

Size spectra

Size spectra were also used to verify the model was producing a detectible (and plausible) ecosystem signal, by considering the size spectra through time inside and outside the modelled reserves. Changes in the slope and intercept of size spectra have been shown to reflect shifts in community composition and food web structure (Rice 2000) and have been used to investigate the effect of fishing and other pressures in many systems (e.g. North sea Jennings et al 2002). They also respond to other changes. For instance they highlight shifts in basal productivity, whether due to natural regime shifts or eutrophication (Caddy and Garibaldi 2000).

Size spectra are formed by calculating the biomass (typically mg N or ash free dry weight) per log(size class) size bin. For ease of presentation in this case the biomass was in mg N m^{-2} and the binning we used had size classes that rose by an order of magnitude from 0.0001-1cm and 5cm after that. This linearised the earliest part of the foodweb (which is quadratic in form and usually omitted in the size spectra used to consider only fish communities, but was included in this case as we have a total system interest) and retained a resolved representation of the larger bodied groups of particular conservation interest. The rigour of this approach was shown by Fulton et al (2005).

The means of interpreting a size spectra analysis is via the slope and axial intercept of the resulting plots. The axial intercept is a straightforward measure of basal productivity, rising with increasing primary production and dropping with declining production. The slope of the size spectra is a measure of system structure and diversity. It becomes steeper if larger bodied animals are rarer (indicating a drop in system diversity and skewed size distribution). It is less clear when smaller or intermediate size classes are removed from a system, as the resulting curve is bumpy rather than linear, but empirical and theoretical studies have shown that also means diversity has dropped and so is in itself a useful system indicator. Consequently, size spectra should be a good means for monitoring the effect of MPAs through space and time – in particular they should be a direct performance measure of whether a closure is meeting size and biodiversity objectives. By creating a coloured contour plot of size spectra through time it is (quite literally) easy to see whether there is a shift in the size distributions (and thus slope of the plot), by looking at whether the relative abundance of larger bodied animals changes.

Figure 15 shows an example from Maria Island when an intensive sampling regime is modelled. Inside the closed area there is a clear increase in numbers of small fish (colour goes from yellow-green to orange through time for 10-15cm fish indicating an increase in biomass of this size class) and intermediate sized fish (as 20-25cm fish go from blue to green through time) and larger vertebrates (35cm and above – very dark blue rises to blue, so there is still not a lot of them but it would be a noticeable increase to a diver in the area). In the immediate area of the MPA there is some decline of 10-12cm fish (it drops yellow/orange to green) and particularly 15cm-20cm fish (where it drops from green to blue) and no recovery is seen in largest fish. This suggests that fishing pressure is impacting more of the age groups and species and having a broader effect on system structure. This is also seen at a distance from the MPA. However the pattern is actually not as clear, as the sampling is starting to include a different set of species to the shallower waters around the closure.

The results presented here show that Atlantis-SM can capture complex differences in spatial dynamics. The outputs are generally consistent with experience (Barrett et al 2007) or at least what one would expect from inside and outside non-take MPAs. They provide a degree of comfort that the model is capturing ecosystem dynamics sufficiently well that the evaluating the performance of indicators at a range of spatial scales is robust.

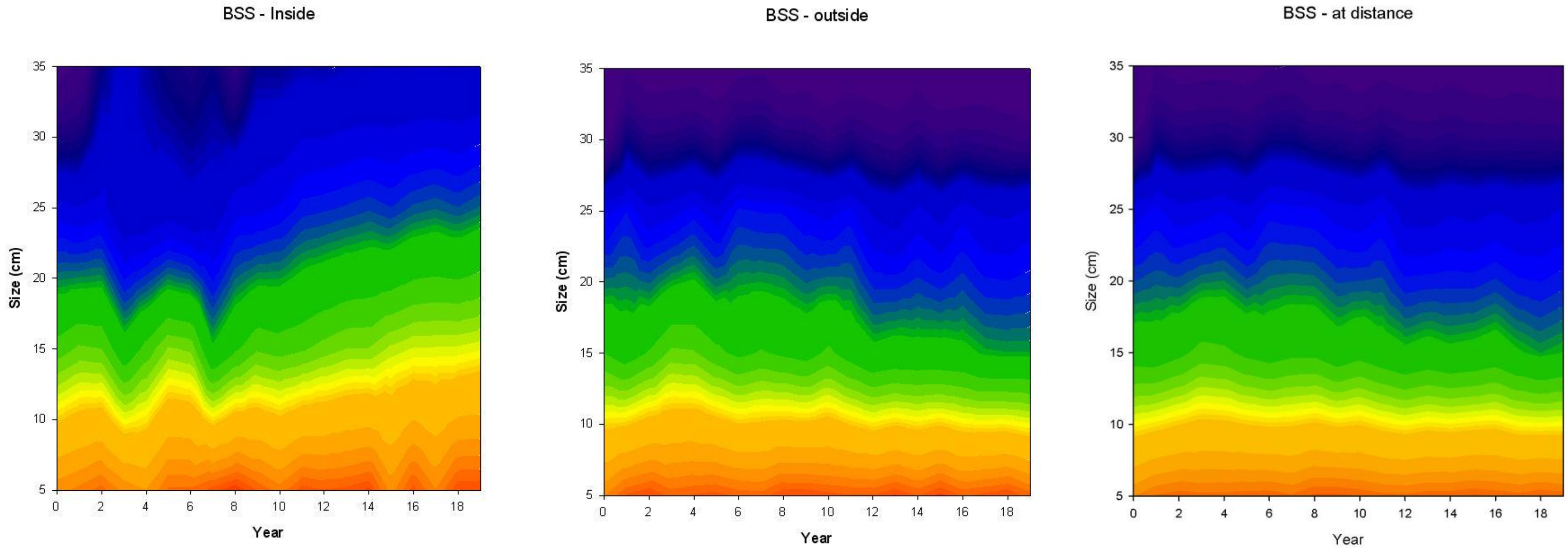


FIGURE 15: SIZE SPECTRA INSIDE, IMMEDIATELY OUTSIDE AND AT SOME DISTANCE FROM THE MARIA ISLAND CLOSURE IN THE INTENSIVE SAMPLING SCENARIO RUN WITH ASM.

Evaluating Indicator Performance

This study required very extensive computer resources. We applied a 3x4x4x4* (productivity x MPA size x sampling schemes x impact type, with the * indicating that one of the impacts (fisheries) was also considered at 3 levels) matrix of specifications and scenarios to assess indicator robustness, giving 432 individual outputs. In addition we also examined several other scenarios at the request of a high-level steering committee established to give guidance and advice from a policy perspective to the project team. As the total modelled output was in excess of 240 GB, this does present some problems in presenting the entirety of the results in a coherent and clear manner.

Consequently, we have summarised the results for signal strength and fidelity in Table 4, rather than providing outputs from each individual run. Example plots for signal strength and fidelity are shown in Figures 16 and 17, respectively. The plots show signal strength and fidelity at each spatial scale (local, regional, broad and global) for robust, intermediate, poor and uninformative outputs for MPA size and different sampling regimes.

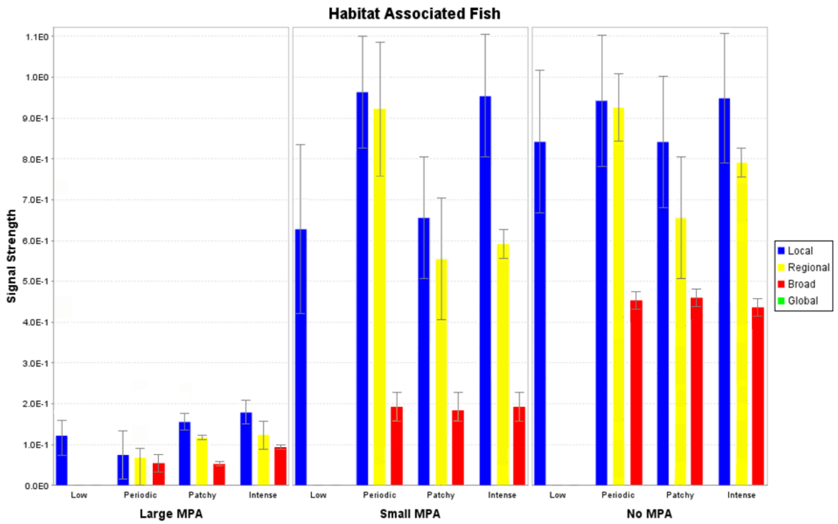
In Table 4 "split" results are recorded where one particular pattern of results held for a majority of scenarios or productivity specifications, but a different pattern was found under a specific (consistent) set of scenarios or specifications. For example, many of the biomass indicators showed intermediate fidelity under low or periodic sampling across all scenarios and specifications, but robust fidelity for the other sampling schemes

TABLE 4. SUMMARY OF THE RESULTS FOR SIGNAL STRENGTH AND FIDELITY

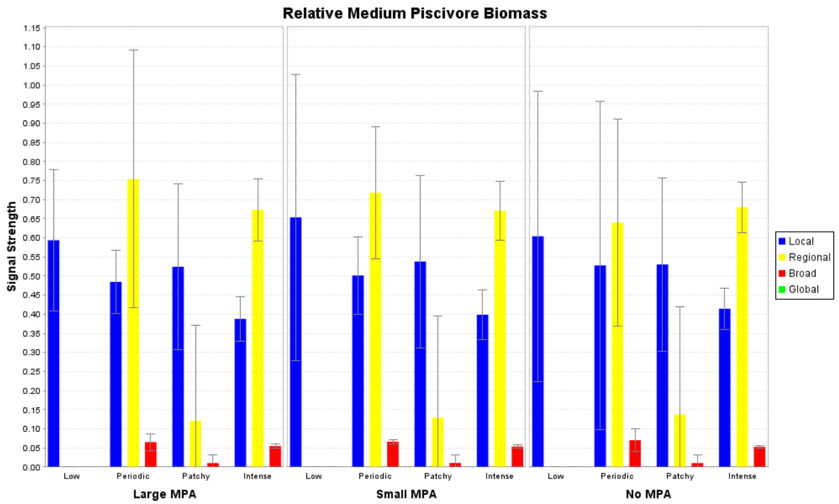
Indicator	Signal strength	Signal fidelity performance	Comments
			KEY: Robust / Intermediate / Poor / Uninformative
BIOMASS INDICATORS			
abalone biomass	Intermediate / Robust	Intermediate / Robust	Robust for low productivity, but under other productivities it shows intermediate performance when using low or periodic sampling design
lobster biomass	Intermediate	Intermediate / Robust	Can be sensitive to sampling design (performance degrades for patchy or periodic sampling).
bluethroat wrasse biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
purple wrasse biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
pink snapper biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
silver sweep biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
zebra fish biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
banded morwong biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
habitat associated fish biomass	Robust	Intermediate / Robust	Intermediate signal fidelity performance if low or periodic sampling
demersal fish biomass	Intermediate / Robust	Intermediate / Robust	Typically intermediate but with pollutants or gauntlet fishing can be a robust indicator. Performance degrades for patchy or periodic sampling
medium piscivore biomass	Poor / Intermediate	Intermediate / Robust	Signal strength performance degrades from intermediate to poor for patchy or periodic sampling. Signal fidelity performance robust under low fishing pressure, but is otherwise Intermediate under low or periodic sampling or when there is low productivity.
demersal shark biomass	Poor / Intermediate	Intermediate / Robust	Signal strength performance degrades from intermediate to poor for patchy or periodic sampling. Signal fidelity performance robust under low fishing pressure, but is otherwise Intermediate under low or periodic sampling or when there is low productivity.
pelagic shark biomass	Poor / Intermediate	Intermediate / Robust	Signal strength performance degrades from intermediate to poor for patchy or periodic sampling. Signal fidelity performance robust under low fishing pressure, but is otherwise Intermediate under low or periodic sampling or when there is low productivity.
small pelagic biomass	Poor / Intermediate	Intermediate / Robust	Signal strength performance degrades from intermediate to poor for patchy or periodic sampling. Signal fidelity performance robust under low fishing pressure, but is otherwise Intermediate under low or periodic sampling or when there is low productivity.
TL4+ biomass	Poor / Intermediate	Poor / Intermediate	Poorer performance with lower productivity or periodic sampling
habitat cover	Poor / Robust	Intermediate / Uninformative	Signal strength performance is poor unless in or immediately adjacent to closed areas. Signal fidelity performance is intermediate in the immediate area, but degrades under extremes of productivity.
proportion mature	Poor / Robust	Robust	Signal strength is poor for short-lived and mobile species and under low sampling cover.
zooplankton biomass	Poor	Poor	

Indicator	Signal strength performance	Signal fidelity performance	Comments KEY: Robust / Intermediate / Poor / Uninformative
STRUCTURAL INDICATORS			
average trophic level			Typically poor as inconsistent or doesn't have strong gradient spatially as move away from closure sites (can be more informative if heavy fishing pressure in broader system)
high/low value biomass			The signal strength performance is a product of habitat associated target species being high value and those living off reefs being less valuable
large:small biomass			Performance degrades if system under pressure
plantivore:piscivore biomass			Not useful under high system productivity, but useful at local scales or if there are intensive sampling regimes
pelagic:demersal biomass			Relationships exist, but they differ based on context (kind of drivers, magnitude of pressure, scale considered, size of spatial closure)
sedentary:mobile fish biomass			
infauna:epifauna biomass			Signal strength can be perturbation indicator, but not useful vs spatial management objectives
NETWORK INDICATORS			
total consumption			
total production			
DIVERSITY INDICATORS			
Reyni-0.1			Signal strength is hard to interpret
Reyni-10			Signal strength is uninformative under high productivity (where less loss of groups with high pressure), can be harder to interpret at larger spatial scales
WATER QUALITY INDICATORS			
Chla			Signal strength is strongly related to system productivity and water quality so useful for interpreting drivers rather than just state
DIN			Very scale dependent, but good as supporting indicator of local water quality (for interpretation when "things go wrong")

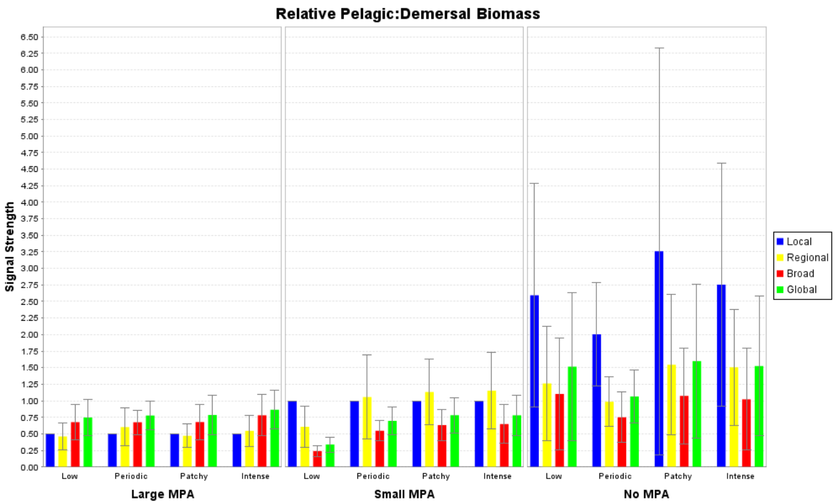
INDUSTRY INDICATORS					
total catch					Signal strength is poor under some sampling designs and under light fishing pressure Signal fidelity is poor for all but high productivity systems (but even then performance is poorer if the closure is large)
abalone CPUE					Signal strength is sensitive to sampling design, with differentiation only occurring at local or global scales (partly due to natural distributions of abalone)
fish CPUE					Signal strength is sensitive to "sampling design" especially if using large closures
lobster CPUE					Signal strength is sensitive to "sampling design" especially if using large closures, but performance can also drop to poor at extremes of productivity
shark CPUE					Signal strength is sensitive to "sampling design" and under high productivity it is a poor indicator if using large closures
total value					Signal strength is sensitive to "sampling design" and shows poor performance for extremes of productivity or under light fishing pressures. Signal fidelity is poor (particularly at distance from closure sites) for all but high productivity systems
AGGREGATE INDICATORS					
State of Nature					Signal strength is sensitive to "sampling design" and also poorer with distance from closures, especially under high productivity
State of Socioeconomics					Signal strength is sensitive to "sampling design" and also poorer with distance from closures, especially under high productivity. Signal fidelity is poor in low productivity systems.



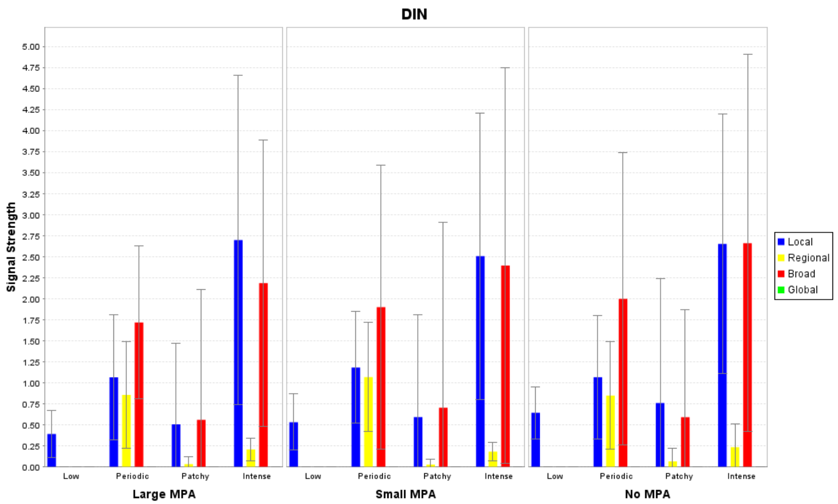
Robust



Intermediate

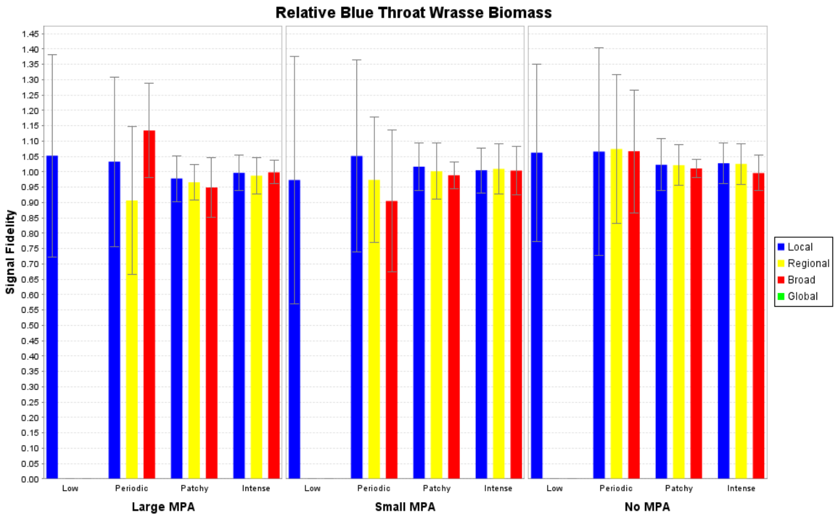


Poor

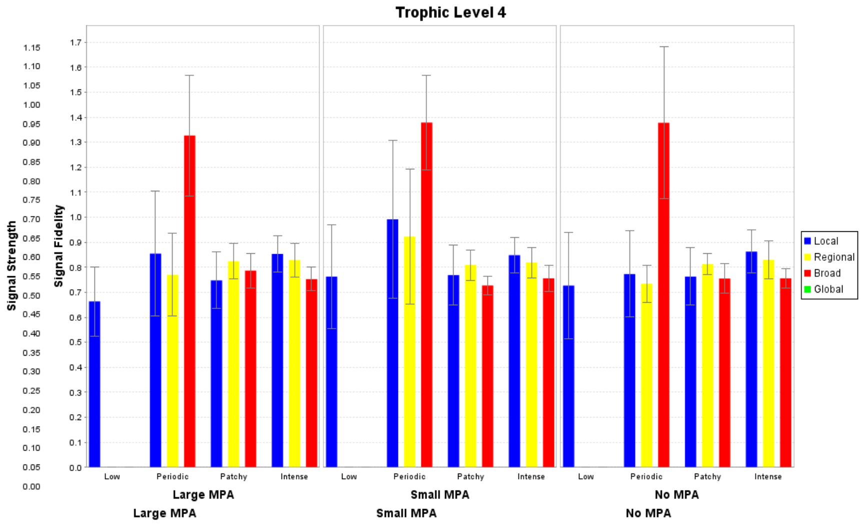


Uninformative

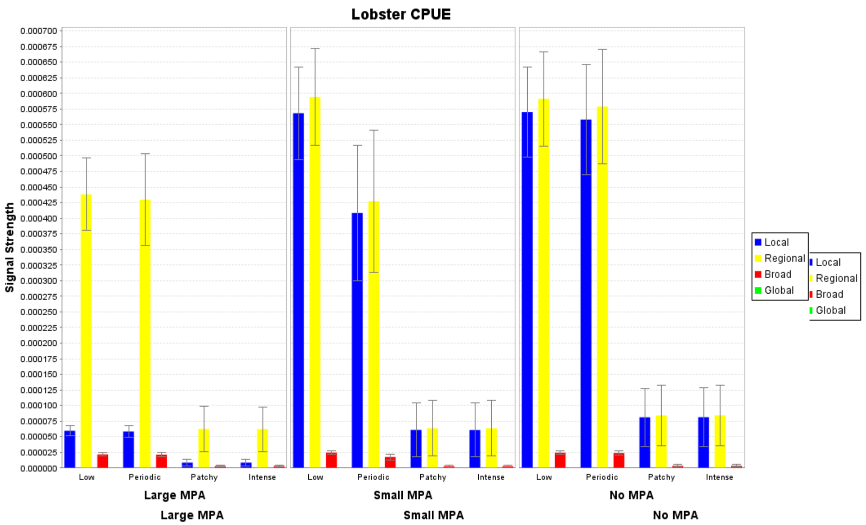
FIGURE 16. EXAMPLE PLOTS OF SIGNAL STRENGTH



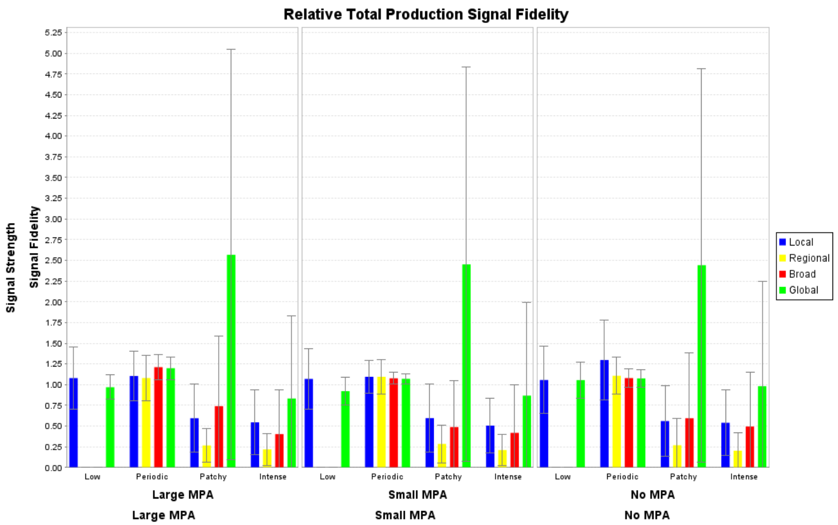
Robust



Intermediate



Poor



Uninformative

FIGURE 17. EXAMPLE PLOTS OF FIDELITY

A summary of how well indicators performed for the three model variations (high productivity, low productivity and medium (baseline) productivity) and under different pressures is given in Table 5. In this table “useful” means the indicator has robust fidelity and signal strength; "Use with care" means the indicator is generally intermediate-robust, but some circumstances may throw the signal off (either degrade its performance, create plateaus instead of linear trends or poorer fidelity); the “not useful” classification is when the indicator is uninformative either based on fidelity or signal strength, can produce non-linear results or would prove too hard to interpret because it has complicated results across different scenarios.

The performance of indicators is also informed by the correlations of indicators with attributes (broad groupings) shown in Table 6. Here useful correlations are those that show moderate to strong linear regressions.

TABLE 5. PRESSURE – PERFORMANCE RESULTS (ASSUMING USING SENSIBLE SAMPLING DESIGN, SUMMARY OF WHAT WORKS IN DIFFERENT CONDITIONS – GREEN = USEFUL, YELLOW = USE WITH CARE, RED = NOT USEFUL)

Indicator	Baseline	Low productivity	High productivity	Nutrient pollution	Climate change	Invading Species	Heavy fishing pressure[1]	Light fishing pressure
BIOMASS INDICATORS								
abalone biomass	Green	Green	Green	Green	Green	Green	Green	Green
lobster biomass	Green	Green	Green	Green	Green	Green	Green	Green
bluethroat wrasse biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
purple wrasse biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
pink snapper biomass	Green	Green	Green	Green	Green	Green	Green	Green
silver sweep biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
zebra fish biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
banded morwong biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
habitat associated fish biomass	Green	Green	Green	Yellow	Yellow	Green	Yellow	Green
demersal fish biomass	Green	Green	Green	Green	Green	Green	Green	Green
medium piscivore biomass	Green	Green	Green	Green	Green	Green	Yellow	Green
demersal shark biomass	Green	Green	Green	Green	Green	Green	Yellow	Green
pelagic shark biomass	Green	Green	Green	Green	Green	Green	Yellow	Green
small pelagic biomass	Green	Green	Green	Green	Green	Green	Yellow	Green
TL4+ biomass	Green	Yellow	Green	Green	Green	Green	Green	Green
habitat cover[3]	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
proportion mature	Green	Green	Green	Green	Green	Green	Green	Green
zooplankton biomass	Red	Red	Red	Red	Red	Red	Red	Red
STRUCTURAL INDICATORS								
average trophic level	Red	Red	Red	Red	Red	Red	Green	Red
high/low value biomass	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
large:small biomass	Green	Green	Green	Yellow	Yellow	Yellow	Yellow	Green
plantivore:piscivore biomass[2]	Yellow	Yellow	Red	Yellow	Yellow	Yellow	Yellow	Yellow
pelagic:demersal biomass[2]	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow

Indicator	Baseline	Low productivity	High productivity	Nutrient pollution	Climate change	Invading Species	Heavy fishing pressure[1]	Light fishing pressure
sedentary:mobile fish biomass								
infauna:epifauna biomass								
NETWORK INDICATORS								
Total Consumption								
Total Production								
DIVERSITY INDICATORS								
Reyni-0.1								
Reyni-10								
WATER QUALITY INDICATORS								
Chla								
DIN[3]								
INDUSTRY INDICATORS[4]								
Total catch[2]								
Abalone CPUE[2]								
Fish CPUE								
Lobster CPUE								
Shark CPUE								
Total value[5]								
AGGREGATE INDICATORS								
State of Nature								
State of Socioeconomics								

1. Includes IUU, gauntlet fishing and heavy fishing pressure
2. Only useful if used at appropriate scales (local or global)
3. Use as supporting indicator to interpret drivers and for water quality information
4. All industry indicators must be treated with extreme care as sensitive to patterns of human behaviour (on face value not as informative as industry independent information, IF that is available)
5. Operationally value will be a concern, but for judging overall objectives this may be of limited value beyond more direct measures of industry based on yield and CPUE; this work would also benefit from more dynamic economic handling

TABLE 6. CORRELATION RESULTS (USEFUL CORRELATIONS = MODERATE-STRONG LINEAR CORRELATIONS)

Key: Strong / Moderate / Weak / None (Nonlinear)

Indicator	Correlations											
	Commercial invertebrates	Habitat dependent or reef associated	Pelagic or mobile groups	Top predators (TL 4+)	Average trophic level	System structure (biomass ratios)	Network indicators	Diversity	Water quality	Total catch	CPUE	Value
BIOMASS INDICATORS												
abalone biomass	Strong	Strong	Strong	Moderate	None	Weak	Weak	Weak	Weak	None	None	None
lobster biomass	Strong	Strong	Moderate	Moderate	None	Weak	Weak	Weak	Weak	None	None	None
bluethroat wrasse biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
purple wrasse biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
pink snapper biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
silver sweep biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
zebra fish biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
banded morwong biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
habitat associated fish biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
demersal fish biomass	Strong	Strong	Moderate	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
medium piscivore biomass	Moderate	Moderate	Strong	Moderate	Moderate	Moderate	Weak	Weak	Weak	None	None	None
demersal shark biomass	Moderate	Moderate	Strong	Moderate	Weak	Moderate	Weak	Weak	Weak	None	None	None
pelagic shark biomass	Moderate	Moderate	Strong	Moderate	Moderate	Moderate	Weak	Weak	Weak	None	None	None
small pelagic biomass	Moderate	Moderate	Strong	Moderate	Moderate	Moderate	Weak	Weak	Weak	None	None	None
TL4+ biomass	Moderate	Moderate	Moderate	Strong	Moderate	Moderate	Weak	Weak	Weak	None	None	None
habitat cover	Moderate	Strong	Moderate	Moderate	None	Weak	Weak	Weak	Moderate	None	None	None
proportion mature	None	None	Weak	Weak	Moderate	Weak	Moderate	Weak	Moderate	None	None	None
zooplankton biomass	None	None	Weak	Weak	Moderate	Weak	Moderate	Weak	Moderate	None	None	None

Indicator	Correlations											
Key: Strong / Moderate / Weak / None (Nonlinear)	**Commercial invertebrates**	**Habitat dependent or reef associated**	**Pelagic or mobile groups**	**Top predators (TL 4+)**	**Average trophic level**	**System structure (biomass ratios)**	**Network indicators**	**Diversity**	**Water quality**	**Total catch**	**CPUE**	**Value**
STRUCTURAL INDICATORS												
average trophic level	Weak	Weak	Moderate	Moderate	Strong	Moderate	Weak	None	None	None	None	None
high/low value biomass	None	None	None	None	None	Moderate	None	None	None	Moderate	Weak	Moderate
large:small biomass	Weak	Moderate	Moderate	Moderate	Moderate	Strong	Weak	Weak	None	None	None	None
plantivore:piscivore biomass	Weak	Moderate	Moderate	Moderate	Moderate	Strong	Weak	Weak	None	None	None	None
pelagic:demersal biomass	Moderate	Strong	Strong	Strong	Moderate	Strong	Weak	Weak	Weak	None	None	None
sedentary:mobile fish biomass	Weak	Weak	None	None	None	Moderate	Moderate	None	Weak	None	None	None
infauna:epifauna biomass	Weak	Weak	None	None	None	Moderate	Moderate	None	Weak	None	None	None
NETWORK INDICATORS												
Total Consumption	Weak	Weak	Weak	Weak	Weak	Weak	Strong	Weak	Moderate	None	None	None
Total Production	Weak	Weak	Weak	Weak	Moderate	Weak	Strong	Weak	Strong	Weak	None	None
DIVERSITY INDICATORS												
Reyni-0.1	None	None	None	None	None	None	None	Strong	None	None	None	None
Reyni-10	Weak	Weak	Weak	Weak	Weak	Weak	Weak	Strong	None	None	None	None
WATER QUALITY INDICATORS												
Chla	None	None	None	None	None	Moderate	Moderate	None	Strong	None	None	None
DIN	None	None	None	None	None	Moderate	Moderate	None	Strong	None	None	None
INDUSTRY INDICATORS												
Total catch	Weak	Moderate	Weak	Moderate	Weak	Weak	None	None	None	Strong	Moderate	Strong
Abalone CPUE	Strong	Moderate	Weak	Weak	None	None	None	None	None	Moderate	Strong	Weak

Key

- Strong
- Moderate
- Weak
- None (Nonlinear)

Indicator	Correlations											
	Commercial invertebrates	**Habitat dependent or reef associated**	**Pelagic or mobile groups**	**Top predators (TL 4+)**	**Average trophic level**	**System structure (biomass ratios)**	**Network indicators**	**Diversity**	**Water quality**	**Total catch**	**CPUE**	**Value**
Fish CPUE	Moderate	Strong	Moderate	Moderate	None	Weak	None	None	None	Moderate	Strong	Weak
Lobster CPUE	Strong	Moderate	Weak	Weak	None	None	None	None	None	Moderate	Strong	Weak
Shark CPUE	Weak	Weak	Moderate	Moderate	None	Weak	None	None	None	Moderate	Strong	Weak
Total value	Weak	Weak	Weak	Weak	Weak	Weak	None	None	None	Strong	Weak	Strong
AGGREGATE INDICATORS												
State of Nature	Moderate	Moderate	Moderate	Moderate	Weak	Weak	Weak	Weak	Weak	None	None	None
State of Socioeconomics	Weak	Moderate	Weak	Moderate	Weak	Weak	None	None	None	Strong	Strong	Strong

The results for each indicator are listed below:

Biomass Indicators

Abalone Biomass

- As noted above, Atlantis-SM did not capture the actual patterns of abalone biomass change seen. In the Maria Island MPA the biomass of surveyed abalone actually decreased rather than the increase the model projects. The current model doesn't capture some of the size dependent and crypsis related behaviours. Nor did it capture the changing predator-prey relationship between abalone and lobsters as lobsters increased in both size and abundance within the MPA
  - With this error in mind the fidelity is generally good (but performs less well with low or periodic sampling)
  - The signal strength of this indicator is robust with low productivity, but with moderate or high productivity the response is non-linear with distance and sampling design can reduce the signal.

Lobster Biomass

- Fidelity is generally robust, though is slightly weaker (falling to intermediate) if there is periodic (and occasionally low) sampling. In the low productivity model fidelity can be reduced by IUU or small closures, but still gives an intermediate level of performance.
- Signal strength is typically clear and at least of an intermediate level in most cases, but can take on a non-linear form once there is periodic or intense patchy sampling. This appears to be due to natural variation, especially in Victorian waters.

Blue Throat Wrasse Biomass

- The signal strength is always robust, though it drops with periodic sampling or if there is IUU or climate change and can be slightly weaker with smaller closures too. This means it performs less well under highly perturbed systems.
- Fidelity is also typically robust, though it drops off to intermediate with periodic sampling; and can be a little weaker (though still robust) with distance from the closure.
- This pattern of the results is typical of all habitat dependent (reef associated) fish groups examined in the model, both at an individual species level and at an aggregate "habitat associated fish" level.

Demersal Fish Biomass

- The fidelity isn't quite as good as for blue throat wrasse, but is still robust, except when there is periodic sampling (and occasionally low sampling) when the fidelity drops to intermediate.
- Signal strength is clear and robust for low or patchy sampling, but takes on a non-linear form once there is periodic or intense sampling (due to natural variation and

hotspots found outside of closure sites); climate change can also degrade the signal somewhat. It appears particularly useful in perturbed systems (e.g. with high level of pollutants, intensive or gauntlet fishing).

Medium Piscivore Biomass

- Fidelity is robust under moderate to high productivity, but drops to intermediate (or worse) if there is periodic sampling or low productivity and can be poor if there is strong gauntlet fishing.
- Signal strength is typically clear and robust, but takes on a non-linear form once there is periodic or intense sampling or the system (due to hitting natural 'hotspots' outside closures) or if the system is highly perturbed by strong fishing pressure or high nutrient contamination (but even then performance is still intermediate)

Shark Biomass (pelagic and demersal)

- Fidelity is robust under moderate to high productivity, but drops to intermediate if there is periodic or low sampling, low productivity (particularly for smaller closures); can be poor if there is strong gauntlet fishing.
- Signal strength is typically clear and robust for low or patchy sampling, but takes on a non-linear form once there is periodic or intense sampling (due to mobility and attractiveness of high productivity feeding sites outside closures causing natural variation across the model domain); performance can also impacted by high fishing pressure and gauntlet or IUU fishing (but still robust-intermediate performance).

Small Pelagic Biomass

- Fidelity is robust under moderate to high productivity, but drops to intermediate if there is periodic sampling and degrades further to poor if there is intense gauntlet fishing.
- Signal strength is typically clear and robust for low or patchy sampling (though under moderate productivity it can fall off even under patchy sampling), but takes on a non-linear form once there is periodic or intense sampling (as local productivity dominates the monitoring signal).

TL4+ Biomass

- Fidelity is not as good for this composite biomass indicator as for individual groups; it is at best of intermediate quality and is often poor (especially at a distance from the closure) for lower productivities or periodic sampling.
- Signal strength is frequently clear and robust for low or patchy sampling, but takes on a non-linear form once there is periodic or intense sampling; it can fall to poor when system is highly perturbed (e.g. under high fishing pressure or if a low productivity system under additional environmental stress from increased nutrient pollution of rising temperatures under climate change).

Habitat Cover

- Fidelity is not consistent at larger scales (though robust in the immediate area of the closure); its performance is also impacted when there is low productivity (as basal production in the system becomes the primary driver on it rather than other anthropogenically related pressures). When there is low productivity levels are depressed, meaning signal strength is poor (as there is insufficient contrast) and rates of recovery at the scale of the box (or spatial closure) are so slow that signal detection is quite difficult regardless of sampling regime. In cases with higher productivity there is a greater potential contrast and rate of detection.

- In those cases where it is representative of the attribute, it drops off very steeply and so is only informative at the local scale (at regional, broad and global scales there is no signal differentiation).

Zooplankton Biomass

- As this indicator responds to conditions on fine scales it has poor signal strength and fidelity with respect to providing information on the performance of spatial closures.

Proportion Mature

- The signal can be clear. It is better for longer lived (and more site attached species) but overall averages actually perform well across all taxa types (though when used in this way more samples are required if the overall averages are to be trustworthy). This indicator needs fairly intense sampling (to create adequate snapshots). It is clearer (and for more species) when there is heavy fishing pressure, when less sampling is required to detect a signal at the group level.

Structural Indicators

Average Trophic Level

- Fidelity is robust close to the closure (at local and regional scales), but drops off at broader and global scales.
- There is inconsistent signal strength for this indicator; patchy sampling shows a discernible signal with distance from the closure, but this is not evident under all sampling strategies.
- Noting that average trophic level can be hard to measure in reality, mean and maximum length are often proposed as an alternative. In this case these two lengths are reasonably to strongly correlated with ATL, although given that this indicator did not perform well in all cases, it is likely lengths would follow the same pattern.

High/Low Value Biomass

- The fidelity of this indicator is not consistent across spatial scales and falls away with distance from the closure.
- Signal strength is strong, but due to the lack of fidelity it's not always trustworthy and only shows such a strong response because many high value species are reef associated.

Large:Small Biomass

- Fidelity is robust at low productivity, but slips to intermediate for the rest.

- Signal strength is robust for large closures in all cases, but degrades to intermediate or poor with smaller closures (mobility of small and large fish degrading any signal introduced by the closure) or poorer overall system state (i.e. system under heavy pressure)

Planktivore:Piscivore Biomass

- With high productivity this indicator is of no value without intense sampling; at lower productivity the ratio has value close to closures, but shows no differentiation with distance from the closure.
- Fidelity is reasonable as the fidelity of the component parts is intermediate.

Pelagic:Demersal Biomass

- While there is often a clear relationship/signal it is inconsistent across sampling schemes and system states and so would be hard to interpret at fine scales. It is very much a context dependent indicator that is dependent on kind of drivers, magnitude of pressure, scale considered, and size of spatial closure.
- Fidelity is reasonable as the fidelity of the component parts is intermediate.

Infauna:Epifauna Biomass

- No clear signal for judging closure performance; it can be a perturbation indicator, but is not useful for spatial management.
- Fidelity is reasonable as the fidelity of the component parts is intermediate to robust.

Sedentary:Mobile Biomass

- Very clear signal as move away from MPAs, but confounded with location of habitat for species belonging to the different groups.
- Fidelity is reasonable in the region close to the MPAs as this contains suitable habitat for the sedentary groups.

Network Indicators

Total Consumption and Total Production

- At high productivity there is some fidelity to these indicators, but on the whole the network indicators show no clear signal strength or fidelity and are untrustworthy for judging the performance of spatial closures.
- Signal strength is poor at best.

Diversity Indicators

Reyni-Low Order

- No clear signal as richness doesn't change by much (through time) across all the scenarios.
- Fidelity is also typically poor.

Reyni-High Order

- A signal can be seen under low to moderate productivity, less so at high productivity.
  - The pattern vs intensity of sampling suggests that there may be some species-area curve issues occurring it is possible that there is a cumulative increase in the Reyni index signal with distance as the samples are drawn from larger area and pull in more groups than occur in the local area of the closure (confounding the signal due to the closure).
- Fidelity is also typically poor.

Water Quality Indicators

Chla

- No fidelity in signal due to box-to-box variation in environmental delivery of nutrients and resulting primary productivity.
- Do see a strong signal with distance, but this could be confounded by average depth and water clarity.
- As this indicator is strongly related to system productivity and water quality it is useful for interpreting drivers rather than just defining the state (so should be included as a supporting though not primary indicator).

DIN

- No fidelity in signal due to box-to-box variation in environmental delivery of nutrients and resulting primary productivity.
- The signal strength can be clear in some instances, but often shows a concave linearity with distance from the closure, particularly at low productivity (more linear change with higher productivity); this is again due to environmental delivery (including the processes of advection and diffusion) and is not useful as direct performance measure for closures (beyond being a means of monitoring water quality in its own right, which would be useful for interpretation when “things go wrong”).

Industry Indicators

Total Catch

- Fidelity drops off with distance from the closure; and is poorer with small closures or lower system productivity.
- Signal strength is intermediate to poor due to the non-linear nature of the signal with distance from the closure (signal is more asymptotic with larger closures); due to magnitude of catches and their distribution, the non-linearity is smaller when there is no closure; poor performance if fishing pressure is light (especially if a high productivity system and there are high pollutant loads in the area).

CPUE (all fished groups)

- This can be impacted substantially by the fishing and productivity scenarios. At its best CPUE is a reflection of relative abundance. However, its quality as an indicator can degrade quite rapidly - as there are many potentially non-linear steps during harvest of the resource and there can be economic and social drivers that deflect catch from matching the distribution of relative biomass. If nothing else is available it can be used with caution, but it should really be used in conjunction with other indicators rather than being the only indicator relied upon.
- Atlantis-SM doesn't really represent the economic and social drivers explicitly, but even without these, the CPUE signal could drop off quality-wise due to the steps of catchability, selectivity etc. In this study we used fairly simple representations of these so we are presenting the best cases.
- This indicator is also sensitive to extremes of productivity, the size of the closures and scale of reporting (due to confounding with patterns of catch and effort vs true distribution of the species).

Total Value

- The fidelity is intermediate, though drops off with distance, particularly with large closures.
- The patterns in signal strength reflect that in total catch.

Aggregate Indicators

State of Nature

- Fairly good, particularly in low productivity, high-pressure situations. If constituent indicators are impacted by natural variability then the aggregate is too, (making intensive sampling schemes something to be wary of when summarising information in this way). In high productivity systems then the information content of the aggregate also breaks down with distance from the MPA (as get clash of information content of different constituents at different distances). Overall the aggregate performs well if the constituents are meaningful.

State of Socioeconomics

- Same results as for the "State of Nature"

Implications for Sampling

It is clear from the above that not all indicators are useful for spatial management and that some are very sensitive to the sampling regimes modelled. To help guide selection of indicators for future sampling designs, the pressure performance classification shown in Table 5 provides some useful insights into which indicators worked in which situation. In that summary, "useful" means the indicator has robust fidelity and signal strength. "Use with care" means the indicator is generally intermediate-robust, but in some circumstances the signal strength is poor (either its performance is degraded, creating plateaus instead of linear trends or fidelity is reduced). These indicators would likely need to be supplemented with additional information to make sure they were being interpreted correctly, or should be dropped altogether if the system conditions change to one of the cases where their interpretation is not as reliable.

Table 6 presents correlation results between indicators and system attributes of interest. The primary aim here is for assessing indicator performance. However, this table, together with Table 7 (which shows correlations between selected indicators) also provide insights that could be used in developing sampling strategies.

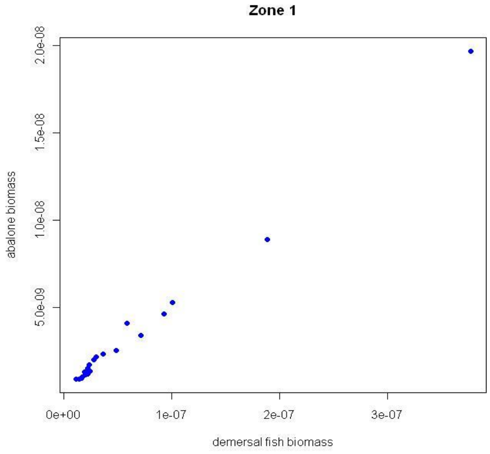
In particular, trimming sampling designs to increase cost effectiveness. For example the correlation between the distributions and levels of different groups means that selective sampling may be a good way of reducing monitoring costs (e.g. follow blue throat wrasse or lobster as proxy for many other reef associated groups). However, as Figure 18 shows, not all correlations are linear (whether strong, moderate or weak), but can also show non-monotonic and bifurcated relationships. This is a reflection of system spatial specificity that suggests that surrogates won't work universally (or their relationships won't be universal) and so should be selected with care.

It is clear from this analysis that many indicators are very sensitive to the type of sampling modelled here and sampling schemes are critical. For several species signal strength is clear and robust for low or patchy sampling, but takes on a non-linear form once there is periodic or intense sampling. This is due to natural variation and hotspots found outside of closure sites caused by, for example, mobility and attractiveness of high productivity feeding sites outside closures causing natural variation across the model domain. Although not explicitly modelled during this study it would appear that stratified sampling designs should help combat these problems. However, as the location of these hotspots may change through time (particularly given the anticipated consequences of climate change), future sampling designs will need to be adaptive to ensure indicators are robust and measure what is intended.

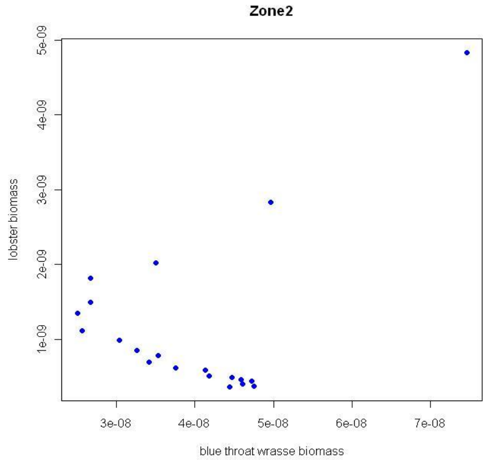
TABLE 7. CORRELATIONS BETWEEN SOME INDICATORS

Indicators	Abalone biomass	Bluethroat wrasse biomass	Demersal fish biomass	Lobster biomass	Medium piscivore biomass	Shark biomass	Small pelagic biomass	Average Trophic level	Infauna/epifauna biomass	total consumption	total production	Chl a	DIN
Abalone biomass	high	none	medium	high	high	high	medium	none	none	none	none	low	low
Blue throat wrasse biomass	none	high	low	none	none	none	none	none	none	none	none	none	none
Demersal fish biomass	medium	low	high	medium	medium	high	low	none	none	none	none	none	none
Lobster biomass	high	none	medium	high	high	high	medium	none	none	none	none	low	low
Medium piscivore biomass	high	none	medium	high	high	high	low	none	none	none	none	none	none
Shark biomass	high	none	high	high	high	high	medium	none	none	none	none	low	low
Small pelagic biomass	medium	none	low	medium	low	medium	high	none	none	low	low	medium	medium
Average Trophic level	none	none	none	none	none	none	none	high	none	none	none	none	none
Infauna/epifauna biomass	none	none	none	none	none	none	none	none	high	none	none	none	none
total consumption	none	none	none	none	none	none	low	none	none	high	high	low	none
total production	none	none	none	none	none	none	low	none	none	high	high	low	none
Chl a	low	none	none	low	none	low	medium	none	none	low	low	high	low
DIN	low	none	none	low	none	low	medium	none	none	none	none	low	high

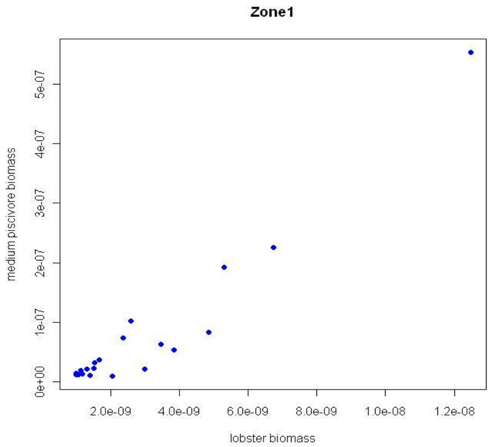
Key
high (r>0.9)
medium (0.7<r<0.9)
low (0.5<r<0.7)
none (r<0.5)



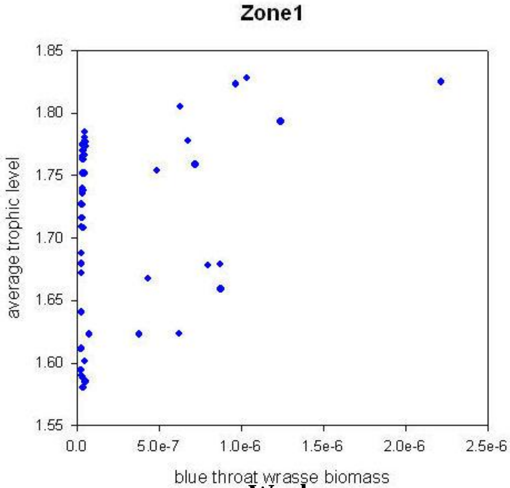
Strong – linear relationship



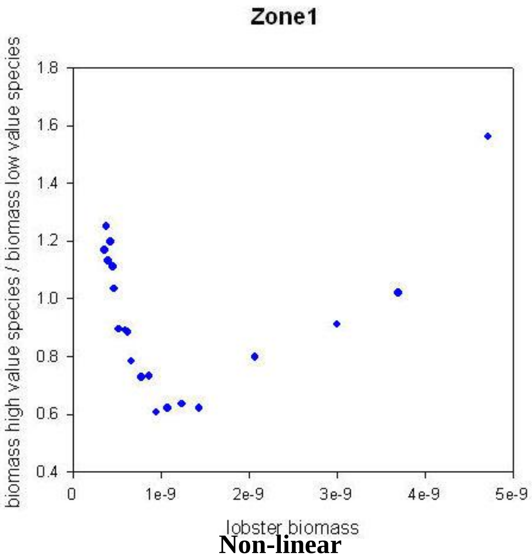
Strong – bifurcation (with time/location)



Moderate



Weak



Non-linear

FIGURE 18 EXAMPLES OF INDICATOR CORRELATIONS

Other results relating to sampling include:

– Chla and DIN are correlated strongly with overall production, but these should be monitored for ancillary or water quality reasons as they are not directly useful for judging performance of spatial closures

– Sampling schemes with low frequency and coverage are acceptable for detecting change inside and outside closures, but have little power to detect signals at broader spatial scales. Low level sampling schemes also need a reasonable time series (i.e. length through time) to enable causes of the signal to be evaluated.

– Fisheries dependent indicators should not be used alone unless there is no alternative (industry independent data is much preferred).

This last point has important consequences. In a brief review undertaken as part of this study, Murphy and Jenkins (Appendix G – later published as Murphy and Jenkins (2010)) review the observational methods used in marine spatial monitoring. Observing marine ecosystems is by nature difficult and costly. Observational methods fall into two broad categories, extractive and non-extractive. The latter methods have particular relevance in spatial zoning arrangements, such as MPAs where extractive sampling methods are prohibited. Field et al (2006) discuss the implications of the latter for fisheries assessment.

Fishery independent trawl surveys, using commercial or research vessels, are commonly used around the world. The objectives of such surveys are to inform management of targeted resources and trends in the impacts of fishing more generally, including changes in trophic structure and biodiversity (Rice and Gislason 1996; Fogarty and Muraswski 1998; Jennings et al 2002; Link 2005; Lauth 2010). Ongoing trawl surveys that could be used as the basis for assessing changes in indicators are uncommon in Australia. Notable examples include an integrated monitoring program in the Northern Prawn Fishery (Dichmont et al 2003) and the proposed fishery independent surveys in the Southern and Eastern Scalefish and Shark Fishery (Knuckey pers comm). In most cases trawl surveys have been once off or of limited duration to respond to specific research questions (eg Williams and Bax 2001; Williams et al 2010). Clearly, trawls surveys are not possible in many areas due to bottom type. Fishery independent surveys also use other commercial or extractive methods such as line, gillnet and trap.

Fishery independent non-extractive methods used for observing marine systems include underwater visual census for shallow water habitats (Edgar et al 2004; Watson and Harvey 2007), underwater video systems often linked with acoustics (Shortis et al 2008, Foster et al 2009), baited remote underwater videos (BRUVS, Harvey et al 2007), acoustics (Kloser et al 2002, 2009, Ryan et al 2009, Schlacher et al 2010), acoustic tagging, and underwater autonomous vehicles (AUVS).

New approaches that might also prove useful include nitrogen isotopes of specific amino acids and genomics (Andy Revill, CSIRO pers comm.).

By using a combination of observational techniques to target specific species or habitats, spatial monitoring surveys can provide information on the whole ecosystem (Murphy and Jenkins 2009). However, the efficacy of many of these methods for the sustained observing required to monitor marine systems has still to be demonstrated. To quote from Australia's Integrated Marine Observing System Five Year Strategy and related documentation:

"From a global ocean observing system perspective, sustained biological observing remains immature. This is recognised as global challenge for the next decade. Within Australia, one of the ten strategic priorities for IMOS is Exploring the potential for whole-of-system approaches. IMOS is recognised internationally as being at the forefront of attempts to integrate from the open ocean, onto the continental shelf and into the coast, and across physics, chemistry and biology. This is a significant challenge that will require ongoing effort and attention over the next decade, and effective collaboration between the observational and modelling communities will be essential"

General Discussion

In this study, Atlantis-SM was developed to evaluate ecological indicators and the power of alternative monitoring regimes in a spatial context. To the authors' knowledge it represents the first such study undertaken. Compared with previous studies, in broad terms the indicators checked show that overall indicator performance identified by Fulton et al (2005) still holds. However, it also highlighted that monitoring for EBM performance may be far from simple. Monitoring schemes with small spatial coverage or infrequent temporal repetition (on the order of 3-5 years or more) had no power to rapidly detect changes in the system; while intensive sampling was confounded by natural system variation and shifts through time, unless carefully planned around stratified sampling schemes. Moreover, indicators such as pelagic:demersal fish biomass, that have been found to be useful across different system types, proved sensitive to scale. These indicators were informative in the immediate area of closures (as the data at this scale is within habitat patches and individual species ranges and so avoid species-scale mis-matches) and globally (because at such large scales the ratio integrates across many species effectively smoothing out any potential mis-matches). However, they do not work at intermediate scales because these exceed the typical spatial range of activity of individual species, but are not yet at a point where they smoothly integrate across sufficient groups.

The ecology of the groups in the system also impacts the performance of individual indicators based on those groups. For example, signals for mobile species (e.g. small pelagics) can be over-stated outside reserves, while signals for more sedentary species (e.g. reef associated demersal fish) decay rapidly with distance from the closures.

At the community scale, Atlantis-SM also suggests that variation in community dynamics between regions can lead to locally specific indicator-attribute relationships; meaning that while indicator signals are representative of the attribute at a specific locale, they may not always be consistent site-to-site. For instance, in two different regions the relationship between the indicator relative lobster biomass and the attribute diversity was linear (with $R^2 > 0.92$), but in opposite directions (in one case there was a positive correlation and in one a negative) at sites less than 300km apart (Figure 19). This difference in direction of response is due to locally specific environmental drivers and community dynamics and has significant implications for monitoring and management, as it shows that an understanding of system dynamics at regional scales will be necessary to understand the signal obtained from indicators. This suggests that universal reference points (analogous to $B_{0.4}$ in fisheries) or directions, which do not take into account local specificity, may not be feasible.

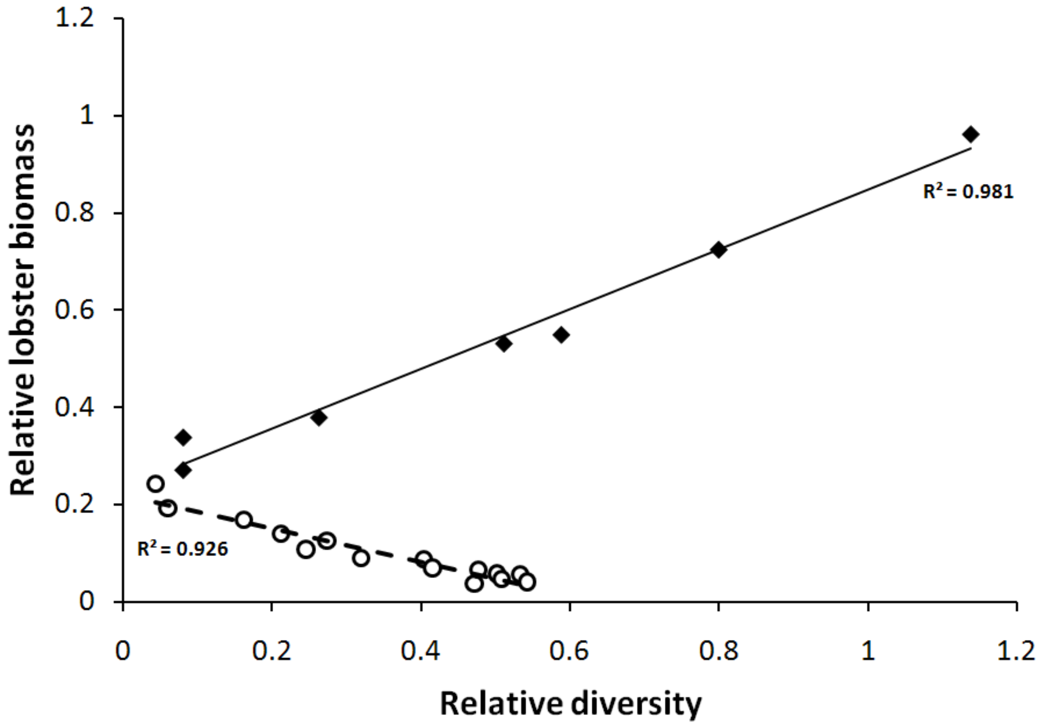


FIGURE 19: EXAMPLE INDICATOR (RELATIVE LOBSTER BIOMASS) VS ATTRIBUTE (RELATIVE DIVERSITY) RELATIONSHIP IN DIFFERENT REGIONS OF ATLANTIS-SM

This finding is at odds with recent literature on indicators (Rochet and Trenkel 2003, Cury et al 2005, Jennings and Dulvy 2005, Trenkel et al 2007, Bundy et al 2010), which not only recommends a definitive set of indicators across many systems and scales but also recommends the use of reference points that are intended to be consistent across systems. Instead suites of indicators drawn from the main general classes of indicators noted above (e.g. relative biomass, biomass ratios, relative habitat cover) will need their associated reference points or directions adjusted to suit status and processes at the locations of interest (and potentially through time as the system changes). Crucially, this also means that a lack of a temporal dimension in monitoring cannot be completely compensated for by periodically applying very intensive surveys across broad spatial scales.

These are strong ecological reasons why a suite of indicators will be needed to capture performance of spatial management. Management objectives typically extend beyond the ecological however. Consequently, inclusion of social and economic objectives is still more reason for a suite of indicators to cover all aspects of the system (and ultimately achieve a triple bottom line outcome).

Both socioeconomic and environmental drivers can affect indicator performance and degrade their signals. For instance, lower productivity and higher fishing pressure can weaken indicator usefulness (impacting fidelity in particular). This is especially so for biomass indicators of groups that have restricted distributions or movement.

Another outcome of this research into performance measurements of spatial management are the implications for monitoring designs. The clearest results from this analysis are that (i) periodic sampling has signal content that can be hard to interpret due to the influence of natural "hotspots" (intensive sampling could be impacted in the same way) and (ii) careful thought must be put into monitoring schemes (e.g. use of stratified sampling at broader scales). There is also an important need to further test and develop cost effective monitoring methods.

Benefits

The main and most direct beneficiaries of this research will be the fisheries and conservation managers responsible for spatial management and the sectors that are affected by these management arrangements. The study shows, and this is generally consistent with experience, that current sampling intensity is adequate to detect signals for inside/outside of closures such as MPAs but has little benefit for broader spatial considerations. This is an important result giving the increasing realisation that off-reserve management is as important as reserve management. It also means that researchers/managers cannot always assume that the expected response in their particular case will match a response found elsewhere. The responsiveness of an indicator might be useful but the reference direction may be site specific. A "recipe book" approach is unlikely to be useful. The study also highlights and that due to logistical constraints it may not be possible to measure trends in all aspects of the system of interest so surrogates will remain important components of monitoring approaches. The study provides useful guidance on this.

Finally, the study contributes to the ongoing debate about reference directions/points for indicators. Our results indicate that for assessing the performance of spatial management such approaches will not be appropriate.

Further Development

The Atlantis modelling framework overall structure was used to make sure it was well suited for use in Management Strategy Evaluation (MSE). This is a simulation approach that enables the consequences of alternative management strategies to be assessed (Smith et al 1999, Fulton et al 2007). In this study, in the strictest sense of MSE, the method used to assess indicator performance was not a full MSE as it does not have an explicit feedback loop. However, it is a closely related variant that plays on the strengths of the approach. It is in essence a variant of the MSE approach that has been used previously. Ludwig and Walters (1981) used a complex population model as operating model and then tried variants of that complex model to see if they performed better in the actual assessment and management of the system than the simple production models for fisheries. Fulton et al (2005) used this approach to identify robust ecological indicators. Further work on indicator performance may be possible using a smaller subset in a fully adaptive MSE framework where adaptive management monitoring programs can be simulated. Such an analysis would not be for the full set of scenarios, because computing requirements could prove prohibitive in that case. However, specific areas of interest could be explored (e.g. base case with small MPA but with different levels of nutrients or climate change etc), as could different levels of sampling.

One area that is beyond the current capabilities of Atlantis (and most other ecosystem models, like EwE) is the explicit handling of biodiversity. While model based indices like Kempton's Q (Kempton and Taylor 1976, Ainsworth 2006) can be applied, more typical indicators such as richness or evenness can not (as there are too few groups explicitly included in the model to represent it effectively). Atlantis requires further development to allow this to be more effectively and dynamically modelled.

Another aspect that would benefit from more sophisticated representation is the handling of socioeconomic components. In the current study fishing pressure was imposed at the current realistic level, at 5 times this level and at half this level. While Atlantis has the capacity to incorporate dynamic fishing fleets, this study is a strategic investigation of the implications of gross shifts in fishing pressure, therefore we simply enforced a constant fishing mortality rate on each fished group. This gives greater control over the proportion of the population that is landed, and removes some of the noise associated with variations in fishers' behaviour that a dynamic fishing model can impose. The implications of this were that we did not explore sophisticated social or economic indicators. However, the ones evaluated are standard and, given the simplicity of the model, are actually "best case" for detecting effects as we removed much of the nonlinearity that would occur. So in this case the model test remains valid. Further development of Atlantis-SM with a full fleet dynamics component rather than the fixed F would be of benefit.

Lastly, it was clear there are clear spatial differences in the dynamics of some species, for example abalone and rock lobster between Victoria and Tasmania. As Atlantis-SM is developed further we need to ensure these spatial dynamics are captured as well as possible. Moreover, Victorian data is needed to validate/verify finding in model that sees different patterns at different locations and at different scales.

Planned Outcomes

This study is, as far as the authors are aware the first 'whole of system' approach to performance evaluation of spatial management. The results of this study should lead to improved effectiveness and efficiency of management using MPAs but more importantly broader spatial management approaches as tools to achieve ESD for marine resources and ecosystems.

An important outcome of the project has been that Atlantis-SM, and what we have learned from its development and application, now provides a "transportable" framework for developing performance measures. The basic approach, both the telescoping treatment of habitats in and around spatial closures and the MSE framework for representing the estimation of indicators, can be applied to other systems via new implementations of the Atlantis model. However, even without going that far it is possible to take the lessons learnt in this case to other systems. In particular it contributes to the policy debate around the implementation of EBM. It highlights the potential for monitoring for EBM performance to be far from simple. Monitoring schemes with small spatial coverage or infrequent temporal repetition (on the order of 3-5 years or more) had no power to rapidly detect changes at the broader system level. In addition, the finding that there is likely to be system specific reference points is particularly important, as the tendency within the literature has been to try and find generic rules and approaches for universal application and broad scale comparisons. The indication from this study, that universal reference points (analogous to $B_{0.4}$ in fisheries) or directions, which do not take into account local specificity, may not be feasible, has important consequences as is at odds with recent literature and suggests that adoption of such approaches could lead to a very misleading interpretation of management performance.

There has been considerable interest in this study. The Marine Policy section of the "Aquatic Biodiversity, Biodiversity & Ecosystem Services" part of the Victorian Department of Sustainability & Environment is looking to use the models and our findings to inform their monitoring and handling of spatial management and conservation of reef systems along the Victorian coasts and in PPB under the "Seagrass and Reef Program for Port Phillip Bay". Scientists in the US are also keenly interested in these results, as spatial management is being proposed as the cornerstone of their approach to ecosystem based management, but as yet they have no means of evaluating the performance of such an approach despite requirements to do just that.

Conclusions

Objective 1: "Through an analysis of monitoring data from existing marine system management regimes (including MPAs) and an identification of observational approaches that are available to be used, develop simple biophysical and management models of impact and response at various spatial scales".

In this study we reviewed the available information for monitoring for spatial management and associated performance measures for programs both in Australia and overseas. The overseas monitoring programs reviewed were from the Philippines, the Caribbean, Indonesia, California, New Zealand, South Africa, Kenya, France and Ecuador. Australian programs reviewed were from Queensland, Tasmania, New South Wales, Victoria, and the Great Australian Bight. The majority of these programs were associated with spatial management of marine protected areas (MPAs). The review also considered monitoring for social and economic objectives of spatial management. We also considered the recent Australian and international literature on ecological indicators and observational approaches for the spatial management of marine systems. A key outcome from the study into performance measurements of spatial management are the implications for monitoring designs.

In terms of model development, the study was able to go further than was expected. The Atlantis modelling framework was developed prior to the commencement of the study, but this framework was extended explicitly for thus project to create Atlantis-SM. This model was calibrated using time series data from Victoria and Tasmania and was able to spatially simulate MPAs in the south east of Australia.

Objective 2: "Use these models to develop and evaluate measures to report performance for specified management objectives particularly in respect of power to detect change".

In this study relative biomass was again found to be a reliable indicator. However, not all species' biomass was equally reliable. In general, using the relative biomass of a highly mobile species as an indicator gave an over-stated signal outside reserves. In contrast, sedentary species were useful for detecting impacts inside a reserve, but under-stated impacts at wider spatial scales. Biomass ratio indicators were similarly impacted by questions of scale. These indicators capture system structure well in the immediate area of closures (because at such fine scales individual species are considered at the scales at which they act) and they work globally (because at such large scales the ratio integrates across many species effectively smoothing out any mis-matches in scale for individual species). Yet they do not work at intermediate scales as these scales exceed the critical lengths scales of individual species but are not yet at a point where they smoothly integrate across sufficient groups. This study suggests that while common classes of indicators are consistently emerging as useful across system types, thought must still be given to appropriate indicators at different spatial scales. In addition, it appears that variation in community dynamics between regions can lead to locally specific indicator-attribute relationships; meaning that while indicator signals were representative of the attribute at their specific locale, they may not always be consistent site-to-site. This has significant implications for monitoring and management, as an understanding of locally specific environmental drivers and community dynamics will be necessary to understand the signal obtained from indicators. This suggests that universal reference points (analogous to $B_{0.4}$ in fisheries) or directions, which do not take into account local specificity, may be infeasible. This finding is at odds with recent indicators literature

(Rochet and Trenkel 2003, Cury et al 2005, Jennings and Dulvy 2005, Trenkel et al 2007, Bundy et al in review), which recommends the use of reference points that are intended to be consistent across systems in terms of reference points (or at least reference directions) and a definitive set of indicators across many systems and scales. Beyond the need to cover sedentary and mobile species and socioeconomic and ecological objectives, this system specificity reinforces the need for suites of indicators whose membership can be adjusted to suit status and processes at different locations (and potentially through time as the system changes). So despite the observation that there may not be one specific reference point indicator, these indicators will likely fall into some of the main general classes of indicators noted above (e.g. relative biomass, biomass ratios, relative habitat cover). The other implication of this work is that long term monitoring schemes will need to be carefully designed to avoid misinterpretation of any signals detected. Crucially, this work clearly showed that a lack of a temporal dimension in monitoring cannot be completely compensated for by periodically applying very intensive surveys across broad spatial scales.

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Appendix A Intellectual Property

No intellectual property has arisen from the project that is likely to lead to significant commercial benefits, patents or licences. Any intellectual property associated directly with this project will be shared between the Fisheries Research and Development Corporation, CSIRO, Department of Primary Industries Victoria and the Tasmanian Aquaculture and Fisheries Institute.

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Appendix C Life History Parameters

Table C.1: Dispersal and movement classes for modelled groups. Note larval dispersal is only appropriate for vertebrate groups

Model Component	Movement Type	Movement Outside Model Domain	Level of Larval Dispersal
Large phytoplankton	Advected	Boundary condition mixing[5]	NA
Small phytoplankton	Advected	Boundary condition mixing	NA
Gelatinous zooplankton	Advected	Boundary condition mixing	NA
Large zooplankton	Advected	Boundary condition mixing	NA
Mesozooplankton	Advected	Boundary condition mixing	NA
Small zooplankton	Advected	Boundary condition mixing	NA
Carvivorous infauna	Advected	Boundary condition mixing	NA
Deposit feeders	Local[1]	Boundary condition mixing	NA
Deep water filter feeders	Site attached[2]	NA	NA
Shallow water filter feeders	Site attached	NA	NA
Urchins	Local	Boundary condition mixing	NA
Deep water megazoobenthos	Local	Boundary condition mixing	NA
Shallow water megazoobenthos	Local	Boundary condition mixing	NA
Meiobenthos	Local	Boundary condition mixing	NA
Macroalgae	Site attached	NA	NA
Seagrass	Site attached	NA	NA
Squid	Broad[3]	NA	Broad
Shallow water herbivores	Limited[4]	NA	Broad
Banded morwong	Broad	NA	Broad
Shallow demersal fish	Limited	NA	Broad
Planktivorous reef fish	Site attached	NA	Broad
Deep demersal fish	Broad	NA	Broad
Zebra fish	Site attached	NA	Broad
Silver sweep	Site attached	NA	Broad
Magpie perch	Limited	NA	Broad
Seahorses, pipefish, gobies	Site attached	NA	Local
Herring cale	Broad	NA	Broad
Purple wrasse	Site attached	NA	Broad
Blue throat wrasse	Site attached	NA	Broad
Blue-eye trevalla, warehou	Broad	NA	Broad
Small pelagic fish	Broad	NA	Broad
Mackerels	Broad	NA	Broad
Shallow piscivores	Broad	NA	Broad
Migratory mesopelagics	Broad	NA	Broad
Non-migratory mesopelagics	Broad	NA	Broad
Pink snapper	Broad	NA	Broad

Model Component	Movement Type	Movement Outside Model Domain	Level of Larval Dispersal
Tunas and billfish	Broad	NA	Broad
Dogfish	Limited	NA	Local (with mother)
Demersal sharks	Broad	NA	Local (with mother)
Large Pelagic sharks	Broad	NA	Local (with mother)
Dogshark	Broad	NA	Local (with mother)
Skates and rays	Broad	NA	Local (with mother)
Baleen whales	Broad	Seasonal migration	Local (with mother)
Dolphins	Broad	NA	Local (with mother)
Orcas	Broad	NA	Local (with mother)
Seals	Broad	NA	Local (with mother)
Abalone	Limited	NA	Local
Prawns	Broad	NA	Broad
Lobster	Limited	NA	Broad
Seabirds	Broad	Seasonal migration	Local (with mother)

1 Local movement includes movement within a box and a very slow diffusion across boundaries consistent with small scale local movements taking some individuals over the boundary between boxes

2. Site attached invertebrates are typically sedentary (literally attached to the substrate), while vertebrates are behaviourally site attached (e.g. to reef-based home ranges)

3. Broadly mobile, capable of crossing the entire model domain within the course of a year

4. Mobile, but not wide ranging over time periods on the order of weeks-months

5. Boundary condition mixing represents local movement processes mixing with boundary conditions around the edge of the model domain.

6. Seasonal migration represents large scale migration by a majority of the population in and out of the model domain seasonally

Appendix D Sampling Model Parameters

The following are the default parameters used in the Atlantis sampling models. Alternatives were tried, but these are the default parameter values were used for the majority of runs.

Table D.1: The default bias and variance sampling parameters

Parameter	Value	Parameter	Value
Proportional bias of sampling mean of		***Error variance (as proportion of the sampling mean) of***	
Salinity	1.0	salinity	0.01
physical properties	1.0	physical properties	0.25
nutrients	1.0	nutrients	0.25
processes (nitrification, denitrification)	1.0	processes (nitrification, denitrification)	0.25
large phytoplankton biomass	1.0	large phytoplankton biomass	0.36
small phytoplankton biomass	1.0	small phytoplankton biomass	0.49
small zooplankton biomass	0.5	small zooplankton biomass	1/0
large zooplankton biomass	1.0	large zooplankton biomass	0.36
cephalopods biomass	1.0	cephalopods biomass	0.36
pelagic bacteria biomass	0.7	pelagic bacteria biomass	0.49
sediment bacteria biomass	0.5	sediment bacteria biomass	0.49
small infauna biomass	0.5	small infauna biomass	2.0
large infauna biomass	1.0	large infauna biomass	1.5
sessile epifauna biomass	1.0	sessile epifauna biomass	0.36
mobile epifauna biomass	1.0	mobile epifauna biomass	0.36
benthic primary producer biomass	1.0	benthic primary producer biomass	0.36
refractory detritus biomass	1.0	refractory detritus biomass	0.36

Parameter	Value	Parameter	Value
labile detritus biomass	1.0	labile detritus biomass	0.36
vertebrate biomass	1.0	vertebrate biomass	0.36
pelagic primary production	1.0	pelagic primary production	0.1
zooplankton production	1.0	zooplankton production	0.1
cephalopod production	1.0	cephalopod production	0.1
pelagic bacteria production	1.0	pelagic bacteria production	0.1
sediment bacteria production	1.0	sediment bacteria production	0.2
small infauna production	1.0	small infauna production	0.2
large infauna production	1.0	large infauna production	0.2
sessile epifauna production	1.0	sessile epifauna production	0.2
mobile epifauna production	1.0	mobile epifauna production	0.2
benthic primary producer production	1.0	benthic primary producer production	0.2
zooplankton consumption	1.0	zooplankton consumption	0.1
cephalopod consumption	1.0	cephalopod consumption	0.1
pelagic bacteria consumption	1.0	pelagic bacteria consumption	0.2
sediment bacteria consumption	1.0	sediment bacteria consumption	0.2
small infauna consumption	1.0	small infauna consumption	0.2
large infauna consumption	1.0	large infauna consumption	0.2
sessile epifauna consumption	1.0	sessile epifauna consumption	0.2
mobile epifauna consumption	1.0	mobile epifauna consumption	0.2
vertebrate weights	1.0	vertebrate weights	0.001
vertebrate production	1.0	vertebrate production	0.36

Parameter	Value	Parameter	Value
vertebrate consumption	1.0	vertebrate consumption	0.36
vertebrate discard rates	0.7	vertebrate discard rates	0.25
vertebrate total catch	0.8	vertebrate total catch	0.25
vertebrate total effort	0.9	vertebrate total effort	0.1
vertebrate total discards	0.8	vertebrate total discards	0.36
counts	1.0	counts	0.25
numbers observed in the catch	0.9	numbers observed in the catch	0.1
selectivity curve fitting[1]	1.0	selectivity curve fitting[1]	1.0
parameters of the selectivity curve	1.0	parameters of the selectivity curve	0.3
Aging	1.0	aging	1.0

1. When set to a value other the one incorrect selectivity curves may be selected for fitting and stock estimation.

Table D.2: Default r_{max} parameter settings for the calculation of potential biological removals in the sampling model

Group	r_{max}	Group	r_{max}
small planktivorous fish	0.6	seabirds	0.4
large planktivorous fish	0.6	pinnipeds	0.35
shallow piscivorous fish	0.4	baleen whales	0.4
deep piscivorous fish	0.4	toothed whales	0.35
tropical piscivorous fish (tunas)	0.4	flathead (*Neoplatycephalus spp*)	0.35
migratory mesopelagic fish	0.6	ling (*Gentyperus blacodes*)	0.15
non-migratory mesopelagics	0.6	orange roughy (*Hoplostethus atlanticus*)	0.07
shallow demersal fish	0.4	southern bluefin tuna (*Thunnus maccoyii*)	0.07
deep demersal fish	0.45	gummy shark (*Mustelus antarcticus*)	0.07
demersal sharks	0.45	cephalopds	0.8
pelagic sharks	0.35		

Appendix E

Measuring the Performance of Spatial Management in Marine Systems: a Review

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November2008

Victoria
The Place To Be

Measuring the Performance of Spatial Management in Marine Systems: a Review

Gregory P Jenkins, Jodie Kemp, Jodi Ryan

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Executive Summary

Monitoring for spatial management and associated performance measures were reviewed for programs both in Australia and overseas. Overseas monitoring programs reviewed were from the Philippines, the Caribbean, Indonesia, California, New Zealand, South Africa, Kenya, France and Ecuador. Australian programs reviewed were from Queensland, Tasmania, New South Wales, Victoria, and the Great Australian Bight. The majority of these programs were associated with spatial management of marine protected areas (MPAs). The only monitoring programs for spatial management reviewed that were not directly associated with MPAs were the Australian Institute of Marine Sciences Long-term Monitoring Program of the Great Barrier Reef and the Victorian Abalone Assessment Monitoring Program. Monitoring for ecosystem management objectives of MPAs considered both within reserve effects, for example biomass accumulation, and outside reserve effects, for example export of accumulated biomass or propagules across the reserve boundary (spillover). The review also considered monitoring for social and economic objectives of spatial management. A description of the methodology and primary outcomes of each monitoring program is provided as well as a summary of metrics (variables) and performance measures used. A range of metrics have been used in monitoring studies for spatial management, the chosen metric to some extent depending on the management objective under consideration.

Overall, the review concludes that management objectives for spatial management, particularly MPAs, tend to be very general and poorly defined. Objectives need to be framed in a way that management performance can be assessed though monitoring. A suite of suitable metrics are available for this monitoring; however, planning for performance assessment must begin at the time of initial planning for the spatial management, rather than relying on *ad hoc* studies once the management regime is in place. In framing management objectives, many agencies have considered a relatively small spatial scale, associated with individual MPAs and adjacent areas. In future, management objectives should be set at a regional scale so that the overall performance can be assessed. There needs to be a strong commitment to performance assessment; for example, many of the effects of MPAs are not evident for at least a decade. Spatial management in the coming years is likely to broaden considerably from a concentration on MPA management, particularly with the increasing focus on spatially-explicit fisheries management and the ecological effects of fishing, and also on environmental perturbations such as climate change. Performance measures for this type of monitoring need to be based as much as possible on sound ecological knowledge of responses to perturbations, rather than the arbitrary setting of limits with little ecological basis.

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Introduction

Management within a spatial context is becoming increasingly common in marine systems as part of a global movement toward ecologically-sustainable development and ecosystem-based management. Areas such as fisheries management are increasingly moving to a spatial context; however, the most profound development in spatial management in recent years has been the global proliferation of Marine Protected Areas (MPAs). There are many claimed benefits of MPAs in terms of both the protected area and also the broader system in which it is placed. Preservation of fauna and habitat from the damaging effects of human activities such as fishing is often one of the primary aims of establishing marine reserves (Halpern 2003). It is also often anticipated that reserves may be effective as a fishery and conservation tool for organisms that have sedentary adult life-stages and exhibit larval dispersion, enabling biomass exportation to the surrounding areas (Nowlis and Roberts 1999). There is, however, a lack of evidence that reserves are effective for fisheries management. Many studies are poorly designed, and any reported increases in density within reserves have low statistical power (Russ 2002; Sale *et al.* 2005). The benefits to fisheries outside reserves is much less studied than effects within reserves, and to date there is little empirical support for such wider regional benefits of MPAs to fisheries (Sale *et al.* 2005).

To truly understand the implications of spatial management, such as MPAs, there needs to be clear ecological, social and economic management objectives that have a system in place allowing measurement of performance of the spatial management in relation to the objective(s). The aim of the present paper is to review monitoring programs that have been initiated to measure the effectiveness of spatial management in relation to stated objectives. The review also aims to assess the range of indicators/metrics that have been used to measure performance, and, as a result, provide recommendations of the most appropriate performance measures to be used in future assessments of spatial management.

This review summarises management objectives, performance measures, and monitoring methodology and outcomes from existing literature and classifies the literature into monitoring that has been conducted 1) *World Wide*, and 2) in *Australia*, and further classifies these domains into the effect of spatial management in terms of ecosystem, social and economic factors. Monitoring that addresses the effects of MPA management on the ecosystem is divided into "Within Reserve" and "Outside Reserve" effects. "Within Reserve" effects would include change in population parameters, assemblage structure, biodiversity and habitat parameters inside the reserve relative to outside. "Outside Reserve" effects include the exportation of biomass (spillover) and recruits (recruitment subsidy) to the outside of a marine park area. In general, short-term studies (1-3 years) were not included except where general principles were illustrated.

Objectives

- A review of existing monitoring programs used in marine spatial management,
- A review of measures that have been used to assess the performance of spatial management in relation to management objectives.

Monitoring for Spatial Management *World Wide*

Philippines

Marine reserves were established at Sumilon Island and Apo island in 1974 and 1982, respectively (Russ and Alcala 1999). Primary management objectives for these reserves included: 1) to protect the habitat of fish in the reserve, 2) to allow the build up of fish biomass in the reserve, 3) to increase the fish yield at the islands by export of adult and larval fish from the reserve to fished areas, and 4) to encourage tourism (Russ and Alcala 1999). Management objectives were met more successfully at Apo Island where there was greater community support leading to long-term reserve protection (Russ and Alcala 1999). Management objectives were less successful at Sumilon Island due to fluctuating levels of community support caused by socio-political factors (Russ and Alcala 1999).

Ecosystem Effects

Within Reserve Effects

Quantification of the effects of marine reserve establishment requires the identification metrics that are optimal as indicators of reserve performance in relation to management objectives. When the Sumilon Island and Apo Island marine reserves were established, a long-term monitoring program was initiated to survey whether the reserve's desired goals were being achieved. Incorporating information from the long-term monitoring program into the research design, Russ and Alcala (1996; 2003) investigated the rate and pattern of density and biomass change within and outside of the reserves. Russ and Alcala (1996; 2003) also determined how quickly any gains, potentially useful to fisheries, are lost if reserves are subsequently re-opened to fishing.

At the Sumilon Island and Apo Island marine reserves, Russ and Alcala (1996; 2003) investigated whether or not 'abundance of large predatory reef fish' was a good indicator of the effects of marine reserve protection. The research species were from four different families (Serranidae, Lutjanidae, Lethrinidae and Carangidae) that all have inherent life history characteristics (long life, slow growth, low rates of natural mortality, and low and variable rates of recruitment) that increase their potential as good indicators of fishing pressure on coral reefs, and are also highly targeted by fisheries (Russ and Alcala 1996; 2003). For an outline of the research methods employed by Russ and Alcala (1996; 2003) see Table 1.

Russ and Alcala (2003) found that there were eight significant increases in density and biomass and four of these increases occurred when reserve status was applied (Russ and Alcala 2003). Of the four significant increases in density and biomass, three of these increases required 4-6 years of protection (Russ and Alcala 2003). Russ and Alcala (2003) found that there were three significant declines in density and biomass and these declines occurred when reserve protection was removed; twice within the Sumilon Island reserve (1985, 1992) and once outside the reserve (1992).

There were substantial temporal changes in biomass of large predators and these trends tended to reflect changes in density. However, initial change in biomass did not increase as rapidly as density following application of reserve status (Russ and Alcala 2003). This indicated that even though fishing mortality was reduced or eliminated in reserves, there appeared to be considerable time delay before populations of large predatory fish attained a size structure with a high mean and modal size (Russ and Alcala 2003). A rapid increase in density relative to biomass was most apparent when 'recruitment pulses' were observed (Russ and Alcala 2003).

Following the removal of reserve status, density and biomass of large predatory fish declined rapidly over 2-3 years (Russ and Alcala 2003). Thus, the rate of loss of density and biomass when a reserve was first opened to fishing was much faster than the rate of gain when the reserve was first closed (Russ and Alcala 2003). These results have considerable management significance, suggesting long-term protection and management is required to achieve fishery benefits (Russ 2002).

To determine the duration of protection required for populations of large predatory reef fish to attain natural states, Russ and Alcala (2004) further investigated density and biomass trends from the long-term monitoring program at the Sumilon and Apo Islands. Russ and Alcala (2004) found that the biomass of fish targeted by fisheries increased exponentially when protected within marine reserves. The biomass of large predatory fish was continuing to increase exponentially after 9 and 18 years of protection at the Sumilon and Apo reserves respectively. There was little evidence that the rate of accumulation of biomass inside the reserves was slowing down even after so many years of protection suggesting that the length of time to full recovery will be considerable (Russ and Alcala 2004). Two assumptions were made in order to estimate this period. First, biomass growth will follow the logistic model; and second, the conservative assumption that biomass had already attained 90% of the local carrying capacity of the environments in the reserves. Russ and Alcala (2004) concluded that the time required for full recovery of protected populations will be 15 and 40 years at the Sumilon Island and Apo Island reserves respectively. These estimated times of recovery appear consistent with known life history characteristics of these fish and with empirical data on recovery rates of heavily-exploited fish stocks (Russ and Alcala 2004).

A further study (Alcala *et al.* 2005) was designed to evaluate whether the Sumilon Island and Apo Island marine reserves affected the build up of biomass of important fishery species within and outside reserves. Alcala *et al.* (2005) used underwater visual census to investigate the biomass trends of five families of reef fish (Acanthuridae, Carangidae, Lutjanidae, Lethrinidae, and Caesionidae) that constitute a large proportion of the fisheries yield at both islands between 1983 to 2001. For an outline of the research methods employed by Alcala *et al.* (2005) see Table 1.

Alcala *et al.* (2005) found that the biomass of target fish increased within reserves 3-4.5 fold over 9-18 years. Biomass outside of the reserves showed no increase. A difference in the fish biomass between reserve and non-reserve sites was apparent at around 6 years of reserve protection at both Sumilon and Apo Island marine reserves. Biomass of targeted fish had a significant positive correlation with years of marine reserve protection inside but not outside the Sumilon Island marine reserve. The biomass of targeted fish (minus one family: Caesionidae) at the Apo Island marine reserve also had a significant positive correlation with years of reserve protection inside but not outside the marine reserve over the full 18 years of the study. The lack of within-reserve effect for the Caesionidae probably reflects their life history characteristics; that is, generally short lived, with high rates of natural mortality and recruitment variability (Alcala *et al.* 2005). These results, at both islands, are consistent with the hypothesis that removal of fishing causes an increase in fish biomass in marine reserves (Alcala *et al.* 2005).

Outside Reserve Effects

While one of the primary objectives of establishing marine reserves is to enhance local fisheries, there is often little empirical evidence that actually quantifies the effect of 'spillover' on fishery yields. A lack of appropriate experimental design appears to be one of the major reasons why very little research has successfully demonstrated the spillover effect from marine reserves. 'Spillover' in this context is defined as the 'net export of adult and juvenile organisms from no-take marine reserves into adjacent waters'. A lack of information on migration patterns, spatial distribution and catch rate of targeted species before and after reserve establishment limits the ability to unambiguously show how marine reserves are contributing to change in adjacent fisheries. Also, few investigations of spillover from reserves have been continued long enough for an effect to fully develop (Russ *et al.* 2003). Incorporating information from the monitoring program at Apo Island, Russ *et al.* (2003) explored the possible role that marine reserves have as net exporters of adult biomass into local and regional fisheries.

Over an 18 year period (1983-2001), Russ *et al.* (2003) monitored the biomass of an exploited surgeon fish (*Naso vlamingii*) both inside and outside the Apo Island reserve. Russ *et al.* (2003) found that there was a significant effect of reserve status and protection time on the biomass of *Naso vlamingii*. The biomass of *N. vlamingii* tripled inside the reserve over 18 years of protection. No clear pattern of change in biomass outside the reserve over the same period was evident (Russ *et al.* 2003). Over time, the biomass of *N. vlamingii* increased 40-fold outside but close to the reserve boundaries (200 to 250m), but increases did not occur at greater distances (250 to 500m) (Russ *et al.* 2003). For an outline of the research methods employed by Russ *et al.* (2003) see Table 1.

Evidence suggested that density-dependent home-range relocation was responsible for this apparent "spillover' of *N. vlamingii* (Abesamis and Russ 2005). Firstly, density inside the reserve appeared to have reached an asymptote after 15-20 years of protection. Secondly, fish inside the reserve were larger than outside and size declined outside the reserve with distance from the boundary. Finally, aggressive interactions were more common in the reserve and these favoured larger fish. Although the data has a number of limitations, it is at least consistent with the concept of density dependent "spillover" from a marine reserve (Abesamis and Russ 2005).

Ecosystem management goals, indicators and resultant performance measures for Apo and Sumilon Islands are summarised in Table 2.

Socio-economic Effects

The main socio-economic objectives of the management plans for the Sumilon and Apo Island marine reserves were, firstly, to increase the fish yield at the islands by export of adult and larval fish from the reserve to fished areas and, secondly, to encourage tourism (Russ and Alcala 1999). Another general objective for the Apo Island Reserve was to implement community development programs to establish working groups of local people for accomplishing marine resource management and alternative livelihood projects (White and Vogt 2000).

Fishery yields

No long-term monitoring program for fishery yields for Sumilon and Apo Islands was established; however, a number of estimates of yield were made intermittently over the period of reserve management (Alcala *et al.* 2005). This allowed a long-term temporal analysis of fishery yield for these islands to be conducted (Alcala *et al.* 2005). The methods for estimating fishery yield are presented in Table 3. When Sumilon Island was opened to fishing in 1984 after 10 years, catches declined by approximately 40% by 1985. Apo Island reserve was continuously protected from 1982 – 2001. Total catch of major fish species increased significantly (40%) over this period (Alcala *et al.* 2005). The results suggest the marine reserves helped maintain, or even enhanced, the local fishery yields over the long-term (Alcala *et al.* 2005).

These results, however, are subject to a number of potential limitations, including that there may have been a general improvement in environmental conditions leading to an increase in biomass within the reserves and, concurrently, an increase in catch outside them (Alcala *et al.* 2005). Also, only five families of fish were studied, not all fishing gear types were surveyed, critical information was often based on a single replicate, and slight variations in methods used to collect catch data over the sampling period (1983-2001) may have influenced comparisons of catch over time (Alcala *et al.* 2005). Nevertheless, Alcala *et al.* (2005) demonstrated that closure to fishing of 10%-25% of the fishing area of two small islands in the Philippines did not reduce total fishery catch long term (two decades). Rather, the evidence suggests that the total catch was either maintained or slightly increased in the long-term (Alcala *et al.* 2005).

Interview surveys of fishers at Apo Island in 1986 and 1992 revealed a positive attitude to the marine reserve (Russ and Alcala 1999). Fishers reported that there had been an increase in their catch since the marine reserve had been set up (Russ and Alcala 1999).

Tourism

The success of the objective of encouraging tourism largely reflects the management histories of Sumilon and Apo Islands (Russ and Alcala 1999). The inconsistent management of the marine reserve at Sumilon Island has resulted in a lack of success in encouraging tourism. A resort was established in 1992 with the intention of attracting Japanese tourists (Russ and Alcala 1999). However, from 1992-1997 the resort failed to attract sufficient visitors to be financially viable and, therefore, tourism did not attract large amounts of money into the local community (Russ and Alcala 1999). In contrast to Sumilon Island, the consistent management of the marine reserve at Apo Island has led to thriving tourism. Two small tourist resorts were built (in 1991 and 1996) on the island and both have targeted tourists interested in middle-low cost accommodation and have been marketed as "eco-tourism" accommodation (Russ and Alcala 1999). In addition to the resort, a visitor centre was built and information brochures were distributed, further encouraging tourism (Russ and Alcala 1999), which has been of considerable benefit to the local economy (White *et al.* 2000). Apo Island is now one of the most popular diving and ecotourist destinations in the world, and provides one of the best examples of community-based management of marine resources in the world (Russ and Alcala 1999). Widely-published information on monitoring of tourism changes, however, is limited to documentation of on-site residence tourism numbers (stay at least one night) and off-site residence tourism numbers (day visitors) for Apo Island (White *et al.* 2000) (Table 3). Interview surveys in 1986 and 1992 also canvassed fishermen as to the benefits of tourism to the local community (Russ and Alcala 1999). Although most acknowledged the creation of revenue from tourism, they were less positive with regard to their own livelihood (some interviewees were concerned about the environmental impact of increased tourism and lack of direct income to the local community) (Russ and Alcala 1999).

Socio-economic management goals, indicators and resultant performance measures for Apo and Sumilon Islands are summarised in Table 4.

Table 1. Monitoring for spatial management in the Philippines.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Sumilon Island and Apo Island Marine Reserves *(Russ and Alcala 1996)* *(Russ and Alcala 2003)* *(Russ and Alcala 2004)*	The effects of marine reserve protection were assessed using the 'density of large predators' as an indicator. Rates and patterns of increase in density and biomass after marine reserves were established were assessed, and how quickly density and biomass gains were lost when reserves were subjected to unregulated fishing. Surveys were conducted within the reserves and at two sites outside the boundary of the reserves in November/December 1983, 1985, 1988, 1990-1995 and 1997-2000.	Underwater visual census of the density of large predatory reef fish (Serranidae, Lutjanidae, Lethrinidae, and Carangidae). Surveys were carried out at two reserve sites (6-17m depth) and two non-reserve sites (9-17m depth). Six replicate 50x20m censuses were made on the reef slopes of each site. Fish were identified to species and counted. Target species were counted 3.5m either side of, and 5m above, the observer. Care was taken to search under ledges. Each replicate census was separated by a distance of ~10m. Total length was estimated for large species (+/- 2cm for Serranidae and +/- 5 cm for Lutjanidae, Lethrinidae and Carangidae). Juveniles (<10-15cm) were not counted. Length-weight relationships were used to convert density and size-structure data into biomass (Russ and Alcala 1996; Froese and Pauly 2002).
Sumilon Island and Apo Island Marine Reserves *(Alcala et al. 2005)*	Surveys were conducted within and outside the boundary of marine reserves during November/December 1983, 1985, 1988-1995 and 1997-2001.	Underwater visual census was conducted on five families (Acanthuridae, Carangidae, Lutjanidae, Lethrinidae, and Caesionidae). See above (Russ and Alcala 1996) for methods. Actual counts and estimation of total length (+/-5cm) of large Acanthuridae and all Carangidae, Lutjanidae, and Lethrinidae were made. Length estimates were not made for small species. The abundance of small species of Acanthuridae and all Caesionidae was estimated in log abundance categories. The use of abundance categories is likely to compromise accuracy of estimates for speed of surveys, however, any potential bias in biomass estimates is likely to be consistent across space and time (Alcala *et al.* 2005).
Philippines – Apo Island Marine Reserve *(Russ et al. 2003)*	The possible role that marine reserves have as net exporters of adult biomass into local and regional fisheries was explored. Over a period of 18 years (1983-2001, except for the years 1984, 1986, 1987 and 1996) the biomass of an exploited surgeonfish was monitored inside a marine reserve and at an adjacent site open to fishing. The change in spatial distribution of biomass outside the reserve over this period, and the spatial distribution of hook-and-line catch was measured.	Under water visual census of *Naso vlamingii* was carried out. See above (Russ and Alcala 1996) for methods. Counts and estimates of total length (+/-5cm) of *Naso vlamingii* were made. Juveniles (<10cm) were not counted. An estimate of biomass was made from density and size structure data and a length-weight relationship estimated for large *Naso* species (Froese and Pauly 2002).

Table 2. Management goals, monitoring objectives, indicators and performance measures for spatial management in the Philippines.

Location and Proposed Management Objectives	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Sumilon and Apo Island: Protection of biodiversity and an increase in fish abundance and biomass to export fish via spillover	Determine if the abundance of large predatory fish is a good indicator of the effects of reserve protection Investigate density and biomass change following the establishment of marine reserves and the subsequent effects when status is removed Determine the duration of protection required for populations of large predatory reef fish to attain natural states	Biomass Density	Family (Five families of predatory fish)	Rate and pattern of biomass and density change (temporal)	(Russ and Alcala 1996) (Russ and Alcala 2003) (Russ and Alcala 2004)
	Monitor biomass of fish families targeted for fishing to determine the effect of reserve status on fish populations	Biomass	Family (Five families of fish targeted by fishing)	Rate and pattern of biomass change (temporal/spatial)	(Alcala *et al.* 2005)
	Investigate biomass change outside marine reserves	Biomass	Species (One exploited species)	Rate and pattern of biomass change (temporal/spatial)	(Russ *et al.* 2003)

Table 3. Socio-economic monitoring for spatial management in the Philippines.

Monitoring Location and Citation Details	***Monitoring Outline***	***Methods***
Sumilon Island and Apo Island Marine Reserves *(Alcala et al. 2005)*	To evaluate whether marine reserves affect fishery yields, reserve status and yield estimates were monitored. Annual trap and gillnet catch was estimated at Sumilon Island (1979-1980, 1983, and 1985-1986) after the establishment of the marine reserves (Alcala 1981; Alcala and Russ 1990). Annual catch by trap, gill net, hook and line, and spear was estimated at Apo Island before (1980-1981) and after (1985-1986, 1997-1998, 2000 and 2001) the establishment of the marine reserves (Bellwood 1988; Maypa *et al.* 2002).	Five families of fish were surveyed (Acanthuridae, Carangidae, Lutjanidae, Lethrinidae, and Caesionidae). The whole reef was surveyed at the Sumilon Island reserve, and catch data was recorded to family level. The fish market and landing sites were surveyed at the Apo Island reserve and catch data was recorded to family or species level.
Apo Island Marine Reserve *(White et al. 2000)*	Tourism numbers at Apo Island were documented	Documentation of on-site residence tourism numbers (stay at least one night) and off-site residence tourism numbers (day visitors)
Apo Island Marine Reserve *(Russ and Alcala 1999)*	Attitudes of local fishermen to the creation of the marine reserve and sanctuary were monitored	Interview surveys were conducted in 1986 and 1992

Table 4. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in the Philippines.

Location and Proposed Management Objectives	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Sumilon Island and Apo Island	To monitor annual catch to determine the effect of reserve status on fishery yields	Annual yield	Family (Five families of fish targeted by fishing)	Temporal trend in annual yield in relation to reserve protection	(Alcala *et al.* 2005)
Increase the fish yield at the islands by export of adult and larval fish from the reserve to fished areas	To determine if fishermen perceive that marine reserve protection benefits fish yield	Interview response	Local fishermen	Level of positive perception	(Russ and Alcala 1999)
Apo Island	To monitor the number of tourists visiting the Island	no. of tourists	Non-residents	Temporal trend in tourism visits	(White *et al.* 2000)
Encourage tourism	To determine if fishermen perceive that marine reserve protection has increased tourism revenue (generally and personal benefit)	Interview response	Local fishermen	Level of positive perception	(Russ and Alcala 1999)

Caribbean – St Lucia

In 1995, a network of marine reserves was established within the Soufriere Marine Management Area (SMMA) at St Lucia, in the Caribbean. They were established primarily to protect and re-build severely over exploited fish stocks and recover fishery productivity, and to contribute to the development of tourism. Management of the area focuses on sustainable use, cooperation among resource users, and collaborative institutional research. The reserves have been biologically surveyed and monitored periodically since 1994.

Ecosystem Effects

Within Reserve Effects

Following the creation of the St Lucia marine reserves, Hawkins *et al*. (2006) examined how coral cover, habitat structural complexity, and sedimentation influence the rate and extent of recovery in fish communities. Commercially important species were the focus of the research (groupers, snappers, grunts, parrotfish, and surgeonfish) and surveys were conducted annually from 1994 -1995 to 2002 (except for 1999). For an outline of the research methods employed by Hawkins *et al.* (2006) see Table 5. Ecosystem management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 6.

A pre-protection census that was conducted in 1994 -1995 revealed that there was virtually no difference between marine reserves and fishing grounds in the total biomass of fish (Hawkins *et al.* 2006). By the final census that was conducted in 2002, however, Hawkins *et al*. (2006) found that total biomass had more than quadrupled in reserves and tripled in fishing grounds with all families increasing significantly in biomass over time at nearly all of the survey sites. While all families, except for grunts, showed a greater increase in biomass within the reserves as opposed to the fishing grounds, the strongest responses were in parrot fish and surgeonfish (Hawkins *et al.* 2006). The trend in species richness showed an initial increase, but leveled off after three years of protection (Hawkins *et al.* 2006).

Although the effect on fish biomass and species richness from protection alone was not significant for any family, the biomass of all families changed significantly over time and almost always increased. Hawkins *et al.* (2006) found that biomass increases were particularly strong amongst herbivorous fish. The increase in biomass for surgeonfish occurred slowly until the final year of the study where the increase was rapid. Protection of predators resulted in a large biomass increase in reserves for groupers and a smaller increase for snappers (Hawkins *et al.* 2006). Significant interactions between location and one or both of protection and time for all families except surgeonfish, indicated the change in biomass differed among the four reserve-fishing ground survey locations. Because biomass change may be strongly affected by time of protection, the divergence of biomass between reserves and fishing grounds may develop slowly and it could take some time before the interaction between protection and time becomes significant (Hawkins *et al.* 2006).

Habitat characteristics and deterioration did not appear to affect the rates of biomass build-up (Hawkins *et al.* 2006). While the reef habitat was suffering from the effects of storms, sedimentation (Sladek Nowlis *et al.* 1997) and coral diseases (Nugues 2002), fish stocks continued to increase. Of the six habitat factors that were tested, Hawkins *et al.* (2006) found that only protection and sedimentation had any significant influence on reserve performance. Protection from fishing was the most important factor responsible for improving fish stocks, explaining 44% of the variance in biomass growth. A further 28% of the variance was explained by sedimentation, a process known to stress reef invertebrates, significantly reducing the rate of increase in biomass (Hawkins *et al.* 2006). Measures of coral cover and reef structural complexity, and their rate of change over time, had no significant effect on the rate of increase in fish biomass.

A limitation of this study was that only one pre-protection census was conducted, where three are generally recommended (Russ 2002). However, the experimental design did incorporate data from both before and after protection was established. A series of reserves were surveyed, and the temporal and spatial changes between reserves and fishing grounds were assessed (Hawkins *et al.* 2006). In summary, Hawkins *et al.* (2006) showed that the biomass of commercially exploited fish stocks increased rapidly in the SMMA, both in reserves and adjacent fishing grounds, after marine reserve protection was established. The rate of change, however, was variable between five families of fish across four different survey locations.

Socio-economic Effects

Fishery Yield

Traditionally, Soufriere households are reliant on fishing as their primary or secondary source of income (Sandersen and Koester 2000; Pierre-Nathoniel 2004). This reliance continues today and fishing techniques include beach seines, gill nets, fish pots and trolling. This exploitation combined with the resource competition from the introduction (in the 1980s) of other tourist-related activities, including yacht anchorage, scuba diving, snorkelling and eco-tourism, have added to the pressure on the Soufriere area (Sandersen and Koester 2000; Pierre-Nathoniel 2004).

A zoning strategy was developed in which 23 separate zones were delineated as 5 marine reserves, 10 fishing priority areas, 4 multiple use/recreational areas and 4 mooring areas (Goodridge *et al.* 1997; Sandersen and Koester 2000; Pierre-Nathoniel 2004). It was thought that by restricting fishing in the marine reserves, fish would recover and gradually there would be spill over of commercially-important reef fish to adjacent fishing grounds (Sandersen and Koester 2000).

Goodridge *et al.* (1997) observed, interviewed and made records of the local fishers' behaviour, fishing effort, fishing methods and catch rates following the zoning (refer to Table 7 for methods). Results showed that nearly a quarter of the total reef-fishing effort, during the year following zoning, was carried out in the areas closed to fishing. Additionally, there were no significant changes in catch rates one year prior to zoning (4.32 kg/gear type/fishing trip) and one year following (4.26 kg/gear type/fishing trip) (Goodridge 1997). Interviews concluded that fishers had not yet accepted the management strategies and believed that catch rates were greater in the marine reserves (Goodridge *et al.* 1997).

Within the SMMA, Roberts *et al.* (2001) investigated whether the marine reserves have enhanced adjacent fisheries. Data was collected on fishing with fish traps for a period immediately after reserve establishment and again after 5 years of reserve protection, and the change in yield was examined. For an outline of the research methods employed by Roberts *et al.* (2001) see Table 7.

Roberts *et al.* (2001) found that within 5 years of creation (between 1995-1996 and 2000-2001), the catches of local fisheries adjacent to the network of marine reserves had increased significantly. Mean total catch per trip for fishers with large traps increased by 46%, and for fishers with small traps by 90%; catch per trap increased by 36% for big traps and 80% for small traps (Roberts *et al.* 2001). Fishing effort was relatively similar for the two periods (Roberts *et al.* 2001).

The findings of this research indicated that the establishment of the SMMA marine reserves has improved the adjacent fishery, despite the 35% decrease in area of fishing grounds (Roberts *et al.* 2001). The difficulty with the interpretation of the results of this study was that there were only two time periods compared. The results cannot therefore be interpreted in terms of the natural inter-annual variability in catch making the significance of the observed change difficult to assess.

In the same study by Roberts *et al.* (2001), interviews with, and surveys on, Soufriere fishers were also conducted to assess whether (in their opinion) the fishery had improved since the SSMA was established (Roberts *et al.* 2001). Results showed that most fishers felt better off with reserves than without, especially the younger fishers. For example, approximately 46 % of all fishers under the age of 45 felt that the fishing was better, ~ 13.5 % thought it was the same, ~ 22% thought it was worse, while the remaining (18%) didn't know or didn't wish to comment (see Table 7 for method of interviewing).

Socio-economic management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 8.

Table 5. Monitoring for spatial management in the Caribbean – St Lucia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
St Lucia, Soufriere Marine Management Area (SMMA) *(Hawkins et al. 2006)*	This research investigates factors affecting the rate and extent of biomass build-up among commercially important groupers, snappers, grunts, parrotfish, and surgeonfish. The SMMA was sampled in December 1994 and January 1995 (before reserve establishment) and again in August and September 1996 (after one year of protection). From then until 2002, the SMMA has been monitored annually except for 1999. Non-reserve "control areas" were also monitored.	Each survey consisted of 114 fish counts in reserves and 83 in surrounding fishing grounds, at two depths (5 m and 15 m). A 10 m tape measure was laid on the reef and used to denote the diameter of a cylinder extending above the reef. For 15 min, the number and estimated size (cm) of all non-cryptic species occurring within or passing through the cylinder was recorded. Whilst laying out the tape, any large, wary species within the counting area were recorded. The presence of cryptic species was also recorded to give an indication of total species numbers. Within each area, estimated percent coral and algal cover and was recorded. Semi-quantitative estimates of reef structural complexity on a scale of 0-5 were made. To measure sedimentation rate (between 1997 and 2001 at 11 locations throughout the SMMA), sediment traps were collected every two weeks over seven time periods of 2-6 months during both wet and dry seasons. At each location there were two traps fixed at 25cm above the reef.

Table 6. Management goals, monitoring objectives, indicators and performance measures for spatial management in the Caribbean – St Lucia.

Location and Proposed Management Objectives	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
St Lucia, Soufriere Marine Management Area (SMMA) *Protection and to re-build severely over exploited fish stocks and recovery of fishery productivity*	Examine how coral cover, habitat structural complexity, and sedimentation influence the rate and extent of recovery in fish communities.	Benthic Percent Cover	Habitat Type	Rate and pattern of biomass and species diversity change in relation to reference areas	(Hawkins *et al.* 2006)
		Habitat Structural Complexity	Categorical		
		Sedimentation Rate	mg/cm^{-2} day^{-1}	Habitat cover, and sedimentation rate trends.	
		Biomass	Family (Five families of exploited species)		
		Species Richness	Total (All species)		

Table 7. Socio-economic monitoring for spatial management in the Caribbean – St Lucia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
St Lucia, Soufriere Marine Management Area (SMMA) *(Goodridge et al. 1997)*	Observe, survey and do questionnaires on fishing effort, fishing methods, fisher behaviour and site preference to assess the effects of the reduction in legally fishable area. Observations were made from 1/11/95 to 31/7/96. Changes in catch rates were also measured and recorded from trips between 1/8/1994 & 31/7/1995 and 1/8/1995 & 31/7/1996.	Between 1/11/95 & 31/7/96 the main Soufriere landing site was monitored 5-6 days a week between 8:00 am & 5:00 pm. All reef fishers, their boat types, type & no. of fishing gear, effort, and numbers of fish caught (by species) were recorded. Between 1/11/95 & 10/2/96, fishing activity in the SMMA was surveyed 5-6 days a week by boat, traversing one end of the zoned area to the other between 7:00 am & 9:00 am and 2-3 times a week between 5:0 pm & 7:00 pm. On each survey, the fisher, the location of fishing activity relative to zone boundaries within the SMMA, fisher identification and the kind of gear being used was recorded. A questionnaire was also supplied to a sample of the Soufriere reef fishers to compare fishing behaviours, gears, methods and the perceptions of the reef fishery before the SMMA was zoned and immediately after the zoning was put into effect. The catches from a total of 1,455 pots hauled in 392 pot fishing trips in 95/96 and catches from a total of 495 pots hauled in 113 pot fishing trips in 94/95 were sampled. Mean catch rates for all pots combined & individual pot types were calculated as kg/gear type/fishing trip.
(Roberts et al. 2001)	The effect on adjacent fisheries of marine reserves was investigated. The reef fishery in the SMMA was studied for two, 5 month periods in 1995-1996, immediately after reserves were created, and in 2000-2001, after 5 years of protection.	Data from two trap-fishing methods was collected. Large traps soaked overnight and small drop and-lift traps, baited and soaked for 1-2 hours. Total fishing effort remained stable over the course of the study.
(Roberts et al. 2001)	Interview local fishers to determine whether they felt that the fishery had improved since the SMMA was established.	71 local fishers were interviewed in Creole (via an interpreter). 23 of those interviewed were aged 15-30, 22 were 31-45, 15 were 46-60 and the remaining (11) were 61-85. Each was asked if the fishery was "better", "the same", "worse" or "don't know or won't say". The percentage of each response was calculated.

Table 8. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in the Caribbean – St Lucia.

Location and Proposed Management Objectives	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
St Lucia, Soufriere Marine Management Area (SMMA)	Observe, survey & do questionnaires on fishers in order to assess the effects of the reduction in legally fishable area	Fishing effort		Level of compliance with restrictions	
		Fishing method			
Prevent fishing in marine reserves and enhance artisanal, subsistence fisheries		Fisher behaviour	Local community		(Goodridge *et al.* 1997)
		Site preference			
		Catch rates (kg/gear type/fishing trip)	All species	Temporal change in catch rates	
	Investigate whether marine reserves enhance adjacent fisheries	CPUE (catch per trap & catch per trip)	Total (All Species)	Rate and pattern of CPUE change outside Park boundary	(Roberts *et al.* 2001)
	Interview fishers to get opinions on whether the fishery has improved (in terms of catch estimates) since the SMMA was established	Interview response	Local community	Has fishing in the reserve stopped & has artisanal fishing been enhanced	(Roberts *et al.* 2001)

Indonesia

The Komodo National Park (KNP), Indonesia, was established in 1980 to protect biodiversity and the breeding stocks of commercial fishes for replenishment of surrounding fishing grounds. The KNP Monitoring Program was established in 1996 to facilitate the management of resources and the control of blast fishing. It has been a vital component in the successful management of the park (Mous *et al.* 2003).

Ecosystem Effects

Within Reserve Effects

The KMP Monitoring Program gathers spatial and temporal information on reef health and recovery both inside and outside the park. It was designed to assess management effectiveness over a wide area rather than for fine-scale biological monitoring. For an outline of the research methods employed by the KNP Monitoring Program (Pet and Yeager 2000) see Table 9.

Results from the program have revealed that management has been successful in decreasing the incidence of blast fishing in the park and show good recovery of coral reef. Average cover of live hard-coral has gradually increased from 15% in 1996 to 24% in 2004 inside the park (Mous *et al.* 2003). This result was statistically significant because outside the park, hard coral cover dropped from 25% to 17% between 2000 and 2002 after initial increases between 1996 and 2000 (Mous *et al.* 2003). It is possible that a crown of thorns starfish (COTS) outbreak in the northeast of the park and continued blast fishing around an island in the northwest caused this decline, but more analysis is needed to confirm this hypothesis. The monitoring has also allowed the detection of biological events that are not strictly the focus of the program, such as the COTS outbreaks and coral bleaching (Mous *et al.* 2003).

Based on monitoring program information that focuses on spawning aggregations of high valued reef fish in KNP, Pet *et al.* (2005) described the temporal patterns in aggregating behaviour, and trends in average body size and numbers of two commercial species of grouper, *Epinephelus fuscoguttatus* and *Plectropomus areolatus*. To corroborate whether aggregation was likely to be associated with reproduction, observations were made on species-specific behaviours thought to occur only during the reproductive season. For an outline of the research methods employed by Pet *et al.* (2005) see Table 9.

Pet *et al.* (2005) found that although 300 sites within the KNP (17 km^2 of reef slopes) were surveyed repeatedly between 1995 and 2000, distinctive aggregating behaviour was identified for the two fish species at only two sites. *E. fuscoguttatus* and *P. areolatus* aggregated at one site, while another site only contained an aggregation of *P. areolatus*. The importance of these two sites in the life histories of *E. fuscoguttatus* and *P. areolatus* in KNP is highlighted by the apparent scarcity of spawning aggregation sites.

Over the five-year monitoring period there was a reduction in mean fish size for *P. areolatus*, and a reduction in numbers of aggregating *E. fuscoguttatus*. Various factors could account for observed patterns in numbers and size of fish at spawning aggregation sites, including size-selective fishing pressure, variation in recruitment strength over time, variation in growth, or variation in sex-specific differences in the timing of migration to or from the spawning aggregation sites (Pet *et al.* 2005). Although conceding that natural variation in fish population dynamics may have caused the observed long-term patterns in numbers and size, Pet *et al.* (2005) concluded that fishing pressure was also a possible explanation for the observed trends. This latter hypothesis was supported by similar trends observed in body size and abundance of another commercially important grouper in the same area over the same five-year period (Pet *et al.* 2005). Furthermore, monitoring of resource utilisation by the KNP authorities has shown that the two spawning aggregation sites identified in this study were more heavily fished than most other areas within the KNP (Pet *et al.* 1999). Despite limited protection, both sites were still being heavily fished by local artisanal fishers, suggesting that managers should increase the level of protection and monitoring in these locations (Pet *et al.* 2005).

Ecosystem management goals, monitoring objectives, indicators and resultant performance measures for Komodo National Park are summarised in Table 10.

Socio-economic Effects

An estimated 20,000 people live in fishing villages inside and directly surrounding KNP and many take reef fish for the lucrative trade in live reef-fish (Pet *et al.* 2005). Management objectives at KNP are not only to protect biodiversity and breeding stocks of commercial fish, but also to replenish the surrounding fishing grounds (Pet *et al.* 2005). Protective management has been enforced to ban destructive fishing methods such as blast fishing, fishing with cyanide and diving on hookah.

Resource use monitoring

Resource use monitoring has been conducted in the KNP since 1996 and consists of regular field surveys to document the types of resource use, and when, where and by whom they are practiced (Mous *et al.* 2004). 'Resource use' is defined as use of marine, renewable resources, including extractive uses (eg. fishing, coral mining) and non-extractive uses (eg. tourism, education) (Mous *et al.* 2004). A team surveys the Park by boat, interviewing fishers and tourists when encountered, and their activities are recorded (see Table 11 for methods). Initially, resource use monitoring focussed on artisanal fishing only, but from 2002 other types of use such as tourism have been recorded (Mous *et al.* 2004).

Results of the surveys allowed the origins of fishers both regionally and locally to be recorded, and changes in the pattern of Park use was tracked (Mous *et al.* 2004). The recent results suggest that the level of marine tourism is low compared to artisanal fishing; however, it is possible that the survey method leads to an underestimate of the Park use by recreational diving and fishing vessels (Mous *et al.* 2004).

Incidences of blast fishing decreased by 90% almost immediately in 1996 following the implementation of a comprehensive conservation program (Mous *et al.* 2004). Throughout the survey period, hook and line fishing and gillnetting were the dominant forms of fishing gear used, comprising approximately 80% of the reef fishing effort observed from 2002 – 2004. No clear trend in fishing effort has been observed over the survey period; however, an increase in the most recent surveys has been noted (Mous *et al.* 2004). For Park residents and fishers from villages around the Park, the spatial patterns of fishing effort are concentrated in areas adjacent to no-take zones (Mous *et al.* 2004).

Socio-economic management goals, monitoring objectives, indicators and resultant performance measures for the Komodo National Park are summarised in Table 12.

Table 9. Monitoring for spatial management in Indonesia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Komodo National Park Monitoring Program *(Pet and Yeager 2000)*	Komodo National Park (KNP) was established in 1980 to protect biodiversity and the breeding stocks of commercial fishes for replenishment of surrounding fishing grounds. The KNP monitoring program was created to gather spatial and temporal information on reef health and recovery.	Since 1996, percent cover of habitat has been estimated from 12 sites per week (185 sites in total). Each complete survey takes 8 to 9 months and is conducted once every two years. Underwater visual survey (UVC) is conducted by snorkeling (4 m depth) and by SCUBA diving (8 and 12 m depth). Five, four minute surveys are conducted at each depth. A hard coral mortality coefficient is calculated for each site from observations of reef status. Grouper and wrasse (12 key commercially important species) spawning aggregation sites are monitored. These species are heavily targeted by fisheries and can therefore serve as indicators for the impact of these fisheries. Numbers and sizes are recorded at 12 selected sampling sites. Since 1998, six sites have been monitored twice a month, once during new moon and once during full moon. Each site is searched for target fish at a specific depth profile established for that site. 200m transects are surveyed. Regression analysis shows that visibility has no significant effect on numbers of fish recorded at each site (Pet *et al.* 1999).
(Pet et al. 2005)	Identified fishery management implications from monitoring program information that focuses on spawning aggregations of high-value reef fish in the KNP. Assessed temporal patterns in aggregating behaviour, and trends in average body size and numbers, of two commercial species of grouper, over a five year period (1995-2000).	UVC of the groupers *Epinephelus fuscoguttatus* and *Plectropomus areolatus* was conducted by the KNP monitoring program at 12 selected sites (see above for methods used for UVC of groupers). Aggregating behavior of *E. fuscoguttatus* and *P. areolatus* was confirmed at only one of the 12 initially selected sites (site A). A second site (site B), also contained an aggregation of *P. areolatus*. Exploratory surveys revealed no evidence for aggregation during moon phases other than full moon and new moon. Therefore, sites A and B were monitored twice a month, Over a three day period that was centered on the full moon and new moon. Behaviours and other signs that are indicative of the reproductive season were scored as present or absent (eg. presence of gravid females and alteration of colour, frequent aggressive interactions, extensive external wounding associated with the aggression, and swimming on the side in a distinct wavering motion (courtship)).

Table 10. Management goals, monitoring objectives, indicators and performance measures for spatial management in Indonesia.

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Komodo National Park: Protection of biodiversity and the breeding stocks of commercial fishes for replenishment of surrounding fishing grounds	Gather spatial and temporal information on reef health and recovery.	Benthic Percent Cover	Habitat Type and Coral Mortality (Categorical)	Rate and pattern of density and biomass change	(Pet and Yeager 2000)
		Biomass	Species (Twelve exploited species)	Habitat cover and coral mortality trends	
		Density			
	Assess the temporal patterns in spawning aggregation behaviour following the establishment of marine reserves.	Biomass	Species Stage (Two exploited species)	Rate and pattern of density and biomass change of spawning aggregations	(Pet *et al.* 2005)
		Density			
		Behaviour	Spawning		
		Density	Species (Range of fish species)		
		Size Distribution	Species (Range of benthic species and kelp)		
		Water Temperature	Hourly		
		Density	Total (All species, categorical)		

Table 11. Socio-economic monitoring for spatial management in Indonesia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Komodo National Park (Mous *et al.* 2004)	Survey by speedboat is undertaken weekly to assess trends/changes in resource use over time. Records have been made during 1996-2001 (Artisanal fishing) and 2002-2004 (all types of resource use)	Survey team use speedboat to make a trip through the Park interviewing vessels as encountered. The target is for surveys to be undertaken weekly, with each survey conducted over two days. Logistical constraints have meant that actual performance has been at approximately 20 – 80% of the target. Since 2002 records have also kept of the survey route, thereby facilitating spatial analysis. Percentage distribution of the types of fishing gear used and of fishing methods per village, and yearly averages of reef fishing effort per sortie, were calculated from results obtained between 1996-2004,. Relative levels of non-artisanal fishing uses were also recorded from 2002. Spatial coordinates of fishing vessels also allowed “home range” analysis to be undertaken of resource use by members of individual villages.

Table 12. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in Indonesia.

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Komodo National Park *Protect biodiversity and breeding stocks of commercial fish, but also to conserve spawning stocks of high-valued commercial species for the replenishment of surrounding fishing grounds*	• Provide input to Park management on the status of the resource use in the area • Contribute to the field presence of Park staff, thereby reducing illegal resource use • Obtain a measure of success for the overall performance of Park Management	% distribution of the types of fishing gear used % distribution of fishing methods per village yearly average of reef fishing effort % contribution of different resource uses	Local & wider community	Temporal trend of resource use characteristics	(Mous *et al.* 2004)

California

Established in 1980 for having unique natural and cultural resources, the Channel Islands National Park, California includes the north and south sides of Santa Barbara, Anacapa, Santa Cruz, Santa Rosa, and San Miguel Islands. The park was created to 1) protect the nationally significant natural, scenic, wildlife, marine, ecological, historical, archeological, cultural and scientific values of the Channel Islands, 2) understand population dynamics and trends in terrestrial and marine ecosystems, and, 3) provide for visitor use on a low-intensity, limited entry basis to assure that adverse impacts on the park resources were negligible.

Ecosystem Effects

Within Reserve Effects

The Channel Islands Kelp Forest Monitoring (KFM) program, established in 1982, was developed to assess the population dynamics of the kelp forest species within the park. The program has helped to control and eliminate invasive alien species, detect and mitigate pollution, recognise and demonstrate unsustainable uses, change fishery management strategies, and develop and evaluate population and ecosystem restoration methodologies (Davis 2005). For an outline of the monitoring methods employed by the KFM program (Davis *et al.* 1997) see Table 13. Management goals, monitoring objectives, indicators and resultant performance measures for the Channel Islands National Park are summarised in Table 14.

Frequent and extensive analysis and synthesis of the monitoring data has facilitated the discovery of new features and characteristics of the Channel Islands marine reserve ecosystems (Davis 2005). Outbreaks of fatal diseases, such as withering syndrome in black abalone (*Haliotis cracherodii*), were previously unknown, in part because no rigorous ecological monitoring took place before the KFM program (Davis 2005). Monitoring revealed that black abalone populations collapsed, and also provided a regional geographic and temporal description of the spread of mortality (Richards and Davis 1993).

Monitoring results enabled the size structure of surviving abalone population to be characterised, showing persistence of large individuals at some sites but not at others. This information exonerated fishing (which took only large abalone) as a proximal cause of the population collapses at some islands, but it implicated fishing as a contributing stress at others (Davis 2005). Data from the monitoring program also showed that adult black abalone ceased to reproduce successfully when population densities fell below 50% of their original values. These quantitative descriptions then directed subsequent research to examine potential infectious agents, rather than toxic pollutants or poaching and other human activities, and led to the discovery of a new species of pathogen (Friedman *et al.* 1995). The monitoring provided an early warning with sufficient information to protect disease-resistant individuals from fishery harvest and thereby helped ensure survival of another generation (Davis 2005).

The monitoring program has also identified three species of invasive, introduced algae in or near the park (*Sargassum muticum, Undaria pinnatifida*, and *Caulerpa taxifolia*). While it appears that these species have yet to impact on park ecosystems, they have the potential to do so quickly and significantly (Davis 2005).

Monitoring fishery-independent data has provided essential information for fishery managers (Botsford *et al.* 1997). Ambiguous data on fishery-landings obscured the catastrophic serial-depletion of five species of abalone (*Haliotis spp.*), and a sea urchin species (*Strongylocentrotus franciscanus*), that supported a commercial diving fleet in southern California before monitoring data were available (Dugan and Davis 1993; State of California 1995; Davis 1998; 2005). As a result, fishing continued on depleted abalone populations before fishery management policies could be changed, and consequently drove at least one species, the white abalone *Haliotis sorenseni,* to the verge of extinction. *H. sorenseni* is the first marine invertebrate to be listed as endangered in the USA (Davis *et al.* 1996; 1998; Davis 2000; Hobday *et al.* 2001; Davis 2005). The monitoring program provided early warnings of population collapses and ecosystem shifts that prompted changes in resource management policy and strategy (Davis 2005).

Table 13. Monitoring for spatial management in California.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Channel Islands National Park (CINP) long-term Kelp Forest Monitoring Program (KFM) *(Davis et al. 1997)*	The KFM program annually (summer) samples 16 rocky reef sites from the north and south sides of Santa Barbara, Anacapa, Santa Cruz, Santa Rosa and San Miguel Islands. Two of these sites occur in the Anacapa Island Ecological Reserve, established in 1978. Long-term trends are assessed. Sites consist of a 100m fixed transect. Goals for accuracy and precision of monitoring were set *a priori* to detect 40% changes in mean values (Davis 2005).	The density of sedentary species (eg. gastropods, urchins, small fish, algae) is recorded 1 x 2 m quadrats, 12 per site. The density of selected rare, clumped, sedentary species (eg. giant kelp, giant seastar) is recorded 1 x 5 m quadrats, 40 per site. The density of rare and clumped organisms not adequately sampled by quadrats (eg. abalone, gorgonians) is recorded 3 x 20 m band transects, 12 per site. The percent cover of selected algal and invertebrate taxa (eg. coralline algae, coral, bryozoan) is recorded random point contact, 40 points (0.5 x 3 m), 15 per site. The density of selected fish species (eg. rockfish, surfperch, wrasse) is recorded visual transect, 2 m (w) x 3 m (h) x 50 m (long), 8 per site (two separate sampling events separated by at least 8 weeks) The estimated density of selected fish species (rockfish, surfperch, goby, wrasse) is recorded roving diver fish count, 20 m x 100 m, 30 min, 4-8 per site. The estimated size of selected kelp forest community species (eg. giant kelp, abalone, sea urchins) is recorded video transects, 2 per site. Hourly water temperature is recorded at each site. Size frequency distributions (eg. abalone, snails, sponges, gorgonians) are recorded, sample size number is from 60-200 (species dependent). Artificial Recruitment Modules (ARMs) are used to conduct standardised size frequency sampling of selected indicator species. The size frequency distributions are used to identify and monitor recruitment events. Relative density of all species is recorded.

Table 14. Management goals, monitoring objectives, indicators and performance measures for spatial management in California.

Location and Proposed Management Objective	***Monitoring Objective***	***Indicator***		***Performance Measures***	***Reference***
		Variable	***Scale***		
Channel Islands National Park: Protection of the nationally significant features of the area; an understanding of populations dynamics and trends; and provide for the development of sustainable tourism	Assess the population dynamics of kelp forest ecosystems.	Density	Species (range of sedentary species)	Temporal trend in density, habitat cover, size distribution, and species diversity. Change greater than 40% considered significant	(Davis *et al.* 1997)
		Benthic percent cover	Habitat type		
		Species diversity and density	Species (range of fish species)		
		Size distribution	Species (range of benthic species and kelp)		
		Water temperature	Hourly		
		Density	Total (all species, categorical)		

New Zealand

Ecosystem Effects

Within Reserve Effects

The Cape Rodney to Okakari Point Marine Reserve (hereafter the Leigh Marine Reserve, 549.16 ha) is the oldest marine reserve in New Zealand, established in 1975. A monitoring program was initiated soon after establishment that was sampled between 1976 and 1978 (Ayling 1978). The same sites were then further sampled in 1994, 1996 (Babcock *et al.* 1999) and 1999, 2000 (Shears and Babcock 2003). For an outline of the monitoring methods employed by these studies see Table 15.

This monitoring showed a transition from urchin barren to kelp dominated habitat within the reserve over the 25 year period (Shears and Babcock 2003). "Barrens" of unvegetated habitat are common on moderately exposed reefs in the area, typically at depths between 3 and 8 metres, maintained by grazing activities of the urchin *Evichinus chloroticus* and some other gastropod species (Shears and Babcock 2002; 2003). After 25 years, urchin numbers have declined in the reserve and the barren habitat has reduced from approximately 30% of reef area to only a few percent (Babcock *et al.* 1999; Shears and Babcock 2003). Primary productivity of rocky reef habitats was estimated to have increased by 58% between 1978 and 1996 (Babcock *et al.* 1999).

Additional studies have shown significantly lower urchin abundances and barren habitat in both the Leigh Reserve and the nearby Tawharanui Reserve (350 ha, established 1982), compared with adjacent non-reserve reference sites (Babcock *et al.* 1999; Shears and Babcock 2003). Abundances of snapper, *Pagrus auratus*, were much higher and individuals were also much larger in reserve areas compared with protected areas (Babcock *et al.* 1999). Likewise, spiny lobsters, *Jasus edwardsii*, were also more abundant and larger in marine reserves compared to non-reserve sites (Babcock *et al.* 1999). The proportion of urchin barren (< 10 m depth) was significantly higher outside reserves (~40%) compared to inside reserves (~13%) (Babcock *et al.* 1999). The results for habitat structure and algal abundance between reserve and non-reserve sites were confirmed in a further three year study (1999 to 2001) (Shears and Babcock 2003). At reserve sites, non-cryptic (openly grazing on the substratum) urchins declined to the point where urchin barrens were completely absent by 2001 (Shears and Babcock 2003). For an outline of the monitoring methods employed by these studies see Table 15.

In summary, abundances of urchin predators are much higher in reserves compared with non-reserve sites, where they are heavily targeted by both commercial and recreational fishermen. Predation rates on tethered urchins in the marine reserves are approximately 7-times higher than at non-reserve sites (Shears and Babcock 2002). Overall, this evidence has led to the conclusion that long-term declines in urchin barrens in the Leigh Reserve are the result of top-down control of urchin numbers by predators, compared to the situation outside reserves where urchins are released from predation due to the effects of fishing (Shears and Babcock 2002). This situation, however, appears to be relatively localised, with many other marine reserves in New Zealand not showing this effect (Shears and Babcock 2004).

The Long Island – Kokomohua Marine Reserve, Marlborough Sounds was opened in 1993, and was the first marine reserve established in the South Island of New Zealand (Davidson 2001). A long-term monitoring study has been carried out focussed on the blue cod, *Parapercis colias*, an important and widespread recreational fishing species in southern New Zealand waters (Davidson 2001). For an outline of the monitoring methods employed by Davidson (2001) see Table 16.

Monitoring was conducted using experimental fishing and underwater visual transects. Over the period of monitoring the mean length of blue cod captured using experimental fishing increased in the marine reserve but declined at the control sites (Davidson 2001). By the end of the monitoring period the proportion of large blue cod > 330 mm in length in the reserve was 35% compared to < 1% at the control sites (Davidson 2001). Catch rates of blue cod in experimental fishing also increased in the marine reserve relative to the control areas. Evidence from behavioural observations and comparison with underwater visual survey results suggested that the increased catch rate was mainly due to an increase in the proportion of naïve fish in the reserve population (Davidson 2001). Underwater visual surveys revealed that towards the end of the study, the abundance of blue cod in the reserve was significantly higher than at control sites, primarily due to an increase in the abundance of larger blue cod (Davidson 2001).

Monitoring of spiny lobster, *Jasus edwardsii*, has been carried in the Tawharanui Marine Park and the Mimiwhangata Marine Park since 1977 (Shears *et al.* 2006). The Mimiwhangata Marine Park (approx. 2000 ha, established in 1984) was closed to commercial lobster potting in 1993 but certain types of recreational fishing are still allowed (Shears *et al.* 2006). This is in contrast to the Tawharanui Marine Park, which has been completely no-take since establishment (Shears *et al.* 2006). Lobster densities were similar in the two marine parks prior to establishment but legal size lobsters were 11-times more abundant and biomass was 25-times higher in the no-take marine park after establishment, compared with no change in lobster numbers in the partially protected park (Shears *et al.* 2006). The results are less than ideal because different years were often monitored in the two reserves and in the case of the Mimiwhangata reserve, there was a large gap between pre-establishment sampling and recent post-establishment sampling. Additionally, the underwater estimation of weight was unusual and the potential inaccuracy of this method was not considered. Overall, the results suggest that in order to achieve recovery of the spiny lobster populations within Marine Parks, a fully no-take level of protection is required (Shears *et al.* 2006).

Management goals, monitoring objectives, indicators and resultant performance measures for New Zealand Marine Reserves are summarised in Table 17.

Table 15. Monitoring for Spatial Management in New Zealand.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Leigh and Tawharanui Marine Reserves *(Ayling 1978; Babcock et al. 1999; Shears and Babcock 2003)*	Examine the temporal change in benthic community structure in the Leigh Marine Reserve from three studies between 1976 and 2000.	Thirteen, 100 m^2 permanent sites established by Ayling (1978) were re-sampled for brown algae and invertebrate herbivores during September 1999 and August 2000 (Shears and Babcock 2003). Sites were grouped among 3 depth strata: shallow (<5 m; n=4), mid-depth (6-8 m; n=5), and deep (10-13 m; n=4). Each site was sampled using 10 haphazardly placed $1m^2$ quadrats. The density and size structure of dominant species were compared with the original 1976-78 data (Ayling 1978), and data from 1994 and 1996 (Babcock *et al.* 1999).
	Differences in benthic communities between reserve and adjacent unprotected sites and the stability of these patterns were assessed.	Fish abundance and size were measured at 18 sites in 1997 using remotely deployed baited video stations (Babcock *et al.* 1999). The video camera was orientated vertically over the bait container on a stand. Fish were filmed in 30 minute recordings and the maximum number observed was used as index of abundance. Fish were measured using marks on the video frame.
		Two sites within and two sites outside each reserve were sampled. Lobsters were surveyed in 1995, within 5 haphazardly placed 50 x 10 m transects at each site (Babcock *et al.* 1999). Abundance and estimated size were recorded.
		Algal abundance was estimated at 21 sites on 1 m wide transects, perpendicular to the shore, and extending from 0 to 10 m below chart datum. Habitat type was classified into pre-defined categories by visual estimate at 1 m intervals (Babcock *et al.* 1999). Productivity was calculated from the area of algae.
		Eight sites within the reserve and in adjacent waters that had sloping reefs with similar topographic complexity were sampled (Shears and Babcock 2003).
		Line transects were haphazardly placed at each site. Transects were run perpendicular with the shore from MLWS (or the top of *Carpophyllum maschalocarpum* band) to the edge of the reef, or 12m depth. Habitat type, rock type, depth, slope and distance from shore were recorded at 5m intervals along each transect (Shears and Babcock 2003).
		At each site, 5 haphazardly placed 1 m^2 quadrats were sampled for macroalgae and macroinvertebrate herbivores adjacent to the transect line at each of the 4 depths (<2, 4-6, 7-9 and >10m). Sea urchins were classified as cryptic or exposed. The density and diameter of urchins (>5 mm) was recorded. Total length of individual thalli of brown algae was measured (+/-5 cm). *Ecklonia radiata* stipe length was measured. Percent algal cover was determined. Algal measurements were converted to biomass.

Table 16. Monitoring for Spatial Management in New Zealand.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
New Zealand - Long Island – Kokomohua Marine Reserve, Marlborough Sounds *(Davidson 2001)*	Experimental fishing for blue cod was carried out from 1993 to 2000 at three reserve and six control sites.	Standardised fishing gear was used, with small, barbless hooks aiming to catch as wide a range of size as possible. Fishing effort (number of fishers and time fished) was recorded at each site. Captured fish were transferred to a holding tank continuously supplied with fresh sea water. Fish were measured and then released at the end of the sampling period.
	Underwater visual transects were undertaken in four or five reserve sites and four control sites to estimate blue cod abundance. Sampling was annual over a 9 year period from 1992 to 2001.	Transects were established parallel to shore in 7 – 12 m depth. Divers counted fish present within an estimated 2 m wide x 2 m high x 30 m long 'tunnel'. Twelve replicate transects were swum at each site except in 1992 when 6 were swum. Blue cod were allocated to three size groups; juvenile < 100 mm, sub-adult 100 – 300 mm and adult > 300 mm total length. Plastic model fish were used to standardise size estimates
New Zealand - Tawharanui and Mimiwhangata Marine Reserves *(Shears et al. 2006)*	Long term but intermittent monitoring has been carried out since 1977. In the Mimiwhangata reserve, 9 sites were surveyed all within the Marine Park while at Tawharanui, 5 sites within and 5 sites outside the Reserve were surveyed.	One permanent transect was established at each site in suitable lobster habitat. Transects ran perpendicular to shore from low tide mark to a depth of 5 – 10 m. Transects were 50 m long and 10 m wide (two divers searching 5 m each side of the transect). Lobsters were visually categorised into legal and sub-legal sizes. Weight of legal-size lobsters was visually estimated.

Table 17. Management goals, monitoring objectives, indicators and performance measures for spatial management in New Zealand.

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Leigh and Tawharanui Marine Reserves *Provide for the development of scientific research programs*	Examine the temporal change in benthic community structure	Abundance Size distribution	Species (dominant brown algae and invertebrate herbivores)	Temporal trend in abundance and size distribution	*(Ayling 1978; Babcock et al. 1999; Shears and Babcock 2003)*
	Examine the spatial pattern in community structure	Maximum number	Species (predatory fish)	Maximum number, density, size distribution, habitat cover in reserve compared with outside reserve	*(Babcock et al. 1999)*
		Density	Species (spiny lobster)		
		Size distribution	Species (predatory fish, spiny lobster)		
		Benthic cover	Habitat type (categorical)		
		Productivity	Species (dominant algae)		
	Examine the spatial pattern in community structure and the stability of these patterns	Density Size distribution	Species (invertebrate herbivores and brown algae)	Density, size distribution, habitat cover and algal biomass in reserve compared with outside reserve	*(Shears and Babcock 2003)*
		Benthic cover	Habitat type (categorical)		
		Biomass	Family (brown algal species)		

Table 17 (Cont.). Management goals, monitoring objectives, indicators and performance measures for spatial management in New Zealand.

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
New Zealand - Long Island – Kokomohua Marine Reserve, Marlborough Sounds *Provide for the development of scientific research programs*	Examine the temporal and spatial change in the abundance and size structure of blue cod	CPUE	Species (blue cod)	Temporal trend in abundance, size distribution and CPUE compared to non-reserve sites	*(Davidson 2001)*
		Size distribution	Species (blue cod)		
		Abundance	Species (blue cod)		
New Zealand - Tawharanui and Mimiwhangata Marine Reserves *Provide for the development of scientific research programs*	Examine the temporal trend in spiny lobster abundance in a fully protected and a partially protected marine reserve	Abundance	Species (spiny lobster)	Temporal trend in abundance and biomass	*(Shears et al. 2006)*
		Biomass	Species (spiny lobster)		

South Africa

The Greater St Lucia Wetland Park, South Africa, consists of five individual ecosystems including a marine ecosystem. The marine reserve was established primarily to protect South Africa's only coral reefs, and to conserve the diverse marine resources within the area (Schleyer and Celliers 2003). The natural systems that are protected within the park are considered to be unique in terms of their biophysical diversity, their hydrological and ecological functioning, and the associated adaptation of the biota.

Ecosystem Effects

Within Reserve Effects

A long-term monitoring program was established for the St Lucia Marine Reserve in 1993 to assess trends in sea surface temperatures, provide information to develop management plans to protect biodiversity, and to ensure sustainable tourism (Schleyer and Celliers 2003). For an outline of the monitoring methods employed by the monitoring program (Schleyer and Celliers 2003) see Table 18. Management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 19.

The results of the St Lucia Reserve monitoring program revealed significant changes in sea surface temperature and community structure over time. A large increase in mean temperature has been measured over the past decade that may be indicative of global warming caused by climate change (Schleyer and Celliers 2003). Outbreaks of the crown-of-thorns starfish (COTS) have resulted in long-term changes in isolated areas, causing a shift from a mixed community of hard and soft corals to one dominated by soft corals at much lower cover (Schleyer and Celliers 2003). Analysis of coral larval dispersal and recruitment data will provide information on reef recovery in the event of future damage (Schleyer and Celliers 2003). The results of the St Lucia monitoring program have wider applications in understanding reef ecology and establishing critical levels for management intervention in the event of reef stress (Schleyer and Celliers 2003).

Socio-economic Effects

In the same monitoring program outlined above, reef damage caused by eco-tourism was also assessed (Schleyer and Celliers 2003). Monitoring of boat launches and scuba dives allowed an assessment of the diving tourism pressure on the reefs (Schleyer and Tomalin 2000). Damage to reefs caused by diving was estimated in 1994 and 1995 using underwater visual census (Schleyer and Tomalin 2000). A quantitative index of damage was derived for each site that was then regressed against diving intensity at the site (Schleyer and Tomalin 2000). The regression indicated that 10% diver damage occurs at 9000 dives per dive site per annum, and an annual precautionary limit of 7000 SCUBA divers per dive site was recommended to avoid reef damage (Schleyer and Tomalin 2000; Schleyer and Celliers 2003). See Table 20 for methods. Socio-economic management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 21.

Table 18. Monitoring for spatial management in South Africa.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
St Lucia Marine Reserve *(Schleyer and Celliers 2003)*	Monitoring was conducted to assess trends in water temperature, coral percentage cover, coral bleaching, and crown-of-thorns starfish abundances.	Hourly water temperature at 17 m (low tide) has been recorded on one of the study reefs (Sodwana Bay) since 1994. Since 1993, eighty 0.25m^{-2} quadrats are photographed annually and subject to image analysis. Percentage cover and bleaching of coral, and the number of crown-of-thorns starfish (COTS) is recorded. Coral larval settlement on experimental plates was studied between 1999-2002

Table 19. Management goals, monitoring objectives, indicators and performance measures for spatial management in South Africa.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
St Lucia Marine Reserve *Protection of South Africa's only coral reefs, and the conservation of the diverse marine resources within the area*	Monitor the steady rise in sea surface temperatures and provide information to develop management plans to protect biodiversity and ensure sustainable tourism	Water temperature Benthic percent cover density	Hourly Reef damage, coral cover and bleaching Species (one introduced species – crown-of-thorns starfish (COTS) and settlement of coral larvae)	Water temperature, reef damage, coral health and bleaching, density of introduced species, and larval dispersal and recruitment trends	(Schleyer and Celliers 2003)

Table 20. Socio-economic monitoring for spatial management in South Africa.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
St Lucia Marine Reserve *(Schleyer and Tomalin 2000; Schleyer and Celliers 2003)*	Coral reef damage and diving intensity was monitored to establish the sustainable diving capacity of reefs	Percentage cover of reef damage was assessed in 1994 and 1995 using drift dives along 2 m-wide belt transects. Twenty-three drift transects were undertaken ranging in length from 20 to 770 m length, with a total length of 4.7 km. A detailed record was kept of the damage encountered and included a visual estimate of the percentage cover affected. Identifiable causes of the damage were also noted The observations were summarised as the sum of percentage damage per cause and per organism type for each transect divided by the transect area Diving intensity at each site was determined from records of total dives from the local conservation authority, and detailed site-specific logs kept by professional dive operators

Table 21. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in South Africa.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
St Lucia Marine Reserve: *Protection and conservation of the coral reefs, and the diverse marine resources within the area*	Monitor diving intensity and damage to coral reefs to determine sustainable diving capacity	% damage per cause per unit area of transect	Assemblage/community	Spatial trend in diving intensity and reef damage	(Schleyer and Tomalin 2000; Schleyer and Celliers 2003)
		% damage per organism type per unit area of transect	Species		
Ensure sustainable tourism		Number of dives per site	Annual		

Kenya

Kenya has a system of fully protected Marine National Parks and partially protected Marine National Reserves. Objectives for the protected areas are:

- Preservation and conservation of the marine biodiversity for posterity
- Provision for ecologically sustainable use of the marine resources for cultural and economic benefits
- Promotion of applied research for educational awareness, community participation and capacity building.

Ecosystem Effects

Within Reserve Effects

A long-term monitoring program was set up in four, fully protected parks that varied in their time of establishment from 1968 to 1991 (McClanahan and Graham 2005). Monitoring of sites was conducted near annually, from 1987 to 2004. By examining data in terms of time since establishment, it was possible to re-construct a pooled time series from 4 years prior to 36 years after establishment (McClanahan and Graham 2005). Fish data were primarily analysed at the aggregated level using size-spectra analysis, although aggregated biomass data was also examined (McClanahan and Graham 2005). Fish communities were quantified using belt transects while benthic communities were quantified using line transects (McClanahan and Graham 2005). See Table 22 for details of monitoring methods. Management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 23.

For size-spectra analysis, individual fish were allocated to body length groupings from 10 to > 40 cm. The slope and mid-point height (an index of the assemblage abundance-biomass) were regressed against time of protection (McClanahan and Graham 2005). Size-spectra slopes were variable and had no significant trend with time. The mid-point height of the size spectra, and the biomass estimates, showed a humped-shaped relationship with time, with maximum height and biomass occurring between 21 and 22 years protection (McClanahan and Graham 2005). Benthic cover (the ratio of calcifying to non-calcifying algae) also increased significantly over time. The results of the study suggested that full recovery of the coral reef fish assemblage required greater than 20 years protection, longer than suggested by some other studies. The trend to calcifying as opposed to non-calcified algae as a result of intense fish grazing may explain the slight reduction in biomass after 25 years protection, possibly due to loss of net primary production (McClanahan and Graham 2005). One potential problem with this study was that for regression analysis against time there was no adjustment for auto-correlation in the time series so that significance levels may have been overstated.

Socio-economic Effects

Fishery Yield

Similar to other developing countries already mentioned, heavy exploitation of fishery resources is also common in East Africa, where destructive techniques such as blasting, poisons and drag nets are used. The Mombasa Marine National Park (MNP) was heavily fished before protective management was established in 1987 (McClanahan and Mangi 2001). When active management started in 1991, 63% of the fishing ground was lost, and effort also declined by 68% (McClanahan and Mangi 2001). Despite a nearly constant number of fishermen (between 25 – 30 per day on an annual basis) and boats (approximately 10 boats per day on an annual basis), total catch has declined (see Table 24 for methods). Thus, the level of spillover from Mombasa MNP was not enough to balance the reduction in fisheries yield that resulted from the creation of a large reserve (McClanahan and KaundaArara 1996; McClanahan and Mangi 2000). Mombasa MNP occupies 6-8 km^2 or around 50-60% of the total fishing area, probably occupying too large an area to supplement fishery-yields substantially. Catch rates at Mombasa MNP have declined the slowest (250 g d^{-1}) when compared to other Kenyan sites (falling at a rate of 310 – 400 g d^{-1}) due to the higher level of management (McClanahan and Mangi 2001).

These results (decreasing catch, despite stable effort) could be occurring for a number of reasons. It could be that increased effort is occurring outside the fishing grounds (such as trawlers working beyond the reef), or that there is a long-term cycle of recruitment or environmental conditions that is driving this pattern. Alternatively, effort could be too high despite being stable, and finally, there may be some forms of environmental degradation that are reducing the fisheries productivity of these reefs (McClanahan and Mangi 2001).

Tourism

Three, fully-protected areas have been active in Kenya for over 25 years and most of them are visited by around 30,000 visitors per year and raise reasonable revenues (McClanahan 1999). There is economic concern, reflected by the trend of visitor numbers to Kenya's parks over time. For example, in 1992, Mombasa Marine National Park and Reserve attracted approximately 60,000 visitors but by 1995 numbers had declined to approximately 20,000 (McClanahan 1999).

One possible reason for declining visitor numbers is impacts on the environment that are external to marine reserves, such as sedimentation due to catchment erosion, high nutrient runoff, and coral bleaching associated with climate variability. Socio-economic monitoring and assessments started in Kenya and other Indian Ocean States immediately following the coral bleaching event in 1998 (as a result of a strong El Niño – La Niña change). Socio-economic assessments were applied to allow managers to take measures to mitigate impacts, such as developing alternative attractions for tourists, assessing new dive sites, and attempting coral rehabilitation (Cesar 2003). The aims of the assessment were: to establish the level of awareness that tourists visiting had about coral bleaching and associated mortality; to evaluate tourist perceptions of the threat of coral bleaching; and to determine the willingness to pay for improvements in reef quality (Cesar 2003). In order to gauge tourist reactions, questionnaire surveys were carried out in 1999, 2000 and 2001 (see Table 24 for methods). Only a limited number of tourists were aware of coral bleaching, although 80 % of those aware said it would actually affect their decision to visit and dive in an area (Cesar 2003). Additionally, tourists visiting Kenya were willing to pay US $59 extra to experience healthier reefs (Cesar 2003).

Socio-economic management goals, monitoring objectives, indicators and resultant performance measures for Kenyan Marine Parks are summarised in Table 25.

Table 22. Monitoring for spatial management in Kenya.

Monitoring Location and Citation Details	***Monitoring Outline***	***Methods***
Malindi and Watamu, Kisite, and Mombasa Marine National Parks *(McClanahan and Graham 2005)*	Fish and benthic communities were surveyed annually at sites within 4 marine reserves (gazetted between 1968 and 1988) annually from 1987 to 2004.	Fish communities were quantified using 2 to 5 replicate, 5 x 100 m belt transects per site. Wet weight estimates of fish were made by classifying each individual encountered in transects to the family, estimating its length, and placing it into 10 cm size-class intervals, up to 40 cm, and with no individuals < 3 cm length recorded. Wet weights per family were estimated from length-weight correlations established from measurements of the common species in each family taken at local fish landing sites. Attached benthic communities were sampled by the line-intercept method using 9 to 27, 10 m line transects at each site per year. Cover of benthic biota > 3 cm length under the line was classified into 9 gross substratum categories (hard coral, soft coral, algal turf, coralline algae, calcareous algae, fleshy algae, seagrass, sand, and sponge), and their lengths were measured to the nearest centimetre.

Table 23. Management goals, monitoring objectives, indicators and performance measures for spatial management in Kenya.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Malindi and Watamu, Kisite, and Mombasa Marine National Parks *Preservation and conservation of the marine biodiversity for posterity* *Provision for ecologically sustainable use of the marine resources for cultural and economic benefits* *Promotion of applied research for educational awareness, community participation and capacity building*	To monitor recovery of biomass and size structure of the fish population as a function of the of the age of the closed area protection	Slope of size spectrum	Aggregated fish-data	Change in slope in relation to years of protection	(McClanahan and Graham 2005)
		Mid-point height of size spectrum	Aggregated fish-data	Change in mid-point height in relation to years of protection	
		Biomass	Aggregated fish-data	Change in biomass in relation to years of protection	
	Monitor cover of benthic biota in relation to years of protection	Ratio of calcifying and non-calcifying algae	Algal functional groups	Change in benthic cover ratio in relation to years of protection	

Table 24. Socio-economic monitoring for spatial management in Kenya.

Monitoring Location and Reference Details	*Monitoring Outline*	*Methods*
Malindi and Watamu, Kisite, and Mombasa Marine National Parks *(McClanahan and Mangi 2001)*	Compare catches on a per man and per area basis. Assess trends in catch and effort in coral reef areas. Data was collected from September 1995 to June 1999.	Fish landing data was collected at 8 landing sites in South Kenya, including sites at Mombasa Marine National Park (MNP) and the Diani area. Wet weights (to the nearest 0.5 kg) of each group of species were estimated using a spring balance. Recorded data included the number fishing and types of gears used at each landing site. The total catch, fish groups and numbers that participated in fishing were recorded each sampling day. Discussions with fishermen revealed that fishing was carried out for 24 days a month, while catch data were collected for 12 days (± 4) for the Kenyatta landing and 3 days (± 1) for each site in the Diani region. Catch on per area & month basis was calculated by adjusting to 24 days & dividing by the area of the fishing ground. These data were analysed to calculate daily, monthly and yearly averages of the catch per individual & area. To describe the fishery, each gears contribution to the catch was calculated for each fish family group.
(McClanahan 1999)	Calculate the number of visitors annually and assess the change over time.	The number of visitors to Kenyan marine parks was recorded and plotted. The number of visitors attending MNP has been plotted since 1990. The number of visitors is reported on an annual basis.
(Cesar 2003)	Questionnaires were conducted from 1990 – 2001 to gauge tourist reactions to coral bleaching and reef degradation.	Questionnaire surveys were carried out in 1999, 2000 and 2001, administered to departing tourists in airports (in Kenya) and selected dive shops and tourist establishments. Tourists were questioned on what they liked and disliked about their stay, and the level of their knowledge about coral bleaching.

Table 25. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in Kenya.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Malindi and Watamu, Kisite, and Mombasa Marine National Parks	Record data on all fishing activities in areas adjacent to marine reserves	No. of fishermen	Local fishermen	Annual trend in number of fishermen, catch and catch rate.	(McClanahan and Mangi 2001)
		No. of boats			
		Types of gear used			
		Catch per area			
		Catch per fisherman			
Provision for ecologically sustainable use of the marine resources for cultural and economic benefits		Total catch			
	Monitor visitors attending the parks	No. of visitors	Wider community	Temporal trend in the number of visitors to marine parks	(McClanahan 1999)
	Establish the level of awareness that tourists visiting have about coral bleaching and associated mortality	Survey response	Wider community	Level of community awareness and perceptions	(Cesar 2003 243)
	Evaluate tourist perceptions of the threat of coral bleaching				
	Determine the willingness to pay for improvements in reef quality				

France

Located in the French Mediterranean, the Cote Bleue Marine Park (CBMP) was established in 1983, primarily to protect marine biodiversity, support social and economic activities linked to the sea, especially fisheries, and promote public education and scientific research (Claudet *et al.* 2006). The CBMP comprises two, effectively enforced, no-take reserves: Carry (85 ha), established in 1983, and Couronne (210 ha), established in late 1995. In addition to the reserves, two kinds of artificial reefs, for protection against illegal trawling, and for biomass production, have been deployed within the park since 1983, with several of them set at the border of the two reserves to ensure trawl exclusion.

Ecosystem Effects

Within Reserve Effects

Claudet *et al.* (2006) conducted a study within the CBMP to investigate whether or not the marine park, together with bordering artificial reefs, was effective in restoring local fish-assemblages. Surveys were conducted at the Cauronne marine reserve during three years: before park establishment (1995), and after (1998 and 2001). To ensure that marine park effects were not confounded with other factors structuring spatial variability of fish, habitat characteristics were considered in the models. Claudet *et al.* (2006) interpreted the model results using a method for identifying indicator species that could be relevant for marine park monitoring and management purposes. For an outline of the research methods employed by Claudet *et al.* (2006) see Table 26. Management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 27.

Claudet *et al.* (2006) found significant differences in the abundance of fish between sites inside and outside of the park, across years, for all groups of fish or species considered, except for unfished species. Interactions were also significant when environmental co-variables were accounted for in the models. Before marine park establishment, small species, species of low fishing value and sedentary species displayed significant differences in abundance between sites inside and outside of the park. Species with low fishing value had, on average, higher abundances outside the park, whereas small species and sedentary species showed, on average, higher abundances within the park. Habitat preferences and/or natural variability could explain these spatial differences, even if habitat variables considered in the study did not allow for this hypothesis to be tested (Claudet *et al.* 2006).

After the CBMP was established, Claudet *et al.* (2006) found that there were clear changes in the spatial patterns of abundance. In 1998, differences in abundance between sites inside and outside of the park were significant for all groups considered, except for small fish and, surprisingly, large species. These differences corresponded to increased abundances within the park. The magnitude of the response to marine park establishment was not clearly related to fishing value at this early stage of restoration (three years after marine park establishment) (Claudet *et al.* 2006). At the fish assemblage level, differences between sites inside and outside of the park were more marked for metrics (total abundance, species richness and diversity) calculated from large fish only. At the species level, all metrics responded to marine park establishment (except for total abundance of *Coris julis*) through increasing abundances within the park. From 1998 onwards, many species belonging to almost all the families encountered in the study were significant indicator species within the park, but no indicator species could be identified for a particular year (1998 or 2001); which in fact would not be desirable for an indicator of protection (Claudet *et al.* 2006).

Claudet *et al.* (2006) found that six years after marine park establishment (2001), differences between sites inside and outside of the park were even more significant than in 1998, except for metrics based on small fish. The contrast between increased abundances of large and medium fish, and stable abundances of small fish, shows that six years after marine park establishment, positive effects mostly related to larger sizes and higher abundances within the park (Claudet *et al.* 2006). Effects linked to population replenishment through generation of offspring were not yet in evidence, at least not from this kind of data (Claudet *et al.* 2006). Furthermore, there was still no clear link between fishing value of species and response to Marine Park establishment. Although the differences in abundance of species with medium to high fishing value were more significant in 2001 than in 1998, this may be explained by demographic characteristics of species or changes in fishing patterns outside the park (Claudet *et al.* 2006). At the fish assemblage level, all metrics (total abundance, species richness and diversity) displayed significant differences between sites inside and outside of the park six years after establishment.

Socio-economic Effects

In the Mediterranean, few data are available on the socio-economic consequences of MPAs (Badalamenti *et al.* 2002). A general increase in tourism activities in Mediterranean MPAs is evident. There is a large increase in the number of divers and vessels using MPAs and this has consequently had impacts on natural benthic communities as a result of diver damage, mooring and the feeding of large fish by divers (Badalamenti *et al.* 2002). Compliance is high in both the no-takes reserves at CBMP as both MPAs were established with the support of users (Claudet *et al.* 2006). There are 40 fishing boats and the number of fishers has remained stable over the last 20 years (Claudet *et al.* 2006). The recreational fishery comprises approximately 60 sailors and most use handlines (Claudet *et al.* 2006).

Table 26. Monitoring for spatial management in France

Monitoring Location and Reference Details	*Monitoring Outline*	*Methods*
Mediterranean - Cote Bleue Marine Park *(Claudet et al. 2006)*	The aim of this study was to test whether the MPA together with bordering artificial reefs was effective in restoring local fish assemblages. Surveys were conducted within and outside of the Cauronne marine reserve (Cote Bleue Marine Park) at the end of summer during three years: before MPA establishment (1995), and after (1998 and 2001).	Two sites within the reserve and two sites outside the reserve (between 14 and 18 m depth) were chosen. Within the reserve, an additional site was sampled at greater depths (between 24 and 26 m). Twelve, 20m long transects (same position each year) were surveyed using underwater visual census techniques at each site. The number and size of each fish observed within a distance of 2.5m on each side of the transect were identified and recorded. Fish sizes were estimated according to three size groups (small, medium, and large); the total fish abundance of a species being the sum of the abundances per size group. For each species, size groups were defined using 33% and 66% percentiles of the maximum size generally observed in the region. All fish seen were recorded but pelagic species (*Sardina pilchardus*) and notoriously cryptic species (e.g. *Gobiidae, Bleniidae, Tripterygiidae*) were excluded from the analyses. For each transect, the complexity of the substratum was coded into three classes: 1) smooth; 2) smooth bottom with a few blocks smaller than 50 cm and not suitable for shelter; 3) more blocks, some higher than 1 m, and a lot of refuges. The percent linear cover of the seagrass *Posidonia oceanica* was estimated along each transect. Although the surveyed sites were very similar, environmental data was collected to assess small-scale spatial variability.

Table 27. Management goals, monitoring objectives, indicators and performance measures for spatial management in France.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Mediterranean - Cote Bleue Marine Park: *Protection of marine biodiversity, support for social and economic activities linked to the sea, especially fisheries, and promotion for public education and scientific research*	Investigate whether or not the marine park together with bordering artificial reefs is effective in restoring local fish assemblages	Density – multivariate (Permanova)	Total (all fish species)	Significant interaction between time and reserve status indicating change in the variable in reserve versus non-reserve area after protection.	(Claudet *et al.* 2006)
			Large, medium, small fish (all species)		
			Large species, medium species, small species		
			Low, high value commercial species		
			Mobile/sedentary species		
		Density – univariate	Total (all fish), large fish		
			Two fished species, one non-fished species		
		Species richness - univariate	Total (all fish), large fish		
		Diversity - univariate	Total (all fish), large fish		
		Substrate complexity	Categorical		
		Benthic percent cover	Habitat type (one marine plant)		

Ecuador

The Galapagos Marine Reserve was established in 1998 to protect the waters surrounding the Galapagos Islands and the resources that they contain. In 2000, a zoning system was introduced to the area to protect marine biodiversity, promote sustainable uses and reduce the conflict between users (tourism, fishing, and scientific research) of the Galapagos Marine Reserve.

Ecosystem Effects

Within Reserve Effects

To assess the importance of baseline information for the evaluation of the effects of marine protected area and zoning establishment in the Galapagos Marine Reserve, Edgar *et al.* (2004b) undertook broad-scale ecological surveys of the reserve in 2000 and 2001. The plant and animal densities on shallow reefs across the marine reserve were determined in order to provide baseline information that could be used to assess changes in different zones over time, and whether the zones were optimally located. For an outline of the research methods employed by Edgar *et al.* (2004b) see Table 28.

The density of sea cucumbers, the most valuable fishery resource, was three-times higher in zones that remained open to fishing compared to the conservation zones that were closed to fishing (Edgar *et al.* 2004b). This result was observed in areas of high population densities; however, there was no significant difference in sea cucumber density between zones with low animal density at other locations within the reserve. Spiny lobster, another important fishery resource, showed a similar pattern with 2.7-times greater abundance in fished zones compared to conservation zones, although the difference between zone types was not significant (Edgar *et al.* 2004b). The density of sharks was five-times higher in tourism zones than fishing or conservation zones (Edgar *et al.* 2004b).

These results reveal the complex social and political interactions and potential bias that can accompany selection of marine protected areas. For example, the fishing industry may attempt to locate protected zones in resource-poor areas, and the tourism industry may argue for protection of areas that contain atypically interesting features (Edgar *et al.* 2004b).

To clarify the broad-scale, marine-biogeographical patterns across the Galapagos, Edgar *et al.* (2004a) investigated the distribution of fauna within the reserve at regional scales. For an outline of the research methods employed by Edgar *et al.* (2004a) see Table 28.

Edgar *et al.* (2004a) found that sub-tidal communities of fishes and macro-invertebrates on shallow reefs differed consistently in species composition across the archipelago. The northern bio-region, that encompasses the Darwin and Wolf Islands, was characterised by high fish species richness, including a high proportion of species of Indo-Pacific origin. There were, however, few endemics or species with distributions that extend south from Ecuador (Peruvian species) present, and relatively low invertebrate species richness (Edgar *et al.* 2004a). In contrast, the western bio-region, that encompasses the Fernandina and Isabella (western) Islands, had high numbers of endemics and temperate Peruvian fish species, and high species richness of invertebrates, but very few species of Indo Pacific origin (Edgar *et al.* 2004a). The Bahia Elizabeth and Canal Bolivar south-western bio-region had more endemics, and fewer species with Peruvian affinities, than observed within the western bio-region (Edgar *et al.* 2004a). The north-eastern bio-region of Pinta, Marchena and Genovesa represented a zone with affinities to both the northern and south-eastern islands. Finally, the south-eastern bio-region included species from a variety of different sources, particularly 'Panamic' species with distributions extending north of Central America (Edgar *et al.* 2004a).

The biogeographical framework for marine reserves is often defined using geophysical approaches, with regions classified in terms of abiotic factors such as salinity, temperature, depth, rock type or sediment particle size that are considered substitutes for biotic factors (Roff *et al.* 2003). However, physical factors rarely act in synchrony, and congruence between physical and biological regionalisations has rarely been assessed (Edgar *et al.* 2004a). This research highlights how biological as well as physical regionalisations can vary greatly, depending on which component of the biota is analysed, and also the importance of incorporating both biological *and* physical factors into the framework for establishing marine reserves.

Management goals, monitoring objectives, indicators and resultant performance measures for the Galapagos Marine Reserve are summarised in Table 29.

Table 28. Monitoring for spatial management in Ecuador.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Galapagos Marine Reserve *(Edgar et al. 2004b)* *(Edgar et al. 2004a)*	Sampling was conducted between May 2000 and November 2001 at sites at 50 islands and islets. Generally, two different depth contours (between 2 and 20 m depth) were sampled at each site. Overall, a total of 579 and 569 depth strata were surveyed for key resource fish and macro-invertebrate species, respectively. The total number of different management zones censused was 11, 40, and 31 for conservation, tourism and fishing zones, respectively.	Fish abundance was estimated from underwater visual census by recording 5m either side of a 50m transect line (50 x 10 m census block). For the majority (59%) of depth strata, two replicate 500 m^2 blocks were surveyed and the data was averaged. Following fish counts, sea cucumbers, spiny lobsters and other large macro-invertebrates were censused one metre either side of the same 50m transect line (see above) ($100m^2$ blocks).

Table 29. Management goals, monitoring objectives, indicators and performance measures for spatial management in Ecuador.

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Galapagos Marine Reserve: *Protection of the waters surrounding the Galapagos Islands and the resources that they contain*	Assess the importance of baseline information for evaluating effects of marine protected area and zoning establishment, and determine whether zones have been optimally located.	Density (univariate)	Major fishery species (Sea cucumbers, lobsters, Bacalao, sharks	Comparison of densities in different management zones	(Edgar *et al.* 2004b)
	Clarify broad scale marine biogeographical patterns.	Density (multivariate)	Assemblage	Separation of different bioregions in multi-dimensional space	(Edgar *et al.* 2004a)

Monitoring for Spatial Management *Australia*

Queensland

The Great Barrier Reef Marine Park was established in 1975 and encompasses the largest series of coral reefs in the world. The system comprises some 3,400 individual reefs where some 400 types of coral, 1,500 species of fish, and 4000 types of mollusc are found. The marine park was primarily established to protect the area's outstanding marine biodiversity and many threatened species while providing for reasonable use. Management focus of the park requires the establishment of appropriate balance between conservation and human-use activities that may have significant impacts on the health of the park. The existing balance between use and conservation is defined by specific and comprehensive zoning arrangements (Hand 2003).

Ecosystem Effects

The Long-term Monitoring Program of the Australian Institute of Marine Science (AIMS) was developed to track the trends in shallow, coral reef communities across much of the Great Barrier Reef to enable assessment of localised changes and development of effective management strategies (Sweatman and Wachenfeld 2003). The research focuses on assessing general status, trends and historical variability of the whole reef system, which is particularly important for understanding and developing responses to large-scale ecological threats such as crown of thorns starfish (COTS) and coral bleaching (Sweatman and Wachenfeld 2003). For an outline of the research methods employed by the AIMS Long-term Monitoring Program (Sweatman and Wachenfeld 2003) see Table 30. The 15-year program provides a broad-scale picture of the status and trends of coral reef communities over a large area of reef. The program has documented large-scale disturbances such as tropical cyclones "Justin" and "Rona", recovery of hard coral cover following tropical cyclones "Ivor" and "Harry", and changes in density and distribution of populations of COTS (Sweatman and Wachenfeld 2003).

Within Reserve Effects

At the Whitsunday and Palm Islands of the Great Barrier Reef, Graham *et al.* (2003) investigated the effect of marine reserve protection on predator-prey interactions of coral reef fish. Previous studies had often lacked dietary information on major predators, and were confounded by differences in habitat complexity between reserve and non-reserve sites. The abundance of prey fish species of *Plectropomus leopardus* (Serranidae), a piscivore and a major target species for fisheries on the Great Barrier Reef, was estimated in protected and fished zones. The prey species were identified from previous detailed studies of the diet of *P. leopardus*. Surveys were focussed on target fish populations and habitat characteristics. For an outline of the research methods employed by Graham *et al.* (2003) see Table 30.

The results of this research suggest a secondary effect of marine reserve protection. Eight of the nine prey species had a higher mean density in fished zones (areas of low *P. leopardus* biomass) than protected zones (areas of high *P. leopardus* biomass) (significant for six species) (Graham *et al.* 2003). The density of all prey fish was twice as high in the fished zone than the protected zone. This suggests a trophic effect of zoning, with a greater density of piscivores within protected areas causing a decrease in prey abundance, and a reduction of piscivores in fished areas resulting in a release of prey populations (Graham *et al.* 2003).

Because there is currently only limited evidence that no-take marine reserves on the Great Barrier Reef have increased abundance of reef fish targeted by fisheries, Evans and Russ (2004) examined the effect of no-take reserves on abundance of species targeted by hook-and-line fisheries around the Whitsunday, Palm, and Keppel Islands. The aims of the study were to compare the density and biomass of *Plectropomus* species (coral trout) and *Lutjanus carponotatus* (Lutjanidae) – both targeted by fishing on the Great Barrier Reef – with two non-target (control) species, *Siganus doliatus* (Siganidae) and *Chaetodon aureofascinatus*

(Chaetodontidae), in protected zones and fished zones of three inshore island groups of the Great Barrier Reef Marine Park. For an outline of the research methods employed by Evans and Russ (2004) see Table 30.

The densities of *Plectropomus* and *L. carponotatus*, both targeted by fisheries, were much higher in protected zones than fished zones for two of the three island groups. Evans and Russ (2004) found that *Plectropomus* were 3.6 and 2.3 times more abundant in protected than fished zones of the Palm and Whitsunday Islands, respectively, and that *L. carponotatus* were 2.3 and 2.2 times more abundant in protected zones than fished zones of the Whitsunday and Keppel Islands, respectively. The biomass of *Plectropomus* and *L. carponotatus* was significantly greater (3.9 and 2.6 times, respectively) in the protected zones than fished zones at all three island groups (Evans and Russ 2004). Legal minimum sizes of *Plectropomus* and *L. carponotatus* are 38 and 25cm TL, respectively. There were significantly higher densities and biomasses of *Plectropomus* >35cm TL (density: 3.8 times; biomass: 5.1 times) and *L. carponotatus* >25cm TL (density: 4.2 times; biomass: 5.3 times) in protected zones than fished zones at all three island groups (Evans and Russ 2004).

Unlike the significantly higher biomass found in protected zones compared with fished zones, the density of *Plectropomus* and *L. carponotatus* clearly varied with island group and this resulted in an overall non-significant difference in density of both species between protected and fished zones. Such a non-significant effect of zoning on *Plectropomus* species density is similar to the findings reported in many other studies on the Great Barrier Reef (Ayling *et al.* 2000; Mapstone *et al.* 2003). Evans and Russ (2004) indicate that the different patterns of density between protected and fished zones at the three island groups may have been caused by differences in recruitment on large scales. For the two control species that are not targeted by fisheries (*Siganus doliatus* and *Chaetodon aureofascinatus*), there was no significant difference in abundance between protected and fished zones (Evans and Russ 2004). Furthermore, there were no significant differences in benthic characteristics between protected and fished zones (Evans and Russ 2004).

The results of this study were confounded somewhat by the sampling times (Evans and Russ 2004). For example, *Plectropomus* species are known to settle onto reefs of the Great Barrier Reef around January-February (Ferreira and Russ 1994; Mapstone *et al.* 2003); however, they may not be detected by visual census methods designed to sample adult fish until much later in the year. Thus, recruitment of *Plectropomus* species in 2002 may have been detected in the Keppel Islands (sampled October 2002), but not in the Palm or Whitsunday Island groups (sampled December 2001 and April 2002 respectively). To reduce the chances of sampling *Plectropomus* species during their new-moon spawning aggregations (Samoilys 1997), Evans and Russ (2004) sampled the Whitsunday Islands during full moons.

Like most other studies of no-take marine reserves, Evans and Russ (2004) lacked data on abundance of fish before protection and therefore it was not possible to show that protection by zoning caused the higher biomass of target fish in protected zones. Williamson *et al.* (2004) examined pre-zoning (1983–1984) data from the Palm and Whitsunday Islands and concluded that protection from fishing was the likely cause of the larger biomass of target groups in protected zones than in fished zones (Evans and Russ 2004).

Management goals, monitoring objectives, indicators and resultant performance measures for the Great Barrier Reef Marine Park are summarised in Table 31.

Socio-economic Effects

As would be expected of the world's largest system of coral reef, the Great Barrier Reef Marine Park (GBRMP) attracts a great deal of tourism, commercial fishing (including trawling) and recreational boating and fishing (Alder 1996). In order to conserve fish stocks in a sustainable manner and ensure equitable resource allocations (between sector groups), the Great Barrier Reef Marine Park Authority undertook a preliminary investigation into the socio-motivational aspects of recreational fishing (Ormsby and Innes 1999). From a survey of 2061 recreational anglers (see Table 32 for the methods of surveys conducted), 50% of the anglers from the Great Barrier Reef regions possessed at least 30 years of fishing experience (Ormsby and Innes 1999). Motivation of the anglers in this region varied, depending on their experience. Less experienced anglers were more interested in the non-catch related benefits to be gained from participating in recreational fishing activities, while the more experienced anglers were more motivated by the skills and challenge involved in catching fish, thus reflecting a greater dependency on the resource (Ormsby and Innes 1999). Seventy-five percent of the anglers in this region spent a single day at a time with approximately 6 persons per boat and spent most of the time line fishing, crabbing and collecting bait (Ormsby and Innes 1999). The information obtained from the socio-economic surveys provides a first step in developing a holistic range of information for managers to draw upon in the management of the park.

For effective management, public acceptance and support of MPAs is just as essential as user compliance with rules and regulations. Therefore, both educational and enforcement programs were also developed and implemented for the GBRMP, and both have had an influential role in maintaining community awareness and involvement (Alder 1996). Educational programs were targeted at the general community as well as specific users within the park. The common theme was to inform the audience of the park's existence, values, issues and management (Alder 1996). From 1985 – 1991, programs were designed for a specific type of user and included additional messages that focused on modifying attitudes and behaviours. For example, television advertisements attempted to motivate recreational fishers to reduce their daily catches (Alder 1996). Apart from word-of-mouth, the most effective education programs (see Table 32 for methods) were user activity guides and television advertising, with 16 % of survey respondents informed of park values and management in this manner (Alder 1996). Displays at annual boat and other shows (5%), and talks at schools and local organisations (5%), were also effective (Alder 1996). Radio, posters and newsletters were the least effective with no respondents having been informed in this way (Alder 1996).

Additionally, a two-phase participation program for the public was used to provide a forum for the community to have a say on the formulation of the original Cairns Section Zoning plan in 1982, and a second zoning plan in 1989 (see Table 32 for methods of survey conducted by Alder (1996). In both planning projects, extensive publicity campaigns were used to encourage residents to write down and forward their concerns about management, and their desires for the future management of the section (Alder 1996). These written submissions were recorded, analysed and used to examine changes in public participation over the study period.

Overall, awareness of the marine park increased significantly ($P < 0.05$) between 1985 and 1991 (Alder 1996). Surveys also indicated that support for restricting certain activities (commercial and recreational fishing, spearfishing, shell collecting, and resort developments) from specific areas remained high, with 80% of people aware and in support of restrictions between 1985 and 1991 (Alder 1996).

Socio-economic management goals, monitoring objectives, indicators and resultant performance measures for the Great Barrier Reef Marine Park are summarised in Table 33.

Table 30. Monitoring for spatial management in Queensland, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Queensland - Long-term Monitoring of the Great Barrier Reef *(Halford and Thompson 1994; Bass and Miller 1996; Sweatman and Wachenfeld 2003; Abdo et al. 2004)*	Monitoring program initiated in 1992. Intensive sampling of each reef occurs once a year. There are permanent sites on 48 "core" reefs with broader scale surveys of about another 50 reefs. The intensive study reefs were chosen to represent the major gradients in reef communities on the Great Barrier Reef: between the ocean and the coast (range 2-200 km); and north to south (over 1,100 km). Sampling is focused on 50 m long transects that follow depth contours at 6-9 m on the northeast side of study reefs. Groups of 5 transects comprise a site, with 3 sites per study reef.	Underwater visual census on MANTA board for two minutes after which observations are recorded. Two teams work in opposite directions around the reef surveying half the reef perimeter each. Density of Crown of Thorns Starfish (COTS), size of COTS (three categories: juvenile <5cm, sub-adult 6-15cm, and adult >15cm), presence of COT feeding scars (three categories: absent, present, common), live, dead and soft coral percent cover (11 categories), and visibility (4 categories) were recorded. Surveys of sessile benthic organisms were conducted. A 30 cm wide belt was recorded along each 50 m SCUBA transect using a digital video camera held 25-30 cm above the substrate. Percent cover of corals and other benthic categories (approximately 200 systematically dispersed points were sampled from each video transect) were recorded. Surveys of the density of reef fishes were conducted by underwater visual census. Fishes of 191 species were counted (0+ year-class was excluded because the surveys spanned the annual recruitment season). Surveys of agents of coral mortality were conducted by underwater visual census in a 2 m wide, 50 m long belt transect. Density of COTS, (three categories: juvenile <5cm, sub-adult 6-15cm, and adult >15cm), COTS feeding scars, *Drupella* spp. feeding scars, unknown scars, percent of coral that is bleached and/or diseased was recorded.

Table 30 (Cont.). Monitoring for spatial management in Queensland, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
(Graham et al. 2003)	This study investigated the effect of marine reserve protection on predator-prey interaction of coral reef fish on the inshore islands of the Great Barrier Reef. Sixteen study sites, that were at least 100 m apart, were chosen randomly at both the Whitsunday and Palm Islands. Eight sites at each location were situated in no-take protected zones and the remaining eight sites were situated in fished zones. Surveys were done in 2001 at the Whitsunday Islands and 2002 at the Palm Islands. The protected reefs had 14 years of zoning protection before this study was conducted.	Five 50 x 6 m replicate belt transects were visually censused at each site within a depth range of 7-11 m. Transects were a minimum of 5m apart. Two divers (counting predators and prey separately) swam side by side along each transect. The prey species of *Plectropomus leopardus* surveyed were five species of pomacentrids, two species of labrid, and two species of caesionid. Precise counts were obtained for all species where possible, or if in too high abundance an estimate was made. Data were collected on the density and size class of predators (P. *leopardus* and *Plectropomus maculates*) at each site. A habitat index incorporating refuge holes was used. The number of different-sized refuge holes was counted within a 1m strip for two, 10 m sections along each transect (see above). Percentage of live coral cover was estimated every 2 m along each transect. Each transect was assigned an arbitrary structural complexity.
Australia, Queensland *(Evans and Russ 2004)*	Underwater visual surveys were used to estimate the effect of no-take reserves on abundance of species targeted by hook-and-line fisheries around the Whitsunday, Palm, and Keppel Islands during 2001 and 2002. Data were collected at six randomly selected sites in each protected and fished zone in the Keppel Islands. Twelve randomly-selected sites were sampled for each protected and fished zone in the Palm and Whitsunday Islands.	Within each of the sites, five transects of 50 x 6 m were surveyed by underwater visual census. Total length was estimated by placing target species of fish into 5 cm size classes. Biomass was estimated from published biomass-length relationships from Fishbase (Froese and Pauly 2002) for *L. carponotatus* and from Ferreira and Russ (1994) for *Plectropomus* species. Surveys did not proceed if visibility was less than 4 m. Benthic habitat data were collected along each transect. Structural complexity of the substratum was estimated (structural complexity index 1-5). A line intercept technique was used to estimate cover of benthic categories (hard coral, soft coral, algae, rubble, sand, and 'other'). Categories were recorded every 2 m along each transect.

Table 31. Management goals, monitoring objectives, indicators and performance measures for spatial management in Queensland, Australia

Location and Proposed Management Objective	*Monitoring Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Great Barrier Reef: Protection of the area's outstanding marine biodiversity and many threatened species while providing for reasonable use	Assess general status, trends and historical variability of the whole reef system, that is particularly important for understanding and developing responses to large-scale ecological threats	Density	Species (COTS)	Coral Percent Cover, density of COTS and reef fish trends (an annual trend of greater than 10% is considered significant)	*(Sweatman and Wachenfeld 2003)*
		Benthic percent cover	Habitat type, coral bleaching		
		Density	Total (all fish species)		
	Determine the effect of marine reserve protection on predator-prey interaction of coral reef fish	Density	Total (all prey species)	Trends in density of predator and prey species	*(Graham et al. 2003)*
		Density	Species (one predator species)		
		Size distribution			
		Habitat complexity	Categorical		
	Examine the effect of no-take reserves on abundance of species targeted by hook-and-line fisheries	Density	Species (exploited and non-exploited species)	Trends in density, biomass and size distribution	*(Evans and Russ 2004)*
		Biomass			
		Size distribution			
		Benthic percent cover	Habitat type		
		Substrate complexity	Categorical		

Table 32. Socio-economic monitoring for spatial management in Queensland.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
The Great Barrier Reef Marine Park (GBRMP) *(Ormsby and Innes 1999)*	During 1997-1998 surveys were conducted to investigate the socio-motivational aspects of recreational fishing.	2061 surveys were completed by recreational anglers from Qld. 1180 fishers were from south-east Qld, 593 were from the Great Barrier Reef region and 288 were western Qld anglers. Information collected included respondent's motivations for participating in recreational fishing, an assessment of their fishing experience, details of their last fishing trip, trip satisfaction, and demographic characteristics.
(Alder 1996)	To investigate the interaction of education and enforcement on influencing behavioural changes relating to compliance with park rules, and to evaluate the effectiveness of education programs in raising awareness of the park, changing attitudes, increasing support for management and participation in management.	From 1985 to 1991 the Qld National Parks & Wildlife Services and the GBRMPA conducted a variety of education and public contact programs (displays at boat shows & coast guard office, user activity guides, talks to schools & local organisations, newspaper articles, TV documentaries & advertising, radio, posters, boat ramp signs etc). The number of people who were directly contacted by the day-to-day management staff was recorded. In August & September 1985, a face-to-face survey of 348 Cairns residents (0.6% of the population) was conducted to gather information on their awareness and attitudes towards management of the marine park. The questionnaire was divided into 3 parts. The first examined the profile of residents (levels of use, activities, age, sex etc). The second part investigated respondent's awareness of the marine park, their ability to identify the park, their knowledge of zoning, their perceptions on who managed the park, and the most effective method for informing residents. The last part examined the attitudes of respondents to various aspects of park management. Similar to the 1985 survey, a face-to-face survey of 454 Cairns residents (0.6 % of the population) was conducted in October and November 1991. Some questions were removed or modified and new questions were added (including how the nature of tourism and recreation changed over the 6 yrs management of the section). A total awareness score (TAS) was developed for both surveys. This provided an overall assessment of the marine park.

Table 33. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in Queensland, Australia

Location and Proposed Effect	*Research Objective*	*Metric*		*Performance Measures*	*Reference*
		Variable	*Scale*		
The Great Barrier Reef Marine Park (GBRMP) *To conserve fish stocks in a sustainable matter and ensure equitable resource allocations (between sector groups)*	Socio-motivational surveys	Survey response	Local and wider community	Temporal trend in awareness and compliance by residents and visitors	*(Alder 1996)* *(Ormsby and Innes 1999)*
To increase awareness of park values, and support for, and participation in, management	Educational programs	Community awareness & support for management Community compliance	Local and wider community	Temporal trend in community awareness	
	Public participation survey	Survey response	Cairns residents	Temporal trend in awareness and support for management	

Tasmania

In 1991 fishing was prohibited in four marine reserves (Maria Island, Tinderbox, Ninepin Point and Governor Island) off the eastern coast of Tasmania. The primary reasons for declaring these reserves were to conserve representative and unique Tasmanian marine habitats, to provide reference locations where the dynamics of marine communities could be observed independently of fishing effects, and to create fish propagation areas (Edgar and Barrett 1997).

Ecosystem Effects

Within Reserve Effects

Initiated in 1992, the Tasmanian Marine Park Monitoring Program collects biotic data from a series of sites within four marine reserves (Maria Island, Tinderbox, Ninepin Point and Governor Island) and adjacent reference sites external to the reserves. The focus of the monitoring program was originally to assess whether creation of Tasmanian reserves affected biodiversity, or the abundance or size distribution of commercially important fishery species, or whether indirect, broad-scale habitat shifts occurred (Edgar and Barrett 2002). Specific factors that were monitored included: 1) fish, macroinvertebrate and plant species richness; 2) density and mean size of rock lobsters (*Jasus edwardsii*), abalone (*Haliotis rubra*), blue-throated wrasse (*Notolabrus tetricus*), purple wrasse (*Notolabrus fucicola*) and trumpeter (*Latridopsis forsteri*); and, 3) cover of the dominant seaweeds. For an outline of the research methods employed by the Tasmanian Marine Park Monitoring Program (Edgar and Barrett 1997; 2002) see Table 34.

Edgar and Barrett (1997) collected data on reef biota from within, and adjacent to, the Maria Island, Tinderbox, Ninepin Point and Governor Island marine reserves during 1992 and 1993, shortly after the declaration of the reserves. Classification of the sites revealed that several of the Maria Island reference sites possessed assemblages quite different from those within the reserve, despite their adjacent locations, and having been selected to match the environmental conditions to sites within the reserve. This discrepancy indicates the need to ensure that external reference sites are truly comparable with internal sites (Edgar and Barrett 1997). Edgar and Barrett (1997) also found that the biota of all sites studied within the Maria Island reserve were relatively homogenous. While the Maria Island reserve was originally proclaimed to protect representative Tasmanian east coast habitats, it apparently contained only a restricted subset of habitat types from that region.

Over 10 years of monitoring, important changes were observed in the marine reserves compared with reference sites (Edgar and Barrett 1999; Barrett *et al.* 2003). One of the most notable results was that the abundance and size of rock lobsters has increased markedly in the Maria Island reserve over 10 years. The abundance in the reserve increased 4 fold in the reserve over the period, mainly attributable to mature animals above the legal size limit that were virtually absent from reference sites (Edgar and Barrett 1999; Barrett *et al.* 2003). Densities of the bastard trumpeter (*Latridopsis forsteri*) also increased significantly in the Maria Island reserve relative to the external reference sites (Edgar and Barrett 1999; Barrett *et al.* 2003). The abundance of larger blue-throated wrasse (*Notolabrus tetricus*), the number of large fish species, and the number of large fishes, has increased significantly at the Tinderbox reserve, which is in an area of high fishing pressure (Barrett *et al.* 2002). A relatively unexpected result of the monitoring was that abalone abundances at Maria Island declined by approximately 50% over the decade, while numbers in the reference sites remained relatively constant (Barrett *et al.* 2003). Analysis of abalone size showed that abundance of large abalone had remained relatively constant, but smaller animals had all but disappeared (Barrett *et al.* 2003). It is possible that the increased size and abundances of predators (ie fish, rock lobsters) has led to greater predation on abalone, or that the presence of these predators has led to a change of behaviour where juvenile abalone emerge from crevices at a larger size (Barrett *et al.* 2003). There was also a trend for decreasing urchin numbers in the Maria Island reserve, consistent with increasing numbers of predators, however this decrease in urchins was not reflected by any change in algal cover (Barrett *et al.* 2003).

In 1995, an interim marine protected area was declared at a group of small seamounts south of Tasmania in which the fishing industry agreed not to trawl for a 3 year period to allow time for research, and the establishment of longer-term management measures. Established to protect the unique and vulnerable benthic communities of seamounts, the Tasmanian Seamounts marine park was established in 1999. Due to the topographically-enhanced currents around their vicinity, seamounts provide a unique deep-sea environment (Roden 1986). The discovery of substantial aggregations of pelagic armourhead (*Pseudopentaceros wheeleri*), *Sebastes* spp., orange roughy (*Hoplostethus atlanticus*) and oreosomatids in these areas has led to increased targeting on seamounts by trawlers throughout the world's oceans (Koslow *et al.* 2001).

To assess the impact of trawling for orange roughy (*Hoplostethus atlanticus*, Trachichthyidae) and the efficacy of the marine protected area, Koslow *et al.* (2001) surveyed the benthic macrofauna of a group of small seamounts south of Tasmania using a dredge and camera. The study examined the influence of depth, as well as the impact of trawling, on the composition of the benthic macrofauna. For an outline of the research methods employed by Koslow *et al.* (2001) see Table 34.

Results revealed that intact coral cover (living or dead) was only found on the un-fished or very lightly fished seamounts within the protected area. There was, however, a high proportion of dead coral aggregate even on un-fished seamounts, the cause of which is not known. Both living and dead aggregates of the dominant coral *(Solenosmilia variabilis)* were most abundant at mid-depths along the slopes of two of the four seamounts (Koslow *et al.* 2001). Maximal cover by combined aggregations of living and dead coral was 27% at 1800 m, and 63% at 1500 m, and decreased to ≤10% both toward the summit and base of the seamounts (Koslow *et al.* 2001). The substrate of the heavily fished seamounts differed markedly from those in the protected area. The substrate of the most heavily fished seamount in the area was predominantly bare rock (>90% at most depths), and the coral material in these areas was either rubble or sand (Koslow *et al.* 2001).

The results of this research reveal the impact of trawling on complex seamount reefs, with the coral substrate and associated community largely removed from the most heavily fished seamounts. The virtual complete loss of this community from the shallow, heavily fished seamounts is not surprising, given the relatively small size of these seamounts, and the hundreds to thousands of trawls carried out on each (Koslow *et al.* 2001). The substrate of heavily fished seamounts in the area now consists predominantly of either bare rock or coral rubble and sand, features not seen on any seamount that was lightly fished or un-fished. The abundance and species richness of the benthic fauna on heavily fished seamounts was also markedly reduced (Koslow *et al.* 2001). Koslow *et al.* (2001) found that most species were widely distributed over the depth range that was sampled, but decreased in abundance at the shallowest and deepest seamounts sampled. The fauna unique to the seamounts appears, however, to be adequately represented within the depth range of seamounts found within the established protected area.

Management goals, monitoring objectives, indicators and resultant performance measures for Tasmanian Marine Reserves are summarised in Table 35.

Table 34. Monitoring for spatial management in Tasmania, Australia

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Australia, Tasmania - Marine Park Monitoring Program *(Edgar and Barrett 1997)* *(Edgar and Barrett 2002)*	Monitoring was established in 1992 at the Maria Island, Tinderbox, Ninepin Point and Governor Island marine reserves. Sampling sites were located within reserves and outside of reserves. Generally, sites are sampled once a year in Autumn.	Fixed transects at sites within each area (4 x 50m contiguous transects). Underwater visual census, generally along 5 m isobath. Fish density and estimated size class was recorded within 5 m either side of transect. Data recorded separately for males, females and juveniles (for species where sex is easily determined). Cryptic fishes and megafaunal invertebrates were counted within 1 m either side of transect (see above). Maximum lengths of abalone and carapace length of rock lobsters were measured. Area covered by macroalgal species is recorded in 0.5 x 0.5 m quadrats at 10 m intervals along a transect line. A method using point intersect (50 points) was used to estimate cover.
Australia, Tasmania - (Koslow et al. 2001)	The aim was to assess the impact of trawling for orange roughy (*Hoplostethus atlanticus*; Trachichthyidae) and the efficacy of a marine protected area, surveying the benthic macrofauna of a group of small seamounts south of Tasmainia. This study examined the influence of depth, as well as the impact of trawling, on the composition of the benthic macrofauna.	The survey was carried out during Jan and Feb 1997. Seamounts were selected to cover as wide a range of depths and fishing effort as possible. Fishing effort was assessed from fishermen's logbook records. Four seamounts covering a range of depths (peaks from 714 to 1580 m depth) and fishing effort (0 to >3000 trawls) were surveyed photographically. Two or 3 photographic transects were carried out on each seamount from the pinnacle to the base. The first 2 transects were oriented orthogonally in east-west and north-south directions. On two of the seamounts a third transect was carried out that approximately replicated a previous transect. Mean transect length was 2652 m (range: 2213 to 3310 m). Photos were generally taken with the camera between 1 and 4 m off the bottom, with the camera facing down. The mean number of photos per transect was 106 (range: 67 to 150). The mean distance between photos on a transect was 27 m (range: 16 to 38 m). The photographs were assessed for percent cover by bottom type (i.e. living and dead coral aggregate, coral rubble, coral sand, mud/silt, rock, and barnacle plates) and for the numbers of each type of non-colonial organism. The benthic fauna was sampled with a Lewis (1999) dredge with a mouth area of 0.72 m^2 (1.2 m wide × 0.6 m high) and a stretched maximum mesh width of 25 mm in the cod end. Seamounts with peaks from 660 to 1700 m below the surface were sampled with the dredge. In all, there were 34 successful dredge samples obtained from 14 seamounts, 6 of which were in the protected area. The dredge samples were sorted at sea into major taxonomic groups (e.g. sponges, crinoids, sea stars, colonial corals, solitary corals, black corals, gold corals, hydroids), weighed by group, and preserved. Most major groups sampled by the gear were identified to species. Drop-lines and traps were deployed for 2 to 7 h at four seamounts to sample the motile fauna (fishes, crustaceans etc.) living in association with the benthic environment. Three trap-types were deployed: fish traps (overall: 86 × 85 × 120 cm, 18 mm square mesh, 2 openings, mouth area ~500 cm^2), crab pots (overall: 70 × 88 × 30 cm; 20 × 40 mm mesh; 2 openings, mouth area 289 cm^2) and small tapered prawn traps (35 cm length × 10 cm diam. at mouth opening, mesh 0.5 cm diam.) to sample small scavengers such as amphipods. About 100 hooks per drop-line were deployed, and the lines were set so the first hook was 2 m above bottom, with subsequent hooks at 1 m intervals. Two hook sizes were used on each line.

Table 35. Management goals, monitoring objectives, indicators and performance measures for spatial management in Tasmania, Australia

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Conservation of representative and unique marine habitats, to provide reference locations where the dynamics of marine communities can be observed independently of fishing effects, and the creation of fish propagation areas.	Assess whether creation of reserves has affected biodiversity, or the abundance or size distribution of commercially important fishery species, or whether indirect broad-scale habitat shifts have occurred	Density	Total (all fish species)	Density, size distribution and habitat cover trends	*(Edgar and Barrett 1997)* *(Edgar and Barrett 2002)*
		Size Distribution			
		Density	Species (cryptic fish and selected megafaunal species		
		Size Distribution			
		Benthic Percent Cover	Habitat type (algal species)		
Protection of the unique and vulnerable benthic communities of seamounts	Assess the impact of trawling for orange roughy (*Hoplostethus atlanticus*, Trachichthyidae) and the efficacy of the marine protected area at a group of small seamounts	Density	Total (All benthic macrofauna)	Trends in species density and biomass with habitat cover trends. Species representation within established marine park.	*(Koslow et al. 2001)*
		Biomass			
		Benthic Percent Cover	Habitat Type		

New South Wales

There are four marine parks in New South Wales (NSW); Jervis Bay Marine Park, Solitary Islands Marine Park, Lord Howe Island Marine Park, and Cape Byron Marine Park. Each of these marine parks is zoned into specific areas: sanctuary zones, habitat protection zones, general use zones and special purpose zones. The primary goal of establishing marine parks in NSW was to create a comprehensive, adequate and representative system to protect marine biodiversity and maintain ecological processes. To achieve its objectives, the NSW Marine Parks Authority is developing a bioregional system of marine parks, where ecologically sustainable use and public recreation may occur if consistent with conservation objectives (Marine Parks Authority 2001).

Ecosystem Effects

Within Reserve Effects

The marine park at Lord Howe Island and Balls Pyramid was established in 2000 and contains a variety of habitats with recognised international significance, including the world's southern-most barrier coral reef and associated lagoon. These marine environments are very distinct from that of the coast and continental shelf of NSW. The primary objective of the park is to protect the seamount system and its conservation values associated with marine biodiversity, habitats and ecological processes (Environment Australia 2002).

Many of the habitats at Lord Howe Island and Balls Pyramid are poorly studied, however, a series of management actions have been designed to acquire a better understanding of these habitats, particularly regarding their resilience and vulnerability to impact. The primary objective for research within the Lord Howe Island marine park is to ensure that management actions are effective. The primary focus during the first five years of management is to survey biodiversity and assess the impacts of existing activities (Environment Australia 2002).

In 2004, the Australian Institute of Marine Science (AIMS) undertook surveys of benthic habitat and fish fauna in the deeper waters around Lord Howe Island and Balls Pyramid, below the depth limits of SCUBA observations (30-200m) where little information was previously available (Speare *et al.* 2004). A towed underwater-camera array was used to obtain video and still images of habitats and epibenthos, and baited remote underwater video stations (BRUVS) were used as a non-extractive method to sample fish and shark fauna. This preliminary, rapid ecological assessment aimed to describe habitats and fish-habitat associations in these deeper waters (Speare *et al.* 2004). Local knowledge from resident fishermen was employed to target areas of interest in terms of bathymetry and fish fauna. For an outline of the research methods employed by Speare *et al.* (2004) see Table 36. Management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 37.

Speare *et al. (2004)* found that Lord Howe Island is surrounded by a significant and extensive fossil coral reef that supports low, sparse stands of brown and green algae. This substratum extended to around 45m depth onto a sandy seafloor devoid of epibenthic structure. The seafloor in deeper waters of the outer-shelf, in depths of 60-100m and below, was predominantly unconsolidated sand that supported communities dominated by gorgonians where rubble, stone or bedrock was exposed and allowed their attachment (Speare *et al.* 2004). The steep shelf slopes had flows of finer, silty sediments between bedrock outcroppings.

The outcrops, walls and overhangs extended down to 200 m and were inhabited by numerous fish (Speare *et al.* 2004). Surveys that were conducted in this habitat, below the limit of sunlight penetration, produced the most sightings of fish. The very high wave and current energies on the rises have flattened the fossil reef, and the topographically complex habitats normally known to support diverse and abundant faunas of filter-feeding invertebrates and fish were most common on the steep shelf drop-off zones (Speare *et al.* 2004). These observations were in accordance with reports from local fishermen (Speare *et al.* 2004).

Socio-economic Effects

After the establishment of the Jervis Bay Marine Park (JBMP) in 1998, the New South Wales Government prepared a draft zoning plan that incorporated data on two conflicting groups; scuba divers and fishers (anglers) that use the park. While conflict resolution was a priority, other factors such as cumulative impacts of users and the protection of the critically-endangered grey nurse shark (*Carcharias taurus*), further complicated planning (Lynch *et al.* 2004).

Research shows that both scuba divers and recreational fishers (anglers) are important recreational users of JBMP. The headlands of the bay appear to be particularly popular sites for marine recreation (Williams *et al.* 1993). Scuba diving has often been perceived as an activity compatible with conservation, however, there are concerns that some heavily dived sites may have visitation rates exceeding the limits of ecological sustainability (Hawkins and Roberts 1996; Treeck and Schuhmacher 1998). While fishing is one of the main activities that is affected by the establishment of MPAs, few studies have investigated the distribution of anglers prior to zoning or the impact of MPA zoning on anglers.

To assess the usefulness of social data in the development of MPA zoning options, Lynch *et al.* (2004) carried out surveys based on both biological information, in particular the protection of an endangered species, and also on the distribution and potential environmental impacts of user groups. The data was used to assess the feasibility of developing a zoning option based on this information. To provide a wider perspective, a comparison between social data collected a decade earlier (Williams *et al.* 1993) and data collected during this study was also undertaken. For an outline of the research methods employed by Lynch *et al.* (2004) see Table 38. Socio-economic management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 39.

Lynch *et al.* (2004) found that the number of logged dives did not increase significantly when the two study periods of 1989–1990 and 1996–1999 were compared (Lynch *et al.* 2004). While dive effort may not have obviously changed between study periods, there was still a considerable diving pressure of approximately 10,000 dives per year, that was mostly concentrated in two small areas (Lynch *et al.* 2004). Although there were 78 recognised dive sites throughout the park, 81% of the diving pressure was concentrated into 16 closely-spaced sites around the two headlands (Lynch *et al.* 2004). Williams *et al.* (1993) also found this general spatial distribution, which suggested that cumulative impact may have a long history at these sites (Lynch *et al.* 2004). In 1999 and 2000 the number of anglers observed in Jervis Bay had doubled during February and July and tripled during April compared to angler numbers observed in 1989 and 1990 (Lynch *et al.* 2004).

It may be possible to control commercial operator entry into parks using a permit system; however, this option may not be economically viable for commercial operators in temperate seas (Lynch *et al.* 2004). Limiting visitor numbers to some sites could also result in the spread of damage to previously un-impacted areas (Lynch *et al.* 2004). Another suggested option is to accept high use at some sights and enhance the sustainability of these areas by establishing structures such as moorings (Lynch *et al.* 2004). The strategy would be to focus both public and park resources into a few already popular areas where environmental degradation could be managed (Lynch *et al.* 2004). Education was also highlighted as a method of reducing environmental impacts, and implementation of an integrated visitor education/interpretation program that uses signage, zoning pamphlets, a user guide, codes of conduct, and ranger patrolling, is currently being prepared (Lynch *et al.* 2004).

Table 36. Monitoring for spatial management in New South Wales, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Lord Howe Island *(Speare et al. 2004)*	A rapid ecological assessment was carried out focusing on representing depth strata between 30 and 200 m. Local knowledge from resident fishermen was employed to target areas of bathymetric and piscatorial interest. Towed video, Baited Remote Underwater Video Stations (BRUVS) and Remotely Operated Vehicle (ROV) were used.	Colour video camera was mounted on a vane and controlled by a winch with 320 m of electro-mechanical cable. Visual imagery of the benthos was recorded. Underwater lights illuminated the field of view. The tape recorder also received GPS data (position, ground speed, true heading, date and time). A computer-based application allowing keyboard classification of substrata, benthos and individual organisms was interfaced with a GPS to facilitate real-time geo-referencing of all data points. The depth and the ship's track were recorded. Average speed over the ground was 1.5 knots. A position resolution of 6 m was achieved. A still camera with external strobe was mounted to the towed body and set to record a still image at 1 minute intervals. This gear was deployed at 21 stations. Each tow was described by the percentage of records assigned to each class of substratum and overlying epi-benthos, and the occurrence of individually marked organisms. Baited remote underwater video stations (BRUVS) were deployed to provide 90 minutes of film recorded at the seabed and were set ~ 300 m apart to provide independence of each replicate unit (Cappo *et al.* 2003; Cappo *et al.* 2004). Interrogation of each tape provided the time the BRUVS settled on the seabed and, for each species, the time of first sighting, time of first feeding at the bait, the maximum number seen together at any one time on the whole tape, and the intra- and inter-specific behaviour. With stationary video, species abundance estimates are limited to the maximum number of individuals sighted in any one field of view, as re-appearances of a single fish image cannot generally be assigned to a specific individual. The use of this method of analysis was reviewed by Cappo *et al.* (2003; 2004) as a conservative measure of relative abundance. Unfavorable sea conditions severely hampered the use of remotely operated vehicles (ROV) for this rapid ecological assessment; however, they were opportunistically deployed around Lord Howe to provide detailed imagery of the benthos for species identification.

Table 37. Management goals, monitoring objectives, indicators and performance measures for spatial management in New South Wales, Australia

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Lord Howe Island: Conserve biodiversity, representative samples of marine ecosystems and habitats, rare or threatened species, and other areas of high conservation value	Survey biodiversity and assess the impacts of existing activities	Percentage of towed video records	Class of substratum and overlying epibenthos	Spatial trends in measured variables	*(Speare et al. 2004)*
		Number	Individual benthic organisms (species)		
		Maximum number of individuals in one field of view (BRUVS)	Fish species		

Table 38. Socio-economic monitoring for spatial management in New South Wales, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Jervis Bay *(Lynch et al. 2004)*	The aim of this monitoring was to develop a zoning option based both on biological information—in particular the protection of an endangered species—and also on the distribution and potential environmental impacts of user groups	Logbooks from the two land-based commercial dive companies were analysed from Jan 1996 to Feb 1999 (37 months). Data were also provided from two "live-aboard" dive vessels. Data extracted from logbooks included: month, number of dive trips, number of individual dives, and location by grid area. Scuba diving sites were categorised into subdivisions based on their size and the location of natural features, such as headlands and beaches. The number of dives within each of these subdivisions was then extracted from the logbook data. The number of dives and dive trips per month were used to compare this study and a previous study (Williams *et al.* 1993). The study area was surveyed by circumnavigating in a powerboat, at a consistent speed and along a defined route, while counting all anglers in front of the survey vessel. A random, stratified sampling design was employed. As in the previous study by Williams *et al.* (1993), the bay was divided into a study grid, although not all grids were surveyed (10 were surveyed). Three sampling months, February, April, and July, were chosen for sampling in 1999 and 2000, as they coincided with months surveyed in a previous study (Williams *et al.* 1993) and coincided with known peaks and troughs in fishing effort. A total of 20 counts were undertaken each month, with 12 counts occurring on weekdays and 8 on weekends. Public holidays were considered as weekends. Commencement of sampling was randomly selected without replacement from four start times (06: 30, 09:30, 12:30, and 15:30 hours). Angling techniques were categorised. Position coordinates of boat and shore angler parties were gathered using detailed notes and a GPS.

Table 39. Socio-economic management goals, monitoring objectives, indicators and performance measures for spatial management in New South Wales, Australia

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	***Scale***		
Jervis Bay *Conserve biodiversity, representative samples of marine ecosystems and habitats, rare or threatened species, and other areas of high conservation value*	The objective of this study was to test the utility of social data in developing MPA zoning options.	Number of individual dives and dive trips per month	Location in bay	Temporal change between 1998-1990 and 1996-1999	*(Lynch et al. 2004)*
		Number of anglers	Location in bay gear type	Temporal change between 1989-1990 and 1999-2000	

Victoria

A representative system of Marine National Parks and Sanctuaries was established in Victoria in 2002 with the role of protecting and conserving representative examples of marine and coastal biodiversity, ecological processes and important natural features. Thirteen marine national parks and eleven marine sanctuaries were established that together covered nearly 54 000 hectares or 5.3 % of Victoria's marine waters.

Ecosystem Effects

Within Reserve Effects

As part of the implementation the marine national parks and sanctuaries, the Subtidal Reef Monitoring Program (SRMP) was initiated in 1998, and the Intertidal Reef Monitoring Program (IRMP) was initiated in 2003, with the primary objective of providing information on the status of Victorian reef flora and fauna (Gilmour *et al.* 2005). The monitoring was focused on macroalgae, macroinvertebrates and fish. This included monitoring of the nature and magnitude of trends in species abundances, diversity and community structure (Gilmour *et al.* 2005). Specifically, the program was designed to allow managers to: 1) compare changes in the status of species populations and biological communities between highly protected marine national parks and sanctuaries, and other Victorian reef areas; 2) determine associations between species and between species and environmental parameters; 3) provide benchmarks for assessing the effectiveness of management actions; and 4) determine the responses of species and communities to unforeseen and unpredictable events (Gilmour *et al.* 2005). For an outline of the research methods employed by the SRMP and the IRMP (Hart *et al.* 2004; Gilmour *et al.* 2005) see Table 40.

Individual areas have been assessed periodically in the SRMP (Gilmour *et al.* 2006; Lindsay and Edmunds 2006b; Lindsay and Edmunds 2006a; Lindsay *et al.* 2006) and the IRMP (Hart *et al.* 2005) in terms of spatial distributions of species (abundance, size, cover) and assemblages. Trends in these variables at individual sampling sites have also been assessed periodically over the monitoring programs. The objectives of these assessments were: to describe the general progress of the monitoring program; to provide general descriptions of the biological communities and species populations at each monitoring site; and to identify any unusual biological phenomena, such as the presence of introduced species (Gilmour *et al.* 2005).

Initiated in 2003, the Victorian Abalone Assessment Monitoring Program is a comprehensive fishery independent monitoring program (Jenkins *et al.* 2005). The monitoring program was established to collect scientific information that will be incorporated into the development and application of the length-based abalone fishery assessment model developed at the Marine and Freshwater Fisheries Research Institute, Queenscliff. Essentially, initial application of the model highlighted the need to obtain new information from more abalone populations than had previously been studied. The collection of data will aid in the development of a model that is specifically suited to the unique characteristics of abalone population dynamics. The biology of abalone varies markedly between different populations and significant differences can be found between populations that are not far apart. For the model to provide reliable assessments it is important to have more information on how such characteristics of abalone populations vary across the State. For an outline of the research methods employed by the Victorian Abalone Assessment Monitoring Program see Table 40.

In addition to collecting information on abalone, transect counts are also made of sea urchins, *Heliocidaris erythrogramma* and *Centrostephanus rogersii*, sea stars, *Coscinasterias muricata*, and turban snails, *Turbo undulatus* (Jenkins *et al.* 2005). A qualitative estimate of cover was also made for major algal divisions, kelp canopy, and crustose coralline algae (Jenkins *et al.* 2005). These groups include important abalone competitors (sea urchins and turban snails), predators (starfish), and habitat (crustose coralline algae). One of the objectives of the Victorian Abalone Management Plan (Anon. 2002) is that ecosystem health is not jeopardised by abalone fishery practices. The plan sets a (lower) limit or trigger reference point of 90% of the average value over the past three years for "ecosystem health indices" which includes abalone predators, competitors and habitat that might be influenced by abalone fishing (Table 41).

Table 40. Monitoring for spatial management in Victoria, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
Victoria: Subtidal and Intertidal Reef Monitoring Program *(Hart et al. 2004)* *(Gilmour et al. 2005)*	The Subtidal Reef Monitoring Program (SRMP) was established in 1998. Underwater visual census techniques were used to obtain the necessary quantitative data for the monitoring program. Sites were established in the vicinity of marine protected areas within Port Phillip Bay (including Port Phillip Heads), Phillip Island, Bunurong, Wilsons Promontory, and along the western and far eastern Victorian coastline. At the beginning of the SRMP the sampling frequency was every six months. After at least four surveys at each of the original four locations, the sampling frequency was changed to once a year. Details of the sampling locations are provided in Appendix 1. Details of the sampling frequency are provided in Appendix 2.	Fixed transects at sites within each area (4 x 50m contiguous transects per site). Underwater visual census, generally along the 5 m isobath. Fish number and estimated size-class are recorded within 5 m either side of transect. Data recorded separately for males, females and juveniles (for species where sex is easily determined). Cryptic fishes and benthic megafaunal invertebrates are counted within 1 m either side of transect (see above). Carapace length of rock lobster and maximum length of abalone measured where possible. Up to 36 abalone individuals are measured for each transect. Percent cover by macroalgal species is recorded in 0.5 x 0.5 m quadrats at 10 m intervals along transect (see above). Point intersect method is used with 50 points to estimate cover. The density of macrocystis plants recorded within 5 m either side of transect (see above). For selected sites (sea urchin barrens of eastern Victoria), selected grazers counted in 0.5 x 0.5 m quadrat at 10 m intervals along transect (eg. urchins, limpets).
	The Intertidal Reef Monitoring Program (IRMP) was established in 2003. Sites were established within Port Phillip Bay (including Port Phillip Heads – Point Lonsdale), Point Addis and Bunurong Marine National Parks, and Point Danger, Barwon Heads and Mushroom Reef Marine Sanctuaries. A reference site was also established in each area.	Five fixed transects up to 200 m long, each running from high to low shore, are spread evenly across the intertidal area. Five fixed sampling locations (2 x 2 m area) are located along each transect. A 0.5 x 0.5 m quadrat is randomly positioned within each sampling location Mobile invertebrates within the quadrat are counted and the shell length of 50 – 100 individuals of abundant gastropod species is measured Percentage cover of macroalgae and sessile invertebrates is measured using point intersection based on 50 points within the quadrat. When present, maximum frond lengths of 50 – 100 *Hormosira banksii* plants are measured. For each quadrat, a digital photo is taken when practical, and substratum complexity is assigned to one of 5, qualitative categories
Victorian Abalone Assessment Monitoring Program *(Gorfine et al. 1997; Jenkins et al. 2005)*	Annual surveys of abalone communities at approximately 250 fixed locations along the coastline. Two annual surveys undertaken to date. Some sites are within Marine National Parks. Details of sampling locations are given in Appendix 3.	Underwater visual census of abalone (juvenile, pre-recruits, adults) and of the major invertebrate species associated with them (eg. selected species of sea urchins, seastars, and gastropods). Six, 30m radial transects starting at a common central point. Species are recorded within a 1 m band transect. Density (separated into juvenile, pre-recruit, adults for abalone) and in some cases length is measured (for abalone). Qualitative estimates of cover of some algal groups are also recorded.

Table 41. Management goals, monitoring objectives, indicators and performance measures for spatial management in Victoria, Australia

Location and Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
Victoria: Protection and conservation of representative examples of marine and coastal biodiversity, ecological processes and important natural features	To provide information on the status of reef flora and fauna (focusing on macroalgae, macroinvertebrates and fish)	Density Size distribution	Total (all fish , benthic mega-faunal invertebrate species)	The nature and magnitude of temporal trends in density, size distribution, species richness, assemblage composition and habitat cover	*(Hart et al. 2004; Gilmour et al. 2005)*
		Benthic percent cover	Habitat type (algal species)		
		Density	Family (*Macrocystis* species)		
		Density	Species (sea urchin species)		
		Species richness/assemblage composition	Total (all fish , benthic mega-faunal invertebrate species)		
Victoria: ecosystem health is not jeopardised by abalone fishery practices	To provide information on abalone predators, competitors and habitat that may be considered as "indicators of ecosystem health"	Density	Species (selected urchins, seastars, gastropods)	Temporal trends in density: (lower) limit or trigger reference point of 90% of the average value over the past three years for "ecosystem health indices"	*(Anon. 2002)*
		Percentage cover	Macroalgal groups, kelp canopy, crustose coralline algae		

The Great Australian Bight

The Great Australian Bight Marine Park (GABMP) was established in 1998 and is one of the largest MPAs in Australia. The park was created to protect habitat for species of conservation significance, particularly the southern right whale (*Eubalaena australis*) and the Australian sea-lion (*Neophoca cinerea*), and the ecological communities and sediments of the seabed.

Ecosystem Effects

Within Reserve Effects

The benthic protection zone (BPZ) of the GABMP was proclaimed in 1998. The BPZ consists of a 20 nautical-mile-wide strip extending from 3 nautical miles from the coast to the edge of Australia's Exclusive Economic Zone (EEZ), 200 nautical miles offshore (Ward *et al.* 2006). The objectives of the BPZ are to: 1) protect the ecological integrity of a large, representative sample of the GAB's unique and diverse benthic flora and fauna; and 2) provide an undisturbed 'sample' of the GAB's benthic habitat that can be used as a reference area for comparison with neighbouring zones that may have been disturbed by trawling or mineral exploration (McLeay *et al.* 2003).

To assess the effectiveness of the BPZ in representing regional biodiversity, Ward *et al*. (2006) conducted the first quantitative survey of the epibenthic assemblages of the continental shelf in the eastern GAB. The objectives of the study were to: 1) identify the epifaunal macro-invertebrates of the area; (2) determine environmental factors (e.g. depth and sediment composition) that might be associated with the distribution patterns of the benthic assemblages; 3) assess the suitability of the BPZ for representing the benthic assemblages of the GAB; and 4) outline a rationale and approach for future performance assessment of the BPZ. For an outline of the research methods employed by Ward *et al.* (2006) see Table 42. Management goals, monitoring objectives, indicators and resultant performance measures are summarised in Table 43.

Bathymetric data showed that throughout much of the eastern GAB the sea floor drops to a depth of 40 m within a few kilometres of the coast. Offshore and south from the Head of the Bight, the seafloor is comparatively flat and featureless, and slopes gently for over 260 km before reaching the shelf edge (Ward *et al.* 2006). Towards the east, the shelf topography is more complex, particularly through the inner-shelf waters (<100 m depth) between Point Bell and Cape Catastrophe. Ward *et al.* (2006) found that water depth plays an important role in determining the distribution of sediment grain-size structure on the shelf, and textural patterns were found to be broadly consistent with bathymetric features. The sediments were generally coarsest in shallow inshore waters and became progressively finer with increasing depth and distance offshore (Ward *et al.* 2006).

Results suggested that the eastern GAB supports one of the world's most diverse soft-sediment ecosystems with a total of 797 species and 10 phyla collected during the surveys (Ward *et al.* 2006). Sessile, suspension-feeding organisms (primarily poriferans, ascidians and bryozoans) dominated the samples, and collectively comprised over 96% of the biomass and 74% of the species collected (Ward *et al.* 2006).

Ward *et al.* (2006) found that there was a significant positive correlation between species richness and biomass and that there was a general decline in both parameters with increasing depth and percentage mud in the sediments. The highest biomass (440 kg tow^{-1}) was found in the inner-shelf waters off the Eyre Peninsula, and in waters near the Head of the Bight (Ward *et al.* 2006). Biomass gradually declined between these two regions and decreased offshore. Similarly, species richness was high (480 spp. tow^{-1}) in near-shore waters off the Eyre Peninsula and at the Head of the Bight (Ward *et al.* 2006). An area of low-diversity extended across the shelf between these two regions, and included most of the central part of the study area.

Six epifaunal assemblages were identified and each of the assemblages was correlated with depth and sediment type (Ward *et al.* 2006). All six assemblages and 54% of the species collected during the surveys were found in the BPZ. These results suggest that the BPZ may effectively represent the epifaunal assemblages of the continental shelf in the eastern GAB (Ward *et al.* 2006). However, this study was confined to shelf assemblages of the eastern GAB, and it is not known whether the BPZ effectively represents and preserves the benthic habitats and assemblages of the western GAB and the continental slope (Ward *et al.* 2006).

By comparing the benthic assemblages of the BPZ and adjacent areas, Ward *et al.* (2006) suggest that they have effectively completed the first stage of a performance assessment. Further, Ward *et al.* (2006) conclude that to provide a basis for measuring both changes in the benthic assemblages within the BPZ over time and the difference between temporal changes that occur inside and outside the BPZ, any future survey must (at least) include re-sampling the same 40 stations that were sampled during their study using the same sampling method (i.e. the epibenthic sled). Although the data obtained by simply re-sampling these 40 stations with the epibenthic sled may provide the basis for completing a preliminary performance assessment of the BPZ, additional information in the form of a power analysis may be necessary to determine if this will be sufficient. For an ongoing system for assessing the performance of the BPZ to be successful, Ward *et al.* (2006) suggest it is essential that analysis of high-quality data on all potentially deleterious anthropogenic activities be conducted; including fishing and mining that occur in the region.

Table 42. Monitoring for spatial management in the Great Australian Bight, Australia.

Monitoring Location and Citation Details	*Monitoring Outline*	*Methods*
The Great Australian Bight *(Ward et al. 2006)*	To assess the effectiveness of the Benthic Protection Zone (BPZ) of the GAB marine park in representing regional biodiversity, quantitative surveys of the epibenthic assemblages of the continental shelf in the eastern GAB were undertaken. The objectives of the study were to: 1) Identify the epifaunal macro-invertebrates of the eastern GAB; (2) Determine environmental factors (e.g. depth and sediment composition) that might be associated with the distribution patterns of the benthic assemblages; 3) assess the suitability of the BPZ for representing the benthic assemblages of the GAB; and, 4) outline a rationale and approach for future performance assessment of the BPZ.	Epibenthos was collected from 85 sites during Apr 2002 (25 sites) and Nov-Dec 2002 (40 sites). To provide a basis for assessing the utility of existing sedimentary data as a predictor of biological assemblages, sampling sites were stratified among nine sedimentary facies recognised for the region (James *et al.* 2001). Five sites, separated by less than 125 km, were sampled within each of the nine sedimentary facies outside the BPZ. A further five sites were sampled in each of four sedimentary facies represented in the BPZ. Samples of the epifauna at each site were collected using a 1.8 m wide by 0.6 m high benthic sled fitted with a 50 mm mesh bag. At each site, the sled was towed across the substrate for 5 min at a speed of 3.5 knots. Samples were later sorted and identified to species or putative taxon. To provide a basis for assessing trophic differences between study sites, taxa were placed into five feeding guilds (suspension- feeders, scavengers, predators, deposit-feeders and grazers) according to their primary-feeding mode (Edgar 2000). Surface sediment samples were collected at each site using a small bucket dredge (20 cm x 70 cm). The dredge was deployed at the end of each sled shot while the vessel drifted. Samples were sieved through an agitated stack of Endecott sieves and the amount of mud (<63 μm) present was determined as a percentage of the total mass sampled. This parameter, together with the mean grain-size and sorting coefficient, were subsequently used to investigate relationships between epifaunal composition and sediment structure. Bathymetric data for the region were obtained from Geoscience Australia.

Table 43. Management goals, monitoring objectives, indicators and performance measures for spatial management in Great Australian Bight, Australia

Location and Proposed Management Objective	*Monitoring Objective*	*Indicator*		*Performance Measures*	*Reference*
		Variable	*Scale*		
The Great Australian Bight: *Protection of habitat for species of conservation significance, particularly the southern right whale (Eubalaena australis) and the Australian Sea-lion (Neophoca cinerea), and the ecological communities and sediments*	Assess the effectiveness of the BPZ (Benthic Protection Zone) of the GAB marine park in representing regional biodiversity	Density	Assemblage (species or putative taxon) Species richness Taxonomic affinity (%) Feeding guilds Mobility	Spatial trends in biomass, community structure Correlation with sediment structure/environmental variables Comparison of assemblages and species within BPZ with broader eastern GAB for representativeness	*(Ward et al. 2006)*
		Biomass	Species Phylum		
		Sediment structure	Amount of mud and sediment grain size		

Implications for the development of performance measures for marine spatial management

Most examples of monitoring for spatial management relate to monitoring of marine protected areas and reserves. These areas represent the most conspicuous form of spatial management in the past few decades and a considerable volume of literature has accumulated related to scientific sampling and monitoring. The management objectives associated with MPAs tend to be very general and poorly defined, and as such designing monitoring programs that effectively measure performance against management objectives becomes difficult. In fact, there appear to be few examples where monitoring programs have been instituted specifically to measure the performance of MPAs relative to management objectives.

The origin of many studies of MPAs is curiosity-based science rather than as a planned component of a management strategy. Such studies are often relatively short-term and may simply represent a comparison of sites within and outside MPAs. Such studies are of little value in determining the effects of MPAs because differences found between reference and protected sites may have pre-existed before protection. Areas selected for protection are often relatively unrepresentative by time compromises are sought amongst stakeholders with conflicting views (Edgar *et al.* 2004b), or indeed reserve areas may have been consciously selected for their unique characteristics of biodiversity or other factors. In many cases, also, sites within a single MPA may be compared with adjacent reference sites, providing the results with little generality in a spatial context. Ideally, to assess the performance of MPAs there is a requirement for monitoring of reserve and reference sites to be undertaken a number of years before protection as well as after protection. This ensures that the existing conditions prior to protection are used as a baseline for any changes that occur post-protection. In practice, it is the interaction between time (pre- and post protection) and space (reserve versus non reserve) that allows the effect of management (MPA declaration) and natural variability to be separated. The monitoring should also be applied to multiple MPAs and associated reference sites to allow maximum generality.

In typical monitoring/sampling programs associated with MPAs the performance measure is the statistical significance of comparisons between protected and reference areas, or ideally the interaction between this comparison and time (pre-/post-protection as mentioned above). As such, these measures are highly dependent on statistical power that is, in turn, a function of sampling design, levels of replication, and the variable or metric selected. A range of metrics have been used in monitoring studies for spatial management, the chosen metric to some extent depending on the management objective under consideration (Table 44). Pilot sampling can be used to assess levels of natural variability and to determine required sample sizes based on a pre-determined effect size. This, however, requires management to set the 'a priori' effect size against which management performance will be measured. This type of approach has rarely, if ever, been used in monitoring of MPAs to measure management performance.

The scale at which MPA management objectives are set, that directly influences the scale at which monitoring programs are targeted, tend to be relatively localised. For example, the type of monitoring discussed above comparing MPAs with reference sites tends to be focussed on objectives such as the rebuilding of depleted fishery stocks. Sometimes, MPA management objectives include the effect of "spillover" where populations within MPAs replenish fishery stocks outside MPAs, either through export of eggs and larvae or movement of juveniles and adults. This broadens the scale of monitoring to both the protected area and (usually) the area adjacent to the MPA. The monitoring required to measure the performance of this objective is less straightforward than for within MPA objectives, and has usually been addressed by a relatively piece-meal approach where a number of lines of evidence are drawn together (i.e. biological monitoring of population density/size, spatial and temporal trends in effort and catch by fishermen). Even here, the scale at which management objectives and associated monitoring is set is relatively small. Management objectives for MPA and associated monitoring have rarely, so far, addressed broader, regional-scale aspects of MPA performance. For example, although spillover may result in an enhanced catch adjacent to MPAs, does this compensate for the reduced fishing area available to the fishery in the regional context (McClanahan and Mangi 2000). This is the scale at which performance needs to be measured in terms of management objectives related to rebuilding fish stocks and enhancing fishing.

The statistics-based performance measures usually associated with MPA management and monitoring are different to the performance measures typically employed in fisheries management (Sainsbury and Rashid Sumaila 2003). In the latter case, the temporal trend in stock abundance or biomass is usually related to a target reference point, which is the level that management would like to see the stock maintained, and a trigger reference point, which is a lower limit that will trigger management action when reached. This type of approach has been used in a few cases of marine spatial monitoring that is not strictly related to MPAs, but rather, is related to monitoring the ecological integrity of the ecosystem. This monitoring is designed to detect environmental or man-induced perturbations to the ecosystem. In the case of the Great Barrier Reef Monitoring Program, the primary aim is to detect the effects of factors such as crown of thorns starfish outbreaks and coral bleaching. In this case, the limit reference is a temporal trend of greater than 10% annual change. In the Victorian abalone monitoring program the perturbation being monitored is the effect of abalone fishing on the environment and the limit reference is an annual change such that key species are below 90% of the average density for the previous 3 years. Unfortunately, these limit reference points are somewhat arbitrary; however, they do provide a quantifiable basis for the triggering of management action. In the case of the effects of fishing, in the future MPAs may be useful in setting target reference points, however, once again the 'a priori' setting of a limit reference point (effect size) will to some extent depend on a subjective assessment by managers.

Social and economic goals for marine spatial management are also mainly related to MPA establishment and tend to be even less well defined than ecological goals. Although some are relatively straightforward, such as the economic goal of increased tourism, other social goals such as increasing community awareness and acceptance are more difficult to measure in terms of performance. Metrics measured can be very simple in terms of factors such as numbers of tourists or dollars spent, or can be complex and subjective in terms of opinions expressed in surveys. In most cases only trends are considered and statistical or target reference comparisons are rare. Some of these issues may stem from the fact that social and economic management goals for MPAs are usually secondary to ecological goals and are therefore given lower priority in terms of measuring management performance. A table of indicator metrics and aggregation levels relative to management objectives was not constructed for socio-economic monitoring due to the generally poor definition of performance measures in the literature.

In summary, management objectives for marine spatial management, particularly MPAs, tend to be very general and poorly defined. Objectives need to be framed in a way that management performance can be assessed though monitoring. A suite of suitable metrics is available for this monitoring, however, planning for performance assessment must begin at the time of initial planning for the spatial management, rather than relying on *ad hoc* studies once the management regime is in place. In framing management objectives, many agencies have considered a relatively small-scale associated with individual MPAs and adjacent areas. In the future, management objectives should be set at a regional scale so the overall performance can be assessed. There needs to be a strong commitment to performance assessment, for example, many of the effects of MPAs are not evident for at least a decade. Marine spatial management in the coming years is likely to diversify considerably from a concentration on MPA management, particularly with the increasing focus on spatially-explicit fisheries management and the ecological effects of fishing, and also on environmental perturbations such as climate change. Performance measures for this type of monitoring need to be based as much as possible on sound ecological knowledge of responses to perturbations, rather than the setting of relatively arbitrary limits with little ecological basis.

Table 44. Summary of indicator metrics and data aggregation levels in relation to management objectives for marine spatial monitoring considered in this review.

Metric	Aggregation Level	Management objective				
		Protecting Biodiversity	Rehabilitation of Population Structure	Export of Biomass	Threatened species protection	Habitat protection
Biomass	Total			X		
	Family	X	X	X		
	Species	X	X	X	X	
	Phylum	X				
Density	Total	X	X	X		
	Family	X	X	X		
	Species	X	X	X		
	Predator/prey species	X			X	
Density - multivariate	Total	X	X			
	Large, Medium, Small fish (all species)	X	X			
	Large, Medium, Small species	X	X			
	Low/high commercial value species	X	X			
	Mobile/sedentary species	X	X			
Size distribution	Species		X	X		
Slope of size spectrum	Total		X			
Mid-point height of size spectrum	Total		X			
Catch	Family			X		
CPUE	Total			X		
	Species			X		
Taxonomic affinity	Total (all species)	X			X	
Species Richness	Total (all species)	X	X			
Species Diversity	Total (all species)	X	X			
Benthic percent cover	Species	X	X			X
	Habitat category	X			X	
	Coral bleaching	X			X	
Habitat structural complexity	Categorical	X	X			X
Coral mortality	Categorical	X	X			
Maximum number (video frame)	Predatory fish species	X	X			
Productivity	Dominant algal species	X				
Settlement/ recruitment	Coral species	X				
Ratio of cover	Calcifying and non-calcifying algae	X				
Percentage of towed video records	Categorical substratum class	X			X	X

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Appendix 1. Subtidal Reef Monitoring Program (SRMP) sites. Co-ordinates are Geodetic Datum of Australia.

Site Code	Site Name	Latitude	Longitude	Depth (m)
Port Phillip Heads				
2801	Point Franklin	-38.3173	144.7173	2
2802	Nepean Offshore	-38.3021	144.6587	2
2803	Nepean Inner West	-38.3041	144.6558	2
2804	South Channel Fort	-38.3069	144.801	2
2805	Shortland Bluff	-38.2753	144.6555	5
2806	Victory Shoal	-38.2802	144.6251	5
2807	Merlan Inner	-38.2873	144.62	5
2808	Nepean Inner East	-38.3042	144.659	2
2809	Lonsdale Kelp Outer	-38.2864	144.6296	7
2810	Merlan Outer	-38.2901	144.6226	5
2811	Lonsdale Kelp Inner	-38.2854	144.6275	7
2812	Annulus (Popes Eye)	-38.2767	144.6976	5
2813	Lonsdale Point	-38.2967	144.613	7
2814	Lonsdale Back Beach	-38.2902	144.5888	5
2815	Lonsdale Pt SW	-38.2941	144.5983	7
Phillip Island				
2901	Nobbies North	-38.5176	145.1099	6
2902	Pyramid Rock West	-38.5255	145.212	6
2903	Pyramid Rock North	-38.5288	145.2228	4
2904	Washing Machine	-38.5588	145.3408	6
2905	Cape Woolamai Mid	-38.5671	145.3586	6
2906	Cape Woolamai East	-38.5651	145.3617	4
Bunurong				
3001	Cape Patterson	-38.6803	145.6096	4
3002	C. Patt. Boat Ramp	-38.6774	145.617	6
3003	Oaks East	-38.6769	145.647	6
3004	Twin Reefs	-38.6789	145.6542	6
3005	Shack Bay West	-38.6767	145.6587	5
3006	Shack Bay East	-38.6718	145.6646	6
3007	The Caves	-38.6643	145.6831	6
3008	Petrel Rock East	-38.655	145.6951	5
3009	Patterson West Deep	-38.6759	145.5873	15
3010	Twin Reefs Deep	-38.6813	145.6527	16
3011	The Caves Deep	-38.6702	145.6912	16
3012	Shack Bay beach	-38.6753	145.6606	7
3013	Petrel Rock West	-38.6604	145.688	6
3014	The Oaks Beach	-38.676	145.6424	7
3015	Boat Ramp East	-38.6762	145.6209	7
Wilsons Promontory				
3101	North Shellback Is	-38.9665	146.2284	10
3102	North Tongue Pt	-38.9916	146.2546	10
3103	Northwest Norman Is	-39.0184	146.2385	10
3104	West Norman Is	-39.0246	146.2418	10
3105	Leonard Pt	-39.0233	146.2839	10
3106	Pillar Pt	-39.0396	146.3047	10
3107	South Norman Pt	-39.0538	146.3205	10
3108	Oberon Pt	-39.0764	146.3245	10
3109	East Glennie Is	-39.0839	146.2347	10
3110	West Glennie Is	-39.0896	146.2322	10
3111	North of Sea Eagle Bay	-39.1024	146.3338	10
3112	Sea Eagle Bay	-39.1118	146.3425	10
3113	North Anser Is	-39.1378	146.3191	10
3114	South Pt	-39.1341	146.3707	10
3115	Roaring Meg Bight	-39.13	146.3822	10
3116	West of West Landing	-39.1295	146.4077	10
3117	East landing	-39.1241	146.4239	10
3118	Fenwick Pt	-39.1131	146.4303	10
3119	Waterloo Pt	-39.0877	146.4407	10
3120	Central Waterloo Bay	-39.0619	146.4446	10
3121	North Waterloo Bay	-39.0642	146.4693	10
3122	North Cape Wellington	-39.0558	146.4799	10
3123	Bareback Bay	-39.0521	146.4754	10
3124	South Refuge	-39.0457	146.4782	10
3125	North Refuge	-39.0361	146.4704	10
3126	Horn Bay	-39.0287	146.467	10
3127	North Horn Pt	-39.0249	146.4714	10
3128	The Hat	-38.9972	146.4474	10

Appendix 1 (Cont.). Subtidal Reef Monitoring Program (SRMP) sites. Co-ordinates are Geodetic Datum of Australia.

Site Code	Site Name	Latitude	Longitude	Depth (m)
Twofolds Region				
3204	Old Jetty Bay	-37.7958	149.2662	4
3206	Hicks Light	-37.802	149.2773	5
3207	Krafts Garden	-37.7975	149.2878	5
3208	Tullaberger Deep	-37.5577	149.8423	7
3210	Gabo Monument	-37.564	149.9048	6
3212	Iron Prince	-37.5193	149.9629	5
3213	Howe West	-37.5099	149.9735	7
3214	Howe Central	-37.5078	149.9761	8
3215	Howe Border	-37.5071	149.9786	10
3216	Durvillaea Flats	-37.796	149.2878	4
3217	Meuller Reef	-37.7861	149.3224	7
3218	Petrel Point	-37.7821	149.3848	8
3219	Prince Wreck	-37.5207	149.9643	6
3220	Cape Howe Perpendicular	-37.5101	149.9736	10
3221	Pt Hicks SW	-37.8003	149.266	8
3222	Point Hicks Joggle	-37.8036	149.2741	5
3223	Beware Reef	-37.8203	148.7869	10
3224	Pearl Point	-37.7942	148.8832	8
Western Victorian Coastline				
3701	Merri	-38.4071	142.4776	8
3702	Breakwater	-38.4052	142.4675	5
3905	Olives	-38.4113	144.2301	7
3906	Ingoldsby Inner	-38.415	144.2122	4
3907	Angelsea reef	-38.4179	144.1952	3
3908	Phyco Reef	-38.347	144.3302	8
3909	Eagle Rock Inside	-38.4704	144.1069	5
3910	Eagle Rock Out	-38.4706	144.1085	7
3911	Marengo Reefs	-38.7787	143.6701	3
3912	Barnham Black	-38.7654	143.6789	6
Port Phillip Bay				
4101	Point Cook Pines	-37.9293	144.7947	3
4102	RAAF Base	-37.9462	144.7687	4
4103	Jawbone	-37.8658	144.8763	3
4104	Point Gelibrand	-37.8658	144.8997	3
4105	Ricketts Point Tea House	-37.9974	145.0295	3
4106	Halfmoon Bay	-37.9719	145.0088	3

Appendix 2. Timing of Subtidal Reef Monitoring Program (SRMP) surveys for each location as of January 2005.

Location	Survey No.	No. Sites	Season	Survey Period
Port Phillip Heads	1	15	Autumn	May 1998
	2	15	Spring	September - October 1998
	3	15	Autumn	May - July 1999
	4	15	Spring	October - November 1999
	5	15	Autumn	May - August 2000
	6	15	Summer	November 2000 - January 2001
	7	15	Autumn	June - July 2001
	8	15	Summer	January 2002
	9	15	Summer	January 2003
	10	15	Winter	July – August 2004
	11	15	Summer	November – December 2004
Phillip Island	1	6	Spring	September 1999
	2	6	Summer	February 2000
	3	6	Summer	February 2001
	4	6	Spring	November 2001
	5	6	Autumn	March – April 2002
	6	6	Autumn	February – March 2003
Bunurong	1	8	Winter	June 1999
	2	12	Summer	January - March 2000
	3	4	Winter	July - August 2000
	4	12	Summer	Dec 2000 - Jan 2001
	5	12	Winter	May - June 2001
	6	12	Summer	February-March 2002
	7	12	Winter	August 2002
	8	12	Autumn	March-April 2003
	9	4+	Summer	December 2004 - Present
Wilsons Promontory	1	28	Early Summer	November - December 1999
	2	22	Autumn	May - June 2000
	3	26	Early Summer	November 2000
	4	22	Autumn	May 2001
	5	21	Early Summer	November - December 2002
	6	21	Autumn	May – June 2002
	7	23	Early Summer	November 2003
	8	18	Winter	August 2004
Twofolds shelf region	1	18	Late Summer	March 2004
West Victorian Coast	1	8	Summer	December 2003 - January 2004
Port Phillip Bay	1	6	Late Summer	March 2003
	2	6	Early Autumn	April 2004

Appendix 3. Sampling sites for abalone monitoring. AFA = Abalone Fishing Area, MNP = Marine National Park, MS = Marine Sanctuary, WAR = Work Allocation Region. WGS84 Datum

Site	Name	Latitude	Longitude	Depth	Type	WAR
	Portland					
138	Passage	-38.4017	141.6472	12	AFA	Portland
139	Inside Nelson	-38.4053	141.5648	12	AFA	Portland
140	Killer waves	-38.4080	141.5598	9	AFA	Portland
141	Washing Machine Rk	-38.4239	141.5570	12	AFA	Portland
142	Cape Nelson	-38.4288	141.5533	18	AFA	Portland
143	Nelson Cave	-38.4305	141.5517	12	AFA	Portland
144	Inside Murrells	-38.4085	141.5234	5	AFA	Portland
145	Murrells #2	-38.4179	141.5233	18	AFA	Portland
146	Murrells	-38.4130	141.5197	18	AFA	Portland
147	Horseshoe	-38.3782	141.4098	6	AFA	Portland
148	Seal Caves	-38.3933	141.4142	9	AFA	Portland
149	Horseshoe #2	-38.3849	141.4102	8	AFA	Portland
150	Cape Bridgewater	-38.3938	141.3954	14	AFA	Portland
151	Pebble beach	-38.3892	141.3880	15	AFA	Portland
152	Cape Bridgewater front	-38.3883	141.3799	17	AFA	Portland
153	South West Bridgewater	-38.3689	141.3640	16	AFA	Portland
154	Petrified forest	-38.3674	141.3643	15	AFA	Portland
155	Watersprings	-38.3643	141.3659	15	AFA	Portland
156	Whites	-38.3505	141.3790	13	AFA	Portland
157	Inside Bridgewater	-38.3505	141.3790	13	AFA	Portland
	Warrnambool					
120	Leavys #2	-38.3950	142.4486	8	AFA	Port Fairy
121	Leavys Beach	-38.3927	142.4443	7	AFA	Port Fairy
122	The Cutting	-38.3542	142.3619	8	AFA	Port Fairy
123	Outside Killarney	-38.3661	142.3272	14	AFA	Port Fairy
124	Killarney	-38.3635	142.3230	7	AFA	Port Fairy
125	Mills Reef	-38.3674	142.2953	9	AFA	Port Fairy
126	Inside Mills	-38.3651	142.2911	6	AFA	Port Fairy
127	Lighthouse	-38.3901	142.2580	7	AFA	Port Fairy
128	Lighthouse back	-38.3945	142.2549	10	AFA	Port Fairy
129	Inside Craggs	-38.3894	142.1395	8	AFA	Port Fairy
130	McKenchie Craggs	-38.3852	142.1360	8	AFA	Port Fairy
131	The Craggs	-38.3851	142.1355	9	AFA	Port Fairy
132	Outside Craggs	-38.3896	142.1346	16	AFA	Port Fairy
133	The Craggs inside	-38.3835	142.1335	9	AFA	Port Fairy
134	N/W Corner LJPIs	-38.4117	142.0161	14	AFA	Port Fairy
135	Lady Julia Percy Island	-38.4167	142.0097	9	AFA	Port Fairy
136	Prop Bay east	-38.4222	141.9961	13	AFA	Port Fairy
137	Prop Bay Centre	-38.4212	141.9954	14	AFA	Port Fairy
294	Big Bay 1km east of Antairs rocks	-38.5613	142.7784	4.1	AFA	Shipwreck
295	West Antairs rocks.	-38.5562	142.7668	3.8	AFA	Shipwreck
296	Childers Cove centre of bay wooden steps	-38.4905	142.6720	3.5	AFA	Shipwreck
297	W'bool Hard against cliffs	-38.4455	142.6009	5.9	AFA	Shipwreck

Appendix 3 (Cont.). Sampling sites for abalone monitoring, AFA = Abalone Fishing Area, MNP = Marine National Park, MS = Marine Sanctuary, WAR = Work Allocation Region. WGS84 Datum

Site	Name	Latitude	Longitude	Depth	Type	WAR
	Apollo Bay					
106	Cape Patton	-38.6941	143.8369	15	AFA	Otway
107	Sugarloaf #2	-38.7008	143.8054	9	AFA	Otway
108	Sugarloaf	-38.7009	143.8046	10	AFA	Otway
109	Bonbory Pt	-38.7626	143.6829	7	AFA	Otway
110	Marengo	-38.7808	143.6785	12	AFA	Otway
111	Bald Hill	-38.7931	143.6516	10	AFA	Otway
112	Pt Lewis	-38.8418	143.5758	10	AFA	Otway
113	Parker RV	-38.8464	143.5670	9	AFA	Otway
114	The tide #4	-38.8579	143.5413	8	AFA	Otway
115	The tide #3	-38.8593	143.5284	8	AFA	Otway
116	The tide	-38.8601	143.5218	11	AFA	Otway
117	The tide #2	-38.8636	143.5197	13	AFA	Otway
118	West of Otway	-38.8566	143.5011	12	AFA	Otway
119	Cape Otway	-38.8539	143.4974	15	AFA	Otway
265	BB 2.1	-38.8268	143.5859	9	AFA	Otway
266	BB 1.5	-38.8298	143.5858	6	AFA	Otway
267	BB 1.2	-38.8322	143.5843	5	AFA	Otway
268	BB 1.1	-38.8326	143.5834	7	AFA	Otway
269	BB 2.3	-38.8270	143.5885	7	AFA	Otway
270	BB 2.2	-38.8270	143.5881	6	AFA	Otway
271	BB 1.4	-38.8300	143.5857	6	AFA	Otway
272	BB 1.3	-38.8300	143.5857	9	AFA	Otway
273	BB 2.4	-38.8256	143.5874	6	AFA	Otway
274	BB 2.5	-38.8254	143.5870	6	AFA	Otway
275	BB 3.1	-38.8240	143.5865	6	AFA	Otway
276	BB 3.2	-38.8241	143.5867	8	AFA	Otway
277	BB 3.5	-38.8222	143.5877	9	AFA	Otway
278	BB 3.3	-38.8230	143.5869	9	AFA	Otway
279	BB 3.4	-38.8229	143.5868	6	AFA	Otway
290	Cats	-38.7387	143.1881	6.2	AFA	Otway
291	Ryans Den	-38.7590	143.2866		AFA	Otway
292	White Cliffs	-38.7582	143.3295	4.9	AFA	Otway
293	Rotten Point	-38.7838	143.4122	0.6	AFA	Otway
	Torquay					
245	SC 1.1	-38.3046	144.3782		AFA	Surf Coast
246	SC 1.2	-38.3048	144.3782		AFA	Surf Coast
247	SC 1.3	-38.3053	144.3779		AFA	Surf Coast
248	SC 1.4	-38.3059	144.3780		AFA	Surf Coast
249	SC 1.5	-38.3058	144.3777		AFA	Surf Coast
280	SC2.1	-38.2894	144.4153		AFA	Surf Coast
281	SC2.2	-38.2887	144.4159		AFA	Surf Coast
282	SC2.3	-38.2880	144.4166		AFA	Surf Coast
283	SC2.4	-38.2871	144.4175		AFA	Surf Coast
284	SC2.5	-38.2866	144.4194		AFA	Surf Coast
285	SC3.1	-38.2903	144.5011		MS	Surf Coast
286	SC3.2	-38.2903	144.5011		MS	Surf Coast
287	SC3.3	-38.2900	144.5022		MS	Surf Coast
288	SC3.4	-38.2900	144.5021		MS	Surf Coast
289	SC3.5	-38.2900	144.5021		MS	Surf Coast

Appendix 3 (Cont.). Sampling sites for abalone monitoring, AFA = Abalone Fishing Area, MNP = Marine National Park, MS = Marine Sanctuary, WAR = Work Allocation Region. WGS84 Datum

Site	Name	Latitude	Longitude	Depth	Type	WAR
	Port Phillip Bay					
98	Fred's Farm	-37.8697	144.8882	3	AFA	Port Phillip Bay
99	Point Cook Homestead	-37.9251	144.8011	3	MS	Port Phillip Bay
100	Sheoak	-37.9427	144.7561	4	AFA	Port Phillip Bay
101	The Stick	-37.8823	144.8574	2	AFA	Port Phillip Bay
102	RAAF Base	-37.9462	144.7673	3	AFA	Port Phillip Bay
103	Kirk's Point Inner	-38.0252	144.5856	2	AFA	Port Phillip Bay
104	Kirk's Point mid	-38.0301	144.5935	2	AFA	Port Phillip Bay
105	Kirk's Point Sth Cardinal	-38.0308	144.5945	3	AFA	Port Phillip Bay
230	K 3.5	-38.0292	144.5681	6	AFA	Port Phillip Bay
231	K 3.4	-38.0224	144.5811	6	AFA	Port Phillip Bay
232	K 3.3	-38.0223	144.5804	7	AFA	Port Phillip Bay
233	K 3.2	-38.0264	144.5698	7	AFA	Port Phillip Bay
234	K 3.1	-38.0273	144.5681	6	AFA	Port Phillip Bay
235	K 2.5	-38.0317	144.5632	4	AFA	Port Phillip Bay
236	K 2.4	-38.0317	144.5631	6	AFA	Port Phillip Bay
237	K 2.3	-38.0316	144.5632	7	AFA	Port Phillip Bay
238	K 2.2	-38.0328	144.5622	6	AFA	Port Phillip Bay
239	K 2.1	-38.0330	144.5620	5	AFA	Port Phillip Bay
240	K 1.5	-38.0375	144.5586	6	AFA	Port Phillip Bay
241	K 1.4	-38.0375	144.5586	5	AFA	Port Phillip Bay
242	K 1.3	-38.0380	144.5576	5	AFA	Port Phillip Bay
243	K 1.2	-38.0380	144.5576	5	AFA	Port Phillip Bay
244	K 1.1	-38.0380	144.5576	4	AFA	Port Phillip Bay
	Cape Schanck					
70	West Head Inner	-38.4912	145.0368	13	AFA	Mornington Peninsula
71	West Head Reef	-38.4891	145.0241	7	AFA	Mornington Peninsula
72	Flinders Back Beach	-38.4879	145.0220	7	AFA	Mornington Peninsula
73	Flinders BB Outside	-38.4917	145.0152	6	AFA	Mornington Peninsula
74	Pinnacles	-38.4876	145.0046	7	AFA	Mornington Peninsula
75	Flinders BB West	-38.4861	145.0006	8	AFA	Mornington Peninsula
76	Blowhole	-38.4871	144.9883	12	AFA	Mornington Peninsula
77	The Blowhole West	-38.4871	144.9851	12	AFA	Mornington Peninsula
78	Glensira Point 1	-38.4880	144.9818	7	AFA	Mornington Peninsula
79	Glensira Point 2	-38.4915	144.9766	13	AFA	Mornington Peninsula
80	Symonds Bay	-38.4898	144.9580	10	AFA	Mornington Peninsula
81	Symonds Bay West	-38.4907	144.9495	8	AFA	Mornington Peninsula
82	The Arch Gully	-38.4958	144.9386	9	AFA	Mornington Peninsula
83	The Arch	-38.4975	144.9305	15	AFA	Mornington Peninsula
84	Picnic Point	-38.4973	144.9267	7	AFA	Mornington Peninsula
85	Picnic Point Cliff	-38.4979	144.9217	16	AFA	Mornington Peninsula
86	Picnic Point Bay	-38.4953	144.9176	8	AFA	Mornington Peninsula
87	Bushrangers Bay Inside	-38.4933	144.9094	10	AFA	Mornington Peninsula

Table 3. (Cont.) Sampling sites for abalone monitoring. AFA = Abalone Fishing Area, MNP = Marine National Park, MS = Marine Sanctuary, WAR = Work Allocation Region. WGS84 Datum

Site	Name	Latitude	Longitude	Depth	Type	WAR
88	Limestone Caves	-38.4920	144.9043	8	AFA	Mornington Peninsula
89	Cape Schanck East 2	-38.4959	144.8971	14	AFA	Mornington Peninsula
90	Bushrangers Bay	-38.4944	144.8948	7	AFA	Mornington Peninsula
91	Cape Schanck East 1	-38.4965	144.8924	11	AFA	Mornington Peninsula
92	Pulpit Rock South	-38.5016	144.8871	14	AFA	Mornington Peninsula
93	Pulpit Rock	-38.4948	144.8840	14	AFA	Mornington Peninsula
94	North West Cape Schanck	-38.4795	144.8819	8	AFA	Mornington Peninsula
95	Fingal Beach	-38.4711	144.8765	4	AFA	Mornington Peninsula
96	Sorrento Back Beach	-38.3594	144.7426	8	AFA	Mornington Peninsula
97	Portsea Back Beach	-38.3240	144.6805	8	AFA	Mornington Peninsula
250	F 3.1	-38.4831	145.0175	7	MS	Flinders
251	F 3.2	-38.4831	145.0174	8	MS	Flinders
252	F 3.3	-38.4835	145.0154	8	MS	Flinders
253	F 3.4	-38.4833	145.0152	5	MS	Flinders
254	F 3.5	-38.4833	145.0151	5	MS	Flinders
255	F 2.1	-38.4863	145.0181	6	MS	Flinders
256	F 2.2	-38.4866	145.0174	4	MS	Flinders
257	F 2.3	-38.4866	145.0174	5	MS	Flinders
258	F 1.2	-38.4828	145.0242	5	AFA	Flinders
259	F 1.1	-38.4841	145.0256	8	AFA	Flinders
260	F 2.4	-38.4866	145.0160	11	MS	Flinders
261	F 2.5	-38.4867	145.0154	12	MS	Flinders
262	F 1.5	-38.4857	145.0202	8	AFA	Flinders
263	F 1.4	-38.4861	145.0209	7	AFA	Flinders
264	F 1.3	-38.4858	145.0222	4	AFA	Flinders
	Phillip Island					
62	Cape Woolamai	-38.5637	145.3464	14	AFA	Phillip Island
63	West Cape Woolamai	-38.5591	145.3407	11	AFA	Phillip Island
64	Pyramid Rock	-38.5314	145.3464	9	AFA	Phillip Island
65	Seal Rocks	-38.5277	145.0995	13	AFA	Phillip Island
66	North Seal Rocks	-38.5251	145.1005	5	AFA	Phillip Island
67	South Nobbies	-38.5205	145.1152	12	AFA	Phillip Island
68	West Nobbies	-38.5176	145.1096	9	AFA	Phillip Island
69	Kitty Millar Bay	-38.5158	145.1727	11	AFA	Phillip Island
	Wilsons Promontory					
54	South East Point	-39.1298	146.4080	14	MNP	Wilson's Promontory
55	Anser Island	-39.1453	146.3268	11	MNP	Wilson's Promontory
56	South West Point	-39.1107	146.3422	12	MNP	Wilson's Promontory
57	Norman Point	-39.0534	146.3198	6	MNP	Wilson's Promontory
58	Pillar Point	-39.0387	146.3073	10	AFA	Wilson's Promontory
59	Tongue Point	-38.9942	146.2583	9	AFA	Wilson's Promontory
60	Norman Island	-39.0251	146.2420	12	AFA	Wilson's Promontory
61	Glennie Group	-39.0871	146.2310	12	AFA	Wilson's Promontory
298	Middle of Grinder bay	-38.9017	145.9558	10	AFA	Wilson's Promontory
299	Arch Rock in close	-38.8469	145.8942	11	AFA	Wilson's Promontory
300	Cape Liptrap west in close	-38.9004	145.9128	5.9	AFA	Wilson's Promontory
301	Cape Liptrap west	-38.8892	145.9066	13	AFA	Wilson's Promontory
	Mallacoota					
1	Cape Howe	-37.5101	149.9736	10	MNP	Mallacoota
2	Iron Prince	-37.5203	149.9640	7	AFA	Mallacoota
3	Iron Prince	-37.5290	149.9595	10	AFA	Mallacoota
4	North-East Gabo	-37.5605	149.9188	12	AFA	Mallacoota
5	Gabo Front	-37.5704	149.9161	9	AFA	Mallacoota
6	Gabo Island North	-37.5529	149.9115	7	AFA	Mallacoota
7	Gabo Harbour 2	-37.5553	149.9073	5	AFA	Mallacoota
8	Gabo Harbour 1	-37.5500	149.9053	3	AFA	Mallacoota
9	Gabo Harbour	-37.5513	149.9048	12	AFA	Mallacoota
10	Tullaberga Island	-37.5568	149.8480	3	AFA	Mallacoota
11	Tullaberga Is. 2	-37.5543	149.8459	6	AFA	Mallacoota
12	Tullaberga Is. 1	-37.5568	149.8415	7	AFA	Mallacoota
13	Bastion Point 1	-37.5761	149.7690	9	AFA	Mallacoota
14	Bastion Point 2	-37.5744	149.7686	9	AFA	Mallacoota
15	Bastion Point	-37.5760	149.7673	7	AFA	Mallacoota
16	Shipwreck Creek 2	-37.6063	149.7272	10	AFA	Mallacoota
17	Shipwreck Creek 1	-37.6427	149.7104	10	AFA	Mallacoota
18	Shipwreck Creek	-37.6513	149.7015	10	AFA	Mallacoota
19	Little Rame	-37.6764	149.6824	8	AFA	Mallacoota
20	Little Rame 3	-37.6835	149.6801	11	AFA	Mallacoota

Table 3. (Cont.) Sampling sites for abalone monitoring. AFA = Abalone Fishing Area, MNP = Marine National Park, MS = Marine Sanctuary, WAR = Work Allocation Region. WGS84 Datum

Site	Name	Latitude	Longitude	Depth	Type	WAR
21	Little Rame 2	-37.6859	149.6778	5	AFA	Mallacoota
22	Little Rame 1	-37.6879	149.6763	10	AFA	Mallacoota
23	Little Rame	-37.6895	149.6727	6	AFA	Mallacoota
24	Benedore 2	-37.6953	149.6432	8	AFA	Mallacoota
25	Benedore 1	-37.6978	149.6359	8	AFA	Mallacoota
26	Benedore	-37.7003	149.6316	12	AFA	Mallacoota
27	Sandpatch Point	-37.7220	149.6000	11	AFA	Mallacoota
28	Sandpatch Point 4	-37.7192	149.6000	8	AFA	Mallacoota
29	Sandpatch Point 3	-37.7257	149.5986	8	AFA	Mallacoota
30	Sandpatch Point 1	-37.7283	149.5928	10	AFA	Mallacoota
31	Sandpatch Point 2	-37.7273	149.5909	10	AFA	Mallacoota
32	The Skerries	-37.7714	149.5167	7	AFA	Mallacoota
33	The Skerries 1	-37.7560	149.5166	9	AFA	Mallacoota
34	The Skerries 2	-37.7527	149.5142	8	AFA	Mallacoota
35	Big Rame 4	-37.7740	149.4918	10	AFA	Mallacoota
36	Big Rame	-37.7755	149.4870	9	AFA	Mallacoota
37	Big Rame 2	-37.7774	149.4847	15	AFA	Mallacoota
38	Big Rame 3	-37.7675	149.4841	15	AFA	Mallacoota
39	Big Rame 1	-37.7770	149.4835	11	AFA	Mallacoota
40	Island Point 1	-37.7811	149.4332	8	AFA	Mallacoota
41	Island Point	-37.7817	149.4288	10	AFA	Mallacoota
42	Island Point	-37.7783	149.4262	10	AFA	Mallacoota
43	Petrel Point 1	-37.7836	149.3921	14	AFA	Mallacoota
44	Petrel Point	-37.7825	149.3847	10	AFA	Mallacoota
45	Petrel Point	-37.7818	149.3790	12	AFA	Mallacoota
46	Whaleback	-37.8040	149.2910	7	AFA	Mallacoota
47	Point Hicks	-37.8038	149.2738	7	MNP	Mallacoota
48	Point Hicks 2	-37.8037	149.2719	15	MNP	Mallacoota
49	Point Hicks 3	-37.8028	149.2699	6	MNP	Mallacoota
50	Point Hicks 1	-37.7986	149.2658	5	MNP	Mallacoota
51	Beware Reef 2	-37.8185	148.7842	13	MS	Mallacoota
52	Beware Reef 1	-37.8205	148.7857	12	MS	Mallacoota
53	Beware Reef 3	-37.8203	148.7865	12	MS	Mallacoota
	Marlo					
201	Point Ricardo	-37.8112	148.6192	7.9	AFA	Marlo
202	French's Narrows FRN2	-37.8086	148.5960	11.3	AFA	Marlo
203	FRN 1A French's Narrows	-37.8082	148.5993	11.6	AFA	Marlo
204	RIC 2 Ricardo	-37.8119	148.6227	9.1	AFA	Marlo
205	Pt RIC 3	-37.8130	148.6192	12.5	AFA	Marlo
206	Pearl Pt.1	-37.7945	148.8874	16.5	AFA	Marlo
207	Pearl Pt. 3	-37.7944	148.8845	11.9	AFA	Marlo
208	Pearl Pt. 2A	-37.7929	148.8854	11.9	AFA	Marlo
209	Pearl Pt. 4A	-37.7918	148.9006	10.4	AFA	Marlo
210	Pearl Pt. 4	-37.7926	148.8994	12.2	AFA	Marlo
211	WC1 (Cape Conran)	-37.8153	148.7257	10.4	AFA	Marlo
212	CC Light (Cape Conran)	-37.8136	148.7313	11.9	AFA	Marlo
213	CC 1A TOE (Cape Conran)	-37.8075	148.7439	9.4	AFA	Marlo
214	EC 4A (Cape Conran)	-37.8056	148.7480	10.7	AFA	Marlo
215	CC Front (Cape Conran)	-37.8163	148.7286	13.4	AFA	Marlo
216	YE East A (Yeerung)	-37.7942	148.7876	9.1	AFA	Marlo
217	YE East B (Yeerung)	-37.7955	148.7897	12.8	AFA	Marlo
218	YE C (Yeerung)	-37.7968	148.7746	13.1	AFA	Marlo
219	YE D (Yeerung)	-37.7973	148.7638	9.8	AFA	Marlo
220	YE West (Yeerung)	-37.7976	148.7616	7.9	AFA	Marlo

Appendix F

The use of telescoping spatial scales to capture inshore to slope dynamics in marine ecosystem modelling

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Abstract

Ecosystem processes function at many scales, and capturing these processes is a challenge for ecosystem models. Nevertheless, it a necessary step for considering many management issues pertaining to shelf and coastal systems. In this paper we explore one method of modelling large areas with a focus at a range of scales. We model the waters off south-eastern Australia using a polygon telescoping approach, which incorporates fine-scale detail at the coastal zone, increasing in scale to a very coarse scale in the offshore areas. The fine-scale resolution of the reef and coastal areas reproduces observed trends in reef fish abundances. This telescoping technique is a useful tool for incorporating a wide range of habitats at different scales into a single model.

Key words

Ecosystem model, spatial structure, Atlantis, model complexity

Introduction

Spatial scale is an important factor in marine ecological studies (Rose & Leggett 1990, Levin 1992, Perry & Ommer 2003, Rahbek 2005, Guisan et al. 2007, Dumont et al. 2008). Community dynamics are mediated by factors that occur at a range of scales from local, small-scale biological interactions to large-scale oceanographic processes (Levin 1992). It is therefore clear that effective management or investigation of marine systems must consider ecological patterns at both regional and local scales, and integrate information over these scales (Williams & Bax 2001).

The fisheries of south-eastern Australia comprise target species in shallow coastal waters (e.g. *Sillago punctata* (King George whiting)) and over large areas of open ocean (e.g. *Xiphias gladius* (broadbill swordfish)). The life histories of some target species traverses these different scales, with the juvenile inhabiting small, inshore nursery grounds before recruiting to the adult offshore fishery (eg. *Mugil cephalus* (sea mullet)) (Kailola et al. 1993). Exposure of these species to the fishery is also being increasingly influenced by the distribution of Marine Protected Areas (MPAs) in both coastal and offshore waters.

Appropriate tools are required in order to effectively manage fisheries and biodiversity at these diverse scales. Ecosystem models are one tool that has seen increased use over the last decade as a means of capturing the dynamics of large marine systems, both at the local and regional scale. However, determining the appropriate spatial definition in these models that will allow the representation of a range of scales remains problematic, in terms of both the spatial resolution and the spatial structure of the model.

Spatial resolution has been shown to strongly influence modelled ecosystem dynamics (Sharov 1996), including competition (Johnson & Seinen 2002), trophic structure, community composition and ecosystem complexity (Fulton et al. 2004). While an increase in spatial resolution can potentially provide more detail and accuracy when modelling an ecosystem, there are also drawbacks. Issues such as increased data requirements for parameterisation and calibration, increased computational power requirements and the increased complexity of model output, make producing and using highly spatially-resolved models difficult (Fulton et al. 2003). As a result, ecosystem models have often been built with either no spatial resolution (Walters et al. 1997, Bissett et al. 1999, Cury et al. 2000) or with large areas represented by relatively few boxes (Baretta et al. 1995, Fulton & Smith 2004).

In addition to spatial resolution, the spatial structure of a model must be considered. The majority of spatially defined ecosystem models use a regular grid (e.g SEAPODYM (Lehodey et al. 2008), Ecospace (Walters et al. 1999), OSMOSE (Shin & Cury 2001)). One of the limitations of this approach is the inflexible nature of the spatial partitioning, there being no option to adjust the level of definition throughout the model. An alternative, but less common approach to the grid structure is the box model, which incorporates multi size and shape polygons to define the spatial domain (eg, ERSEM, (Baretta et al. 1995)). The flexibility of the polygon approach allows the modeller to scale the model to exactly the dimensions that are most useful for the questions being addressed, and differentiate the level of spatial complexity throughout the model domain. While curvilinear grids (e.g. (Trottier et al. 1983) also vary through space, their structure is highly constrained compared to polygonal grids.

Both one-way (Beckers et al. 1997, Zavaterelli & Pinardi 2003) and two-way (Ginis et al. 1998, Barth et al. 2005) nesting approaches have been used in oceanographic models to represent small systems at a high level of spatial resolution, while also including the dynamics of large-scale systems. While one-way nested models are limited by the assumption that regional dynamics do not affect large-scale processes, two-way nested oceanographic models have been shown to work successfully and allow predictions to be made over both large and small scales. This approach, however, has not been explicitly extended to ecosystem models, where large- and small-scale biological processes interact.

In this paper we create a box model with a structured, telescoped spatial geometry, in order to address some of the problems associated with modelling a large spatial domain with high resolution at key points. We use a polygon structure to define a fine-scale spatial zoning inshore, gradually increasing the scale to give an intermediate grain along the upper slope – and moving to a coarse scale for the lower slope and oceanic zones. Using existing biogeographical descriptions, we partition the model into biologically meaningful spatial units across the domain. One of the motivating factors behind this work was to allow us to model a larger area, even when the available data to parameterise that area was sparse. The trade-off between model uncertainty and data uncertainly is handled by providing greater model detail where more data is available (therefore trying to find a closer match to data), and less detail in areas where there is a higher degree of data uncertainty (so less emphasis is put on matching model output to sparse or uncertain data).

Our objective is to create a model that realistically represents of the overall dynamics of the Eastern Victorian and Tasmanian coastlines of Australia. Large-scale processes and abiotic influences out to the continental shelf and upper slope are incorporated into the model. While these deeper areas are not the main focus of the model, we incorporate them in order to account for the potential impacts of broad-scale ecosystem dynamics upon local communities. We aim to capture the broad system dynamics, rather than exact detail. In order to get an idea of the model performance we measure it against recent abundance trajectories of selected reef species in order to determine whether overall patterns can be reproduced.

Modelling context

Models have many different purposes and the ultimate use of a model contributes significantly to its form and judgements of whether or not this form is acceptable. This model was developed to allow the authors to explore potential monitoring schemes and effective indicators of the performance of spatial management. The intent was not to create a highly tactical "assessment" model, but to create a simulation environment that allowed for the application of the management strategy evaluation approach to spatial management. Management strategy evaluation (MSE) is a simulation technique based on modelling each part of the adaptive management cycle. It was originally developed more than 20 years ago to consider implications of alternative management strategies applied to natural resources (e.g. fish stocks). The method is now widely accepted as a best practice approach for single stock and ecosystem-level management questions (Plaganyi 2007) and has recently been adopted for multiple use questions as well (McDonald et al. 2008). The strength of the MSE approach is that it does not try to find some optimal solution based on a single model. Instead alternative strategies are evaluated using a simulated system to check for the robustness of the results. As such the model does not need to be exact or highly constrained, rather the most important feature of an MSE model is that it capture non-linearities, patterns and feedbacks typical of those seen in the real system.

Methods

Atlantis framework

The Atlantis Spatial Management (Atlantis-SM) model uses the Atlantis ecosystem modelling framework (Fulton et al. 2004). The biophysical sub-model of Atlantis tracks nutrient flows through the main biological and detritus groups within marine ecosystems. Processes such as production, consumption and growth, habitat dependency, reproduction, movement and large-scale migration, predation and other forms of mortality and waste production are all handled explicitly. The trophic resolution is typically at the functional group level, although some age-structured single species groups are also included. The outputs of the model consist of deterministic time series for each biological component in each spatial cell in the modelled ecosystem.

Model domain

The Atlantis-SM domain covers approximately 640,035 km^2 off the south-east coast of Australia. The domain extends from north of the New South Wales-Victorian border to just west of Port Phillip Bay, and around the Eastern side of Tasmania, including all of Bass Strait (Figure 1). Soft sediment habitats, including sand, mud and seagrass, form the dominant component of the area’s inshore environment. Rocky reefs and kelp forests also play important roles in the ecology of the region. Offshore, the model incorporates both shelf and open ocean environments. Ecologically, the area is highly diverse and contains a high proportion of endemic species.

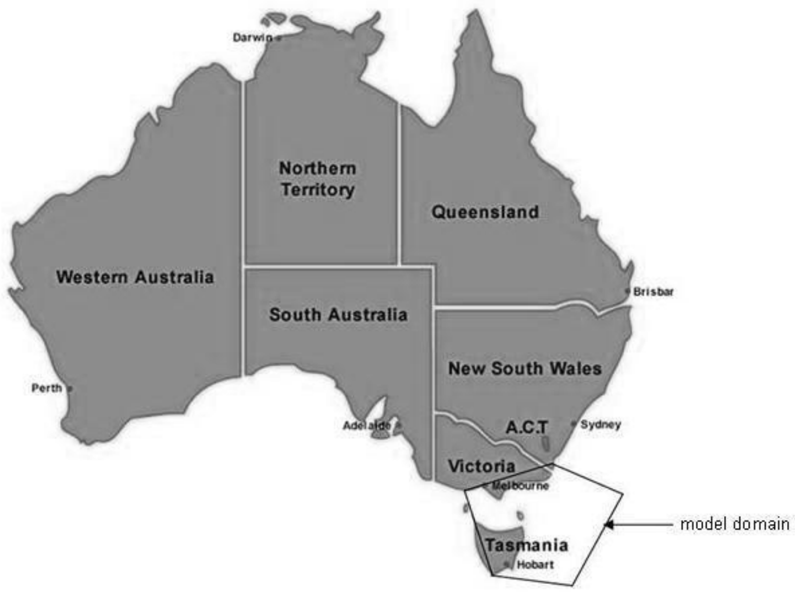


Figure 1: Model domain for the Atlantis-SM model.

Spatial geometry

The horizontal box geometry is based on the major geographical, physical and ecological properties of area. The spatial geometry was determined using information from a demersal bioregionalisation (IMCRA 1998, Butler et al. 2001, Lyne & Hayes 2005) and an independent pelagic analysis based on the CSIRO Atlas of Regional Seas (CARS). The work of Williams and Bax (2001) was used to create the telescoping detail, particularly along the Victorian coast. Beyond Victoria the methods given in Lyne and Hayes (2005) were used to create the same level of telescoping around other spatial management zones (particularly in Tasmanian waters). This began by basing the gross form of the most distant boxes on the geomorphology and depth of the slope was used to distinguish boxes from shelf edge, upper and mid to lower slope. Further detail along the shelf and inshore along the coast was created based on geomorphological characteristics of the area, in combination with overlying water properties, and ecological knowledge pertaining to habitat associations, environmental preferences and realised spatial distributions of known species across the shelf and around existing MPAs in Victoria and Tasmania. This spatial distribution data was available through monitoring programs in and around MPAs in these states. This meant the boundaries of the Point Hicks, Cape Howe and Maria Island Marine National Parks were explicitly represented by the final model structure. The existence of the monitoring data (which has been collected around the MPas and surrounding reefs over the past 10 years) .made the fine-scale box geometry feasible and informative. The slope and offshore boxes were more coarsely defined, in contrast, because of less available data and also because they were not the focus of the study. This sliding scale of resolution is an automatic outcome of the hierarchical scale provided by the method of Lyne and Hayes (2005), which makes it a natural fit with modelling that explicitly matches resolution with the level of data availability and uncertainty.

The spatial geometry was defined by a polygonal box geometry of 80 boxes with up to 6 water column layers per box (plus 1 sediment layer). The maximum depth of the model was 2400m. If the depth of a polygon was less than 2400 m, the water column layers were truncated to match, so that shallower boxes had fewer layers. In boxes where bottom depth exceeded 2400m, waters below this depth were omitted and the bottom layer was treated as having an open lower boundary with regard to exchanges. The number of depth layers per box, domain bathymetry and bioregions are shown in Figure 2.

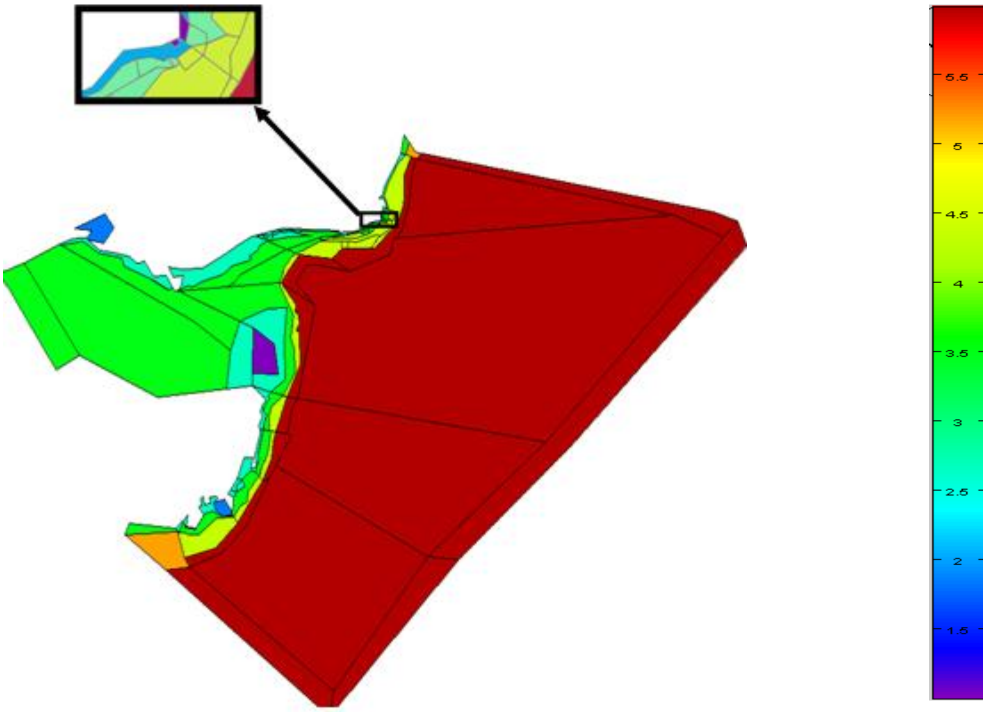


Figure 2: Atlantis-SM model domain showing the number of layers in each box, with inset showing fine scale detail around Cape Howe Reserve.

The polygons defined for the model ranged widely in size. The smallest box was 3.98 km^2 and represented a small unprotected area inside Cape Howe Marine National Park. The largest box was 209,000 km^2, which was one of four large offshore boxes on the eastern side of the model. The average polygon size for coastal regions was 587 km^2, while slope boxes ranged from 1,160 km^2 to 5,100 km^2. The least spatially resolved area was offshore, with box size ranging from 35,100 km^2 to 209,000 km^2. The overall structure of the box geometry followed the depth contours of the area. Within this structure, bottom type, bioregion and marine protected area boundaries were taken into account to form the fine-scale definition.

Physical transport

Vertical and horizontal exchanges between boxes were calculated from archived current velocities generated by the global ocean model OFAM (Oke et al. 2005), which had a spatial resolution of 0.1° over the box model domain. The exchanges were calculated by integrating the daily normal component of currents over each depth band of each box face (using realistic bathymetry to ensure face sectional areas are accurate). To account for the non-uniform nature of the box structure, exchanges were corrected for hyper diffusion within boxes. As a first approximation, the east-west exchanges were divided by the longitudinal scale of the box and north-south exchanges were divided by the latitudinal scale of the box. A conservative tracer was then used to check flows through the system, with tuned box-specific flow scalars used to remove any remaining hyper diffusion effects. If this combined correction was not made the flows in larger boxes were overstated by orders of magnitude as once in a box any tracer was assumed to be equally accessible throughout the box, which artificially inflated the flows; this effect was removed by the correction (and this approach was more tractable than inverse modelling over such large and complex domains).

Biological movement / habitat dependence

To capture the effects of heterogeneity within a single polygon, a sub-grid scale model was used to represent the effects of within-box benthic habitat patchiness. Habitat dependency is incorporated into the parameterisation of the model to reflect whether a group can access an area containing a specific habitat type. If a group can’t access a habitat type, then it is also not possible for it to access any prey biomass associated with that habitat type. During trophic interactions the habitat usages of predator and prey are compared to see if the two groups can be in the same small scale patches and thus able to interact directly. Habitat dependency is only relevant in the water layer that interacts with the sediment, in higher water layers the biological groups are not limited by habitat association. This allows for differing behaviour with vertical movements, such as feeding on mesopelagic prey layers.

Biology and initial conditions

The biological components of the model provide a representation of the entire foodweb; inshore and offshore, pelagic and demersal and from bacteria and phytoplankton up to top predators. The model includes 3 types of detritus, 3 types of primary producer, 21 invertebrate and 31 vertebrate groups, some represented at the species level and others as functional groups (Table 1 shows the complete list along with initial biomass for each group). The vertebrate, abalone, lobster, cephalopod and prawn groups are age-structured, but all other groups are handled as biomass pools. Those groups represented as functional groups were aggregate groups of species with similar size, diet, predators, habitat preferences, migratory patterns and life history strategy. In addition to these living biological groups, pools of ammonia, nitrate, silica, carrion, labile and refractory detritus are also represented dynamically.

Data for biological parameters such as initial abundance, seasonal distribution, fecundity and timing of reproduction, growth and habitat preference, were obtained from a variety of sources including: the databases of the Central Ageing Facility, Fisheries Victoria; unpublished data (Fisheries Victoria, TAFI), the Fishbase database (www.fishbase.org); re-parameterised from ecosystem models that encompassed the study domain (Fulton et al. 2007) and literature on the region (Kuiter 1993, Gomon et al. 1994, Edgar 1997, Taylor & Willis 1998, Edgar & Barrett 1999, Greely et al. 1999, Ewing et al. 2003, Edgar et al. 2004, Lyle et al. 2004, Barrett et al. 2007). As a single set of biological parameters is used across the model domain, fitting of the model must be done simultaneously across each group and spatial area.

Table 1: Functional groups in VMPA Atlantis and their initial biomass values

Model Component	**Group Composition**	**Initial Biomass (t)**
Large phytoplankton	Diatoms	4975869
Small phytoplankton	Picophytoplankton	4448839
Gelatinous zooplankton	Salps, coelentrates	95029941
Large zooplankton	Krill and chaetognaths	2930676
Mesozooplankton	Copepods	1901186
Small zooplankton	Heterotrophic flagellates	95108706
Carvivorous infauna	Polychaetes	5417350
Deposit feeders	Holothurians, echinoderms, burrowing bivalves	2.42E+08
Deep water filter feeders	Sponges, corals, crinoids, bivalves	500249
Shallow water filter feeders	Mussels, oysters, sponges, corals	64833
Urchins	Echinoidea	6914
Deep water megazoobenthos	Crustacea, asteroids, molluscs	1500159
Shallow water megazoobenthos	Stomatopods, octopus, seastar, gastropod, crustacea	103891
Meiobenthos	Meiobenthos	6299170
Macroalgae	Macroalgae	417926
Seagrass	Seagrass	21352
Squid	*Sepioteuthis australis*, *Notodarus gouldi*	94893
Shallow water herbivores	e.g. *Girella tricuspidata*, *Liza argentea*, *Dactylophora nigricans*	51114
Banded morwong	*Cheilodactylus spectabilis*	511
Shallow demersal fish	*Pagrus auratus*, Labridae, *Chelidonichthys kumu*, *Pterygotrigla*, *Sillaginoides punctata*, *Zeus faber*	200654
Planktivorous reef fish	e.g. Atypichthys strigatus, *Enoplosus armatus*, *Trachinops caudimaculatus*	8020
Deep demersal fish	Oreosomatidae, Macrouridae, *Zenopsis*	68611
Zebra fish	*Girella zebra*	824
Silver sweep	*Scorpis lineolata*	284
Magpie perch	*Cheilodactylus nigripes*	584
Seahorses, pipefish, gobies	Syngnathidae, Gobiidae	11011
Herring cale	*Odax cyanomelas*	1823
Purple wrasse	*Notolabrus fucicola*	2978
Blue throat wrasse	*Notolabrus tetricus*	3817
Blue-eye trevalla, warehou	*Hyperoglyphe antarctica*, *Seriolella*	13919
Small pelagic fish	*Engraulis*, *Sardinops*, sprat	142795
Mackerels	*Trachurus declivis*, *Scomber australisicus*	34906
Shallow piscivores	e.g. Sphyraena novaehollandiae, *Arripis truttacea*, *Pomatomus*	281896
Migratory mesopelagics	Myctophidae	173976
Non-migratory mesopelagics	Sternoptychids, cyclothene	403340
Pink snapper	*Pagrus auratus*	30656
Tunas and billfish	*Thunnus*, *Makaira*, *Tetrapturus*, *Xiphias*	15158
Dogfish	Squalidae	476245
Demersal sharks	*Heterodontus portusjacksoni*, Scyliorhinidae, Orectolobidae	290432
Large Pelagic sharks	*Prionace glauca*, *Isurus oxyrunchus*, *Carcharodon carcharias*, *Carcharhinus*	106812
Dogshark	*Centrophorus*	184953
Skates and rays	Rajidae, Dasyatidae	17672
Baleen whales	*Megaptera novaeangliae*, *Balaenoptera*, *Eubalaena australis*	4423
Dolphins	Delphinidae	675
Orcas	*Orcinus orca*	1028
Seals	*Arctocephalus pusillus doriferus*, *Arctocephalus forsteri*	643
Abalone	*Haliotis*	5510
Prawns	*Haliporoides sibogae*	390124
Lobster	*Jasus*	21646
Seabirds	Albatross, shearwater, gulls, terns, gannets, penguins	261

Trophic Connections

Estimates for trophic availability were based on details in Fulton et al. (2007) and published information (O'Sullivan & Cullen 1983, Gales et al. 1993, Kuiter 1993, Gales & Pemberton 1994, Smale 1996, Edgar 1997, Bulman et al. 2001, Bulman et al. 2002, Uchikawa et al. 2002, Hume et al. 2004). The final values used were the result of estimated values modified through model calibration so that (i) the resultant realised diet composition matched the available data, and (ii) the time series trajectories generated by the model matched the available time series of observational data.

Fishing model

Fishing pressure was simply represented as a constant fishing mortality (F). The fishery mortality of fish, cephalopods and crustaceans from commercial fisheries was incorporated into Atlantis-SM using average daily catch values from annual catch statistics from 1990-2004 by both federal and state fisheries (DPIWE 2004, Fulton & Smith 2004). These values were modified slightly in the calibration process (within the underlying data uncertainty) so as to achieve a stable model biomass (i.e. with no evidence of numerical instability) that matched the available biomass trajectories that have been observed in the ecosystem over the past 10 years (Barrett et al. 2007).

Model calibration

Time series trajectories of both biomass and abundance of reef fish and some invertebrate groups were constructed from available observational data (Gilmour et al. 2005, Barrett et al. 2007). These time series showed 10 year biomass trajectories for the reef species, the first 6 years in these data sets were used to calibrate the trajectories of the reef groups in our model.

This calibration was done simultaneously across all model groups throughout the model domain using a modified form of pattern-oriented modelling ((Fulton et al. 2007, Kramer-Schadt et al. 2007). Previous sensitivity and factor analyses (Pantus & Dennison 2005, Fulton et al. 2007) have identified the most sensitive parameters and these are then calibrated using the following criteria (i) minimisation of deviation from observed biomass time series across all groups in all spatial boxes simultaneously, subject to the constraint that the shape of the time series must reflect the observed time series in the majority of boxes (this is because it is possible for a flat line to have a smaller deviation than a curve with the correct shape that has a small phase shift relative to the observations); and (ii) observed catches must be sustained without pushing any model group to extinction. For the groups where no time series data were available (e.g. the off-shore pelagic groups), biological parameters were calculated simply to achieve a stable ecosystem within the range of biomass values reported for these groups in the literature. The parameters modified in the calibration process included: predator and prey availability, mortality, reproduction and growth. Care was taken during calibration not to take any parameter beyond the plausible range of values defined by the literature (where available) or expert advice from researchers active in the region.

The “best fit” parameterisation presented in this paper is only one of a small set that are acceptable under the calibration criteria used. While only one (representative) parameterisation is presented here, the entire set of parameterisations were retained to form one dimension of any scenario analysis. A broader sensitivity analysis (not presented here) showed that model performance fell off substantially as the parameterisation moved further from the set of parameterisations found in the calibration process. The breadth of scales, data sources used and the clear degradation in performance outside the bounding set provide confidence that the calibrated parameter set do represent a structurally realistic set (at least in the local region of phase space). Previous work (Little et al. 2005, Fulton et al. 2007), has shown that it is sufficient when using models to provide broad strategic management advice. Over determined simulation models are not appropriate for tactical management (e.g. annual setting of total allowable catch), but they provide good platforms for exploring alternative broad management strategies and defining monitoring schemes in an adaptive management framework (Dichmont et al. 2000, Lassen & Medley 2000, Sainsbury et al. 2000).

Results

Comparisons with observations

A comparison of the modelled time series with survey data for the individual species groups is presented below (Figures 3-5). Plots are presented only for those locations where sufficient survey data were available for thorough comparison. It must be remembered that the fisheries component in the model is static, therefore we did not expect to see exact matches on an annual basis. Rather, the focus was on whether the overall directional trend in the biomass of the single species groups can be captured by the model. Where sufficient data was available, we have presented both calibration and validation output against the survey data. The six years of model calibration are presented as open squares in figures 3 and 4, while the validation data is presented as stars.

Habitat associated fish

Looking first at the reef fish (Figure 3), *Notolabrus tetricus* (blue throat wrasse) performed well in the Victorian coastal region, but less well in Maria Island and Port Phillip Bay . This was almost certainly due to the large variability seen in the survey results for the small regions of Maria Island and Port Phillip Bay, in comparison with the other coastal regions. *Notolabrus fucicola* (purple wrasse) showed the opposite trend to blue throat wrasse, in that the smaller regions of Maria Island and Port Phillip Bay performed better than the Victorian coastline, although again the large variability in the Maria Island survey data could not be reproduced by the model. The pattern holds that the model captures long term trends, but not necessarily interannual variation for all species.

Odax cyanomelas (herring cale) also showed mixed performance. The model performed best for this group in Port Philip Bay. A spike in 1999 on the Victorian coastline was not reproduced; and while the Maria Island data diverged from the modelled data early in the run, there was some convergence in relative biomass later in the simulation period. *Cheilodactylus nigripes* (magpie perch) showed a poor fit along the central Victorian coastline, which continued into the long-term model run trajectories. The Port Phillip Bay trajectories, however, showed quite a good match. *Girella zebra* (zebra fish) performed well in both of the Victorian regions for which there were sufficient survey data to make a comparison. Although there was a difference in the timing and rate of decline on the central Victorian coast, it did converge again after only a few years of observations. *Scorpis lineolata* (silver sweep) performed the poorest of all the reef fish groups, although there was only sufficient data to compare the model output to the survey results in the one region. The poor fit may be due to errors in the survey data rather than inaccurate model results. Survey data for this species is often variable and unreliable as a measure of total biomass within the system due to the schooling nature of this species. This behaviour means that divers swimming along survey transects will either encounter a school of silver sweep or none at all, therefore counts may be unrealistically low or high for the area. *Cheilodactylus spectabilis* (banded morwong) showed a similar trend predicted in both the Victorian coastline regions and the Maria Island region. A large spike in the observed biomass in Maria Island in 2005 was not reproduced by the model, but was regarded as an exception by the experts in the region and may have been a result of survey inaccuracies

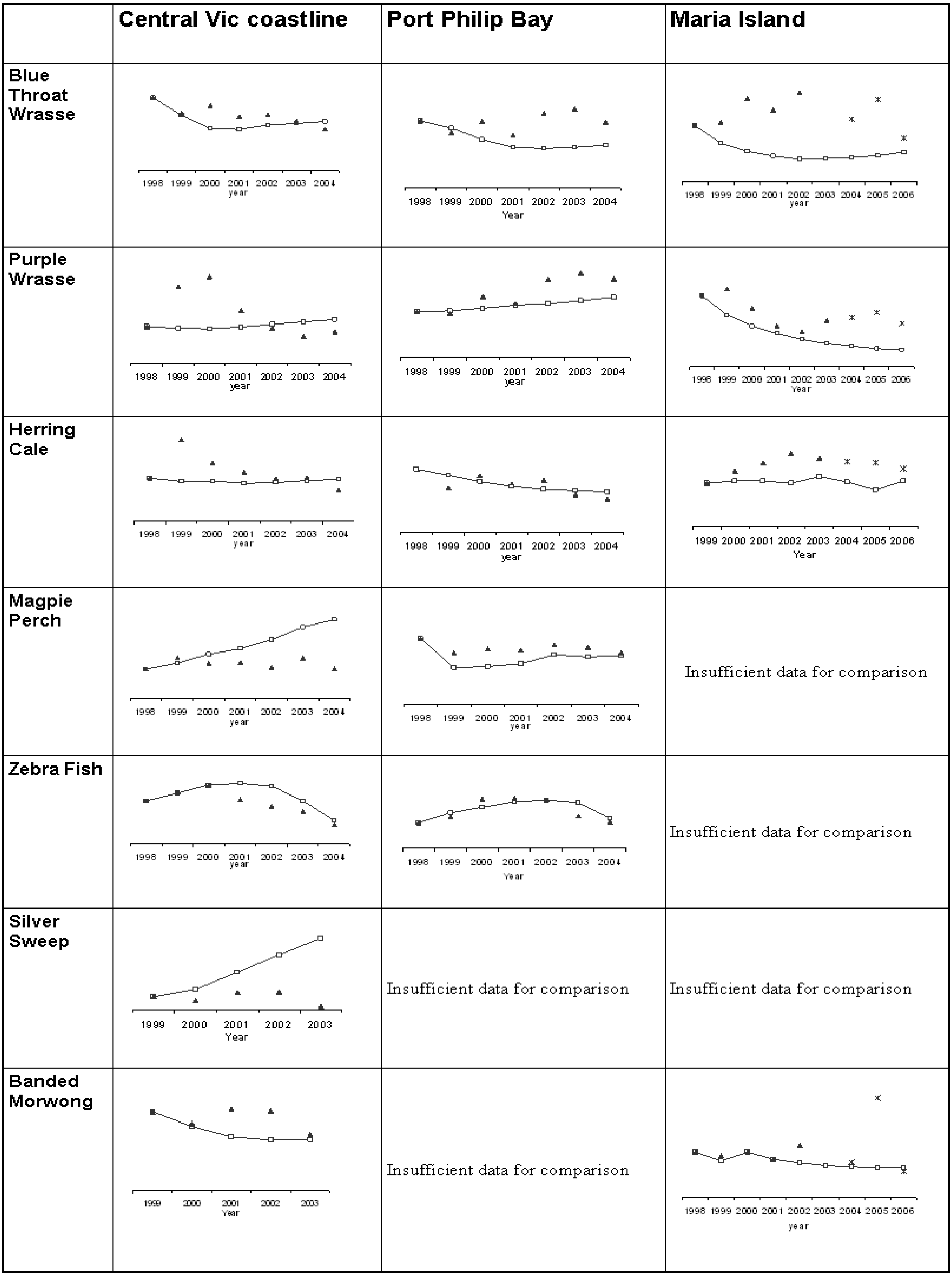


Figure 3. Comparisons of survey data (black triangles) and model time series and for reef fish. Open squares show model calibration years, black stars are test set data.

Invertebrates

The comparison of the modelled time series and survey data for invertebrates is shown in Figure 4. The modelled *Jasus sp.* (lobster) performed well against the Victorian coastline survey data, but diverged from survey data in the Maria Island box over time. *Haliotis sp.* (abalone) performed well in the Port Phillip Bay region, but less well in the Maria Island area, where a sudden drop in biomass between 2001 and 2003 was not reproduced by the model. There is a possibility that the decline in observed numbers in the Maria Island regions may have been a result of behavioural changes in abalone due to the increased size of lobsters in the area after marine park protection. The abalone may have become more cryptic and therefore less likely to be counted in diver surveys of the region (Barrett, pers. com.). This very fine scale change in behaviour is beyond the simple behavioural model used in Atlantis, although preliminary results from more refined representations does result in emergent behaviour such as crypsis, which leads to a concurrent fall in “observed” biomass (Fulton unpub).

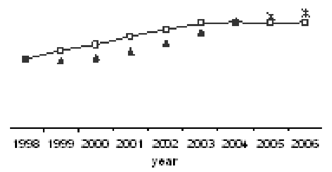
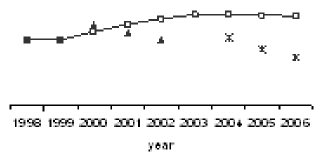
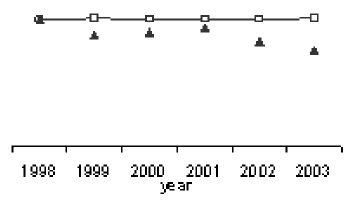
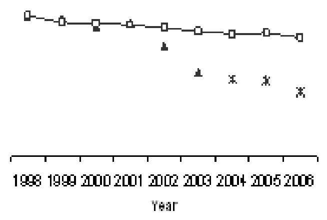
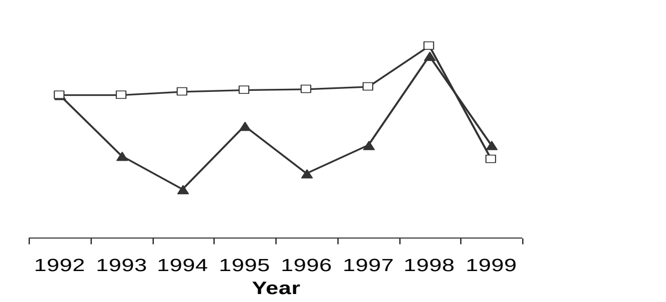
	Central Vic Coastline	Port Philip Bay	Maria Island
Lobster		Insufficient data for comparison	
Abalone	Insufficient data for comparison		

Figure 4: . Comparisons of survey data (black triangles) and model time series and for invertebrates. Open squares show model calibration years, black stars test set data.

Non-habitat associated fish

Detailed time series for the off shore, non-habitat associated groups was not available due to a lack of long-term survey data for these species. Some groups, however, had limited fisheries catch per unit effort (CPUE) data available. Although this data is subject to a high level of uncertainty and does not give highly reliable biomass trajectories, we have provided comparisons for two groups with the best available data. The deep demersal fish group and the demersal shark group is shown with a comparison of relative CPUE rates for the offshore areas (Figure 5). Although there was much more variation in the CPUE data, the trend shown by the modelled trajectories does not vary drastically from the overall CPUE trajectories.

Deep Demersal Fish



Demersal Sharks

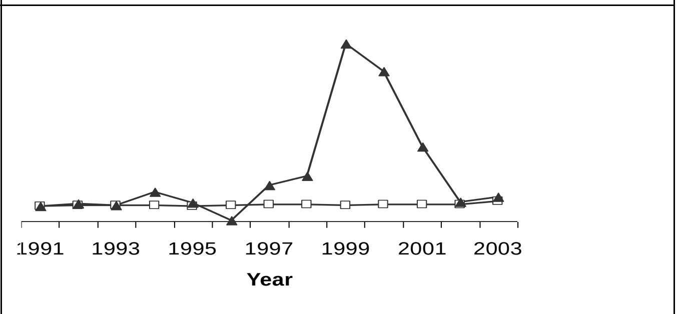


Figure 5: Comparison of CPUE (black triangles) and modelled time series (white squares) of relative biomass for off-shore groups.

Comparison with non-telescoped model

The Atlantis-SM model domain is a sub-domain from the larger model, Atlantis-SE (Fulton et al 2007). The Atlantis-SE model covered most of South Eastern Australia and it did also use polygons to represent the model geometry, it did not use the dedicated telescoping of box sizes used here. Consequently it could not consider the finer scale dynamics around specific spatial management zones (the intended use of Atlantis-SM). Moreover, Atlantis-SE had a deeper shelf to slope focus, although to allow for the representation of the juvenile life history stages of key commercial species shallower shelf species were included in the model. Thus the two models do overlap to some degree both ecologically and spatially and so it is informative to compare them to show the advantages presented by using the telescoping approach when spatial data uncertainty or particularly spatially detailed questions must be addressed within an ecosystem context. . Figure 6a shows the distribution of squid in Atlantis-SM, while Figure 6b shows the corresponding area of Atlantis-SE with its spatial distribution of squid. Similarly, Figure 7a shows the distribution of shallow demersal fish in Atlantis-SM, and Figure 7b shows the distribution in Atlantis-SE. There is broad agreement between the two models for the distributions of both squid and shallow demersal fish along the south east of Australia.

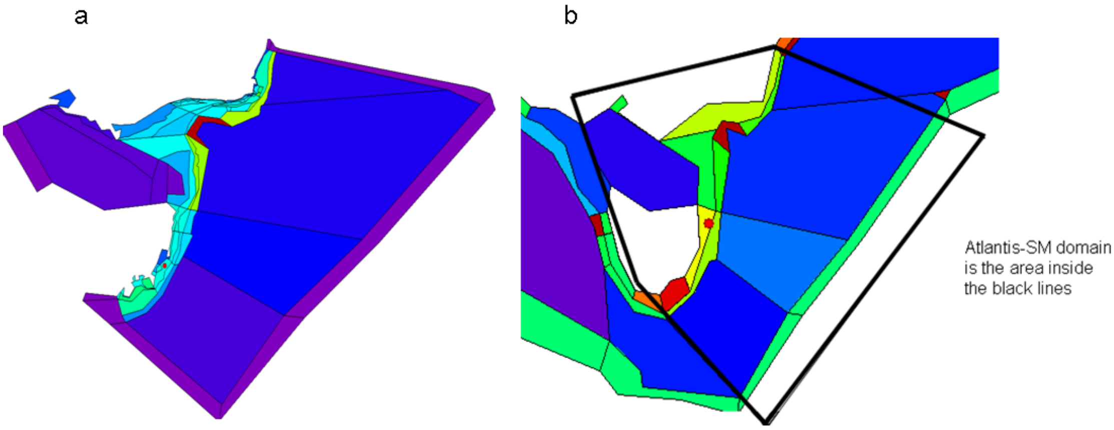


Figure 6. Distribution of squid in a) Atlantis-SM, and b) Atlantis-SE (colour scale is purple to red, with purple being lowest density and red being highest).

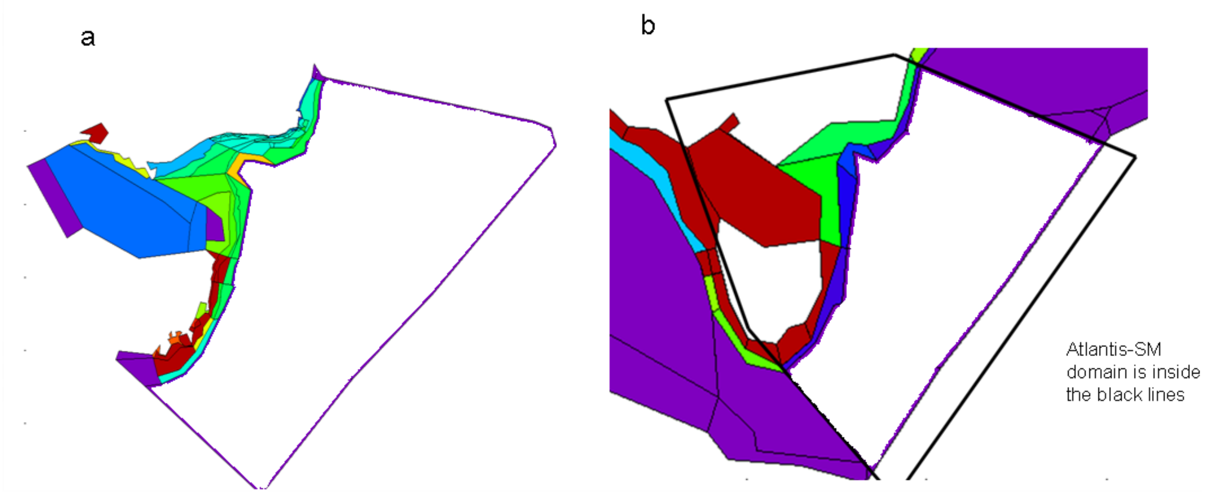


Figure 7. Distribution of shallow demersal fish in a) Atlantis-SM, and b) Atlantis-SE . White areas area either land or areas where shallow demersal fish do not occur. Colour scale is purple to red, with purple being the lowest density and red being the highest.

Figures 8a and 8b show the distribution of lobster in Atlantis-SM and Atlantis-SE respectively, with the Tasmanian east coast expanded to show the finer telescoping detail. Again there is broad agreement, however, the coastal portion of Atlantis-SM shows the variation from slope to coastal areas, whereas Atlantis-SE aggregates these areas, which leads to a loss of detail. However, by using the telescoped modelling approach of Atlantis-SM, a more detailed description of the cross shelf distributions of both groups is gained, with the subtle changes in density from the coast to the shelf much more apparent than in the Atlantis-SE model.

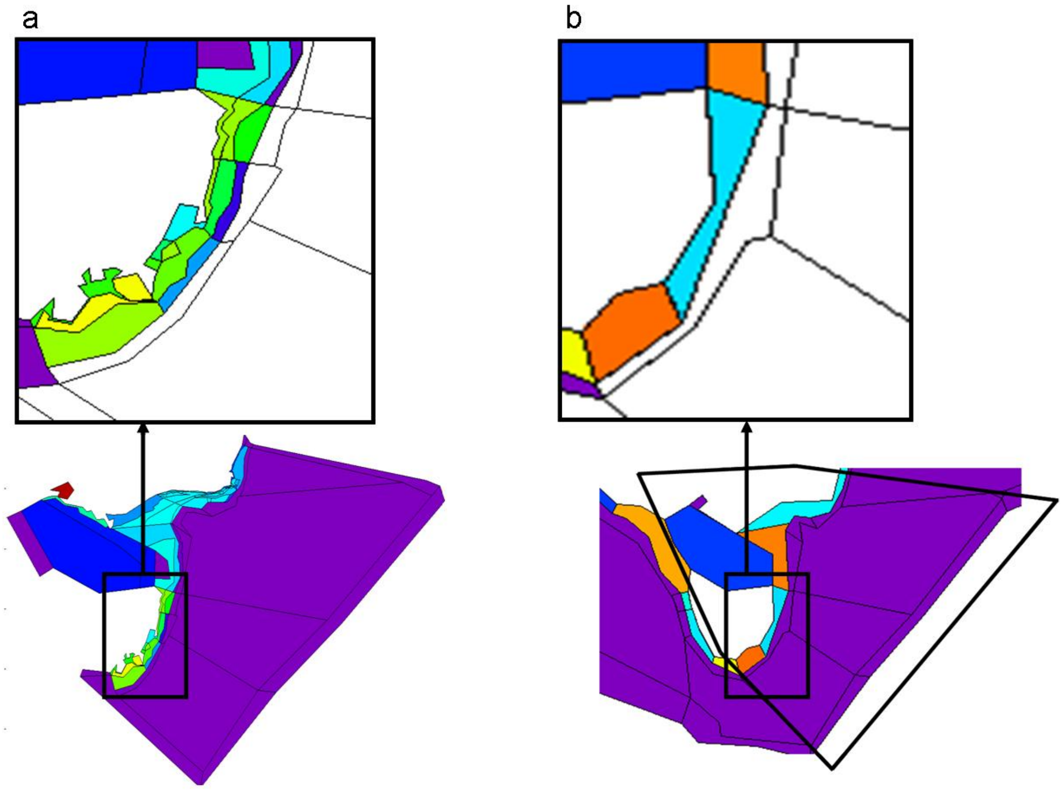


Figure 8. Distribution of lobster in a) Atlantis-SM, and b) Atlantis-SE. The East coast of Tasmania is expanded to show further telescoping detail. Colour scale is purple to red, with purple being lowest density and red being highest (Areas where lobster do not occur have been made white in expanded squares).

Not only does the telescoping approach allow much finer spatial resolution than the more coarse structure of Atlantis-SE, but it also enables location specific responses to be assessed. As the original purpose of the model was to assess indicators of spatial management, it was important to be able to capture differences in dynamics inside and outside of MPAs. Figure 9 illustrates location specific responses to the MPAs of lobster in this model. The time-series in the lower, left hand box shows an increase in lobster biomass with time in the MPA in comparison with the stable biomass in surrounding areas. On the other hand, the time series in the upper, right hand box shows a stable biomass of lobsters within the MPA with the surrounding areas showing a decline. These very fine scale, spatial differences cannot be captured in the more coarsely structured Atlantis-SE.

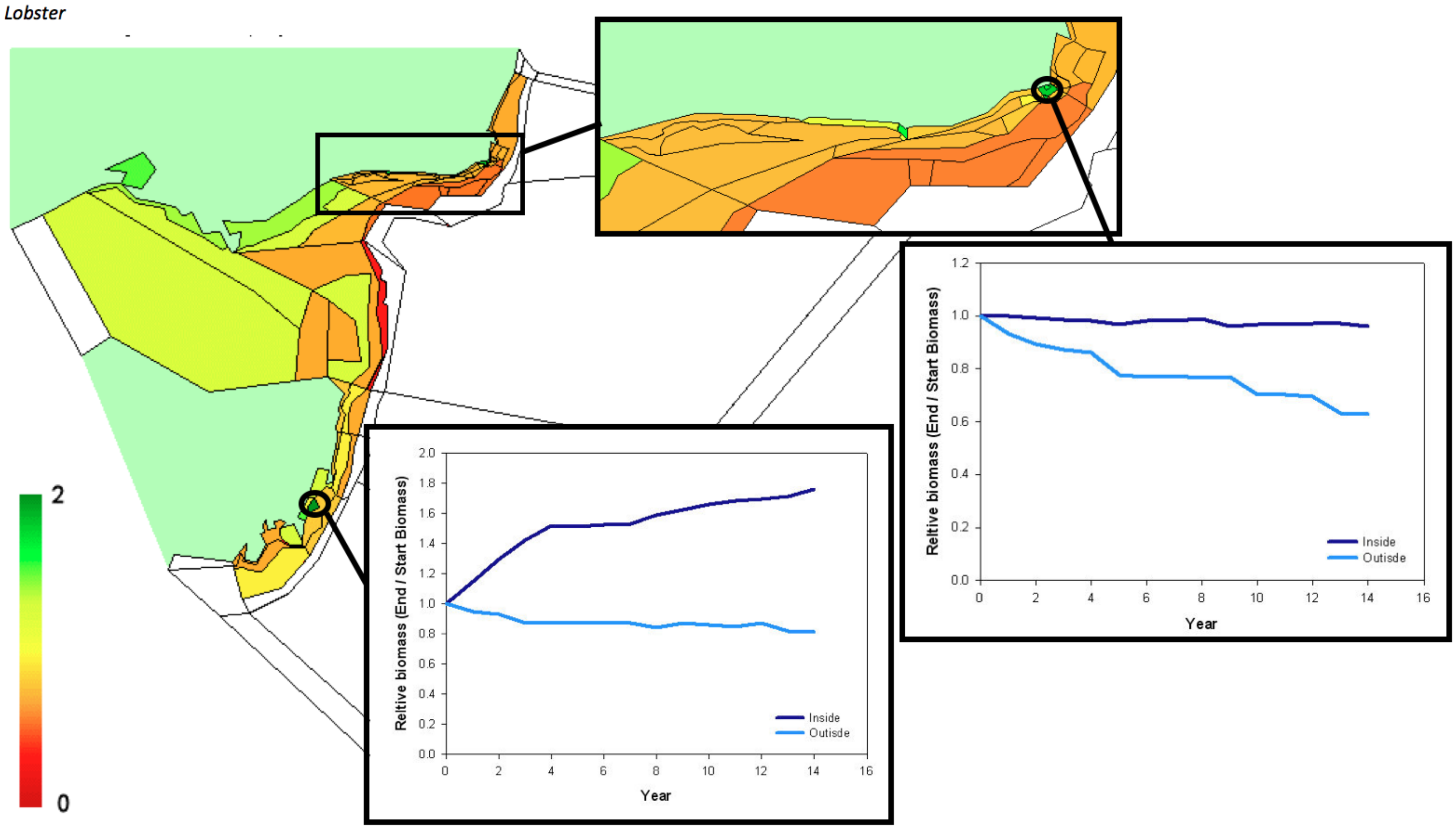


Figure 9. The relative abundance of lobster in each polygon at the end of the 20 year projection period

The general distributions and biomass dynamics of these groups (and others like them) in Atlantis-SE were of acceptable magnitude and form overall across the 3.7million km^2 that model covered (a much larger area than was the focus of Atlantis-SM), allowing it to be used for strategic whole-of-system MSE (Fulton et al 2007). Such general distributions are not sufficient though for addressing fine scale spatial management questions on the shelf, therefore a more detailed approach is obviously needed. Atlantis-SM shows that one viable solution is telescoping, which provides a higher level of detail in the areas of interest.

Discussion

The spatially structured telescoping approach described in this paper allows large, diverse ecosystems to be modelled reasonably well, although some limitations restrict the performance of the model. The static biological parameterisation across all regions, and the static fisheries parameterisation across time and space, meant that exact dynamics of many biological groups cannot be captured at all scales. Where data were available to support spatially-resolved biological parameterisation, we found that model performance improved. Similarly, forcing a realistic fisheries time series also improved model performance in the boxes and for the groups for which it was applied. However, since this detail of data cannot be supplied for all groups in all boxes, the extra complexity cannot generally be justified and so was not used in the standard runs presented here. Despite these issues, the reasonable performance for most species demonstrates that the telescoping approach can produce a comprehensive representation of processes across all scales relevant to regional management.

While our approach is appropriate for the purpose of this model (which is to support strategic management investigations around monitoring of spatial management), it will not be so for all purposes. In cases where a model is to be used for tactical management decisions, use of dynamic fishing pressure, as well as non-uniform parameterisation across all boxes would be justified. In all model development, the purpose of the model has to be clear at the outset so that an assessment can be made as to wether the extra expense (in terms of money, time and human resources) that would be incurred in the development of a more detailed model is necessary and therefore justified.

One difficulty that occurs when large polygons are used is the representation of realistic distribution and movement of both biological and physical components of the system (Lenser & Constable 2007). Although models such as ERSEM (Baretta et al. 1995) use a spatially structured 3D approach, biomass is considered to be uniform throughout each polygon. While this is not a problem in small polygons, or where biological components are planktonic and show small horizontal gradients, it may become problematic over large polygons, or where actively swimming animals are included (such as in Atlantis-SM).

The inclusion of parameters that influence or restrict movement between boxes and depth layers allows Atlantis-SM to move closer to a realistic representation of movement and distribution, even in the large polygons. The sub-grid model that defines habitat patches provides some representation of heterogeneity within these large polygons. Incorporating the habitat patchiness, in conjunction with habitat dependency parameters, allows Atlantis-SM to effectively capture dynamics such as density-dependent behaviour occurring at lower densities in boxes with low levels of appropriate habitat. This level of large- and fine- scale movement means that species' distributions can impact on trophic interactions, causing more realistic diet limitations and system dynamics. This approach differs from habitat preference parameters (as are represented in models such as Ecospace (Walters et al. 1999), where a biological group may be less likely to move to a less preferable habitat, but is not restricted from doing so.

The spatial dynamics included in Atlantis-SM include both horizontal and vertical structure. In comparison with 2D models, such as Ecospace (Walters et al. 1999) and Osmose (Shin & Cury 2001), the additional dimension of depth creates further spatial complexity, which in turn allows more detailed habitat preferences (ie, groups can have depth preferences), that can act as both refuge and to limit access to resources.

Regardless of the strategies used to represent distribution and movement in large polygons, it must be recognised that where a detailed representation of the dynamics and heterogeneity in a small area is required, then the more effective technique is to use fine-scale spatial structure (Fulton et al. 2004). Large polygons are useful as a compliment, for representing either relatively homogenous areas, or areas where capturing the differences within the box is not necessary for the questions being addressed. This is where the most valuable aspects of the telescoping approach lie; via the representation of ecological processes external to the coastal system, but which are none-the-less drivers of that system, without incorporating excessive spatial complexity.

By using polygons based on observed bioregional structures, the telescoped modelled domain represents the heterogeneity found along coastal regions, and also benefits from reduced computational demands in the ocean environment where species distributions are not so strongly constrained spatially. In this way we have been able to create a model of a large area without unnecessarily increasing the level of spatial complexity in the offshore regions, thus tailoring it according to both the data available, and the dynamics of each part of the system that are relevant .

Spatial telescoping provides a means to manipulate spatial complexity in a model in order to suit both the data availability and the system dynamics, whist also facilitating the inclusion of important large-scale ecosystem impacts. By using this technique we are more likely to find a balance between the level of model spatial complexity, and the inclusion of important, ecosystem drivers at both large and fine scales.

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Appendix G

Observational methods used in marine spatial monitoring

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Hannah Murphy and Gregory P. Jenkins

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Executive Summary

Observational methods used in marine spatial monitoring studies include both fishery-independent and –dependent techniques. Fishery-independent spatial monitoring techniques include underwater visual census, underwater video, remote sensing, acoustics, and experimental catch and effort data. Fishery-dependent data include catch, effort, and catch per unit effort calculated from fisher surveys, catch landings, and gear types used. Both fishery-independent and -dependent data are used to determine the abundance and distribution of target species in and around management areas. This review summarises the applications, advantages, and biases of each of these observational categories.

Combining observational methods has been shown to be an effective means of enhancing our ability to monitor marine ecosystems. At the same time, there is also a need for the continued development of non-intrusive monitoring technology.

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Introduction

Observational methods currently used in marine spatial monitoring studies include underwater visual census, underwater video, aerial and satellite images, acoustics, and fishing catch and effort data. While these forms of observational techniques provide the framework for the majority of monitoring studies, various adaptations and combinations of these techniques are also used, including baited remote video, acoustic tagging, and combining, for example, underwater visual census and video techniques.

Observational methods can be classified as fishery-independent if they rely on research monitoring, or fishery-dependent if the monitoring information comes directly from the fishery. Some observation methods are used in both fishery-dependent and fishery-independent situations.

This review provides a summary of the current observational methods used in spatial monitoring studies, including their advantages and biases. The review also identifies additional observational techniques and adaptations of current techniques that could enhance our ability to monitor marine ecosystems.

Results

Fishery-independent spatial monitoring

Underwater visual census

Underwater visual census (UVC) is a fishery-independent technique used to quantify the distribution, cover, species richness, abundance, and sizes of flora and fauna in shallow marine habitats (Edgar *et al.* 2004a; Watson and Harvey 2007). UVC is a fast, cost-effective, non-intrusive means of obtaining data for monitoring studies.

Transect surveys

There are three types of transect survey used in UVC: strip (belt) transect, time transect, and line transect (Edgar *et al.* 2004a). Strip transect, where the diver estimates the density of the target species by swimming along a strip of known or estimated length and width, is the most widely used technique (Edgar *et al.* 2004a). Strip transects are commonly used for fish surveys, where both species identification and total species length are determined by one or two divers who swim at a constant speed while sweeping their eyes from side to side in front of them (eg. Russ and Alcala 1996; Davidson 2001; Russ *et al.* 2005; Willis *et al.* 2006). Fish identification is usually to the species level and total length estimates can vary from broad categories of small, medium, large (Claudet *et al.* 2006; Azzurro *et al.* 2007) to size classes (Edgar *et al.* 2004a; McClanahan *et al.* 2007) or absolute size estimates (Russ *et al.* 2005). Strip transects are also used in tag-recapture monitoring surveys, where divers in and around management areas count and measure tagged animals in order to obtain information on their mobility patterns (eg. Chapman and Kramer 1999; Cole *et al.* 2000). Strip transects have also been used to survey spiny lobsters (Kelly *et al.* 2000; Davidson *et al.* 2002; Shears *et al.* 2006), macroinvertebrates (Edgar *et al.* 2004b), and scallops (Beukers-Stewart *et al.* 2005).

Time transects allow for a diver to record the number of animals sighted during a specific time interval (Edgar *et al.* 2004a). Time transects are used to estimate relative abundance rather than density of target species, as time transects are not constrained within a specified width or along a line (Syms 1995; Edgar *et al.* 2004a; Cox and Hunt 2005). Time transects can be used for monitoring gregarious animals with patchy habitat distribution, such as spiny lobsters (Cox and Hunt 2005), small, cryptic fish (Syms 1995; Beldade *et al.* 2006), and gastropods (McClanahan 2002).

Line transects (also called distance sampling), where the diver swims along a line and estimates the distance and direction of the animal from the transect line, is occasionally used to survey fish in management areas (Kulbicki and Sarramegna 1999; Edgar *et al.* 2004a). Divers are not restricted by transect width and can count as many fish as possible while recording the distances of the sightings. This method takes into account the distribution of distances of fish species from the transect line, which may increase both the counts and the detection of vagile or cryptic species that are usually missed in the restricted confines of the strip transect (Kulbicki and Sarramegna 1999).

Harvey *et al*. (2004), however, found that divers could possibly underestimate by 82% and overestimate up to 194% the actual area surveyed, which would influence estimates of fish densities. Consequently, line transects are not as widely used for density counts of mobile fauna as strip transects (but see: Amand *et al.* 2004; Ferraris *et al.* 2005; Pet *et al.* 2005; Pink *et al.* 2007).

Line transects are also used to quantify the percentage cover of coral (e.g. Cellier and Schleyer 2002; Nadon and Stirling 2006; Leujak and Ormond 2007), relative abundance of abalone (Miner *et al.* 2006), and general habitat features (Russ *et al.* 2005). There are two variations of the line transect used for surveying sessile flora and fauna: line-intercept transects and line-point transects. Line-intercept transects involve lying a transect line over the substrate, and the percentage cover of target species are estimated as the fraction of the total length of the transect line that crosses over each species of interest (Leujak and Ormond 2007). Line-point transects are a quicker method to quantify percent cover than line-intercept transects as the substrate is only sampled under specific points along the line transect, for example every 10 cm or 1 m (Miner *et al.* 2006; Leujak and Ormond 2007). Leujak and Ormond (2007) found that line-intercept and -point transects over-estimated the percent cover of some species; however, these two techniques are quick, cost-effective means of determining percent cover.

Other forms of UVC
Point counts are another commonly used survey method for estimating fish densities. This technique is much faster than transect surveys, as only one diver is needed to scan 360° from a fixed point or while descending to a fixed point (Edgar *et al.* 2004a). The radius of point counts range from 5 m to 15 m, with the most common being 10 m. Point count surveys are normally timed, ranging from 10 - 15 minutes, and species identification and length estimates are also recorded. This technique allows for increased replication at monitoring sites.

In the rapid visual technique (RVT), the diver ranks species in order of encounter in a timed sampling session (Demartini and Roberts 1982; Edgar *et al.* 2004a). The diver swims in a zigzag fashion for a specific period of time and records all encountered species (Demartini and Roberts 1982). Variations of RVT have been applied to snail censuses (McClanahan and Muthiga 1992; McClanahan 2002) and fish censuses (Demartini and Roberts 1982; Schmitt *et al.* 2002). Schmitt *et al*. (2002) found that RVT works well in conjunction with strip transects as RVT can be used to census shy and cryptic fish species while strip transects can census the other fish species.

Quadrat surveys are used for benthic organisms, such as sea urchins (Shears and Babcock 2003; Davis 2005; Guidetti 2006), sea cucumbers (Schroeter *et al.* 2001) and coral recruits (Cinner *et al.* 2006), and used in general habitat surveys to determine percentage cover of flora and fauna (Friedlander *et al.* 2003; Pillans *et al.* 2007; Santin and Willis 2007). Quadrats can be set up along a line transect, or they can be either fixed or haphazardly placed in an area of interest. Quadrat sizes range from 0.02 m^2 –5 m^2, depending on the survey criteria and size of the flora and fauna of interest. Quadrats can be a more time consuming method of monitoring sessile flora and fauna than line-intercept transects, especially when quadrats are hand-mapped by a diver and all species are measured and identified (Leujak and Ormond 2007). But using video and photographs of quadrats can decrease the time needed in the field to count and measure target species (Leujak and Ormond 2007). Once in the laboratory, a point counts technique can be applied to the video and photography images, so only substrate that falls under a set of random points are counted to determine percent cover (Ryan *et al.* 2007). Also, tracing the outlines of the substrates in the images can be used to calculate area (Ryan *et al.* 2007).

Several studies have used modified UVC techniques in order to suit a target species' life history traits. Stewart and Beukers (2000) used a baited point count to count cryptic, piscivorous fish. Pulverized bait was dropped into the middle of a circle, which was marked by glow tape to reduce diver's bias, and the diver counted cryptic fish that entered the radius for 15 minutes (Stewart and Beukers 2000). This method counted significantly more cryptic fish than strip transects. Gilbert *et al*. (2005) used a modified strip transect to census moray eels (a nocturnal predator). Transects were 25 m long and 5 m wide and the diver actively searched the strip by zigzag swimming and searching in all the habitat crevices. Milazzo *et al*. (2005) used a circular transect technique because of the rugged topography in the marine protected area (MPA). A diver swam in a 10 m radius from the centre of the reef, rather than along a straight line (Milazzo *et al.* 2005).

Comparison of UVC methods
UVC survey methods have been compared in order to evaluate which method gives the best estimate of species densities. Strip transects and point counts have been compared often as they are both commonly used survey methods. Buxton and Smale (1989) found that point counts were better suited for mobile species while strip transects were better suited for sedentary species. Samoilys and Carlos (2000) found that density estimates were no different between these two methods, but that point counts were faster, especially when the radius is 7 to 7.5 m. Colvocoresses and Acosta (2007) found that strip transects detected a greater species richness but point counts detected higher species densities, and Consoli *et al*. (2007) found that the mean number of species recorded by strip transects were greater than point counts, but that the mean abundance of fish species was higher using point counts. The findings from these comparison studies indicate that the use of strip transects or point counts should depend upon the suite of species studied and the question asked by each study.

Other UVC methods have also been compared. Nadon and Stirling (2006) compared strip transects, line transects, and line-point transects in monitoring coral reef status, and found that line-point transects were the most cost-effective and quickest monitoring technique. Baron *et al*. (2004) compared strip transects, point counts, and RVT, and found that while RVT identified a more complete species list, strip transects recorded higher species abundances and densities compared to point counts. Pilot studies that compare UVC techniques may provide site- and species-specific information on the most efficient survey technique for a spatial monitoring study.

Biases of UVC

The biases of UVC are numerous and well documented (Watson and Harvey 2007). Biases include under or over-estimation of the size-structure of fish populations (Edgar *et al.* 2004a), observer's experience (Williams *et al.* 2006), temporal variability of fish species (Willis *et al.* 2006), under or over-estimation of species density (Kulbicki 1998; Willis 2001; Edgar *et al.* 2004a), and the influence of a SCUBA diver on fish behaviour (Cole 1994; Watson and Harvey 2007). Edgar *et al*. (2004b) found that divers' estimates of fish lengths are 7% greater than measured lengths, with an increasing tendency to inaccurately estimate fish lengths as they deviated from 300 mm. Attempts to reduce this bias include intensive training of divers using plastic or wooden models of varying sizes and shapes (e.g. Chapman and Kramer 1999; Davidson 2001; De Girolamo and Mazzoldi 2001; Ordines *et al.* 2005; Williams *et al.* 2006) and using scale-bars or rulers attached to the slate that can be used in the field to compare observed fish estimates with a calibration source (Buxton and Smale 1989; Edgar *et al.* 2004a; Barrett *et al.* 2007). Obtaining accurate and precise fish length data is important in estimating fecundity or biomass of fish species, and small errors can have a large effect on weight estimates (Harvey *et al.* 2002a; Watson and Harvey 2007). Using well-trained divers and the same divers for all UVCs in a study helps reduce observer's bias (Williams *et al.* 2006). In order to reduce biases in species density counts, small, cryptic species can be excluded from species counts (e.g. Garcia Charton *et al.* 2002), as it has been shown that UVC underestimates cryptic species densities. It may also be advantageous to do pilot studies of UVC methods to determine which survey method has less impact on the behaviour of the target species (De Girolamo and Mazzoldi 2001). Ultimately, however, the aversion or attraction of fish to divers is dependent on the species and its previous encounters with divers. An alternative census method, such as remote video deployment, may be a more appropriate survey technique for specific target species.

Video

Video is a non-extractive monitoring method for marine flora and fauna. Video provides similar monitoring advantages of UVC, including speed, non-intrusiveness, and repeatability, while avoiding some of UVC's inherent biases (Cappo *et al.* 2003).

Underwater video

Underwater video is used to reduce the impact of divers on fish behaviour (Abdo *et al.* 2006) and to survey areas at depths and times that are dangerous for divers (Jury *et al.* 2001; Cappo *et al.* 2003; Morrison and Carbines 2006). Like UVC, underwater video is used to monitor habitats of interest by providing data on species abundances, lengths, and taxonomy. Underwater video has been deployed remotely and used in a handheld version by divers. In remote deployments, underwater video cameras have been attached onto sleds and towed behind boats (Stoner *et al.* 2007), attached to remotely operated vehicles (Rochet *et al.* 2006), and attached onto submersibles (Collins *et al.* 2002) to provide spatial and density data on deep-water flora and fauna. Remotely deployed underwater video has also been used to obtain behavioural data on marine fauna, including nocturnal data on sleeping snapper (Morrison and Carbines 2006), predator/prey interactions of lobsters (Mills *et al.* 2005), and deployed near a reef to continuously monitor fish behaviour (Jan *et al.* 2007).

The limitations of remotely deployed video include limited visibility due to turbidity (Cappo *et al.* 2003) and difficulty in measuring fauna that are not orthogonal to the camera (Harvey *et al.* 2002b). The difficulty of measuring faunal lengths has been addressed by using stereo-video techniques (Harvey *et al.* 2002b). Stereo-video uses two cameras to record synchronous pairs of frames with a calibration rod or grid to measure the length of the fish present in the frame (Harvey *et al.* 2002b; Costa *et al.* 2006). Technology is constantly improving and the use of remote underwater video systems will become more prevalent as traditionally shallow-water fisheries are moving further offshore due to the depletion of inshore stocks (Watson *et al.* 2007).

Handheld underwater video cameras are used by divers during transect surveys, and allow divers with limited survey skills to conduct monitoring surveys. Video transect surveys are faster than UVC surveys, and permanent records of the surveyed flora and fauna can be used for future reference (Nadon and Stirling 2006). However, underwater video requires extensive time after the survey to analyse the video images, the lens has a restricted field of view compared to the human eye, the video camera records a large number of zero counts of fish, and it is difficult to measure fish length when the fish is not orthogonal to the camera (Cappo *et al.* 2003; Costa *et al.* 2006; Dunbrack 2006).

Baited Underwater Video

The use of baited underwater video (BUV) in monitoring studies decreases the occurrence of zero counts and increases the similarity between surveys of predator/scavenger marine fauna (Cappo *et al.* 2003), and additionally allows for an assessment of predator/scavenger behaviour (Cappo *et al.* 2003; Watson and Harvey 2007). Like remotely deployed underwater video, BUVs commonly use stereo-camera techniques in order to obtain 3D images of marine fauna (Shortis *et al.* 2007). In the more recent literature, baited remote underwater video stations (BRUVS) are used in monitoring studies. BRUVS are self-contained structures that record data for, on average, 30 to 45 minutes before being retrieved from the sea floor (Shortis *et al.* 2007). The time for the bait plume to diffuse is estimated at 20 to 30 minutes, but not a lot is known about bait plume mechanics, which affects density estimates of marine fauna (Watson *et al.* 2005; Harvey *et al.* 2007; Malcolm *et al.* 2007). Numerous BRUVS can be dropped and left in a desired habitat approximately 350-400 metres apart, allowing replicates to be independent of each other (Cappo *et al.* 2004).

There are biases and unknown parameters involved in BUV deployment. BUVs are biased towards species that are attracted to bait, such as predators; however, BUV have recently been shown to survey herbivorous and planktivorous fish equally well as non-baited underwater video systems (Harvey *et al.* 2007; Malcolm *et al.* 2007). More information is needed on bait plume mechanics and inter- and intra-specific competition at the bait (Malcolm *et al.* 2007). The development of a software package to process and measure video images, in order to reduce time spent in the laboratory after deployment, is greatly needed (Shortis *et al.* 2007).

BUVs are commonly used to monitor spatial changes in species richness, abundance, and biomass between management areas and fished areas, especially in habitats that are too deep for diver surveys and for target species that are difficult to survey with UVC due to behavioural characteristics (Malcolm *et al.* 2007). Some recent applications of BUV include monitoring the change in snapper populations after the implementation of MPA status (Willis and Babcock 2000; Willis *et al.* 2003; Denny *et al.* 2004); monitoring the changes in fish assemblages between MPA and fished areas (Watson *et al.* 2007; Kleczkowski *et al.* 2008); and providing information on spatial (Cappo *et al.* 2007; Malcolm *et al.* 2007) and temporal trends (Malcolm *et al.* 2007) of reef fish assemblages. BUVs have been adapted to float in the water column in order to obtain behavioural information and abundance estimates of pelagic fish (Heagney *et al.* 2007), and have proven to be a simple, robust method for detecting patterns in spatial trends of predators and scavengers in marine habitats (Cappo *et al.* 2007).

The quality of the data obtained from BUVs has been compared to data acquired from other common observational survey techniques, such as UVC and non-baited video systems. The main conclusions from comparisons of UVC and BUV are that:

- BUV should be used to compliment UVC or used when UVC is not available as BUV have inherent biases towards predators/scavengers (Stobart *et al.* 2007)
- UVC, BUV, and non-baited camera survey methods should be combined in order to counteract their inherent biases (Watson *et al.* 2005)
- choosing a specific survey method, such as UVC, BRUVS, or angling, should depend on the life history traits of the target species (Willis *et al.* 2000).

Remote sensing and spatial monitoring

Marine habitat monitoring by satellite imagery and aerial photography is limited to water depths where light can penetrate the water column and reflect back to the optical sensor (wavelengths: blue, green, red) (Hernandez-Cruz *et al.* 2006; Ball and Blake 2007). Because of this limitation, extensive ground-truthing of the images using observational methods, such as UVC and underwater video, is used for identification and classification of the remotely-sensed image.

Satellite Imagery
Satellite sensors that have been used in spatial monitoring studies include IKONOS (1-4 m resolution) (Yamano *et al.* 2006; Lu and Weng 2007) and Landsat ETM+ (30 m resolution) (Andrefouet *et al.* 2005; Lu and Weng 2007). It is important to correct satellite images for water depth, water quality, and atmospheric effects to compare images over time and between locations (Andrefouet *et al.* 2002).

High resolution satellite sensors, such as IKONOS, can provide images with high spatial detail of remote, inaccessible reef habitats (Andrefouet *et al.* 2003). However, the low spectral resolution of multispectral satellite sensors (3 water-penetrating wavebands and 1 infrared waveband) reduces their ability to monitor coral bleaching events as the spectral signature of dead coral and sand is similar (Yamano and Tamura 2004) and the spectral signature of dead coral covered with macroalgae is very similar to live coral (Clark *et al.* 2000). There is a need for a hyperspectral space-borne sensor that is specifically designed for coral reef monitoring, which will allow for the identification of coral to the species level without additional field work (Hochberg *et al.* 2003; Kutser and Jupp 2006; Kutser *et al.* 2006).

Regional- scale Landsat sensors produce spatial images of coral reefs and seagrass meadows at landscape scale, but Landsat sensors are less able to monitor changes at the local scale, due to their coarse resolution (Andrefouet *et al.* 2001; Ferwerda *et al.* 2007). Since images have been archived from the 1970s (Ferwerda *et al.* 2007) and have been shown to accurately map live coral, sand, seagrass, and rubble (Andrefouet *et al.* 2001; Mumby *et al.* 2004; Mishra *et al.* 2006), Landsat has been able to monitor changes in coral reef geomorphological structure (e.g. Vanderstraete *et al.* 2006) and changes in seagrass meadow stands (e.g. Gullstrom *et al.* 2006) over time. Landsat images are affordable, provide objective data compared to potentially biased UVC or video surveys, and are useful for landscape-scale spatial monitoring projects (Call 2003; Dekker *et al.* 2005; Benfield 2007).

Aerial Photography
Aerial photography has been used to map and monitor seagrass meadows, other forms of submerged aquatic vegetation (SAV), such as macroalgae, and coral and rocky reefs. These habitat types are often included in management areas, due to the presence and extent of the habitat type itself or as a habitat for a specific species (Frid *et al.* 2008). Protecting these marine habitats has been recognized as a key element in ensuring ecological sustainability (Frid *et al.* 2008).

Aerial photography, with its high resolution images (<1 m) and targeted acquisition times to correspond with optimum weather and tide conditions, is the preferred method for remotely sensing the spatial extent of seagrass meadows (e.g. (Kendrick *et al.* 2000; Leriche *et al.* 2006; Ball and Blake 2007). However, aerial photography is not able to differentiate between seagrass species, as epiphytes on seagrass can mask subtle spectral differences between seagrass species (Ball and Blake 2007). Historical aerial photographs can provide a baseline for seagrass monitoring studies (e.g. Ball and Blake 2007)) and historical images provide high spatial and temporal replication at a variety of study sites (Hernandez-Cruz *et al.* 2006).

Aerial photography has also been used to monitor the short and long-term health of remote coral reefs. For example, aerial photography was used to assess a large-scale coral reef bleaching event in the Great Barrier Reef, but the images underestimated the extent of coral reef bleaching as dead coral covered in macroalgae and live coral were not distinguishable in the aerial images (Berkelmans and Oliver 1999; Andrefouet *et al.* 2002). The aerial photographs used for these studies had a resolution of 2 m, where a finer resolution could have provided more information on individual bleaching events (Andrefouet *et al.* 2002). Historical aerial photographs have also been used to monitor changes in coral reef ecosystems; for example, Lewis (2002) determined that there was a statistically significant reduction of reef area of the fringing reefs of Barbados since the 1950s.

Acoustics
While satellite images and aerial photographs are limited by water depth and spectral wavebands, acoustic techniques provide an alternative method to monitoring deep water habitats and their associated biological assemblages. Acoustic methods can also be used to monitor the movement of animals in relation to spatial management.

Single-Beam Sonar
Ground-discriminating, single-beam echosounders, such as RoxAnn and QTCView, produce habitat maps based on the reflective characteristics of the seabed floor (Kenny *et al.* 2003). Acoustic Ground Discriminating Software

(AGDS) integrates components of the returned echoes to extract three indices (depth, roughness, and hardness) to map seabed features (White *et al.* 2003).

Echosounders have been used in a variety of monitoring studies of management areas and habitats of interest. For example, AGDS and single-beam echosounders have been used to monitor marine conservation areas along the coast of the United Kingdom (Brown *et al.* 2005) and scientific echosounders in conjunction with bottom trawls have enabled researchers to quantify the abundance and distribution of snipefish in Portugal (Marques *et al.* 2005). In addition, the acoustic ground discrimination system QTCView has been integrated with a fisheries scientific echosounder to provide details on changes in the seabed while the echosounder simultaneously recorded temporal changes in the distribution of the target fisheries species (Freeman *et al.* 2004).

Integrated acoustic techniques can be used to monitor target species and their environment in deep, inaccessible habitats (Freeman *et al.* 2004). The limitations of single-beam echosounders, however, including incomplete seabed coverage and high variability between surveys due to differences in vessel speed, restricts the use of single-beam echosounders as a monitoring tool (Lindenbaum *et al.* 2008).

Side-scan Sonar

Side-scan sonar is a broad-acoustic beam (swathe) system that produces a wide area, continuous high resolution image based on returning echoes from the seabed floor with information on sediment texture, topography, and bedforms (Kenny *et al.* 2003). The main advantage of side-scan sonar is its ability to generate a photo-realistic picture of the seabed (Kenny *et al.* 2003), which can then be used to identify habitat features of interest. Side-scan sonar is attached to a tow fish and towed at a constant height above the seafloor, which allows the sonar to be independent of water depth.

Side-scan sonar has the potential to be used as a spatial monitoring tool for deep water habitats. It has been used to accurately map the spatial patchiness of oyster beds (Allen *et al.* 2005); as a monitoring tool to assess the 3D structure of a coral reef after a bleaching event (Collier and Humber 2007); has mapped abalone habitat where it was too deep for divers to conduct surveys (Butler *et al.* 2006); and, has mapped MPAs (Cochrane and Lafferty 2002). Side-scan sonar maps provide clear and spatially accurate images of the true patchiness of a given habitat (Allen *et al.* 2005), and, therefore, have the potential to be used for monitoring a variety of marine species and habitats.

Acoustic camera

Dual-frequency identification sonar (DIDSON) uses sound to produce near video-quality images of fish at ranges up to 15 m from the camera at high frequency settings and 40 m from the camera at low frequency settings (Holmes *et al.* 2006). DIDSON cameras emit horizontal, line-focused beams that produce images at sufficiently high resolution to allow for the identification of different classes of objects, including fish and structures (Moursund *et al.* 2003; Holmes *et al.* 2006). DIDSON acoustic cameras can record up to 7 frames a minute, which allows for real time observations of fish movement and behaviour (Moursund *et al.* 2003). The limitations of the DIDSON camera include a null zone in front of the camera, which occurs when the object directly facing the camera is lost from view, and the need for a flat habitat as large structures can block the beams (Rose *et al.* 2005).

This system provides high resolution images of multiple targets and accurate direction of travel information (Holmes *et al.* 2006) and has been shown to be a very effective tool for fisheries assessments where conventional underwater video is limited by low light and high turbidity (Moursund *et al.* 2003). The DIDSON camera could therefore be used in small-scale monitoring studies in and around deep, turbid management areas.

Acoustic tagging

Acoustic tagging is used to determine home-range behaviour, movement patterns, migration, use of space, diel activity patterns, and site fidelity of marine animals (Lowe *et al.* 2003; Parsons *et al.* 2003; Egli and Babcock 2004; Heupel *et al.* 2006; Meyer *et al.* 2007b). Acoustic tagging can also be used to evaluate the effectiveness of management areas in providing protection to target species and spillover to surrounding fished areas (Eristhee and Oxenford 2001; Zeller *et al.* 2003; Egli and Babcock 2004; Starr *et al.* 2005; Pecl *et al.* 2006) and three main methods are in common use, as described below.

To obtain fine-scale movement data over a short period of time, active tracking of the acoustically tagged animals can be done from a boat using directional hydrophones and acoustic receivers (e.g. Zeller 1998; Meyer *et al.* 2000; Starr *et al.* 2002; Lowe *et al.* 2003; Wetherbee *et al.* 2004; Popple and Hunte 2005; Afonso *et al.* 2008).

A passive monitoring system for fine-scale movement data of high accuracy is the radio acoustic positioning telemetry (RAPT) buoy system (Jackson *et al.* 2005; Jorgensen *et al.* 2007). The RAPT buoy system reports on individually coded, acoustically tagged animals within the RAPT system range to provide continuous data on residency and movement patterns (Jackson *et al.* 2005; Jorgensen *et al.* 2007). Acoustic receivers on each buoy communicate with a base station by radio signals, or alternatively by cable, and provide positioning information of very high accuracy (± 2 m) (Jorgensen *et al.* 2007).

For large-scale tracking of animals for up to two years, the majority of spatial management studies use independent acoustic receivers to record the movement patterns of acoustically tagged animals (Egli and Babcock 2004; Starr *et al.* 2005; Garla *et al.* 2006; Topping *et al.* 2006; Meyer *et al.* 2007a). Arrays of acoustic receivers can be deployed in a variety of patterns including "curtains" that animals must pass through to reach an area of management interest, such as spawning grounds (eg. Pecl *et al.* 2006). The acoustic receivers record the time, date, and identity of the uniquely coded animals when the animals move within the detection range of the receiver (Heupel *et al.* 2006; Pecl *et al.* 2006).

The advantages of using acoustic tagging for movement and home-range studies include: the ability to monitor multiple animals at the same time, the provision of spatial and temporal data on movement patterns, and the capacity to obtain data without the need to physically recapture the animals (Jackson *et al.* 2005). There are, however, some drawbacks to acoustic tagging studies. Active monitoring of tagged individuals by boat can be time consuming and produce gaps in the movement data collected (Topping *et al.* 2006). Passive monitoring is limited by the need for the acoustically tagged animals to stay within range of the array of receivers and the position accuracy of the RAPT system decreases with bad weather conditions and complex seabed morphology (Jackson *et al.* 2005). The amount of noise and physical disturbance in a study area can also affect the detection range of independent receivers (Heupel *et al.* 2006). Acoustic tags are expensive and have to be surgically implanted into the target species, which reduces the sample size of studies (Zeller 1999). Because of these small sample sizes, acoustic tagging studies are sometimes augmented by using other observational techniques to study the movement patterns of the population, such as mark-recapture and UVC (Zeller and Russ 1998; Meyer and Holland 2005; Garla *et al.* 2006; Afonso *et al.* 2008).

Fishing

Traps

Fish traps provide information on the densities and movement patterns of animals in and around management areas. Traps are especially useful in targeting species that live in structurally complex habitats where other survey techniques, such as UVC and video, may not be successful (Wells *et al.* 2008). However, trapping can be biased towards species with high catchability (Arreguin-Sanchez 1996) and traps are also size selective (Millar and Fryer 1999). Traps can be placed along transects at a set distance apart both inside and outside a management area (McClanahan and Mangi 2000; Kaunda-Arara and Rose 2004). Animals caught in traps can be tagged and mark-recapture studies using traps can provide information on the spatial movement of animals inside and outside of the management area (Chapman and Kramer 2000; Zeller *et al.* 2003; Kaunda-Arara and Rose 2004). For example, Chapman and Kramer (2000) used mark-recapture and trapping to determine that their study species had high site fidelity and a large home range, which were important considerations for implementing a management plan. Trapping has been used in spatial monitoring of crabs (Pillans *et al.* 2005), lobsters (Rowe 2001; Davidson *et al.* 2002; Goni *et al.* 2006), and fish (e.g. Kaunda-Arara and Rose 2004).

Hook and line

Remote, surface based sampling, such as experimental angling, can be used to counteract behavioural biases of fish to divers (Willis *et al.* 2000). The advantages of hook and line include simultaneous sampling of the area, accurate measurements and identification of fish species, and tag-recapture opportunities (Willis *et al.* 2000). Using hook and line as a monitoring tool has some inherent biases, including variation in catchability of species (Arreguin-Sanchez 1996) and size selectivity of hooks (Millar and Fryer 1999). Willis *et al.* (2000) used hook and line in a grid system to produce area estimates of relative densities of snapper and blue cod and compared these results to BUV and UVC surveys. Abesamis and Russ (2005) used hook and line at set distances from the boundary edge of a management area with eight replicates at each distance with each tide to determine spatial distribution of fish and the effectiveness of management protection. Zeller *et al.* (2003) used hook and line to determine movement patterns of target species.

Trawling
Commercial trawling is an extractive process that can be used to describe patterns of distribution and abundance of species (Cappo *et al.* 2004). Trawling has been used in restricted zones of management areas, such as the Great Barrier Reef (Cappo *et al.* 2004), and in monitoring a target species in habitats of interest, including the Gulf of Mexico (Wells *et al.* 2008). Cappo *et al*. (2004) compared trawling and BRUVS, and found that trawls and BRUVS sampled different species, but trawls recorded higher species richness, especially at night. Wells *et al*. (2008) found that trawling sampled numerically the most red snapper per unit area compared to trapping and BUV, but the trawl was size selective towards smaller red snapper. While trawling samples small, cryptic species well compared to most other observational techniques, it is an extractive method that can damage the seafloor and should be used sparingly when monitoring an area.

Fishery-dependent spatial monitoring

Fishery-dependent spatial data, which are measured as catch, effort, and catch per unit effort (CPUE), are commonly based on fish landings, gear types, fishing time, and fishing location. Fishery-dependent data cannot be used to monitor study species within a fishery exclusion area; however, this data can provide valuable information on trends occurring outside no-take management zones (Alcala *et al.* 2005). Moreover, some forms of spatial management allow certain types of fishing but exclude others, allowing for the possibility of using fishery-dependent sampling inside spatial management areas (McClanahan and Mangi 2001).

Catch
Catch of target species by fishers in and around management areas can be used to provide data on the distribution and abundance of target species. Catch can be measured by logbook entries by fishers (Jennings and Polunin 1996), annual trap, gillnet, hook and line, and spear catches (Alcala *et al.* 2005), and landed catches (Mangi and Roberts 2007). Relying on fishery-dependent data, like catch, can reduce the ability of fisheries managers to estimate the biomass and distribution of the target species (Mayfield *et al.* 2008) due to biases inherent in catch rate data. These biases include differences in catchability of target species (Arreguin-Sanchez 1996), size selectivity of different gear types (Millar and Fryer 1999), and unrecorded discards and by-catch.

Effort
The spatial distribution of effort is an important consideration in fisheries spatial management objectives (Daw 2008). Spatial monitoring of effort includes locations of fishing boats in an area (Lynch 2006), time spent fishing and number of fishers (Hawkins *et al.* 2006), catch landings and gear used (Mangi and Roberts 2007), logbook entries (Jennings and Polunin 1996), and interviews with fishers (Daw 2008). In-depth interviews with fishers allows for their fishing positions to be mapped using GPS, and these effort maps can be used in planning management areas (Daw 2008).

Catch per unit effort
For many fisheries stocks, fishery-dependent data, mainly CPUE, is the only means to estimate relative stock size (Schroeter *et al.* 2001). CPUE surveys collect information from fishers on species, numbers, and lengths of their catches, location of catches, number of boats, and gear types used (e.g. McClanahan and Kaunda-Arara 1996; McClanahan and Mangi 2000; Abesamis and Russ 2005). While CPUE can be used as a relative measurement of abundance and distribution of the target species (e.g. Mayfield *et al.* 2008), it uses potentially biased catch and effort data, which can produce inaccurate estimates of the biomass of the fishery. For example, Schroeter *et al*. (2001) found that CPUE did not provide an accurate abundance measurement, as the CPUE was higher than the actual abundance due to high effort.

Combination of fishery-dependent and -independent methods
A combination of observational methods and traditional fishery-dependent surveys may provide a more complete picture of the available biomass, the spatial distribution of the fishery, and the potential effects of catch, effort, and catch per unit effort on fishery stocks in and around management areas.

For catch estimates, a combination of fishery-dependent catch data, UVC surveys, and line-intercept methods for measuring the rugosity of the substratum were used to monitor the changes in catch rates after the implementation of spatial management (McClanahan and Kaunda-Arara 1996; Obura *et al.* 2002; Russ and Alcala 2004) and determine the selectivity of fisheries around a management area (Russ and Alcala 1998). These studies have highlighted the need to complement catch data with fishery-independent surveys of the actual available biomass in the fisheries in order to provide meaningful information on the population trends of target species.

Conclusions

Combining observational methods has proved to be an effective means of reducing inherent biases in each technique and increasing the range and detail of data obtained from the surveys (e.g. Zeller and Russ 1998; Babcock *et al.* 1999; Willis *et al.* 2000; Watson *et al.* 2005). By using a combination of observational techniques to target specific species or habitats, spatial monitoring surveys can provide information on the whole ecosystem. The most common combination of observational techniques is UVC and underwater video, where UVC is used to survey fish species and video is used for a benthic survey (Guidetti 2006; Cheal *et al.* 2007; Whitfield *et al.* 2007). Other combinations include acoustic tagging and UVC (Zeller and Russ 1998; Meyer and Holland 2005; Garla *et al.* 2006) and BUV and BRUV (Watson *et al.* 2005). Emerging technologies include the DIDSON acoustic camera, which can count dense schools of fish in turbid waters (Holmes *et al.* 2006); BRUVS, which have recently been shown to sample not just predators and scavengers, but a variety of marine fauna, including herbivorous and planktivorous fish (Harvey *et al.* 2007; Malcolm *et al.* 2007); and integrated acoustic techniques to map the seafloor and monitor its associated marine fauna (Freeman *et al.* 2004). These emerging technologies could decrease the amount of field time required for monitoring surveys and increase our knowledge of ecosystem functions. There is a need for continued development of non-intrusive technology for marine monitoring studies.

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