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Ecological Indicators of the Ecosystem Effects of Fishing: Final Report

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CONTENTS

EXECUTIVE SUMMARY

The potential for fishing to impact ecosystem components (either directly or indirectly) is now widely recognised. In the context of the current global focus on ecologically sustainable development (ESD) and particularly on ecosystem-based fisheries management (EBFM) this means that fisheries, and fishery management agencies, need to be able to demonstrate publicly that they are achieving their objectives within an ecosystem perspective. This in turn requires the definition of clear operational objectives, as well as the monitoring of performance relative to those objectives. Unfortunately, operational objectives and quantitative performance measures for EBFM are the exception rather than the rule, so at this stage the search for robust ecological indicators is likely to be a search for indicators that reliably tell us something about the ecological impacts of fishing on various components of the ecological system, such as individual species, communities, habitats, and whole ecosystems. Of the many hundreds of potential indicators that exist or have been proposed, it is not yet clear which ones achieve this purpose. Consequently, the development (or identification) of robust indicators for the ecological impacts of fishing has become a clear priority.

There are an enormous number of ecological indicators that have been put forward as candidates to detect and describe the effects of fishing on marine ecosystems. Over the last decade there have also been a number of reviews and framework documents laying out the criteria that may be used to best judge the utility of indicators. Despite all of this work, there have been few formal attempts to evaluate more than a handful of the many candidate indicators. This study uses two methods (one model-based and the other GIS-based) to evaluate the performance of a suite of ecological indicators that were considered good candidates for measuring impacts of fishing.

The simulation study employs the management strategy evaluation approach, whereby “operating models”, which attempt to capture the critical complexities or behaviours of real ecosystems, are used to generate fishery-dependent as well as fishery-independent pseudo-data from which the indicators can in turn be calculated. Indicator performance is then measured by their ability to detect or predict trends in key variables of interest ("attributes"), the true values of which are known from the simulation models. If indicators can not perform well in this setting they are very unlikely to perform acceptably using real data (where uncertainties will potentially be much larger). The performance of each indicator is evaluated across a range of ecological and fishing scenarios. “Robust” indicators are those that reliably and consistently predict the trends in one or more key attributes.

The indicators evaluated in the simulation study cover species, assemblages, habitats, and ecosystems, and include empirical indicators as well as those derived from simple models such as ECOPATH. The operating models used to generate the pseudo-data in this study are biogeochemical ecosystem models (based on the Atlantis framework) of two marine systems off south-eastern Australia (a large marine embayment, and an EEZ-scale regional marine ecosystem). These models include dynamic representation of the main trophic functional groups and ecological and physical processes thought to be important in temperate and sub-tropical marine ecosystems. They also incorporate the effects of fishing from a wide variety of fishing gears, as well as the confounding impacts of other broad-scale pressures – such as increased nutrient loads, coastal habitat degradation, and increases in average water temperature.

Results from the simulation study suggest that indicators at the community level of organization appear to be the most robust, in the sense defined above, and that it is necessary to use a variety of ecological indicators to detect the range of impacts from fishing. Several key functional groups are also found to provide a good representation of broader ecological changes in most instances. The broad categories these groups fall into are described in the guidelines given below, but specifically they include groups such as gelatinous zooplankton, cephalopods, seagrass, small-bodied planktivores, and top predators (e.g. large sharks, seabirds and marine mammals).

The GIS component of this study takes some of the most promising candidate indicators identified in the literature review (Fulton *et al.* 2004a) and the subsequent simulation study, and attempts to apply them to a real world ecosystem. The ecosystem chosen was the northwest shelf of Australia (NWS). Although well studied by Australian standards, it is representative of many fished ecosystems in term of its biological diversity, the presence of strong human pressures and environmental drivers, and in terms of the kinds of data issues (quality, uncertainty and availability) that affect many systems. This real world application of indicators proved to be a nice counterpoint to the rigor and depth possible in the simulation testing, as it highlighted the kinds of pragmatic decisions that are necessary when applying ecological indicators to real fished ecosystems.

The successful application of the GIS approach for a number of indicators showed that this approach is feasible, though not without its problems. Existing, “off the shelf” software can be readily used for data manipulation, analysis, and visualisation of a range of ecological and fisheries indicators at many spatial and temporal scales. The simple indicators applied to the currently available NWS data demonstrated that these kinds of easily calculated and relatively transparent diagnostic indicators can provide useful insights into whether shifts in communities and ecosystems may have occurred. More importantly, this work highlighted the very real benefit from (and need for) fisheries independent data. Trends in the values calculated for an indicator over time must be put in context to be really meaningful. As was found in the simulation study, this can be achieved by employing a suite of indicators using different data and focused on different parts of the ecosystem.

Comparing the performance in each method of analysis (simulation and GIS) of those indicators considered in both analyses there is a clear consistency (if an indicator performs well in one method it performs well in both, and similarly for poor or data quality dependent performances). The comparison of performance across the analyses also reinforces the general findings outlined above – that simple community-level indicators are likely to be the most robust and that the application of ecological indicators in a management context would benefit from the collection of fisheries independent data. Thus, whether in theory or in practice, the same conclusions have been reached and a few general guidelines on criteria for selecting ecological indicators for the effects of fishing can be provided.

1) Ideally, indicators should be easily measured, cost effective, precise, and be easily understood and interpreted. In reality, a pragmatic approach will be required for indicator selection.

2) There is no definitive set of indicators that will work in every situation – some degree of system dependency will always remain (even if only in the definition of desirable reference levels and directions). However, suites of simple indicators (e.g. relative

biomass of indicator groups, biomass ratios, size spectra, and maximum length of fish in the catch) appear to be robust measures of the overall state of an ecosystem. In contrast, indicators that require large amounts of data or employ intermediate models (or multiple steps) in their calculation, are unlikely to be consistently reliable in any system, as they are prone to error that can obscure the underlying signal. These kinds of indices (and models) may be very valuable in other contexts, but it would be ill-advised to use them as the sole focus of on-going long-term monitoring schemes designed to measure and manage the impacts of fishing on marine ecosystems.

3) Taking into account efficiency, reliability, and the need to identify the causes of overall changes in the ecological properties of ecosystems, a suite of indicators should be chosen that span a wide range of processes (with different associated rates), biological groups, and indicator types ("tactical" and "strategic", "early warning" and "integrated system state"). Such a suite will almost certainly require data that span many temporal and spatial scales. It is likely that the best combination of indicators for signal detection and system characterisation is achieved by monitoring groups from each of the following biological categories: groups with fast turnover rates (whose responsiveness may make them good "early warning" indicators), groups targeted by fisheries (as their sections of the food web are potentially the most heavily impacted, but also of significant human interest), habitat-defining groups, and charismatic or vulnerable groups (as they usually have slow dynamics and integrate across long-term and system-wide impacts).

4) In the longer term, indicators will play a role in the broader adaptive management processes associated with moves towards EBFM. They will be used both for performance reporting against management objectives (when those are fully operationalized) and as part of feedback decision rules that determine and set management arrangements to mitigate negative ecological impacts of fishing. Indicators of various types and levels, but particularly at the population and community level, will be required. Population-level indicators will be used more tactically (such as those currently used by fisheries management) and community level indicators may play a more strategic role (that can put signals from the populations into a broader ecosystem perspective). It is also likely that these uses of indicators in management will benefit from the establishment of unfished reference areas, and the collection of fisheries-independent information on other parts of the ecosystem, as signals are easier to detect and interpret if there are natural baselines or reference values available, and if there is a broader understanding of the overall system dynamics.

Even using these criteria, it will be impossible to escape some degree of system dependency when creating lists of ecological indicators for use in EBFM. However, it is possible to identify a set of robust indicators that are a good "straw man" for beginning a system-specific consideration of appropriate indicators for use in each Australian regional marine ecosystems and for specific fisheries. A suggested default set would include:

- relative biomass of demersal fish, small pelagic fish, marine mammals, piscivorous fish, scavengers, cephalopods, gelatinous zooplankton - which may be followed directly or used to calculate biomass ratios (e.g. PS:ZP, pelagic:demersal fish) that also give insight into web structure
- proportional cover of habitat defining epifauna and macrophytes

- simple diversity indices (number of communities in bioregionalisations or simple taxonomic counts)
- size spectra – the slope of these spectra can be a particularly strong indicator of overall changes at the community level
- maximum (or mean) length of the catch (across species)
- average trophic level or trophic spectra – while good quality diet data are preferred, it is possible to obtain useful relative measures for these indicators by applying "rules-of thumb" categorisations of animals
- size at maturity of main target, bycatch and top predators
- physical system characteristics (e.g. temperature, turbidity and chlorophyll a)

It must be remembered that this is only a preliminary set of recommended indicators and that further work checking whether these are appropriate in the system of interest (and what their associated reference levels/points/directions are) is needed.

"Rules of thumb" and careful use of fisheries data will provide some potentially useful indicators, but are unlikely to provide a complete picture. More worryingly, any signals such data do contain may be weak and hard to detect until large scale changes have already occurred. The first step in correcting for this problem is to ensure wider observer coverage and the collection of those kinds of data (size, species composition etc) associated with sound commercial fisheries and bycatch monitoring programs (many of these are already collected to some degree in Commonwealth fisheries). The greatest information content will be gained from indicators if fisheries independent data are brought into the mix. A first step is the inclusion of climate and ocean colour information from satellite data, but fisheries independent biological data would be most useful. To have a long-term future, given resources available in Australia, any such survey scheme would have to be carefully planned and implemented. Much can be learned in this area from the success and failure of fishery independent monitoring by Australian states and overseas fishery agencies. For instance, the northeast US has been monitored for four decades and they now have a highly efficient means of value adding to their basic trawl samples to include community-level and non-target species monitoring. While wholesale adoption of their methods is beyond Australian means, insights from their experience could be used to frame a scheme within the logistical constraints faced by Australian management agencies. One potential solution may be to use rotational or periodic "strategic" surveys in combination with the collection of a wide range of biological and environmental data in collaboration with the fishing industry (similar to what is already done in the Western Australian rock lobster fishery). Any actual survey scheme would need much deeper consideration before any moves are made to adopt it.

Whether fisheries independent surveys are introduced or not, not all marine ecosystems will be equally easily characterised using ecological indicators. Ecosystems in shelf and slope areas will probably require a larger set of indicators and finer scale data for characterisation than purely pelagic systems, but the resulting signal strength will be much stronger than in the pelagic systems. This means that ecosystem-based management of fisheries operating in shelf or slope waters (e.g. trawl, trap and demersal line fisheries) may be much easier than open water fisheries (e.g. tuna and billfish fisheries). It is also important to remember that management may be most easily implemented on the scale of a fishery, but ecosystems often span many fisheries, so the calculation of ecological indicators and the core of integrated EBFM must also happen at that larger scale.

Looking to the future, the conclusions from this study, and the recommendations on robust indicators, need further verification based on real data and the study of real systems. A very useful and instructive exercise would be for each major fishery to allocate resources to attempt a compilation of the default set of indicators listed above. At the very least, this would help synthesise existing data, and better inform future monitoring needs.

In addition to verification of performance, indicators need to be linked to reference levels (of some kind) and to management actions. This will be a difficult, but necessary, process. Some promising approaches are emerging (e.g. the use of reference directions in place of reference points, and the concept of a reference area based on the use of principle components analysis (Link *et al*. 2002)), but they are still at early stages of development and testing. Once the use of ecological indicators is better integrated into adaptive management processes, the next stage will be to test those adaptive strategies using methods of management strategy evaluation. The application of MSE methods to testing the robustness of ecological indicators is a step in this direction.

1. BACKGROUND

Ecologically sustainable development (ESD) has formed the basis of the management of natural resources in Australia since 1992 when the National Strategy for ESD was released. ESD principles and objectives are explicitly referred to in most Australian fisheries legislation (Sainsbury *et al*. 2000), with considerable advances having been made at the target species level in the application of ESD principles in fisheries management. To date, less attention has been paid to the wider ecological impacts of fishing.

Globally, changes in the focus of public and scientific scrutiny have seen the concept of ecologically sustainable development evolve to incorporate wider ecological impacts of fisheries on marine ecosystems (Constable, 2001; Sainsbury and Sumaila, 2003). Environmental management agencies, environmental NGOs, news media and politicians are starting to pay much closer attention to ecological impacts of fishing such as the direct effects of fishing on non-target species, indirect effects on related species and the food chain, and impacts on habitats. In turn this has meant that fisheries, and fishery management agencies, are responding to these issues, but they are also being held accountable for outcomes. As a result, the fisheries and management agencies need to be able to demonstrate publicly that they are achieving successes in the area of broad ESD and complying with key objectives of the Acts under which they operate. This necessitates setting clear objectives and monitoring performance relative to those objectives. Unfortunately, there is an ever increasing breadth of ecological objectives in fisheries and environmental legislation and it is not clear what performance measures to use. Thus, the development of robust ecological indicators has become a clear priority.

The project “Ecological indicators for fishery management: non-target species, habitats and food chains” has the issue of robust ecological indicators at its heart. The project is jointly funded by the Australian Fisheries Management Authority (AFMA), Agriculture, Fisheries and Forestry - Australia (AFFA), Environment Australia (EA), and CSIRO Marine Research (CMR). The project was initially developed in June 2000, and forms part of a broader strategy to improve the set of tools available to fishery and environmental managers in moving towards ecosystem-based fishery management. The project’s two key objectives are:

1) to review the current use of ecological indicators in fisheries management, and to identify a range of candidate indicators for further evaluation (the results of this work can be found in Fulton *et al.* 2004a).

2) use management strategy evaluation (MSE) methods to evaluate indicators, and to identify robust ecological indicators and reference points for use in fisheries management.

This final report has been written as part of the latter objective.

2. INTRODUCTION AND SCIENTIFIC APPROACH

2.1 Potential Effects of Fishing

The potential impacts of fishing on target species have been recognised for many years now, though research continues in fields such as the potential genetic effects and their evolutionary implications (Heikinheimo and Mikkola 2004). Consideration of broader ecological impacts began decades ago (e.g. Ketchen 1947), but did not receive wide attention until relatively recently and the publication of several reviews of the ecological effects of fishing (Goni 1998, Hall 1999, Jackson *et al.* 2000, Dayton *et al.* 2002, Pauly *et al.* 2003). These reviews show that the ecological impacts of fishing can be divided into direct and indirect. The most obvious effect of fishing on target and non-target species is direct mortality. This mortality can occur through targeted fishing, bycatch, incidental capture, ghost fishing (by fisheries related debris), or by the culling of species regarded as competing with fishing operations (e.g. the shooting of seals). The indirect effects of fishing are often (but not always) more subtle and include:

1) modification of habitats – either physically (e.g. removal of habitat defining epibenthos) or chemically (e.g. changes in nutrient loading due to waste water discharged from processing plants);
2) restructuring of trophic linkages – due to the removal of predators or prey;
3) competitive release – due to the direct or indirect removal of competitors by fisheries activities, which may lead to indirect effects on other species (e.g. fishing may lead to an increase in the abundance of bio-eroders that can damage reefs);
4) change in ecosystem productivity – fisheries activities may cause production to decrease (e.g. if there is direct removal or damage of the phytobenthos by trawling), but when primary producers are effected by the mechanisms listed in (2) and (3) production can increase (if there is competitive, or predatory, release of primary producer species that are competitors, or prey, of species harvested by fisheries);
5) modification of energetic pathways – alternative trophic subwebs can be enhanced by shifts in community structure and feeding links due to discarding or the increased vulnerability to predation, disease, or stress of flora and fauna damaged by fishing activities;
6) species invasions (e.g. organic discards from fishing vessels can attract scavenger species);
7) loss of mediation events (e.g. without tuna forcing their prey to the surface during feeding, seabirds can not access this potential food resource);
8) change in the demographic structure of harvested and bycatch species (e.g. in comparison with unfished values, sex ratios and age- or size-structure may be skewed);
9) loss of genetic diversity or change in genetic frequencies – fishing not only removes some genetic stocks, it can also act as a selective pressure resulting in a change in genetic frequencies (e.g. in contrast to a large size acting as a refuge from predation, fish maturing at smaller sizes may have a selective advantage when under fishing pressure, as fisheries tend to target larger fish);

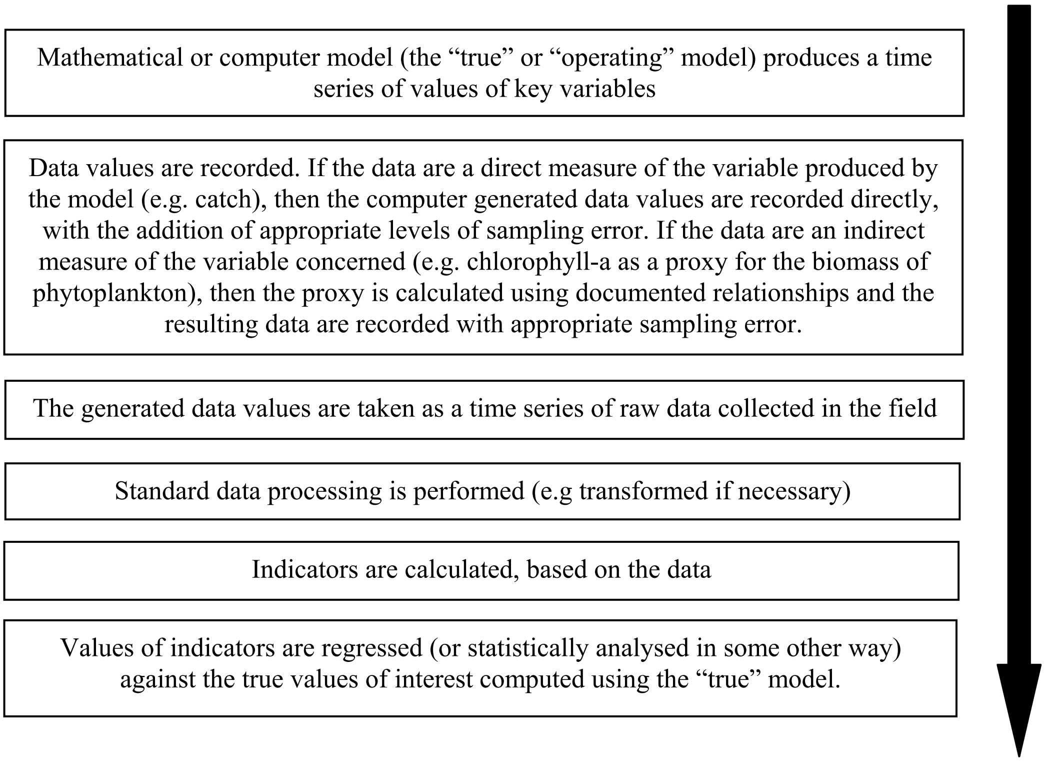
10) reduction in reproductive success – either due to a lack of available prey resources during the reproductive season (e.g. seabirds will abandon nests if prey are scarce), modifications to the size structure of the breeding population (fisheries usually target the larger, more productive, size classes), or because of removal of larval or juvenile habitats (e.g. epibenthic structures);
11) behaviour modification (e.g. fishing operations may disrupt feeding or reproductive aggregations);
12) increase in disease or a reduction in condition – injuries caused by fishing gear, increased supplies of nutrients and detritus (due to discarding), increased toxicant contamination (from resuspended sediments) can all increase the prevalence of pathogens, increase stress (which reduces the immune system response), and create conditions that may facilitate transmission of disease; and
13) increased susceptibility to environmental fluctuations – when fishing acts synergistically with environmental factors, it can accelerate or amplify the impacts of environmental change (i.e. a rate of fishing mortality that is acceptable in a pristine or relatively constant environment may cause or hasten the collapse of a stock in an environment where there are large scale changes in climate over time).

2.2 Evaluation by Simulation

The data and knowledge required to evaluate potential management methods, such as indicators of ecological impacts of fishing, are often very large. One of the most cost-effective ways of addressing this issue is to evaluate the performance of potential methods (e.g. indicators) using computer generated (rather than real) data. Given that there is perfect knowledge about the true trends, properties, structure and sampling variance in computer generated data, if the indicators cannot perform well with this data then they are extremely unlikely to perform acceptably using real data (where uncertainties are large). This approach has been employed successfully in many settings over the last two decades, especially in fisheries (Butterworth and Punt 1999, Sainsbury *et al.* 2000), and was adopted here as a major part of the examination of the potential utility of a broad range of indicator-types for the identification of the ecological effects of fishing. As no operational objectives linked to indicator exist as yet, the evaluation of the indicators in this study is done in a diagnostic sense. That is, which indicators can effectively communicate trends in the properties of ecosystems?

A simple outline of the main steps of the general approach is given in Figure 1.1. The “operating” models used to generate the data values (i.e. the models that represent “truth”) do not necessarily have to be complicated, but they must have as much resolution as needed to estimate the indicator values. In this context, resolution not only refers to spatiotemporal resolution, but also to the number and types of variables included in the models. This means the models must, at the very least, include the key relationships thought to underlie the functioning of the system of interest. For example, to investigate ecological indicators at the ecosystem level, the model(s) will need to include feeding relationships, movement, habitat characteristics, and anthropogenic pressures (e.g. fishing or nutrient release). A full description of the models used in this study is given in Chapter 3 and the associated appendices.

FIGURE 2.1: General steps of the simulation testing approach. The model data generation and recording steps are generally repeated many times to integrate across uncertainty in processes, functional relationships and parameters, and may even be based on several alternative “true” models.



Mathematical or computer model (the “true” or “operating” model) produces a time series of values of key variables

Data values are recorded. If the data are a direct measure of the variable produced by the model (e.g. catch), then the computer generated data values are recorded directly, with the addition of appropriate levels of sampling error. If the data are an indirect measure of the variable concerned (e.g. chlorophyll-a as a proxy for the biomass of phytoplankton), then the proxy is calculated using documented relationships and the resulting data are recorded with appropriate sampling error.

The generated data values are taken as a time series of raw data collected in the field

Standard data processing is performed (e.g transformed if necessary)

Indicators are calculated, based on the data

Values of indicators are regressed (or statistically analysed in some other way) against the true values of interest computed using the “true” model.

2.3 Attributes and Indicators

Attributes are the aspects of an ecosystem that people (scientists, managers or society in general) are interested in. Ideally attributes are derived from high-level objectives/concepts, such as maintaining “ecosystem health”, “ecosystem integrity”, “ecosystem resilience”, or “the conservation of biodiversity” (Punt *et al.* 2001 attached as Appendix B). In many cases attributes will not be directly measurable and as a result proxies (indicators) that track (or predict) the state of the attribute must be used. By following the trend in an indicator, or evaluating an indicator relative to some criterion, it is possible to gather information about one or more attributes of a system that are not easily perceived in other ways (Landres 1992, Vandermeulen 1998, Jackson *et al*. 2000, Seijo and Caddy 2000).

Many types of indicators exist (qualitative as well as quantitative), but an important rationale for selecting an indicator is that it is more cost-effective than following many individual species separately, and sufficiently responsive to indicate changes in the system without being overly sensitive to natural variation (Hellawell 1986, Landres 1992, Rapport 1992, Seijo and Caddy 2000, ICES 2001, Rochet and Trenkel 2003, Nicholson and Jennings 2004). The natural complexity of ecosystems and ecological processes means that this may not hold true for any single indicator in all circumstances and using a suite of indicators may be more robust

(Bernstein 1992, Schneider 1992, ICES 2001, Adams *et al*. 2002, Fulton *et al. submitted*). Research across a number of fields (Rapport 1992, Seijo and Caddy 2000, Xu *et al*. 2001a, Adams *et al*. 2002) – as well as the work presented here – indicates that the most effective integrated suite of indicators contains indicators that

- screen for a loss of system integrity (i.e. breakdown in system function);
- highlighting changes in system structure;
- diagnose the cause of the change in the system; and
- indicate system recovery.

Using a suite of indicators also provides robustness and guards reaching erroneous (or even contradictory) conclusions about the state of an ecosystem based on a single indicator – an approach were indicator selection becomes critically important (Eiswerth and Haney 2001) and is ultimately destined to fail given natural uncertainties and typical financial constraints.

In the marine realm, environmental indicators have been traditionally used to monitor water quality and contamination. It has only been within the last 10-15 years that indicators have begun to be used in the assessment of ecological and ecosystem attributes (e.g. “ecosystem health”). The use of indicators in this new role is still under development and assessment and it is not as well tested or understood as the other uses for indicators (Landres 1992, Shin and Cury 2004). In the remainder of this report we present our analysis of a wide variety of univariate and multivariate indicators – some with a long history of use or extensive testing and others that show great promise, but require additional verification of their performance in the field.

2.3.1 Definitions

The following are a number of definitions required for complete understanding of terms that are associated with indicators and used repeatedly in the body of the report.

2.3.1.1 Indicator species

Species that are readily identified, cosmopolitan and for which simple measures (e.g. biomass) are easily sampled, have low variability and characterise system state, or indicate the cause of a change in the system, are known as “indicator” species. Typically, indicator species are drawn from the macrophytes (e.g. seagrass), benthic invertebrates (often a preferred indicator group in water quality monitoring), fish (which are particularly useful bioindicators in estuaries), and birds and mammals (which integrate across scales and the trophic web) (Hellawell 1986, Carignan and Villard 2002). While selection of key species from across the foodweb can be informative, selection of indicator species must be done carefully, as focusing attention on restricted parts of the trophic web (e.g. higher trophic groups) can lead to unbalanced results (Carignan and Villard 2002, Fulton and Smith *in press*). The key is to capture contemporaneous information on a range of elements (biological, physical and chemical) across a range of scales and habitats that display a wide range of sensitivities to system modification (Carignan and Villard 2002, Whitfield and Elliott 2002).

2.3.1.2 Species of Interest

A species of interest is any species drawing scientific or management interest. They may be harvested, key-stone, bycatch, pest, threatened, “flagship”(public conservation targets), "indicator", or "umbrella" species (species that are area, dispersal, resource or process limited and whose persistence requirements encapsulate those of a number of associated species or groups; Carignan and Villard 2002).

2.3.1.3 Hierarchical Classification of Indicators

In addition to the characterisation of indicators as "empirically measured" versus "model-dependent", or "physical" versus "ecological", it is possible to classify indicators based on the hierarchical biological level they correspond to and the response scales they consider: (1) individual level (concentrating on behavioural and metabolic responses); (2) population level (focusing on behavioural, demographic, metabolic and genetic responses by individual species); (3) community level (emphasising species richness, diversity, ecological processes, community composition, trophic and habitat structure); and (4) ecosystem level (considering entire ecosystem processes, such as production, overall structure, cycling and diversity). Studies undertaken by scientists concerned with monitoring water quality indicate that sensitivity to natural variation drops with the progression from individual to ecosystem level indicators (Figure 2.2a), while responsiveness to anthropogenically based impacts follow a non-linear relationship (Figure 2.2b) – suggesting that population or community level indicators may be the most useful (Frost *et al.* 1992). A similar finding has resulted from the assessment of environmental and ecological indicators of the effects of fishing reported here, where community level indicators were often the most effective.

The following is a brief summary of the use to date of indicators of the various levels

Individual level

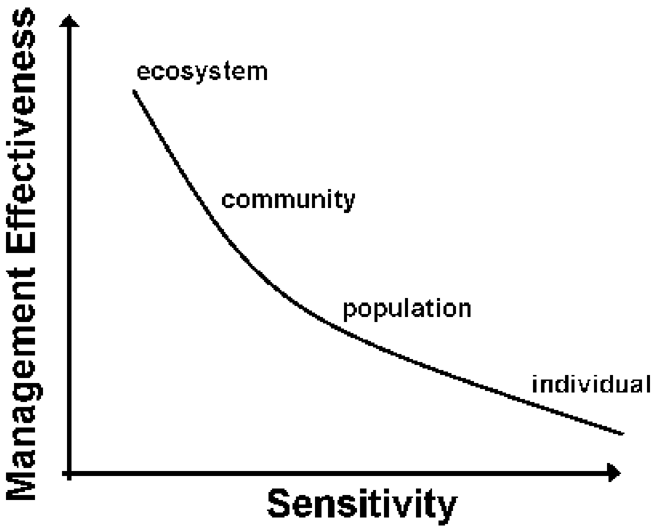
The use of individual level indicators is effectively non-existent in fisheries. While the condition or reproductive and provisioning rates of individuals from species of interest could theoretically be used as indicators of the effects of fisheries, these indicators are usually used at the population level.

Population level

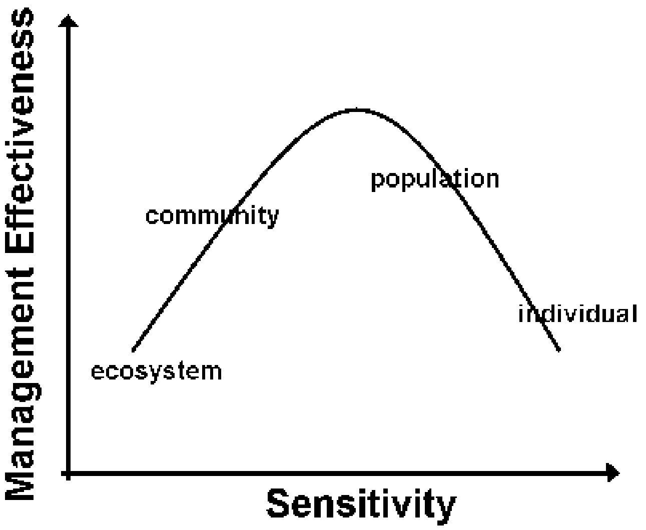
A number of indicators, including biomass, production and size-structure have been used to consider population "health" in the past and they may also have potential as proxies for community and ecosystem level processes. Measures of single species or suites of species of interest have a long history of being used as indicators in single and multispecies fisheries management; and indicators based on threatened species already feature in some of the legislation concerning fisheries activities. Studies considering candidate indicators for immediate application in ecosystem-based fisheries management (ICES 2001, Rochet and Trenkel 2003) have also put population level indicators forward as the most operational – because: they are more easily interpreted; there is already a good understanding of their properties and how they are affected by fishing; and some researchers believe it is difficult to think of an ecosystem property that is not preserved if the individuals species in the ecosystem

Figure 2.2: Schematic diagrams of the relationship between sensitivity (to natural and human induced influences) and effectiveness for indicators at different biological levels of organization (ecosystem, community, population and individual).

(a) sensitivity to natural variability versus indicator effectiveness for overall system evaluation



(b) sensitivity to human activities versus indicator effectiveness for overall system evaluation



are conserved (ICES 2001). Despite these attractive features, more scientific understanding is required before their true worth as long-term indicators of ecosystem attributes is known. The utility of population level measures as indicators for the state of particular species is not under question, but their usefulness as proxies for attributes further up the hierarchy is uncertain. Research into indicator performance, such as the model studies reported here, has show that population level indicators can be unreliable with regard to capturing the state of ecosystem attributes. In addition, some population level indicators require data that are exceedingly difficult to collect for non-commercial species and as a result these indicators are not practical (Rochet and Trenkel 2003). Overall, while the extension of the use of population level indicators to ecosystem-based management is attractive, it must be done with caution (Die and Caddy 1997, Bax *et al*. 1999, Lindenmayer *et al*. 2000).

Community level

While not used widely as yet in fisheries management, some of the most interesting and promising indicators measure community level responses to the effects of fishing. Indicators of habitat complexity, diagnostic assemblages and structural indicators are all showing strong correlations in the field with impacts of fishing (Sainsbury 1988, Auster *et al.*1996, Collie *et al.* 1997, Poiner *et al.*1998, Schwinghammer *et al.*1998, Auster and Langton 1999, Bax *et al.* 1999, Freese *et al.* 1999, Jennings *et al.* 1999, Lindholm *et al.* 1999, Eleftheriou 2000, Jackson *et al.* 2000, Nicholson and Jennings 2004, Shin *et al. submitted*). One of the most promising types of community level indicators is size-based indicators. Changes in the ecosystem causes shifts in community composition and foodweb structure and size-spectra analysis can detect these shifts, in particular, the slope and intercept of the size-spectra curves can convey a lot of information about the dynamics of the shifts (Rice 2000). In addition, the types of data required by are already readily (and easily) collected by fisheries and scientific research vessels. While size-spectra do not respond exclusively to the effects of fishing (being affected by natural regime shifts and eutrophication as well; Caddy and Garibaldi 2000), they are not as susceptible to confounding by shifts in market forces and technological advances as other trophic indices. The greatest issue with community level indicators is that reference points are not always easily identified (though reference directions have been defined for many of them; Link *submitted*, Shin *et al. submitted*) and unfished reference areas (which can be rare for exploited marine systems).

Ecosystem level

Fisheries and ecological research at the ecosystem-level is typically rare in reality and as a result many of the most promising ecosystem level indicators stem from ecological theory or from models, such as ECOSIM (Walters *et al*. 1997, Christensen *et al*. 2000). As a result they are tied to the assumptions (and quality) of the theories and models, so their true information content and limitations are not well known and evaluation of the robustness of these indicators is essential (ICES 2000). The experience of workers in other fields (e.g. water quality monitoring) does show that ecosystem level indicators will not make good “early warning” indicators (Rapport 1992) or “rapid assessment” tools (Bernstein 1992). This is because they are less sensitive to change due to compensatory dynamics within their constituent parts (Frost *et al* 1992) and associated time lags in their overall responsiveness (Bernstein 1992). This means that if ecosystem level indicators are used they will need to be used in the correct context –

to describe high level, often long-term, attributes such as "ecosystem health" (Rapport 1992).

2.3.2 Selection Criteria

Many bodies have put out lists of criteria for indicator selection (Landres 1992, Rapport 1992, Jackson *et al*. 2000, ICES 2001, Tegler *et al*. 2001, Rochet and Trenkel 2003, Rice and Rochet 2004), but the basic features common to all is a need for indicators to be:

- of sound conceptual basis (to allow for clear understanding of its behaviour and its relationship with the state and dynamics of the ecosystem, and to allow for the realistic setting of objectives and reference points or directions);
- feasibly implemented (i.e. easily and accurately measured over a large proportion of the area to be managed);
- responsive to manageable human activity (in a predictable fashion and with tight temporal linkage);
- exclusive (i.e. minimal responsiveness to variation induced by other causes);
- comprehensive;
- communicable to non-scientists

Guided by these criteria Fulton *et al.* (2004a) evaluated a long list of potential indicators of the effects of fishing – in fulfilment of objective 1 of this project. The resulting list was put forward as potential candidates for testing by the second phase of the project (the results of which are reported here).

2.3.3 Selected Indicators

Literally hundreds of potential ecological indicators can be found in the literature. The literature review completed by Fulton *et al.* (2004a) reduced this set to a smaller number of potentially useful indicators. Of these, those that could feasibly be tested using simulated data, rather than field data collected from real systems were selected for testing. This list was augmented with a limited set of indicators whose potential only became apparent during the testing period, as a result of on-going work by other researchers from around the world who are also investigating the topic of indicators (e.g. Miller and Cury 2003, Chassot and Gascuel 2003). The final list of indicator types considered in this study is:

- Biomass or stock size
- Total mortality (actually catch / PBR-type catch limit)
- Size-structure
- Age-structure
- Catch-rate
- Discard rate
- Size-spectra
- k-dominance curves
- Coefficient of variation for total biomass
- Fishery In Balance (FIB)
- Average trophic level
- Diversity index (count of groups present)
- Multi-dimensional scaling
- Rate of damage

- Benthic habitat complexity
- Biomass (or % cover) of cover-defining species or species groups
- Reproductive success (of birds, mammals; especially endangered groups)
- Piscivore biomass / planktivore biomass
- Pelagic fish biomass / Demersal fish biomass
- Chlorophyll-a
- Throughput
- Production / Biomass
- Primary Production Required (PPR)
- Omnivory Indices and the System Omnivory Index (SOI)
- Dominance of detritus
- Relative ascendency
- Residence time
- Ecotrophic Efficiency (EE)
- Average maximum length in the catch
- Absolute maximum length in the catch
- Total values of network characteristics (e.g. total consumption, total production, total detrital flow, total export, total import)
- Trophic efficiencies
- Rate of denitrification
- Trophic spectra

Further details (e.g. definitions) for these indicators are given in section 4.4.2.

2.4 GIS Analysis

Another tool only recently available in fisheries science is the use of spatial and geostatistics within Geographic Information Systems (GIS). These were originally used for exploratory data analysis (e.g. Kemp and Meaden 2002), but are increasingly being employed in a fisheries management role (e.g. Rueda and Defeo 2003). A series of GIS analyses of real data from the Northwest Shelf of Australia are used here to complement the simulation tests.

It was decided that a real world application of the most promising indicators would give insight into whether trends in some of the most promising indices could be detected in real data, with all its inherent constraints and uncertainties that are not necessarily present in simulated data. These real world calculations could have been done simply at a whole of ecosystem-level. However, it was felt that given the availability of at least three large-scale spatially resolved data-sets for the NWS system, the increasingly common use of GIS in the marine realm (Wright and Bartlett 2001, Valavanis 2002, Fréon *et al* 2004), and the ability of GIS software to process information (and thus calculate indicators) at almost any spatial scale (data permitting) meant that retaining the spatial dimension in the analyses was too good an opportunity to pass up.

3. CASE STUDY LOCATIONS

Two modelled and one real system are used to investigate indicators in this study (Figure 3.1). The two modelled systems are based on real systems (Port Phillip Bay and the southeast region of Australia), but they are not matched to the systems exactly. Instead they were modified to be more representative of temperate marine ecosystems in general. The differences between the modelled systems and the real ecosystems occurred primarily in the parameterisation of the biology (average parameter values from the literature were used rather than those for specific biota from the real ecosystems). The decision not to match the case-studies exactly to the biology of the real ecosystems was so that the findings would be more general. This approach has been used successfully in the past (e.g. Fulton 2001).

The real system included in the study (analysis based on actual rather than modelled data) is the Northwest Shelf of Australia. This system was chosen for the GIS analysis as it is a well studied system that has been scientifically sampled for over twenty years and it has also been fished with varying intensity for decades. Moreover, while it is well studied by Australian standards, it is representative of many fished ecosystems globally in term of its biological diversity, the presence of strong human pressures and environmental drivers, and in terms of the kinds of data issues (quality, uncertainty and availability) that affect our ability to understand the system.

A description of each study area is given below and for the modelled ecosystems the modifications made to make the ecosystems more representative are discussed in Chapter 4.

3.1.1 Port Phillip Bay

One of the modelled ecosystems is a temperate enclosed bay based on Port Phillip Bay, Melbourne, Australia (A in Figure 3.1). The bay is shallow, its (approximately) 1,930 km2 area is mostly less than 8m deep, though it reaches 24m at its deepest points. It is fed by 8 drainage basins and is nearly enclosed (a large sand bar delta near the bay mouth restricts exchange with Bass Strait and the open ocean). Approximately 15% of Australia's population lives within the catchment area of Port Phillip Bay (~3 million people), and a number of fisheries have operated, or are currently active, in the bay. These fisheries include: a large recreational fishery with annual effort of ~670,000 angler hours and landings of ~470t of fish (MAFRI 1996); commercial finfish fisheries with a total annual take of between 700 and 2,000t (worth about $3 million AUD wholesale); and fisheries for a number of invertebrate species such as cultured mussels (annual landings of 600t, $1.5 million AUD) and wild abalone (annual landings 50t, $1 million AUD). In the past, scallops have also been harvested intensively in the bay, with up to 10,000t (shell weight) being landed annually. The bay has also been the subject of a number of environment studies (e.g. Harris *et al.* 1996), which means good quality information spanning many years is available for the ecosystem.

3.1.2 Southeast Australia

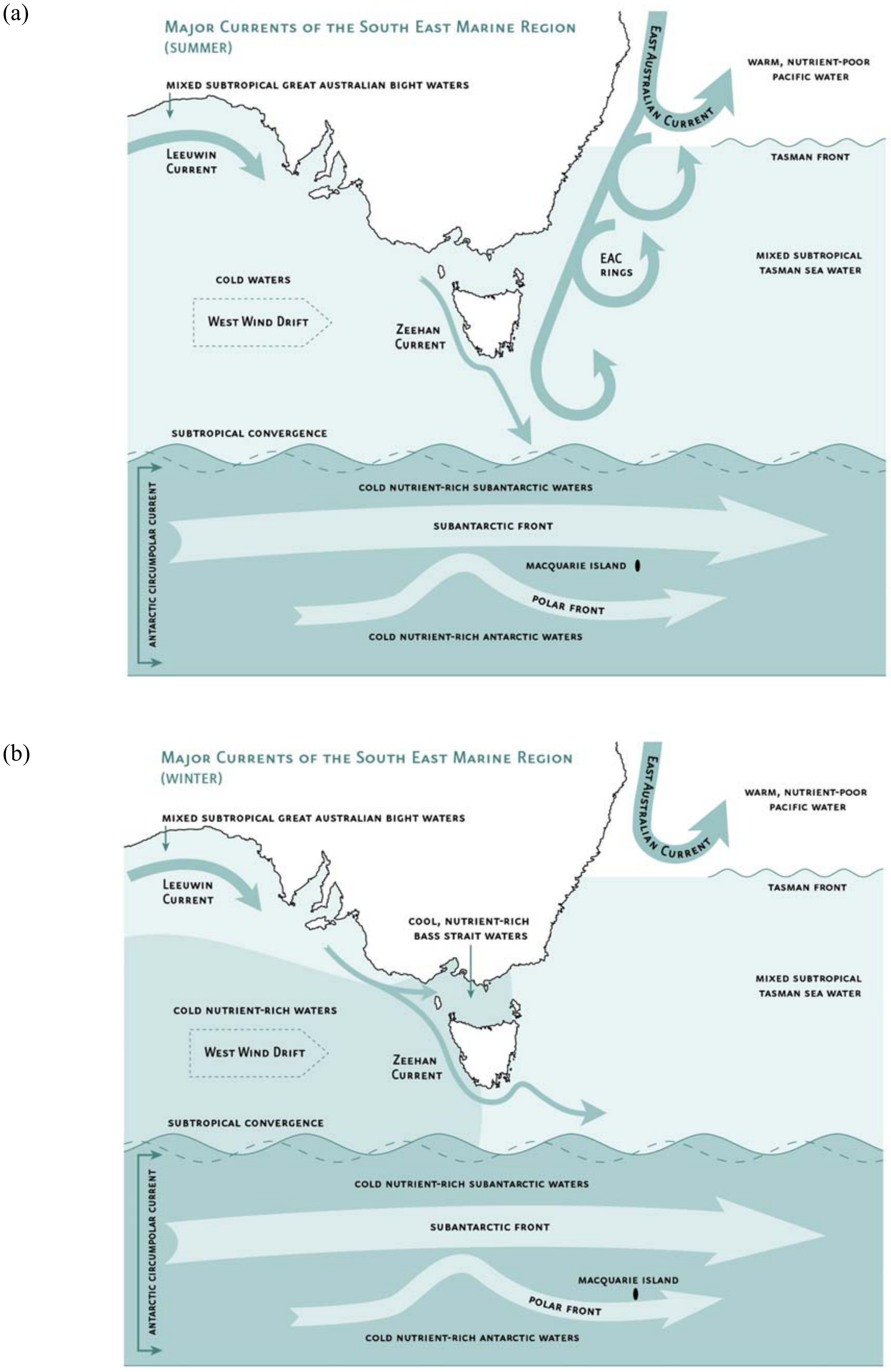
The other modelled ecosystem is a large-scale coast-to-open ocean system based on the EEZ for much of eastern and southern Australia (B in Figure 3.1). It covers an area over 3.7 million km^2 and the area modelled explicitly stretches from the surface down to 1800m. It includes large bays and gulfs, the continental shelf and slope, sea mounts, submerged canyons and open ocean; it spans tropical, subtropical, cool temperate and subantarctic environments. Its immense area means that it includes a wide variety of bottom types (e.g. silts, oozes, material of terrestrial origin, gravel, rocky reefs, sands, and exposed limestone bedrock); and it includes some of the most productive locations in Australian waters. The influence of the Leeuwin and East Australian currents on the region also make it highly diverse as they bring in biota from other areas. The strongest signal in the oceanography of the region is the seasonal pattern of current strength (Figure 3.2) – particularly the Zeehan and East Australian currents extents – which has strong implications for sea surface temperature, the creation of upwellings and the supply of nutrients, and ultimately the productivity and distribution of the biological components of the ecosystem. Not surprisingly, the large area under consideration, the wide range of habitats included, and the many influences upon the area mean that it is quite diverse (including many relic and endemic species, but also highly migratory species or species that are only seasonal visitors).

The majority of Australia's population lives along the shores of this southeast area and there have been substantial pressures put on it by fishing (both state and commonwealth fisheries), industrial and agricultural contaminant release, and habitat modification due to coastal development. The area has been the focus of some of Australia's oldest (e.g. rock lobster has been fished commercial since the 1850s), largest (e.g. Great Australian Bight pilchard fishery, which is Australia's largest fishery, by volume (ABARE and FRDC 2004), with a TAC of 36000t) and most lucrative (e.g. abalone) fisheries. The area produces over 50% of the gross value of Australia's fisheries production (ABARE and FRDC 2004). There are at least 148 species harvested commercially in the area – including invertebrates (e.g. abalone, rock lobster, prawns and squid) and fin-fish – spanning a range of life histories from short lived pilchards to slow growing Orange Roughy, and species caught in shallow water (e.g. King George Whiting captured in Port Phillip Bay) to those caught on the open ocean (e.g. broadbill swordfish). In total the landed catch (of over 126000t weight wet) is worth in excess of $870 million. The concentration of a large percentage of Australia's population in the area has meant that some parts of it have been well studied, though as a whole it has never been systematically investigated in an integrated way.

3.1.3 Northwest Shelf of Australia

The ecosystem used to test indicators with real world data is the Northwest Shelf (NWS) of Australia (C in Figure 3.1). This is a large region of Australia (95000 km^2) that is characterised by a broad shelf, a tropical hydrographic regime (Wyrtki 1961, Condie *et al.* 2003, Condie *et al. in prep*), and a sharp distinction between naturally turbid inshore waters and clearer offshore waters. Its seabed is made up mostly of calcareous sands and fine muds (AGSO etc refs) and has patchy coverage of reef and sponge beds (CMR and DEP 2002, Althaus *et al. in prep*a). Biologically it has reasonably high productivity (Tranter 1962, Kabanova 1968 and Motoda *et al.* 1978), a diverse Indo-West Pacific fish fauna (Sainsbury *et al.* 1997), and diverse and productive crustacean populations (Sainsbury *pers. com.*, Bulman *in prep*). Research has also shown that the form of the natural environment (the biogenic habitat) plays a large part in structuring the distribution of biological stocks in the area (Sainsbury *et al.* 1997, Althaus *et al. in prep*b). There are a large number of industries that put pressure on the environment on the NWS, including petroleum exploration and extraction, tourism, coastal development, salt

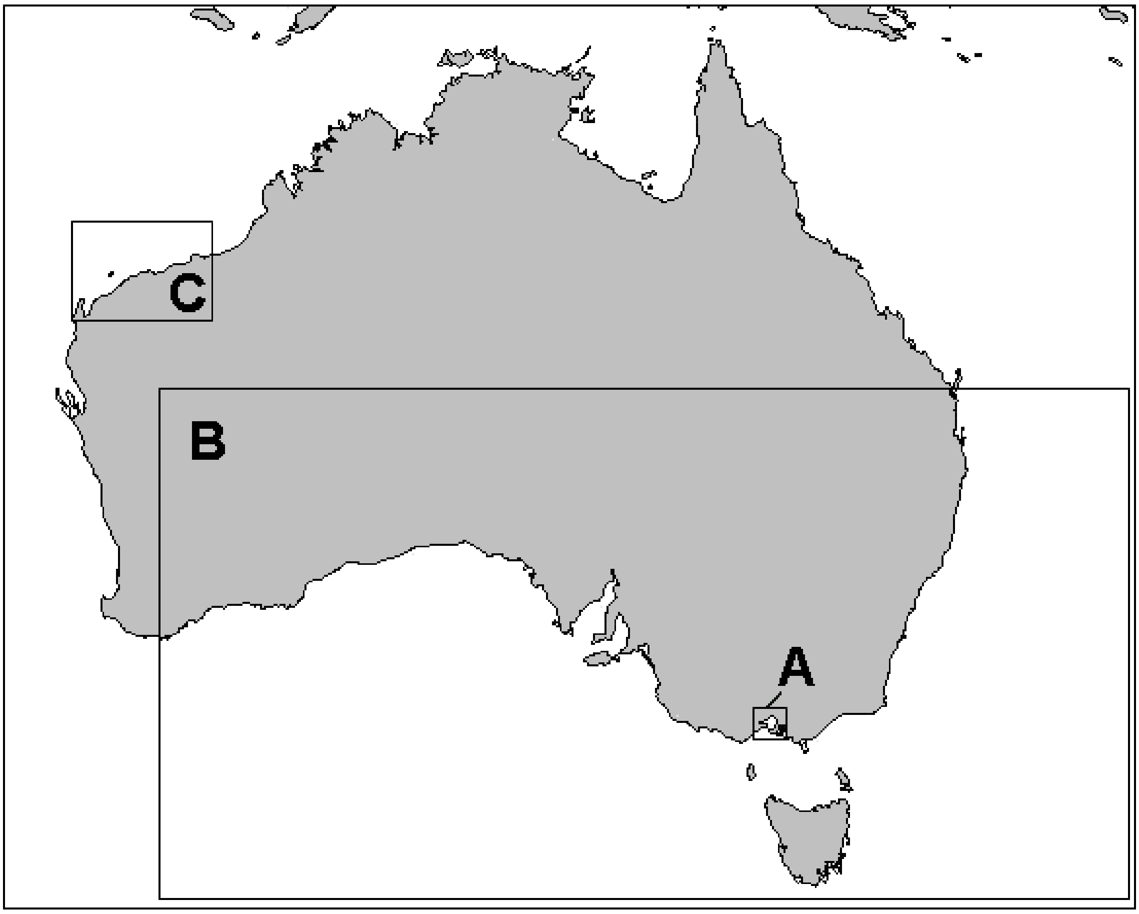
Figure 3.2: The main surface currents and water masses of the south-east region: (a) in summer and (b) in winter. Arrows indicate strength of current flow and dark shaded areas are typically high nutrient locations. Reproduced with permission from NOO 2004.



production, port operations and fisheries. While fisheries is not the biggest sector on the NWS economically it has had significant effects on the biota of the NWS (Sainsbury 1987, 1988).

The NWS has been exploited by a number of fishing operations over the last thirty years: a Japanese trawl fishery targeted *Lethrinus* between 30-120m from 116°E to 117°30'E in the years 1959 to 1963; a Taiwanese pair trawl fishery took many species (mostly *Nemipterus*, *Saurida*, *Lutjanus* and *Lethrinus*) between 30 and 120m from 1972 to the early 1990s; the current domestic Australian trap fishery started in 1984 and targets *Lethrinus*, *Lutjanus* and *Epinephelus* down to 80m in areas that had previously seen little trawling; and the domestic Australian trawl fishery began in 1989 and takes mainly *Nemipterus*, *Saurida*, *Lutjanus* and *Lethrinus* between 30 and 120m and east of 116°45'E. Between 1960 and the present there have also been a number of research surveys carried out in the area, providing information on fish community composition and seabed cover (Sainsbury 1987, Althaus *et al. in prep* a and b). This history of exploitation and study means there are multi-decade timeseries of fisheries and supporting scientific data to test the indicators on.

Figure 3.1: Map of Australia indicating the location of the three case study areas. A and B were the basis for modelled ecosystems, while C was the site of collection for data used during the GIS analysis. A is Port Phillip Bay, Melbourne; B is the southeast region; and C is the Northwest Shelf.



4 SIMULATION TESTING – ATLANTIS FRAMEWORK

4.1 Operating Model

As mentioned in Chapter 3, to allow the modelled ecosystems (and hence the outcomes of the study) to be more representative, the modelled systems were based on the corresponding real systems, but did not match the systems exactly. Similarly, the fisheries (and other human sectors) represented in the modelled systems are not exact reproductions of their real world counterparts. The modelled fisheries and the trends in fishing effort were selected to produce the maximum amount of contrast in the ecological impacts in the generated data set. This contrast was necessary to properly test the flexibility and robustness of the potential indicators. This approach may result in overestimation of the performance of some indicators, but it will more quickly identify indicators with little potential utility (if an indicator does not perform well under these circumstances it will generally be of little practical use) and so screen poor indicators from further consideration.

In contrast to the biology and anthropogenic model components, the physical aspects of the operating models replicate the hydrographic conditions found in the real ecosystems as closely as possible. This was done to ensure that realistic rates of water movement and transfer were retained, even when other forcing conditions (e.g. temperature or nutrient levels) were modified to represent the impacts of anthropogenic pressures.

4.1.1 Atlantis

The Atlantis framework was used for the operating models in this study. Atlantis consists of two models: a biogeochemical ecosystem model (which produces deterministic time-series for each component in the modelled ecosystem) and a sampling model (which “collects” the data from the ecosystem model and calculates the indicators). The sampling model takes the data time-series from the ecosystem model output and adds sampling error and other stochastic elements so that the final data sets resemble as closely as possible those collected by scientific surveys and commercial fisheries. Further details of these models are given below and the equations are given in the appendices.

4.1.2 Biological submodels

Atlantis is a revised version of Bay Model 2 (BM2) (Fulton 2001, Fulton *et al.* 2004b). The original version of BM2 was built as part of a modelling study investigating model complexity, and has proven to be a useful tool for simulating temperate marine bays (Fulton 2001). Atlantis is a deterministic model that tracks the nutrient (nitrogen and silica) flow through the main biological groups found in temperate marine ecosystems, and three detritus groups (labile detritus, refractory detritus and carrion). The invertebrate and primary producer groups are simulated using aggregate biomass pools, while the vertebrates are represented using age-structured models. The primary processes considered in Atlantis are consumption, production, waste production, migration, predation, recruitment, habitat dependency, and natural and fishing mortality. A summary of the chief assumptions is given in Table 4.1, and the general mathematical formulations for the phototrophic, invertebrate consumer, bacterial, and vertebrate groups are provided in Appendix C.

In a few cases (particularly for the shallow bay ecosystem) the literature suggests that aspects of the real ecosystems the operating models are based on are not representative of the majority of that kind of ecosystem globally. In those cases, alternative ecological scenarios are run with different parameter sets (one for the default starting ecosystem structure and another that is more general) – see *Ecological Scenarios* under section 4.2 below.

Case study 1 - Shallow Bay

The food web modelled in the shallow bay operating model contains 21 invertebrate groups (including microfauna) and 8 vertebrate groups (listed in Table 4.2). These groups represent the main functional groups in temperate bays and are defined using size, rates of turnover and shared predators and prey. The resulting foodweb is given in Figure 4.1.

To consider the implications of structural uncertainty two of the key ecological assumptions are varied. The first is the connectivity of the foodweb. In one case the strength of the linkages is very high, while in the other case the majority of the links are weak to moderate in strength. Two other major assumption which was varied was the degree of closure of the ecosystem. The main variation in model formulations between open and closed systems was the form of the vertebrate stock structure. This was varied by using two vertebrate submodels, which cover the spatial structure scenarios typically seen in large temperate bays:

1) A Beverton-Holt stock-recruitment relationship is used to determine the number of young-of-the-year added to each population in each year (see Appendix C.1.4 for details of the equations used). Using this submodel means the ecosystem is closed, as the vertebrate species is fully contained within the modelled system with little if any feed in from outside the system.
2) Constant levels of young-of-the-year are maintained regardless of the size of the population within the bay. In this case the modelled ecosystem is open, as the vertebrate stocks are fed by recruits from a breeding population that is external to the Bay. Only post-recruitment factors (e.g. mortality once in the population) are important in this submodel, whereas pre- and post- recruitment factors impact the population dynamics in the first vertebrate submodel.

The only vertebrate group whose reproduction was not varied between the two cases was the wading bird group. Due to their biology, wading birds, which breed elsewhere, and only spend part of their year around the bay, are considered to have an external breeding population regardless of the recruitment function used for the other vertebrates.

Case study 2 – Coast-to-open-ocean

The food web modelled in the coast-to-open-ocean operating model contains 19 invertebrate

Table 4.1: The assumptions and formulations of Atlantis.

Feature	Assumptions and/or formulation notes
General features	
biomass units	mg N/m3
input forcing	nutrients, temperature and physics on interannual, seasonal, tidal frequencies
level of group detail	functional group (with a small number of individual species)
resolution of the formulation used for the invertebrate groups	follow the dynamics of the entire biomass pool of the functional group (or species) in the cell
resolution of the formulation used for the vertebrate groups	follow the biomass dynamics (structural and reserve weight) of the ‘average individual’ for the functional group (or species) in the cell and the number of individuals in the cell
time step	adaptive* daily or diurnal time step
Process related	
bioturbation and bioirrigation	yes, simple exchange between layers
consumption formulation	type II (asymptotic), with an availability parameter which can be habitat dependent
equations	five general sets of rate of change equations used (autrophs, invertebrate consumer, vertebrate consumer, bacteria, inanimate)
formulation detail	general: only growth, mortality and excretion explicit
light limitation	optimal irradiance fixed
mixotrophy	yes, for dinoflagellates (if present)
nutrient limitation	external nutrients determine uptake
nutrient ratio	Redfield
oxygen limitation	yes
sediment burial	very low background rate included
sediment chemistry	dynamic, with sediment bacteria
shading of primary producers	yes
spatial (or habitat) limitation	yes (for benthic or demersal groups (and species))
spatial structure	flexible with the potential for multiple vertical and horizontal cells
temperature dependency	yes
transport model used for hydrodynamics flows	yes
Model closure	
top predators represented by static loss terms	some top predators are included explicitly, but predators not explicitly included in the foodweb are represented using quadratic mortality terms
mortality terms	linear and quadratic
Vertebrate and fisheries related	
age structure for the vertebrate groups	multiple age classes (or stages, which equate to life phases), with final age class of each group a “plus group”
fishery discards	target and bycatch groups (and species)
incidental mortality due to fishing	yes
invertebrate fisheries	yes
management	variable, may be via effort limitations, gear limitations, minimum legal size, area or temporal closures and may be based on target or endangered stocks
stock-recruit relationship	Beverton-Holt, productivity-based or constant recruitment
stock structure	depends on recruitment function chosen – may be internal (all the stock within the bay and self-seeds) or external (the reproductive stock outside the bay produces the recruits and the oldest age classes migrate out of the bay to join this stock)

* Time steps at this scale (daily or diurnal) may cause instability in variables with fast dynamics (e.g. phytoplankton groups). These groups (all at the base of the foodweb) use smaller time-steps (adaptive in the sense that to optimise computational efficiency they are as large as they can be without causing instability) which are repeated until full model level time step has been completed. In contrast the higher trophic level groups (and the physical submodels) employ only model level time-steps.

Table 4.2: The components included in the shallow bay Atlantis operating model and their code names. An asterisk (*) indicates that the component also has an internal silicon pool

Components	Code name	Components	Code name	Components	Code name
Pelagic invertebrate groups		*Benthic invertebrate groups (cont.)*		***Nutrients***	
diatoms*	PL	infaunal carnivores	BC	dissolved organic nitrogen	DON
autotrophic flagellates	AF	benthic deposit feeders	BD	ammonia	NH
picophytoplankton	PS	meiobenthos	OB	nitrate	NO
dinoflagellates	DF	aerobic bacteria	AEB	dissolved silicate	Si
free-living pelagic bacteria	PFB	anaerobic bacteria	ANB		
pelagic attached bacteria	PAB			***Physical***	
heterotrophic flagellates	HF	*Vertebrate groups*		light	IRR
microzooplankton	ZS	planktivorous fish	FPS	salinity	SAL
large omnivorous zooplankton	ZL	piscivorous fish	FVS	sediment grain types	PHI
large carnivorous zooplankton	ZLC	demersal fish	FD	bottom stress	STRESS
cephalopods	CEP	demersal herbivorous fish	FG	porosity	POR
		wading birds	WAD	volume	VOL
Benthic invertebrate groups		seabirds	SB		
seagrass	SG	large sharks	SH	***Fishery fleet types***	
macroalgae	MA	marine mammals	MAM	purse seine	PUR
microphytobenthos*	MB			line fishery for piscivores	LIN
macrozoobenthos	MZ	***Dead***		line fishery for sharks	LL
(epifaunal carnivores)		labile detritus	DL	pot	POT
benthic (epifaunal) grazers	BG	refractory detritus*	DR	dredge	DGE
benthic suspension feeders	BF			dive	DIV
		Gaseous		jig	JIG
		dissolved oxygen	O2	trawl	TWL

Figure 4.1: Foodweb structure for the biological and fisheries components of the shallow bay Atlantis operating model. Only direct feeding, waste production, nutrient uptake, and environmental links are shown. The codes are listed in Table 4.2, except for Temperature (T) and Wind (W) which have impacts in the model, but are not explicitly modelled. Dissolved Nutrients (ammonia (NH), nitrate (NO) and dissolved organic nitrogen (DON)) are shown here as a single pool (DN) for convenience and clarity. Bycatch groups for each fishery are listed in Table 4.3.

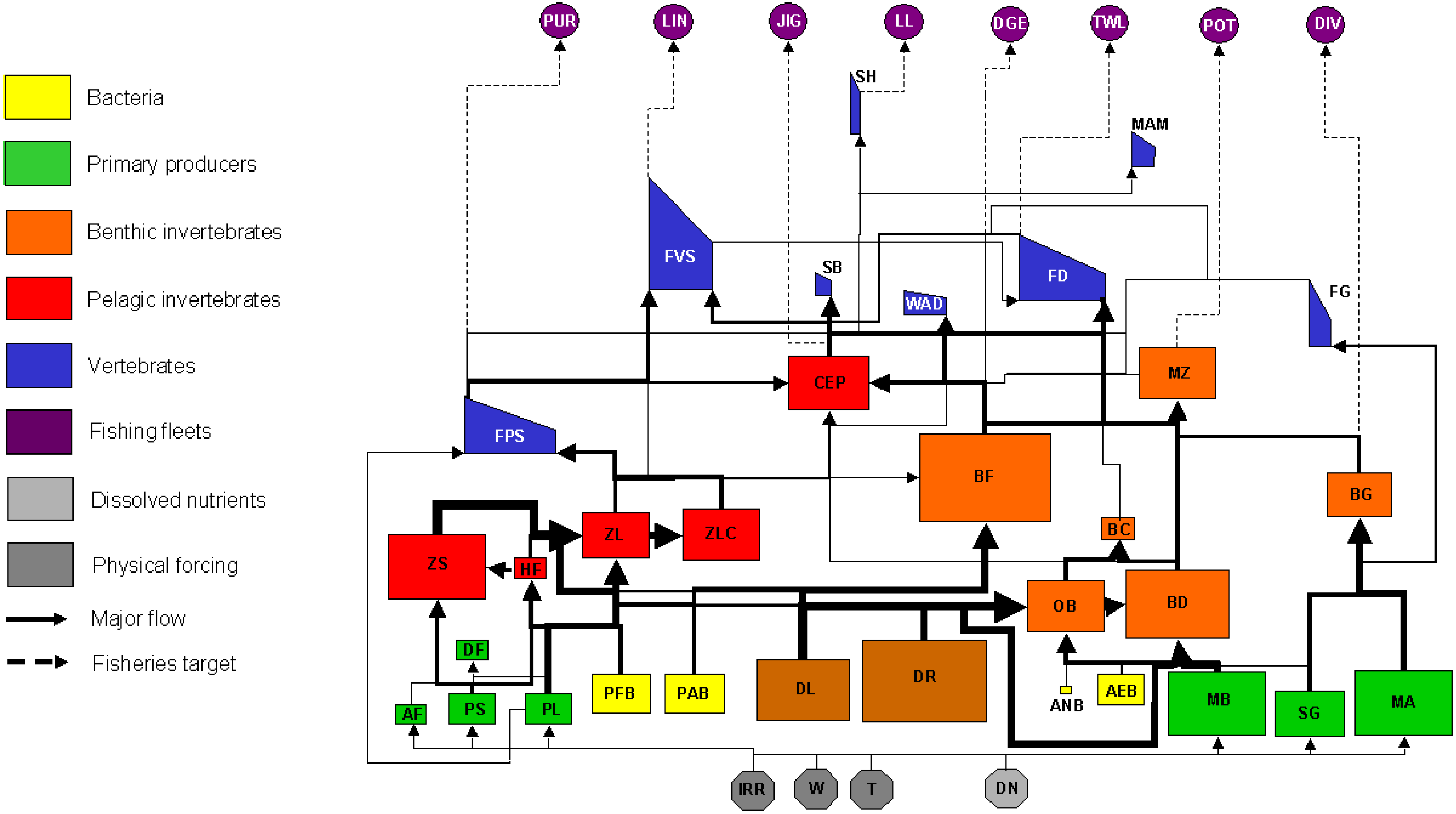


Table 4.3: Target and bycatch groups for the fleets represented in the shallow bay operating model

Fleet	Target groups	Bycatch groups
purse seine	planktivorous fish	marine mammals
Long fishery for piscivores	piscivorous fish	seabirds large sharks
line fishery for sharks	large sharks	seabirds
pot[1]	macrozoobenthos	benthic suspension feeders benthic (epifaunal) grazers infaunal carnivores benthic deposit feeders demersal fish demersal herbivorous fish
dredge	benthic suspension feeders	macrozoobenthos benthic (epifaunal) grazers infaunal carnivores benthic deposit feeders demersal fish demersal herbivorous fish
dive	benthic (epifaunal) grazers	-
jig	cephalopods	-
trawl	demersal fish demersal herbivorous fish	macrozoobenthos benthic suspension feeders benthic (epifaunal) grazers infaunal carnivores benthic deposit feeders

1. See text under fisheries submodels for explanation for the number of bycatch groups included for this fishery.

Table 4.4: The components included in the coast-to-open-ocean Atlantis operating model and their codenames. An asterisk (*) indicates that the component also has an internal silicon pool and the number in parentheses after each fleet type is the number of fleets of that type, differentiated based on what they target.

Components	Codename	Components	Codename	Components	Codename
Pelagic invertebrate groups		*Vertebrate groups (continued)*		***Nutrients***	
large phytoplankton	PL	migratory mesopelagic fish	FMM	dissolved organic nitrogen	DON
small phytoplankton	PS	non-migratory mesopelagics	FMN	ammonia	NH
small zooplankton	ZS	shallow demersal fish	FDS	nitrate	NO
mesozooplankton	ZM	deep demersal fish	FDD	dissolved silicate	Si
large carnivorous zooplankton	ZL	demersal sharks	SHD		
gelatinous zooplankton	ZG	pelagic sharks	SHP	***Physical***	
cephalopods	CEP	seabirds	SB	Light	IRR
		pinnipeds	PIN	Salinity	SAL
Benthic invertebrate groups		baleen whales	WHB	sediment grain types	PHI
seagrass	SG	toothed whales	WHT	bottom stress	STRESS
macroalgae	MA			Porosity	POR
pelagic bacteria	PB	*Vertebrate species*		Volume	VOL
sedimentary bacteria	SB	flathead (Neoplatycephalus spp)	FSF		
meiobenthos	BO	ling (Gentyperus blacodes)	FSL	***Fishery fleet types***	
deposit feeders	BD	orange roughy	FSO	purse seine (3)	PUR
carnivorous infauna	BC	(Hoplostethus atlanticus)		demersal line (3)	DLN
benthic grazers	BG	southern bluefin tuna	FST	pelagic line (1)	PLN
deep filter feeders	BFD	(Thunnus maccoyii)		dredge (1)	DGE
shallow filter feeders	BFS	gummy shark	FSG	dive (1)	DIV
deep macrozoobenthos	BMD	(Mustelus antarcticus)		jig (1)	JIG
shallow macrozoobenthos	BMS			demersal trawl (5)	BTW
		Dead		midwater trawl (3)	MTW
Vertebrate groups		labile detritus	DL	trap (2)	TRP
small planktivorous fish	FPS	Refractory detritus*	DR	recreational (1)	REC
large planktivorous fish	FPL	carrion	DC	gillnet (2)	NET
shallow piscivorous fish	FVS			danish seine (1)	DSN
deep piscivorous fish	FVD	Gaseous		cull (1)	CUL
tropical piscivorous fish (tunas)	FVT	dissolved oxygen	O2		

Figure 4.2: Foodweb structure for the biological and fisheries components of the coast-to-open-ocean Atlantis operating model. Only direct feeding, waste production, nutrient uptake, and environmental links are shown. The codes are listed in Table 4.4, except for Temperature (T) and Wind (W) which have impacts in the model, but are not explicitly modelled. Dissolved Nutrients (ammonia (NH), nitrate (NO) and dissolved organic nitrogen (DON)) are shown here as a single pool (DN) for convenience and clarity. Bycatch groups for each fishery are listed in Table 4.5.

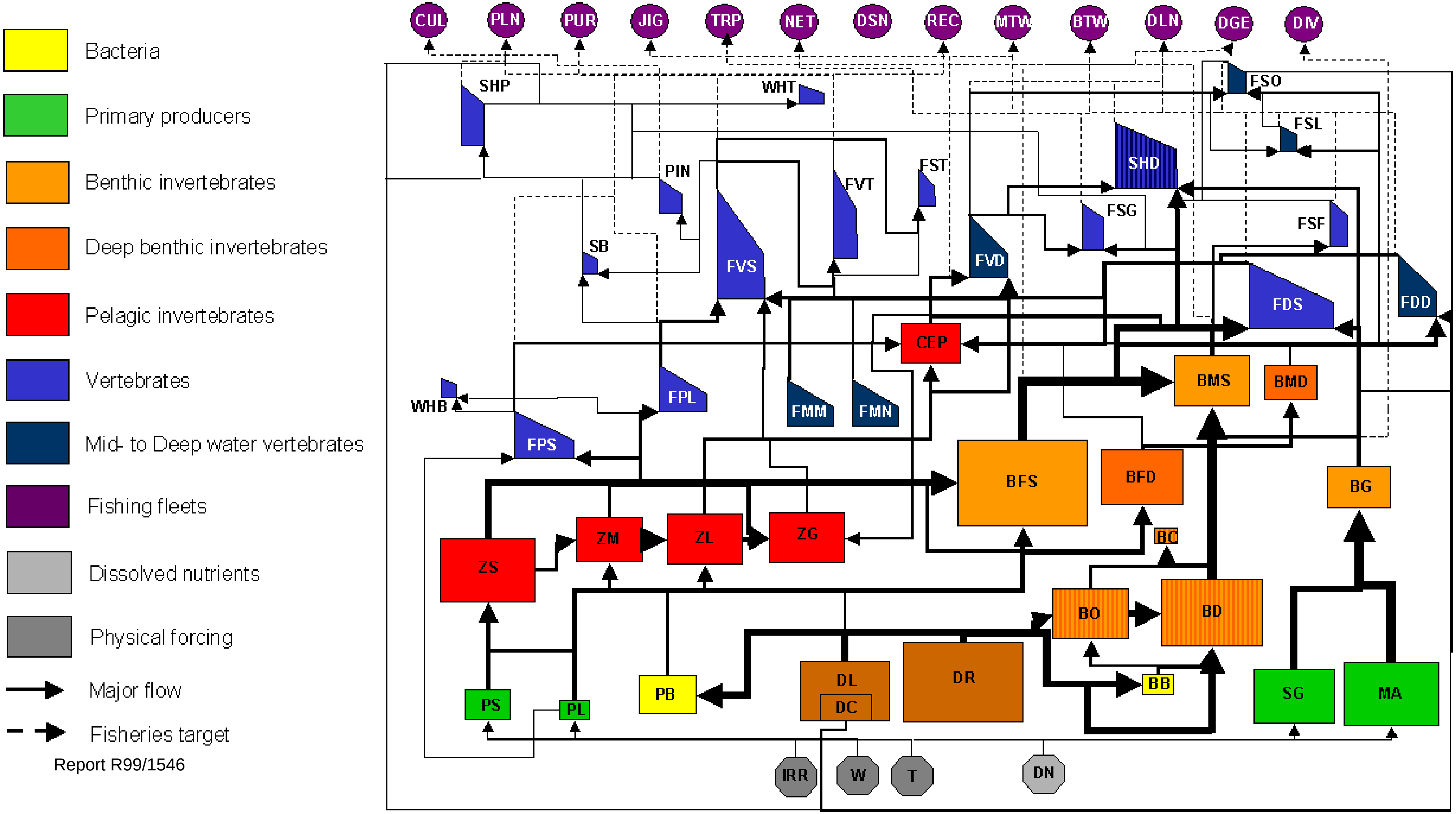


Table 4.5: Target, by-product and bycatch groups for the fleets represented in the coast-to-open-ocean operating model.

Fleet	Target group	By-product groups	Bycatch groups
purse seine for tunas	southern bluefin tuna tropical piscivorous fish		toothed whales
purse seine for planktivores	small planktivorous fish large planktivorous fish		toothed whales
purse seine for piscivores	shallow piscivorous fish		
demersal line deep demersal fish	deep demersal fish ling deep piscivorous fish		
demersal line for piscivores	shallow piscivorous fish		
demersal line for demersal sharks	gummy shark demersal sharks		
pelagic line	southern bluefin tuna tropical piscivorous fish pelagic sharks		seabirds
dredge	shallow filter feeders		macroalgae seagrass
dive	benthic grazers		
jig	cephalopods		
demersal trawl for macrozoobenthos	shallow macrozoobenthos	shallow demersal fish	small planktivorous fish demersal sharks carnivorous infauna macroalgae seagrass deposit feeders gelatinous zooplankton
demersal trawl for cephalopods	cephalopods		

Fleet	Target group	By-product groups	Bycatch groups
demersal trawl for deep demersal fish	deep demersal fish deep piscivorous fish ling	gummy shark shallow macrozoobenthos	migratory mesopelagic fish non-migratory mesopelagic fish demersal sharks deep filter feeders deep macrozoobenthos carnivorous infauna deposit feeders gelatinous zooplankton
demersal trawl for flathead	flathead shallow demersal fish	large planktivorous fish shallow piscivorous fish cephalopods shallow filter feeders	gummy shark demersal sharks pinnipeds shallow macrozoobenthos carnivorous infauna macroalgae seagrass deposit feeders gelatinous zooplankton
demersal trawl for orange roughy	orange roughy		deep filter feeders carnivorous infauna deposit feeders
midwater trawl for cephalopods	cephalopods		
midwater trawl for planktivores	small planktivorous fish large planktivorous fish		pinnipeds
midwater trawl for deep demersal fish	deep demersal fish deep piscivorous fish		migratory mesopelagic fish non-migratory mesopelagic fish
trap for macrozoobenthos	shallow macrozoobenthos		
trap for deep demersal fish	ling deep demersal fish		
gillnet for demersal fish	deep demersal fish shallow demersal fish ling		pinnipeds baleen whales toothed whales

Fleet	Target group	By-product groups	Bycatch groups
gillnet for demersal sharks	gummy shark demersal sharks	pelagic sharks	baleen whales
danish seine	flathead shallow demersal fish	large planktivorous fish shallow piscivorous fish	gummy shark
cull	pinnipeds		
recreational	shallow piscivorous fish flathead shallow macrozoobenthos shallow demersal fish cephalopods shallow filter feeders	large planktivorous fish small planktivorous fish tropical piscivorous fish demersal sharks pelagic sharks benthic grazers	

groups (including microfauna), and 20 vertebrate groups (listed in Table 4.2). These groups represent the main functional groups, and 5 species of interest, in coastal and open waters around southern and eastern Australia. As in the shallow bay case, the functional groups are defined using size, rates of turnover and shared predators and prey. The species of interest selected for inclusion in the operating model are all species of particular commercial significance that have been (or are expected to be) particularly vulnerable to intensive fishing pressure. The resulting foodweb is given in Figure 4.2.

4.1.3 Physical submodels

The hydrographic submodel included in Atlantis (and BM2 before it) is a transport model, modified from that developed for the Port Phillip Bay Integrated Model by Murray and Parslow (1999) and Walker (1999). Horizontally, it resolves the modelled area into discrete polygons (Figures 4.3 and 4.4). The size of each polygon reflects the extent of spatial homogeneity in the physical variables represented in the model (depth, seabed type (reef or flat), canyon coverage, porosity, bottom stress, erosion rate, salinity, light and temperature). Atlantis is also vertically structured and is comprised of multiple layers (up to five water column layers, an epibenthic layer, and a sediment layer (Figures 4.3 and 4.4)). The fluxes driving the transport model are either derived from a spatially and temporally finely resolved three-dimensional non-linear, variable density hydrodynamic model (Walker 1999) or from satellite data. The model is also driven by seasonal variation in irradiance and temperature, as well as by nutrient inputs from point sources, atmospheric deposition of dissolved inorganic Nitrogen (DIN), and exchanges with an oceanic boundary boxes. If the deepest water column box contacts the sediments, a sediment chemistry submodel deals with nutrient remineralisation and oxygen exchange. Otherwise, the bottom of the deepest water column box is also treated as an open boundary.

Case study 1 - Shallow Bay

The modelled shallow temperate bay ecosystem, based on Port Phillip Bay, is represented using 59 polygons and 3 layers (Figure 4.3). In any polygons shallower than 25m the water column layer was truncated appropriately (e.g. a polygon 8m deep would have a single 8m water column layer). The transport model is driven by files containing only four years of flows, light levels, evaporation, precipitation and point source inputs (of nutrients, freshwater etc) that were taken from a full three dimensional hydrodynamics model of Port Phillip Bay, or timeseries of data collected in Port Phillip Bay during the mid 1990s. Once the model reaches the end of a four-year period it rewinds and repeats the four years of physical forcing. In addition to allowing model scenarios to span long time periods, this also allowed for some regular long-term environmental cycles (analogous to ENSO) in the environmental forcing functions.

The standard scenarios run for the first case study span 90 years (after model burn in), with output of data values every 28 days.

Case study 2 – Coast-to-open-ocean

The modelled coast-to-open-ocean ecosystem, based on the southeast region of Australia, is represented using 59 polygons and up to 7 layers (Figure 4.4). If the depth of a polygon is less than 1800 m, the water column layer(s) are truncated to match (e.g. a box in that is 100m deep would have 2x50m water column layers). Any open ocean cells that are deeper than 1800m have no epibenthic or sediment layers, and are treated as having an open boundary under the deepest water column layer. The transport model is driven by files containing a single year of flows (estimated from satellite data, Dunn *unpublished*), light levels, evaporation, precipitation and point source inputs generated artificially, but based on real time series collected from the Southeast of Australia from the mid to late 1990s. As described in the first case study the forcing files a repeated through time (the flows annually, the other files every four years).

The standard scenarios run for this case span 120 years (after model burn in), with output of data values every 90 days. These times do not match those for the first case study. The length of the run was extended to accommodate the long-term dynamics of some of the long-lived vertebrate groups and the possible trajectories of fisheries development over such a large area. The lower temporal resolution in the output data was necessary due to data storage limitations. It does omit some fine scale variation in the groups with fast turnover dynamics, but it is still sufficient to capture seasonal patterns, variability and the dominant patterns in the output (Fulton 2001), especially given that the forcing is estimated from running averages of satellite data and the large volume of the model cells (Fulton *unpublished*).

4.1.4 Fisheries submodels

The Atlantis operating models include the main types of fleets typical of those kinds of ecosystem. Each fleet is characterised by its target, byproduct and bycatch groups, gear type (and associated selectivity curve and habitat impacts), habitat dependency, discarding, and effort allocation submodels. The alternative fisheries submodels are detailed below after the fleets in each operating model are described.

Case study 1 - Shallow Bay

Eight fisheries are included in the bay ecosystem operating model (Table 4.2): purse seine, longline, pot, dive, dredge, trawl, jig and shark line fisheries. These fisheries are differentiated by their primary targets, but may share bycatch groups (Table 4.3). These fisheries represent the primary fisheries operated in temperate bays. The recreational fisheries are not explicitly represented, but are lumped with the commercial fisheries using the same gear. Also the “pot”, fishery actually represents all the fisheries targeting the macrozoobenthos (e.g. crabs). These include fisheries using drop nets, set nets, pots, trawls, haul nets, beach seines, beam tide trawls, fish traps, and wading. Consequently, the range of non-target groups affected by the ‘pot fishery’, as implemented in the bay ecosystem operating model, extends beyond those usually associated with a pot fishery.

Case study 2 – Coast-to-open-ocean

Thirteen types of fisheries are included in the coast-to-open-ocean operating model (Table 4.3): purse seine, demersal line, pelagic line, dredge, dive, jig, demersal (bottom) trawl, midwater trawl, trap, gillnet, Danish seine, recreational and pinniped cull fisheries. There are multiple fleets from some of the types of fisheries, which represent fleets with different target or byproduct groups (Table 4.5). There is overlap in the groups taken (either as targeted catch, byproduct or bycatch) by the various fisheries.

Figure 4.3: Horizontal and vertical spatial geometries used to define the temperate bay ecosystem based on Port Phillip Bay. The fine black lines indicate the boundaries of model polygons, thick black lines mark the edges of management zones, and sampling locations are indicated by black dots.

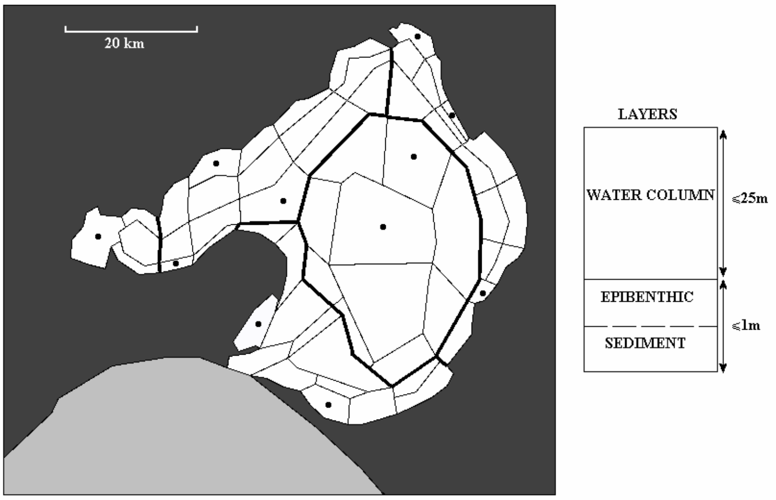


Figure 4.4: Horizontal and vertical spatial geometries used to define the coast-to-open-ocean-ocean ecosystem based on the southeast region of Australia. The fine black lines indicate the boundaries of model polygons, thick black lines mark the edges of management zones, and sampling locations are indicated by black dots.

Alternative Fisheries Submodels

Several alternative fisheries submodels are included in the operating models. These include alternative bycatch, habitat dependency, selectivity, discarding, and effort allocation models. All of these can change through time - representing changes in fishing practices, advances in gear, alternative management strategies and the like.

For each of these types of submodels (i.e. bycatch or selectivity or discarding or effort allocation etc) each fleet can only use one of the alternative formulations at a time, but the formulation used can change through the course of a run. In addition, different fleets can be using different alternatives for the various submodels. As the different types of submodel are not mutually exclusive, multiple combinations of the various submodels can be activated in any one model scenarios. For instance, in the basic fishing scenarios the dive fishery uses constant selectivity, constant discarding rate, prescribed effort matrices, a TAC, and has no incidental impacts on the environment; while the demersal trawl fishery targeting deep demersal fish uses the dynamic fleet model, a normal selectivity curve, variable discarding, adaptive management, some TACs, has byproduct groups and causes incidental impacts on the biogenic habitats.

Effort Allocation

The wide range of effort allocation submodels available in Atlantis vary from very simplistic constant or prescribed effort through to more dynamic fleet models. While none of the alternative submodels are as sophisticated as fleet dynamics models that model the behaviour of individual vessels (e.g. Little *et al*. 2004), the more elaborate submodels in Atlantis do represent the dynamics of aggregate fleets and allow for behavioural responses to effects such as effort displacement due to the depletion of local stocks or the creation of Marine Protected Areas. In contrast, the simplest submodels allow for an exploration of fishing effects when there is little information beyond location and magnitude of catch taken. The alternative effort allocation submodels are:

- constant fishing pressure;
- constant effort with the fishing pressure per cell adjusted for area;
- constant effort distributed spatially based on CPUE of previous record period (day, week, season etc);
- prescribed effort distribution (effort per cell per quarter specified by user and model linearly interpolates between them from day-to-day to avoid numerical discontinuties);
- dynamic effort based on distribution of CPUE in previous record period;
- fleet dynamics model incorporating ports (which become operable and developed for fishing at different times), distance (with a fuel penalty function), exploratory fishing and the distribution of CPUE in the previous record period;
- human population based recreational fishing effort (from operating ports);
- effort displacement due to MPA placement or low CPUE; and
- effort which can change through time, and the change can be persistent or pulsed.

The formulations for these submodels are given in Appendix D.

Selectivity Curves

A number of selectivity curves have been included in the Atlantis framework (equations are given in Appendix D). The specific curve used can differ between fleets taking the same group and across groups taken by the same fleet. This allows for a wide range of gear-species interactions and impacts to be modelled. The selectivity curve types included in Atlantis are:

- constant;
- cohort specific constant;
- size based logistic curve;
- size based normal curve;
- size based lognormal curve;
- size based gamma curve;

The selectivity submodel also allows for a vertical mismatch between the group being fished and the gear deployment depth.

Discarding submodels

Each fleet discards a proportion of the catch of each species taken as waste (to represent offal etc). In addition to this there is explicit discarding defined for each group taken by each fishery – to account for high grading or the discarding of undersize individuals. There are three alternative discarding submodels:

- constant;
- cohort specific constant; and
- size based (minimum legal size etc).

The discarding submodel also allows for illegal retention of undersize fish. The formulations for these discarding submodels are given in Appendix D.

Bycatch submodels

The alternative bycatch submodels deal with alternative assumptions regarding the level and distribution of the incidental impacts of fishing on non-target groups (listed in Tables 4.3 and 4.5). Incidental impacts on the ecosystem components due to fishing can either be in the form of groups taken as bycatch (e.g. marine mammal take in purse seines), or in the form of habitat destruction due to the passage of fishing gear (e.g. the crushing of epibenthos). The three alternative submodels are:

- the passage of fishing gear causes no incidental damage to non-target groups (i.e. fishing only impacts the harvested groups);
- the amount of incidental impact on the ecosystem is determined by the simple percentage overlap of the area (habitat types) occupied by the group and the area (habitat types) the fishery can access; or
- the amount of incidental impact on the ecosystem is determined using a variation of the probabilistic model of trawl distribution developed by Ellis and Pantus (2001).

The formulations for these bycatch submodels are given in Appendix D.

4.1.5 Management submodels

The management submodels included in the Atlantis framework form a two-part hierarchy.

At the higher management level, fisheries can be managed using non-adaptive methods, or they can use adaptive methods which respond to reference limit triggers. In the adaptive case, if the estimated stock size of target, vulnerable or endangered groups falls below a reference limit, then a more specific management measure can be activated. The measure can be introduced over a user defined time period (so it can be effective immediately or have a lag of many years in implementation) and it can be reviewed periodically to see if the measures can be relaxed or need to be tightened further. The estimated stock size can be based on fisheries data only, or on scientific research surveys (the latter is more commonly used in the scenarios included in this report). The date taken as the virgin biomass reference point can also be varied, to evaluate the effects of historical ratchets (e.g. Pauly's ratchet, Pauly 1995) on stock and ecosystem properties (e.g. "health").

The lower level of the management hierarchy is made up of the range of management measures available in Atlantis. These measures are:

- Total Allowable Catches (TACs) which are set per target or vulnerable group (usually only target group) and which can have multispecies considerations – multispecies fisheries may be allowed to exceed their TACs in a user defined number of species before more stringent management measures (e.g. closures) are imposed;
- gear control (e.g. changes in tow length or mesh size, which are represented respectively via changes in swept area or selectivity curves);
- level of escapement (which may either be a constant, a cohort specific constant, or size based);
- temporal closures that may be temporary or permanent and which may be complete closures for multiple years or only seasonal closures (with the length of the seasonal closure fixed at its initial value or dependent on annual stock estimates); and
- zoning and Marine Protected Areas (so that some areas are open to all fishing, others only accessible by certain gear types, and others defined as completely "no take" zones), which can be of variable sizes, seasonal, in fixed locations or rotating, and have associated rates of infringement and can cause effort displacement to other areas or depths.

The type of management used (e.g. whether a fishery is seasonal and uses TACs) can change through time. Different fleets may also be managed in different ways. For example, an MPA may prevent trawl fishing in a particular area, but purse seining for planktivores may occur anywhere and is only halted if limit reference points for endangered groups are exceeded.

4.2 Scenarios

To investigate the implications for indicator performance of alternative fisheries developments, management actions, environmental and ecological conditions and the simultaneous impacts of other anthropogenic activities, a number of model scenarios were considered for each case study (full list given in Table 4.6). Some of these are based on real events from Australia and elsewhere, while others are simple "what-if" ideas. In some cases the amount of pressure put on the system is beyond that seen in Australian system to date, but it is representative of what has

Table 4.6: Fisheries, ecological and environmental scenarios run in the shallow bay and coast-to-open-ocean operating models. Codes for the scenarios that are used elsewhere in the report (e.g. tables of results) are also given.

Operating Model	Scenario	Code
Shallow Bay		
	Closed system that is unfished (other than research surveys), with fixed levels of nutrient loading[1]	SCN1
	Closed system with eutrophication – as for unfished, but with fivefold increase in nutrients	SCN2
	Closed system with sequential fish down and no incidental mortality	SC1
	Closed system with sequential fish down, no incidental mortality and changing discarding practices	SC2
	Closed system with sequential fish down and homogeneous incidental mortality	SC3
	Closed system with sequential fish down and patchy incidental mortality	SC4
	Closed system with sequential fish down, patchy incidental mortality and changing discarding practices	SC5
	Closed system with sequential fish down, patchy incidental mortality and effort management based on stock size of target groups	SC6
	Closed system with sequential fish down, patchy incidental mortality and spatial management (zoning) in place	SC7
	Closed system with sequential fish down and no trawl Marine Protected Areas (at fixed locations)	SC8
	Closed system with sequential fish down and no take Marine Protected Areas (at fixed locations)	SC9
	Closed system with sequential fish down and fivefold increase in nutrients	SC10
	Open system with sequential fish down and no incidental mortality	SO1
	Open system with sequential fish down and patchy incidental mortality	SO2
	Open system with sequential fish down, patchy incidental mortality and effort management based on stock size of marine mammals and seabirds	SO3
	Open system with sequential fish down and mammals recovering from past large scale harvesting	SO4
Coast-to-open-ocean		
	Unexploited system – no anthropogenic nutrient release, coastal degradation, mean temperature increases or fishing (other than research surveys)	ON1
	Unfished system with intensive levels of all other anthropogenic pressures – fivefold increase in nutrient loads, habitat degradation or clearing around human settlements and 10% (about 2 degree) increase in average temperature	ON2
	Sequential fisheries with light fishing pressure	O1
	Sequential fisheries with intensive fishing pressure (some groups pushed to commercial extinction)	O2
	Sequential fisheries with light fishing pressure and episodic recruitment pulses	O3
	Sequential fisheries with intensive fishing pressure (some groups pushed to commercial extinction) and mammals recovering from past large scale harvesting	O4
	Sequential fisheries with light fishing pressure and mammals recovering from past large scale harvesting	O5

Sequential fisheries with intensive fishing pressure and coastal habitat degradation (around human settlements)	O6
Sequential fisheries with intensive fishing pressure and changing discarding practices	O7
Sequential fisheries with intensive fishing pressure and effort management based on vulnerable groups (particularly marine mammals and seabirds)	O8
Sequential fisheries with intensive fishing pressure and gear creep	O9
Sequential fisheries with intensive fishing pressure and gear limitation by management	O10
Sequential fisheries with intensive fishing pressure and no take Marine Protected Areas (at fixed locations)	O11
Sequential fisheries with intensive fishing pressure and no take Marine Protected Areas (at rotating locations)	O12
Sequential fisheries with intensive fishing pressure and no trawl Marine Protected Areas (at fixed locations)	O13
Sequential fisheries with intensive fishing pressure and no trawl Marine Protected Areas (at rotating locations)	O14
Sequential fisheries with intensive fishing pressure and no incidental habitat destruction	O15
Sequential fisheries with intensive fishing pressure and episodic recruitment pulses once per decade	O16
Sequential fisheries with intensive fishing pressure and episodic recruitment pulses twice per decade	O17
Sequential fisheries with intensive fishing pressure and adaptive management based on changing length of fishing season	O18
Sequential fisheries with intensive fishing pressure and site attachment for all but those groups/species known to be highly migratory (e.g. tunas)	O19
Sequential fisheries with intensive fishing pressure and changing coverage (access) of demersal trawl fleets – to represent improvement in targeting due to GPS and acoustic mappers and also switches to gear that can access rougher ground	O20
Sequential fisheries with intensive fishing pressure and management via Total Allowable Catches (TACs)	O21
Intensive levels of all anthropogenic pressures – fisheries, fivefold increase in nutrient loads, habitat degradation or clearing around human settlements and 10% (about 2 degree) increase in average temperature	O22

1. Note that the scenario for the unfished open system with fixed levels of nutrient loading was run. As this scenario was tuned to give the same levels of biomass in this unfished state as for the closed system and as the subsequent indicator analysis showed that the results for the unfished open system were no different to those for the unfished closed system it has not been included in the list of scenarios discussed in this report.

occurred elsewhere in the world. It is also what was required to produce good contrast in the data generated for indicator testing.

The most informative scenarios were those where the ecosystem underwent large scale ecological change overall, or where there was a lot of contrast created in at least a few key ecosystem components or attributes. Informative scenarios were given priority in the sensitivity analyses, as it was not feasible (due to the size of the task) to repeat all possible combinations of scenarios and parameterisations. Ideally sensitivity to all process formulations would have been considered, but this was not possible in the time available so previous experience with the Atlantis framework (from Fulton 2001) was used to guide the parameterisations and processes selected for testing.

Fisheries Scenarios

Most of the alternative scenarios dealt with different fisheries assumptions, or parameterisations and alternative fisheries development and effort trajectories. By combining various forms of the many fisheries submodels, over fifty scenarios were run for the shallow bay case study (Fulton *et al*. 2003). Evaluation of the biomass trajectories of these scenarios indicated that many of them were very similar and only the thirteen most informative were included in the analysis by Fulton *et al*. 2003. More scenarios from the first case study have been included in the analysis discussed in this report, but it is still only a subset of the full set of scenarios run. The biomass trajectories in those scenarios omitted from the analysis duplicate those from scenarios included in the analysis and so would not add any extra information.

The full set of scenarios run in the shallow bay case study was not run in the coast-to-open-ocean case study. The scenarios run for the second case study consisted of the most informative combinations of submodels from the bay case study as well as a few scenarios which did not make sense at the scale of a single bay, but were important for ecosystems of the scale of the one represented by the coast-to-open-ocean operating model. These latter scenarios are all concerned with details of effort build-up and distribution through time. It made no sense to consider these scenarios in the shallow bay operating model as boats can steam across an ecosystem of that size in much less than a day. In the larger ecosystem though, they are very important as it can take days for a boat to transit to and from potential fishing sites at this scale, which can lead to very different resulting fleet and system dynamics.

Overall, the types of fisheries scenarios run and analysed are:

- no commercial fishing (research surveys only);
- fixed levels of commercial fishing;
- sequential development (described below) of vertebrate and invertebrate fisheries with fixed gears, no incidental impacts of fishing and with/without adaptive management;
- sequential development (described below) of vertebrate and invertebrate fisheries with incidental mortality due to fishing and with/without adaptive management (in the second case study these scenarios were run with a range of parameterisations for the rate of exploration and effort displacement);
- sequential development (described below) of vertebrate and invertebrate fisheries with incidental mortality due to fishing and changing gears, habitat dependencies and discarding practices, and with/without adaptive management;

- sequential development (described below) of vertebrate and invertebrate fisheries with incidental mortality due to fishing and spatial management zoning (multiple scenarios were run with the size of any “no take zones” varying between large (multiple model boxes) to very small (small % of a single model cell), which may be fixed or rotating, and with/without adaptive management; and
- sequential development (described below) of vertebrate and invertebrate fisheries with incidental mortality due to fishing and a background increase in pinniped (marine mammal) population size as it recovers from past large scale harvesting that had reduced the population to a small fraction of its virgin biomass.

Note that when fishing caused incidental mortality due to fishing, multiple scenarios were run using the alternative incidental fishing submodels.

In the scenarios with active commercial fisheries there are usually five phases of development expressed (Figure 4.5). In phase one increasing effort sees the fishery transitions from exploratory (negligible effort levels) to a viable commercial fishery. In phase two there is a gradual but consistent increase in effort until the fishery is putting intense pressure on the stock and low return CPUEs or management measures stall further fleet expansion. In phase three there is continuous intensive fishing pressure and often an expansion into byproduct targeting. In phase four the fishery either collapses or is reduced in size substantially by management actions aimed at restoring stock sizes. In the final phase there is a slight recovery in effort levels as the fishery rebuilds with the recovering stock. Depending on the rate of initial growth and the life history characteristics these phases may take more (or less) time and some fisheries may never reach the final stages, plateauing at relatively constant levels while in phase 2 or 3 (Figure 4.5) – this is more common for fisheries on cephalopods or planktivores). In the sequential fish down scenarios there are staggered starts for the different fleets. The fisheries targeting the higher trophic levels in shallow or easily accessible waters begin first, followed by high value invertebrates. Those fisheries targeting lower trophic levels or stocks in deepwater or harder to access habitats only commence once the CPUE for the easily accessible stocks of higher trophic levels dips (locally or over the entire model domain) (Figure 4.6). These development patterns are representative of those seen in reality (Pauly *et al.* 1998, Pauly *et al.* 2003).

Ecological Scenarios

In combination with the fisheries scenarios listed above, alternative ecological scenarios are considered. Given the very different nature and scales of the two ecosystems modelled it makes little sense to use the same ecological scenarios in both cases. In particular the scenarios used in the bay case study occurred in some parts of the coast-to-open-ocean without requiring specific parameterisations. For instance, while it made sense to consider the implications of open and closed forms of the bay, the source-sink dynamics of some geographical regions of the coast-to-open-ocean system meant that different areas of the model were already open (or closed).

In the first case study the fisheries scenarios are run using constant recruitment (i.e. an open system with stocks external to the bay supplying the recruits) and Beverton-Holt-like recruitment (i.e. a closed system where the stocks in the bay supply all the new recruits) (see Appendix C). The other type of ecological scenario considered in the first case study is the number of strong trophic links in the foodweb. The number of strong links is varied across a wide range, but the resulting systems are substantially different only when the number of links

Figure 4.5: Example effort trajectories showing the fisheries phases. P1 = transition from exploratory to viable commercial fishery; P2 = gradual increase in effort until pressure is intense and further expansion stalls; P3 = continuous intensive fishing pressure; P4 = fishery collapse or substantial effort reduction; and P5 = recovery in stock and fishery rebuilding. Dashed lines are bounds on possible levels of effort; red line is an effort trajectory showing eventual overexploitation and stock decline; and blue line is an effort trajectory showing a sustainable fishery. The trajectory used for each fishery can vary with the scenario considered and the exact bounds and length of the phases through time can vary fishery to fishery.

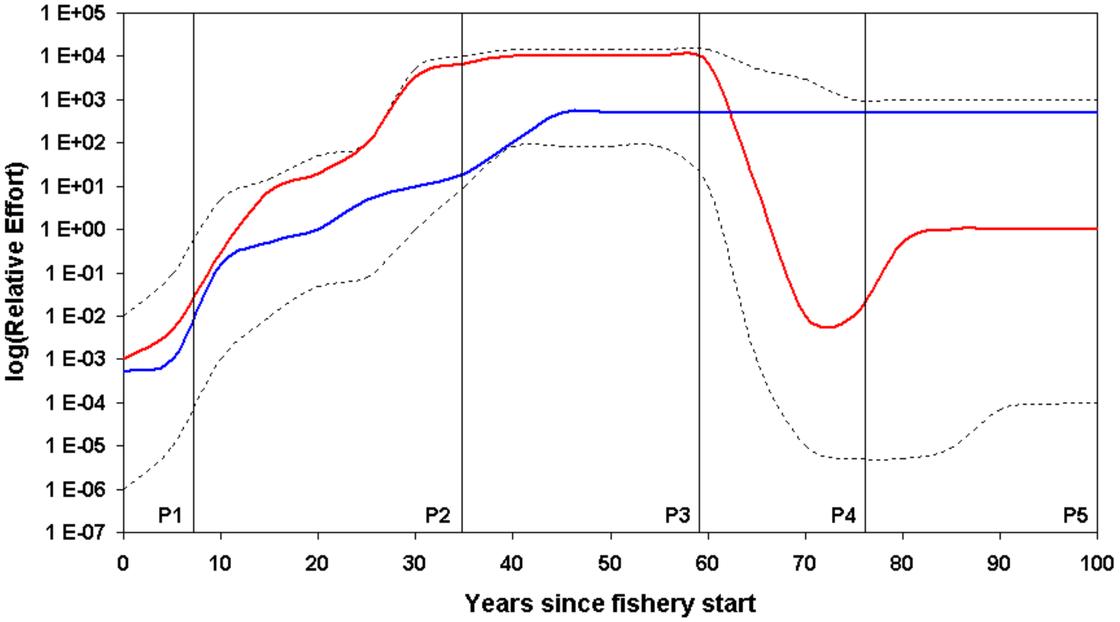
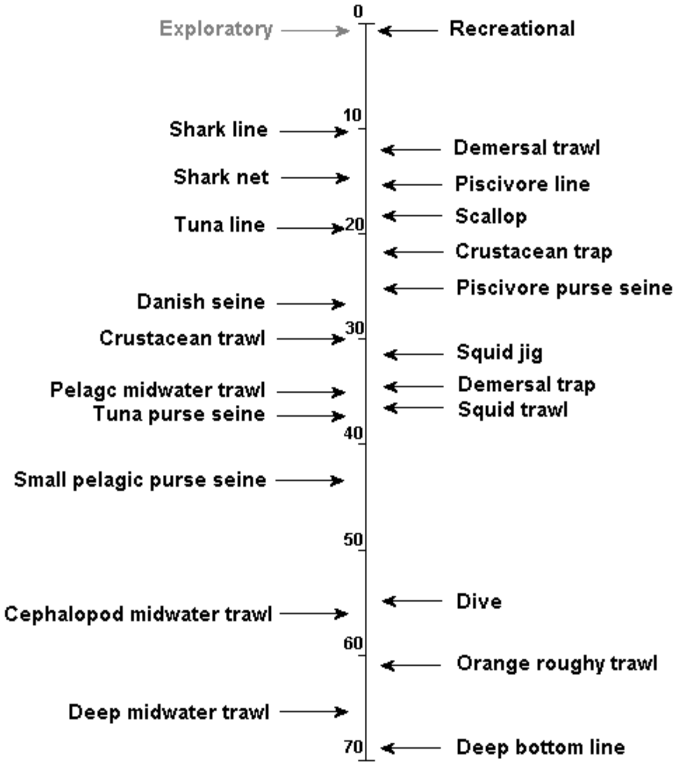


Figure 4.6: Example of staggered starts (fishery in P1 phase in Figure 4.5) of fisheries in sequential fish down scenarios (time axis in years relative to start of first large scale fishery – the recreational fishery). Arrows indicate start time of the fishery. Exact timing can vary depending on scenario definitions, and grey exploratory fishery indicates that exploratory and small scale fisheries exist before the start dates given for the commercial fisheries.



is increased from the default setting of few strong links and many links to an alternative structure where the web links are all much tighter.

In the second case study the ecological scenarios consider issues of dispersal, connectivity and recruitment. The specific ecological scenarios considered are: periodic recruitment of large age classes; variability in recruitment; the degree of larval dispersal (whether local or more widespread); the amount of vertical spread in the distribution of surface dwelling and mid-water fish groups; rates of migration in to or out of the model domain; and the alternative forms of fish movement, whether its fixed seasonally, has a high degree of site fidelity, or is density- and forage-dependent.

Environmental Scenarios

The environmental scenarios all consider anthropogenic changes to the environment due to industries or processes other than fisheries. These were run in combination with the fisheries and ecological scenarios to consider the impacts of simultaneous changes in fishing pressure and other environmental conditions.

The first environmental scenario is a contaminant scenario where point source inputs of nutrients are increased fivefold (a level known to cause eutrophication in coastal boxes). The second environmental scenario is the degradation of shallow benthic habitats due to coastal development and associated activities (e.g. dredging). The last environmental scenario is a general increase in temperature.

In each of the environmental scenarios the change in conditions is produced by multiplying the baseline input files used in the transport model by a scalar (s) that changes linearly with time (t):

$$s = m \cdot \frac{(t - t_0)}{t_p} \tag{4.1}$$

where m is the relative amount by which the value increases over the period; t_0 is the day of the scenario at which the change begins; and t_p is the period over which the change comes into effect.

4.3 Model behaviour

Atlantis captures the essential features of temperate marine ecosystems and captures ecological processes, such as resource partitioning and assemblage formation.

4.3.1 Regional assemblages

The physical, ecological and biogeochemical processes included in the operating models lead to the formation of regions within the modelled area which are characterised by specific groups, or assemblages. While there is variation through time and from cell-to-cell, there are general patterns that hold regardless of the finer details. These patterns can be discerned directly from maps of the model generated biomass distributions, but they are quickly and clearly identified using the method described in Fulton *et al.* (2004b) – where assemblages are determined by (1) considering the fourth root transform of the average biomasses of all groups in each box on a two-dimensional non-metric Multidimensional Scaling (MDS) plot derived from a Bray Curtis similarity matrix; and then (2) examining the average values of the physical variables and the biomass per group (using the SIMPER routine of the Primer software package (Clark and Warwick 1994)) to ascertain which groups determine the clustering seen in the MDS. This analysis identifies areas (groups of boxes) in the model output that share biological and physical characteristics.

Case study 1 - Shallow Bay

The unfished form of the shallow bay operating model predicts two types of assemblages: central and edge (Figure 4.7). There is a single central assemblage that is marked by a distinct plankton community (mainly small phytoplankton and large zooplankton) and while there is some epifauna, it is dominated by infauna, particularly hardy microfauna (meiobenthos, microphytobenthos, and bacteria). The edge assemblages are all of similar form, though details differ. These assemblages are dominated by one of two alternative plankton assemblages, and a rich assemblage of fish, macrophyte and benthic macrofauna.

Case study 2 – Coast-to-open-ocean

The unfished form of the coast-to-open-ocean operating model predicts the presence of at least 15 assemblages in this large area, which fall into five general forms. The form of the assemblages is largely related to depth and nutrient levels (and mixing). The bay assemblages (at A, B and C in Figure 4.8) are shallow water, high nutrient and high light assemblages supported primarily by benthic primary producers and detritus and featuring shallow living fish (or juveniles of groups that move deeper with age). The shelf assemblages (the grey-blue, purple, royal blue, light blue, pink, cyan and yellow areas in Figure 4.8) are similar to the bay assemblages, but with the addition of a few groups (or age stages) that live in slightly deeper (or open) water. The differences between the shelf assemblages are due to changes in local productivity, with those areas fed by upwellings (e.g. the aqua zone in Figure 4.8) including more groups at higher biomasses than the less productive shelf areas (e.g. the purple area). The slope assemblages (cream, maroon and dark blue areas in Figure 4.8) are strikingly different to those on the shelf. They have none of the shallow water groups and no photosynthetic producers (though the detritus and zooplankton supporting the webs there may be sourced from the photic zone), but do include all the deepwater groups (e.g. ling and orange roughy). The maroon area (which includes the slope and adjoining underwater plateau) is the most productive of these areas. The seamount assemblage (the orange areas) are intermediate between the shelf and slope assemblages. The seamount assemblage does not contain all of the shelf groups (e.g. benthic primary producers are missing), but as planktivores, tunas and the higher order predators are attracted to these locations, it contains many more groups than the slope assemblages. The final assemblage is the pelagic (open ocean) assemblage (the grey area in Figure 4.8). This contains no demersal or benthic groups and is dominated by the smaller plankton groups, as well as gelatinous zooplankton and vertebrates with planktivorous or piscivorous diets. As these assemblages were identified using temporally aggregated model output a single pelagic assemblage was identified, but if the data had been considered at a finer temporal resolution (e.g. seasonally) then this large area would have been split - with a

Figure 4.7: Assemblages predicted by unfished shallow bay operating model. Boxes of the same colour contain the same assemblage. Foodwebs for some assemblages are given (not all assemblages are shown for clarity). The foodwebs for the grey, pink and maroon areas are similar to that for the green area, and the assemblage for the dark blue area is similar to that for the light blue area.

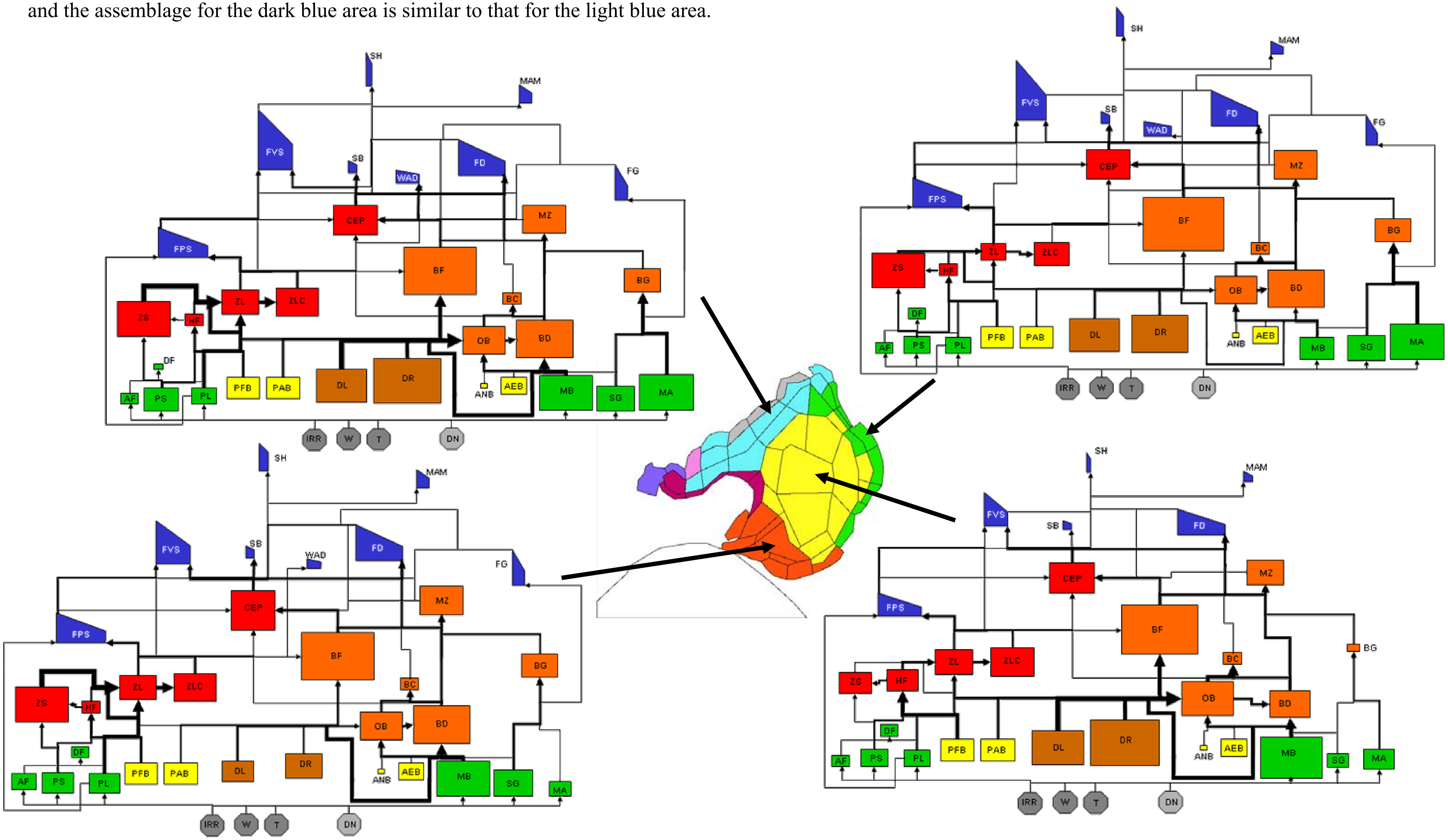
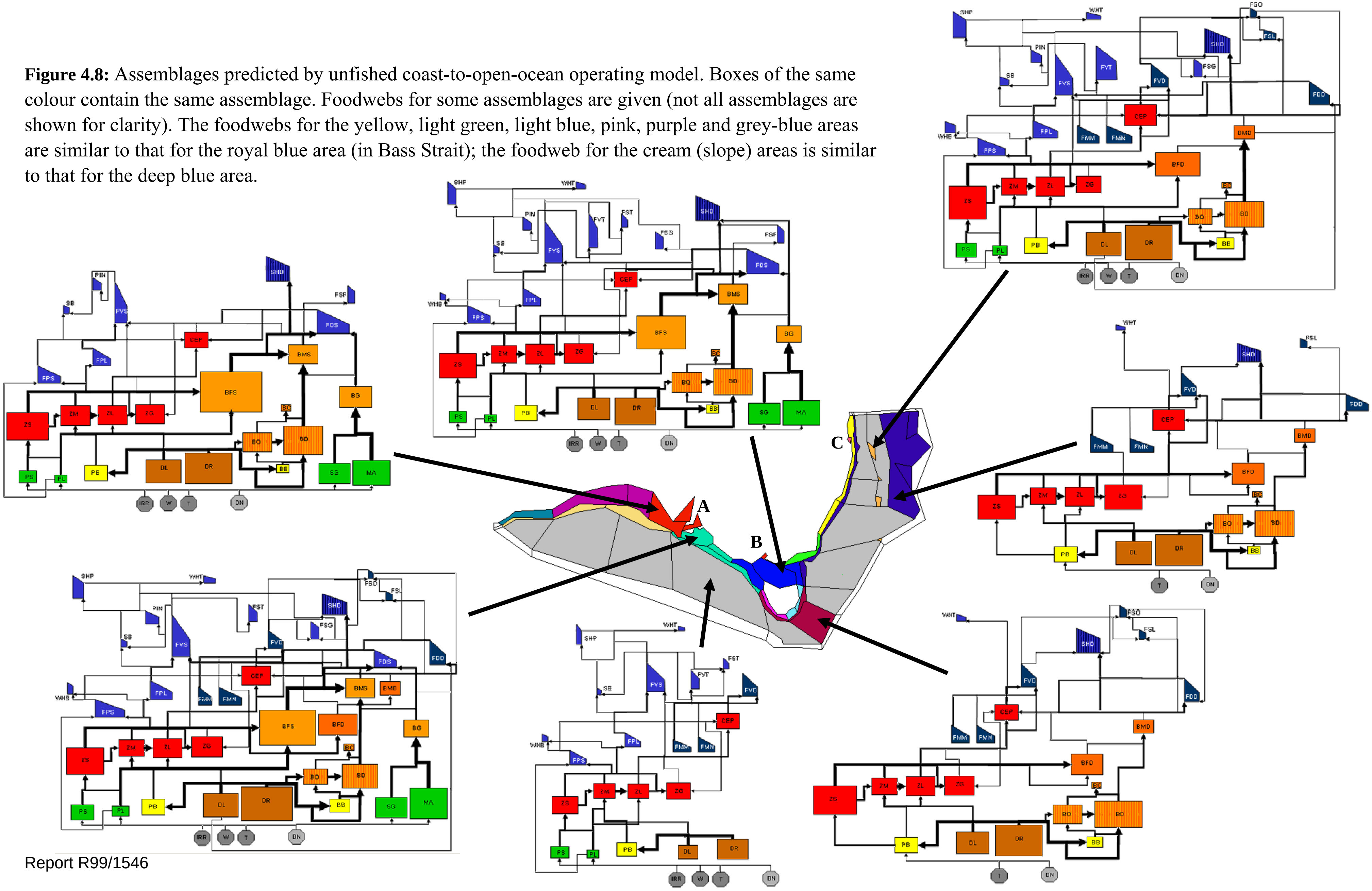


Figure 4.8: Assemblages predicted by unfished coast-to-open-ocean operating model. Boxes of the same colour contain the same assemblage. Foodwebs for some assemblages are given (not all assemblages are shown for clarity). The foodwebs for the yellow, light green, light blue, pink, purple and grey-blue areas are similar to that for the royal blue area (in Bass Strait); the foodweb for the cream (slope) areas is similar to that for the deep blue area.



different assemblage recognised either side of the maroon area (i.e. at least one pelagic assemblage in the open ocean boxes along the bottom of the modelled area and then at least another one in the ocean boxes up the eastern side of the modelled area).

4.3.2 Perturbed states

When fishing (or other anthropogenic pressure) is applied to the modelled ecosystems there are shifts in the assemblages. Different groups can dominate and the differences between regions can drop (or rise) substantially.

Case study 1 - Shallow Bay

The general form of the assemblages remains the same when anthropogenic pressures (e.g. fishing and nutrient pollution) are introduced into the shallow bay operating model. However, there is less differences between the edge assemblages and the area covered by the central assemblage expands, particularly under increased nutrient loads (Figure 4.9). Within assemblages the effect of fishing was largely to reduce those groups directly targeted or taken as bycatch, though groups that could also benefit from fishing due to new food sources (discards) or release from predation or competition (e.g. planktivores, cephalopods and benthic deposit feeders) often increased despite being simultaneously negatively effected by fishing. Examples of the form of assemblages before and after heavy fishing pressure has been applied in the shallow bay operating model are given in Figure 4.10. The assemblages also shifted with changes in nutrient loading, with the epifauna, seagrass and large-bodied fish groups declining and the biomass of small zooplankton and large phytoplankton increasing substantially.

Case study 2 – Coast-to-open-ocean

In contrast to the shallow bay operating model, where the effects of nutrient loading can have the greatest impact on model dynamics (Fulton and Smith *in press*), the anthropogenic activity with the greatest impacts on the predicted ecological state of the ecosystem state is fishing. Coastal development and nutrient pollution do have some effects, particularly in the bays and in the shelf boxes directly effected by the processes, but fishing had much wider impacts. As with the shallow bay operating model, the change in assemblages with fishing saw many areas become more similar. However, the areas on the edges of the model domain (in particular the blue-gray, purple and cream areas) become more different to the other areas as they do not suffer the same levels of exploitation. While some highly mobile groups are depleted even there (as the movement of individuals into fished areas sees the group “globally” decline) other less mobile groups either do not decrease at all or do not decrease to the same extent as seen in the fished areas.

As in the shallow bay case those groups directly affected by the fisheries often decreased in dominance in the assemblages, particularly the higher trophic level vertebrate groups that are targeted by fishing. While those groups (e.g. baleen whales) that only spent part of their time in the model domain were only relatively lightly impacted (often due to incidental mortality or loss of preferred prey groups); and the lower trophic level groups (e.g. planktivores, Mesopelagics, gelatinous zooplankton and cephalopods) increased dramatically (see the examples in Figure 4.11). The loss of the larger (older) individuals in the target vertebrate

Figure 4.9: Pattern of assemblages predicted when increased (fivefold) nutrient levels are applied in the shallow bay operating model.

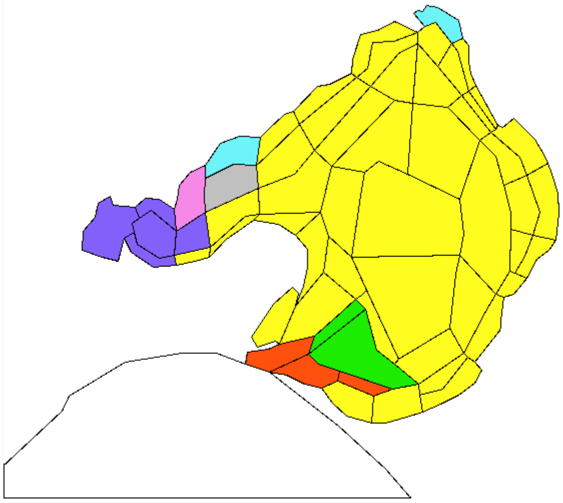
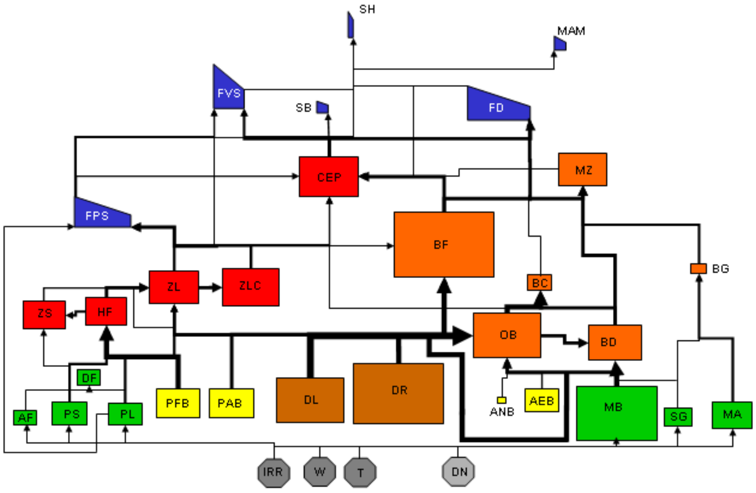


Figure 4.10: Examples of assemblages from the shallow bay operating model before and after intensive fishing pressure is applied: (a) central assemblage, (b) edge assemblage (from the light blue area in Figure 4.7)

(a) BEFORE



AFTER

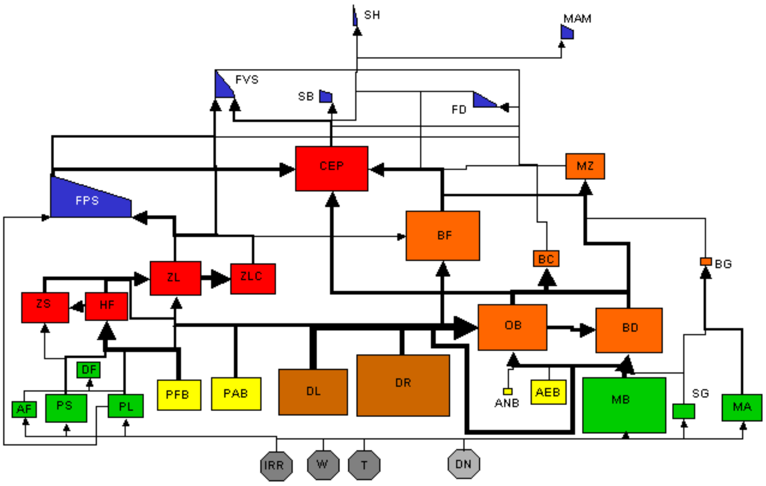
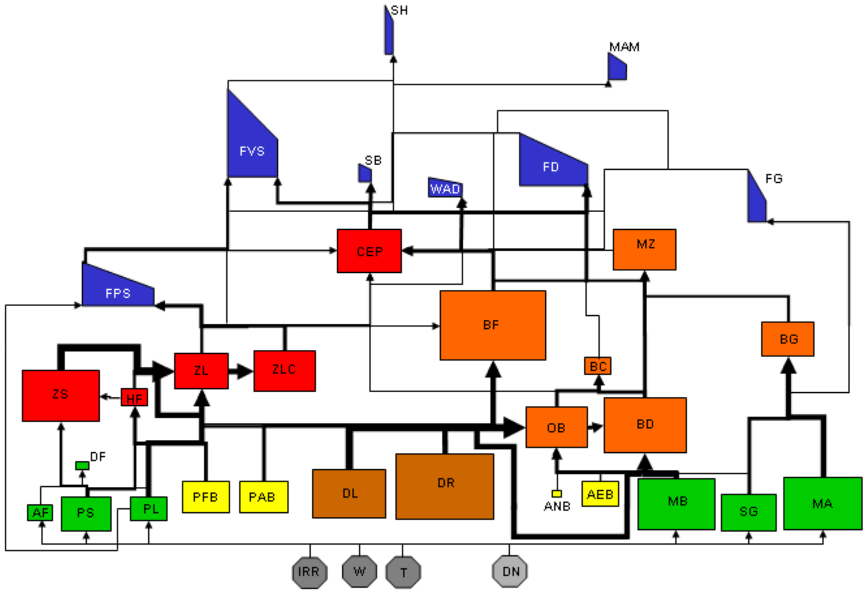


Figure 4.10: continued.

(b) BEFORE



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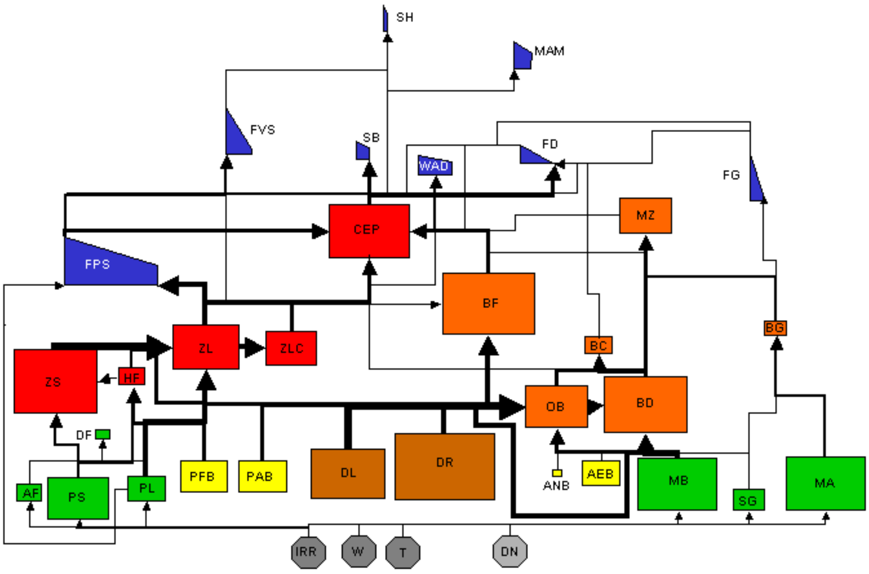
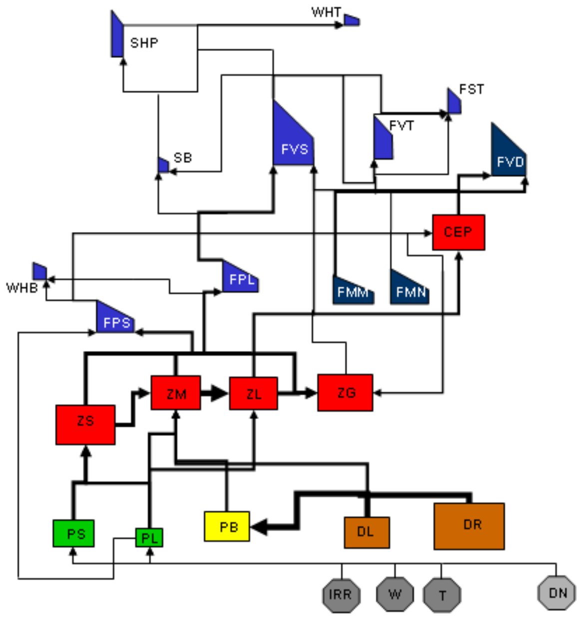


Figure 4.11: Examples of assemblages from the coast-to-open-ocean operating model before and after intensive fishing pressure is applied: (a) pelagic assemblage, (b) heavily fished shelf assemblage (from the royal blue area in Figure 4.8), (c) lightly fished shelf assemblage (from the purple area in Figure 4.8) and (d) deepwater assemblage (from the maroon area in Figure 4.8).

(a) BEFORE



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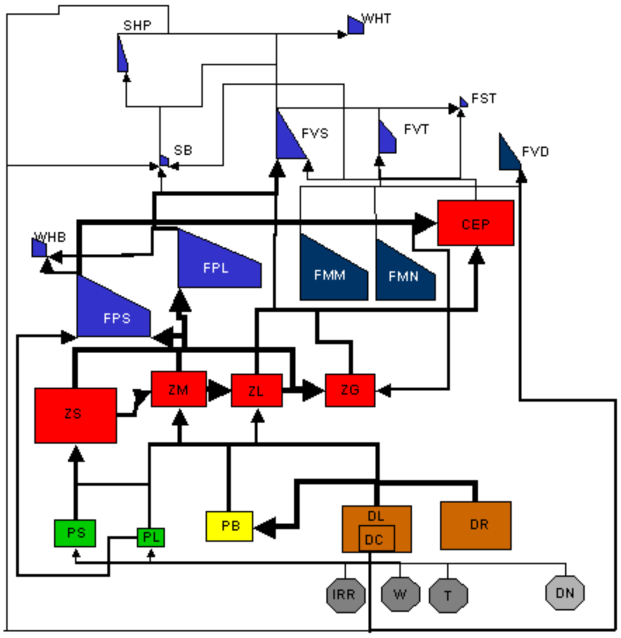
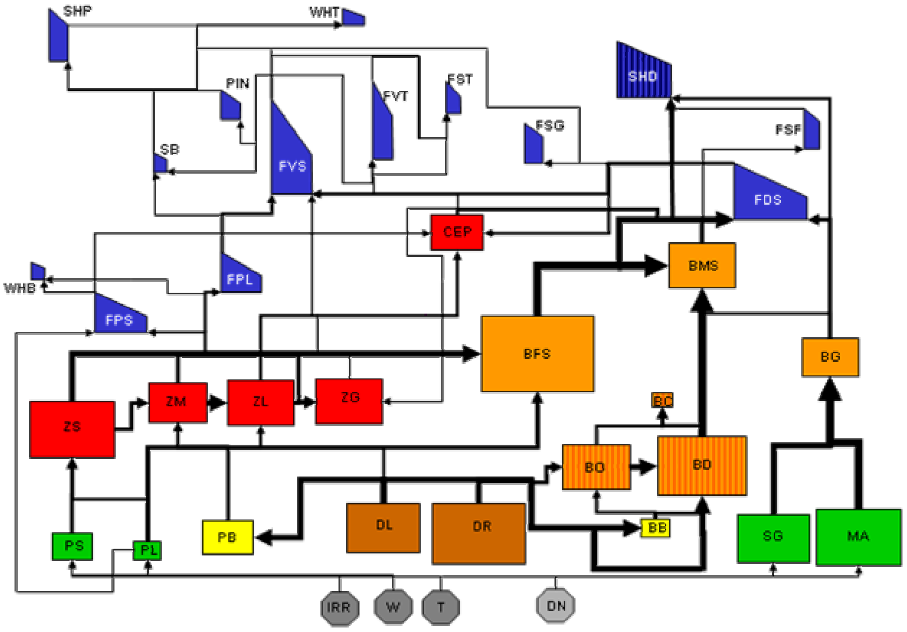


Figure 4.11: continued.

(b) BEFORE



AFTER

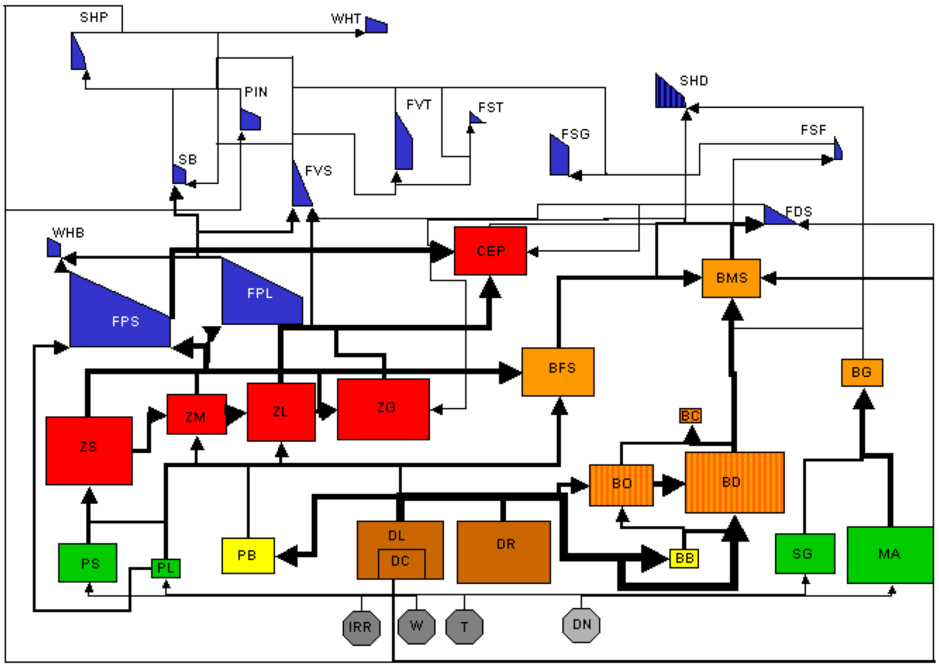
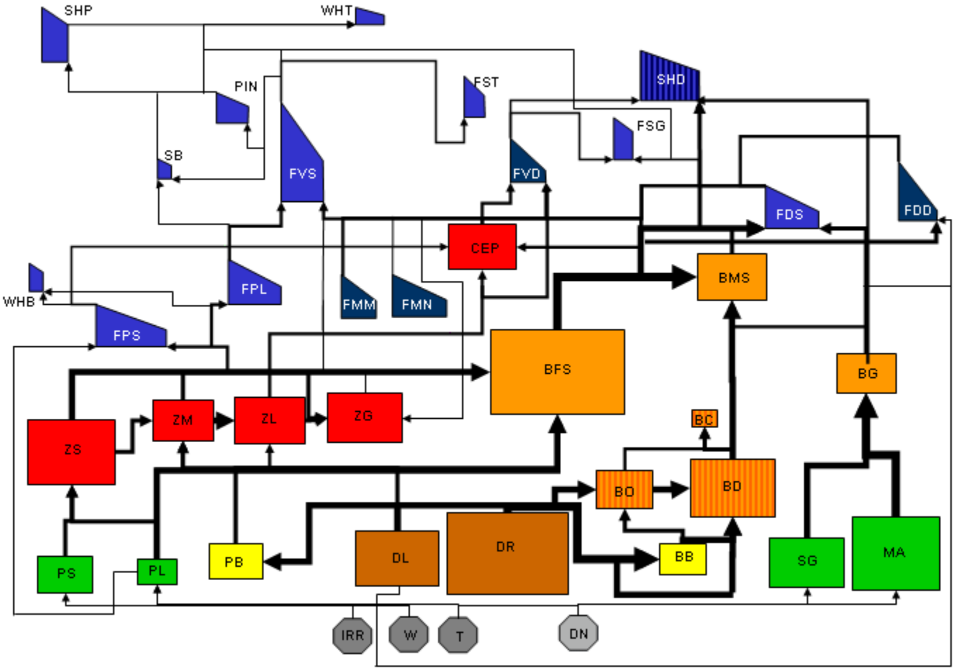


Figure 4.11: continued.

(c) BEFORE



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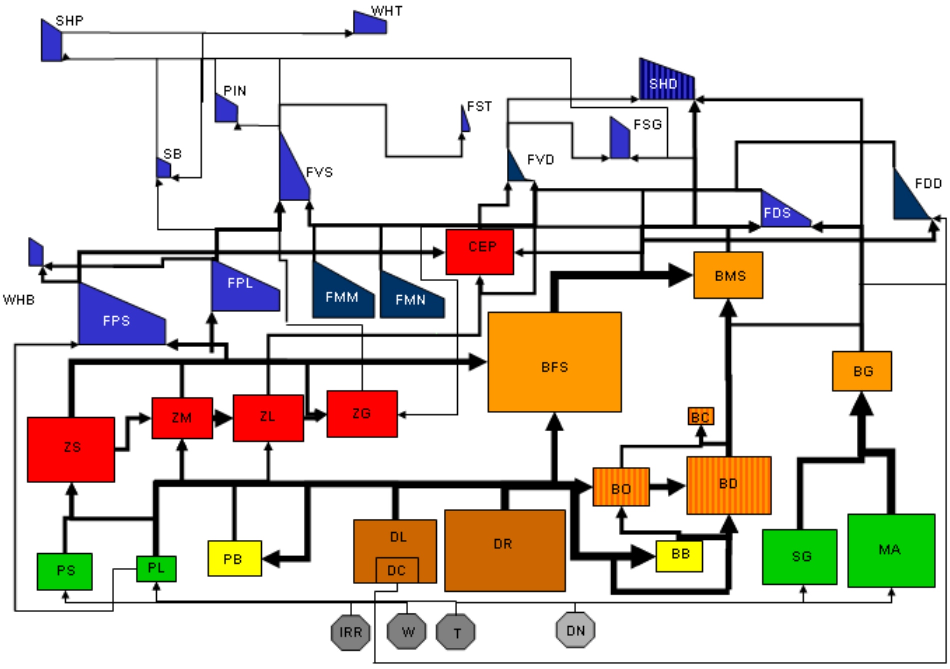
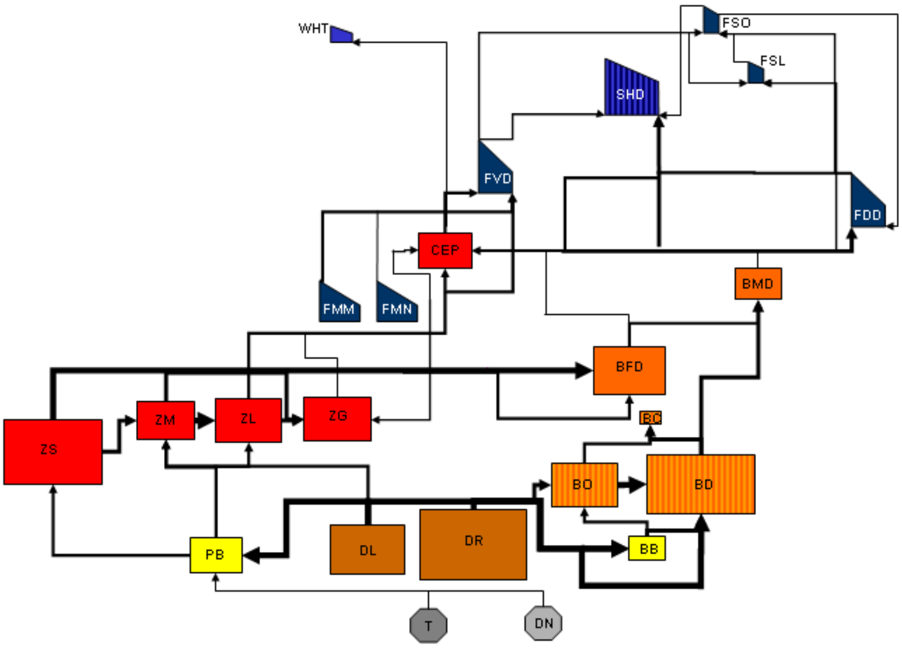
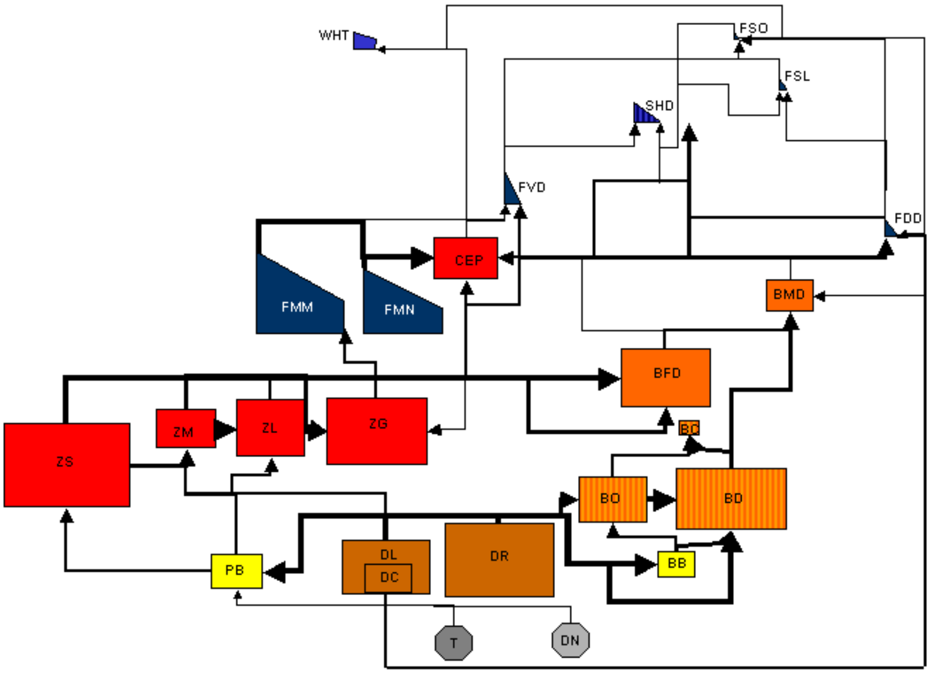


Figure 4.11: continued.

(d) BEFORE



AFTER



groups is also much more evident in this version of the operating model (see Figure 4.11), though it is obvious in the shallow bay case when it is treated as a closed system.

4.3.3 Scenario differences

The general patterns described above are found in the various scenarios, but there are differences scenario to scenario (which tests the utility if the indicators in the alternative situations) and these are outlined below.

Case study 1 - Shallow Bay

Closed system that is unfished (other than research surveys), with fixed levels of nutrient loading (code: SCN1)

As described above, the unfished system shows distinct central and edge assemblages (shown in Figure 4.7). The central assemblage marked by a plankton community dominated by small phytoplankton and large zooplankton and a benthic community with a large proportion of microfauna (meiobenthos, microphytobenthos, and bacteria) and the edge assemblages dominated by finfish and benthic primary producers and macrofauna.

Closed system with eutrophication – as for unfished, but with fivefold increase in nutrients (code: SCN2)

As outlined above, in the eutrophied system the epifauna, seagrass and large-bodied fish groups decline and the biomass of small zooplankton and large phytoplankton increasing substantially.

Closed system with sequential fish down and no incidental mortality (code: SC1)

While largely similar to the scenario where there is a fish down, incidental mortality and spatial management (described below), there are a few notable differences. The higher trophic level target groups (e.g. piscivorous and demersal fish) are much more heavily impacted (declining to less than 1% of their unfished levels). Amongst the non-target groups many show the same patterns as for the scenario with incidental mortality included (as prey are still released and scavengers can still access discards), though those groups that are not directly fished tend to stay closer to their biomass levels in the unfished scenario. Interestingly, those groups most directly effected by change in nutrient regimes (in particular seagrass) are still impacted by the intensive fishing. This is due to changes in nutrient regimes as a result of discarding and the associated biological processes.

Closed system with sequential fish down, no incidental mortality and changing discarding practices (code: SC2)

The system structure predicted in this scenario differs very little from that in the scenario where there is a fish down, incidental mortality and spatial management (described below). It does however, have a much stronger scavenger subweb. The difference in strength developing after the shift to size-based discarding (“high grading”).

Closed system with sequential fish down and homogeneous incidental mortality (code: SC3)

While similar to scenario where there is a fish down, incidental mortality and spatial management (described below) the target groups in this scenario without spatial management are depressed to less than a tenth of that with zoning in place. Even those target groups that benefit (to some degree) from fishing (e.g. the macrozoobenthos which scavenge discards) are impacted by the fishing, though no where near to the extent of those species which do not directly benefit from fishing (this is particularly the case for the benthic invertebrates).

Closed system with sequential fish down and patchy incidental mortality (code: SC4)

This system predicted by this scenario is similar to that for the case with homogeneous incidental mortality, except that the more site attached (and sessile or sedentary) groups are less effected by fishing – having biomasses between 10% and 100% greater than in the homogeneous case.

Closed system with sequential fish down, patchy incidental mortality and changing discarding practices (code: SC5)

This scenario is largely the same as the scenario for the closed system with sequential fish down, no incidental mortality and changing discarding practices (described above). The differences between these scenarios are that (i) the strength of the scavenger web is not as pronounced in this scenario (as there is less bycatch and so less discarding in total), and (ii) the system is more erratic. The scenario with homogeneous incidental mortality shifts in state toward that of the scenario with all human impacts in place and is sufficiently perturbed to remain in that state year round. Whereas the scenario with patchy incidental mortality fluctuates with the activities of those fisheries with the highest rates of discarding – shifting between a "healthier" state (lower biomasses for "bloom" and scavenger groups), when these fisheries are the least active, and the state permanently present in the case with homogeneous incidental mortality, when these fisheries are most active. It is likely that this fluctuation is symptomatic of a transient event in the ecological structure of the ecosystem that would eventually collapse into one of the two states if the scenario was run for an extended period.

Closed system with sequential fish down, patchy incidental mortality and effort management based on stock size of target groups (code: SC6)

This scenario is very similar to that where there is a fish down, incidental mortality and spatial management (described below), except that the benthic invertebrate groups are more heavily depleted.

Closed system with sequential fish down, patchy incidental mortality and spatial management (zoning) in place (code: SC7)

As described above, in this scenario the target and directly impacted groups generally decline, while any groups that substantially benefit from fishing due to the provision of new food sources (discards) or the release from predation or competition increase.

Closed system with sequential fish down and no trawl Marine Protected Areas (at fixed locations) (code: SC8)

While similar to the scenario fish down, incidental mortality and zoning, this scenario which stops all trawling in all the protected areas has higher levels of the higher trophic level target species (by up to twofold). As a result of increasing fishing pressure in the non-trawl fisheries and the higher biomass of predator groups, groups such as the planktivores have biomass levels lower than in the other fished scenarios.

Closed system with sequential fish down and no take Marine Protected Areas (at fixed locations) (code: SC9)

The ecological structure of the ecosystem predicted in this scenario is closer to the structure seen in the unfished scenario than to any of the fished scenarios. Although localised effects of fishing in those areas open to fishing do cause substantial shifts in biomass (towards the levels seen in the other fished scenarios) in the benthic invertebrate and site associated vertebrate groups.

Closed system with sequential fish down and no take Marine Protected Areas (at fixed locations) (code: SC10)

The system structure predicted in this scenario is a mixture of those seen in the scenario with fishing and zoning and the unfished scenario with a fivefold increase in nutrient loading. While the target groups are depleted, the effects of the nutrient loading on the system are often stronger than the effects of fishing, and where the two impacts overlap rather than cancel out the effects can be substantial. For instance, while small zooplankton and large phytoplankton increase substantially as a result of the eutrophication, the large-bodied fish groups (that are impacted heavily by shifts in the web due to eutrophication as well as targeting by fishing) drop to less than 1% of their virgin levels.

Open system with sequential fish down and no incidental mortality (code: SO1)

While the spatial and biomass patterns for the invertebrate groups in this scenario are largely the same as for the scenario with no incidental mortality and a closed system, the vertebrate groups show a lot less of an impact of fishing (as there is restocking from outside of the model domain).

Open system with sequential fish down and patchy incidental mortality (code: SO2)

In comparison to the analogous scenario in the closed system the target groups do not decline to the same degree (remaining at 60% of virgin biomass or greater rather than dropping to as little as less than 1% of the virgin biomass levels). In addition, as the target species are not as heavily impacted by fishing, fishing continues at very high levels and the benthic and site associated groups that are not replenished from outside stocks are more heavily impacted than is the case in the scenario for the closed system (most have biomass levels less than half of those seen in the closed system case).

Open system with sequential fish down, patchy incidental mortality and effort management based on stock size of marine mammals and seabirds (code: SO3)

There scenario is very similar to that for the scenario where there is an open system with sequential fish down and patchy incidental mortality (described above).

Open system with sequential fish down and mammals recovering from past large scale (code: SO4)

The system state predicted in this scenario is similar to that in the scenario where there is an open system with sequential fish down and patchy incidental mortality (described above). As may be expected given the scenario specifications, the noteworthy differences in this scenario are centred on the marine mammals. The increase in biomass of marine mammals as their stocks recover eventually lead to biomass levels greater than seen in any other scenario – including the unfished scenario. While this increased biomass does lead to a slight drop in the biomass of their prey species (particularly those also targeted by fishing), the biomass of mammals even in this scenario is still not large in absolute terms and so it does not cause large cascading effects on the other groups.

Case study 2 – Coast-to-open-ocean

Unexploited system – no anthropogenic nutrient release, coastal degradation, mean temperature increases or fishing (other than research surveys) (code: ON1)

As described above, the unfished form of the coast-to-open-ocean operating model predicts the presence of five general forms of assemblages: the bay assemblages (supported primarily by benthic primary producers and detritus and featuring shallow living fish); shelf assemblages, which are similar to bay assemblages, but also feature groups (or ages) that live slightly deeper; slope assemblages that lack all the shallow water groups and photosynthetic producers, but do include all the deepwater groups; seamount assemblages, which contain a mix of shelf and slope groups; and the pelagic (open ocean) assemblage that contains no bottom associated groups and is dominated by the smaller plankton groups, gelatinous zooplankton and planktivorous or piscivorous vertebrates.

Unfished system with intensive levels of all other anthropogenic pressures – fivefold increase in nutrient loads, habitat degradation or clearing around human settlements and 10% (about 2 degree) increase in average temperature (code: ON2)

Localised effects of coastal habitat degradation and nutrient pollution. In those impacted areas the epifauna decline and planktonic groups increase (particularly the large phytoplankton and small zooplankton).

Sequential fisheries with light fishing pressure (code: O1)

Similar to the scenario with intensive fishing, but with less depletion of the target groups; and in turn less of an increase in groups released from predation and competition. For example, while the increase in the biomass of planktivores still occurs it is only a 20th of that seen when there is intensive fishing.

Sequential fisheries with intensive fishing pressure (some groups pushed to commercial extinction) and spatial management (zoning) in place (code: O2)

As mentioned above, groups directly effected by the fisheries usually decrease in dominance in the assemblages, particularly any fisheries targeted higher trophic level vertebrate groups (or age stages). In contrast, those groups (e.g. baleen whales) that only spend part of the year (or life cycle) in the model domain are only lightly impacted; and the lower trophic level groups: e.g. planktivores, mesopelagics, gelatinous zooplankton and cephalopods) increase dramatically.

Sequential fisheries with light fishing pressure and episodic recruitment pulses (code: O3)

Similar to that for light fishing without episodic recruitment, but the form of the system remains closer to the unexploited system for longer, particularly in the pelagic. However the bycatch groups on the fishing grounds are more heavily impacted locally as fishing pressure continues longer and these groups do not all benefit from their own episodic recruitment.

Sequential fisheries with intensive fishing pressure (some groups pushed to commercial extinction) and mammals recovering from past large scale harvesting (code: O4)

Similar in overall form to the scenario with intensive fishing and zoning, except that the piscivorous and pelagic groups have slightly lower biomasses and the main shallow water demersal fish groups have higher biomasses (though they are still effectively commercially extinct).

Sequential fisheries with light fishing pressure and mammals recovering from past large scale harvesting (code: O5)

The biomass levels and system structure seen in this scenario is largely similar to the scenario with light fishing but no mammal recovery. Notably, as for the shallow bay operating model, the biomass of marine mammals increases to a point that is higher than that seen in any other scenario, including the unfished system. The effects of this increase are greater in this operating model though as the biomasses attained are much larger in absolute terms and so their prey groups are more heavily impacted. This impact is most obvious in those groups also targeted by fishing. While these groups in no way decline to the extent seen in the scenarios with intensive fishing their biomass does decline to levels lower than in the scenario with light fishing pressure and no large scale mammal recovery.

Sequential fisheries with intensive fishing pressure and coastal habitat degradation (around human settlements) (code: O6)

While similar in overall form to the scenario with intensive fishing the only form of anthropogenic pressure on the ecosystem, the biomass of shallow water benthic and demersal groups in this model are all lower in this scenario. These shifts in the ecological structure of the ecosystem also lead to some impacts on pelagic groups as the top level piscivores switch to more pelagic prey groups as their demersal prey groups decline. These effects are not evident in the west of the model domain (where there is little human habitation), but are particularly evident in the east where human settlements are found in all the shallow water model cells.

Sequential fisheries with intensive fishing pressure and changing discarding practices (code: O7)

While similar to the scenario with intensive fishing pressure and zoning (described above) the shift in discarding practices in this scenario means that groups that consume discards (including the shallow water demersal fish groups) have biomass levels that are higher than in the scenario with zoning despite the continued intensive fishing pressure.

Sequential fisheries with intensive fishing pressure and effort management based on vulnerable groups (particularly marine mammals and seabirds) (code: O8)

While the system predicted in this scenario is largely similar to that in the scenario with intensive fishing pressure and zoning (described above) the higher trophic level target groups (piscivorous and demersal fish) are not quite as depleted (though they are still heavily impacted by fishing). The direct result of this slightly higher biomass level for the predatory groups is that the biomass of their prey species does not increase as much as in the scenario with zoning.

Sequential fisheries with intensive fishing pressure and gear creep (code: O9)

The final biomass levels from this scenario are similar to those for in the scenario with intensive fishing pressure and zoning (described above). The major difference between the scenarios is that the depletions occur more quickly and the final biomass levels (particularly for benthic and demersal groups) are slightly lower than in the scenario with zoning.

Sequential fisheries with intensive fishing pressure and gear limitation by management (code: O10)

The system structure predicted by this scenario is again largely similar to those for in the scenario with intensive fishing pressure and zoning (described above). While there is some increases in biomasses for the target groups, the gear modifications usually lead to changes in size structure of the stocks rather than leading to substantially higher biomasses. The most notable biomass increases in this scenario are actually in those species (e.g. seabirds and marine mammals) which directly benefit from mitigation devices.

Sequential fisheries with intensive fishing pressure and no take Marine Protected Areas (at fixed locations) (code: O11)

The final biomasses predicted in this scenario are similar to those for the scenario with intensive fishing pressure and zoning (described above), but there are notable differences. All of the site associated benthic and demersal groups have much higher biomasses in this scenario (more than 10x higher than those for the scenario with zoning), but the highly mobile pelagic piscivore groups are even more depleted (due to effort displacement and shifts in targeting and the consumption of their juveniles in shallow waters by resident, protected, demersal groups).

Sequential fisheries with intensive fishing pressure and no take Marine Protected Areas (at rotating locations) (code: O12)

While this scenario is very similar to that when the no take MPAs are at fixed locations, the biomass levels of the target groups are all at lower levels in this case.

Sequential fisheries with intensive fishing pressure and no trawl Marine Protected Areas (at fixed locations) (code: O13)

Similar to the system for the scenario with no take MPAs in fixed locations, but with all target groups and non-trawl bycatch groups (e.g. seabirds) at lower biomass levels.

Sequential fisheries with intensive fishing pressure and no trawl Marine Protected Areas (at rotating locations) (code: O14)

While this scenario is very similar to that when the no trawl MPAs are at fixed locations, the biomass levels of the target groups are all at lower levels in this case.

Sequential fisheries with intensive fishing pressure and no incidental habitat destruction (code: O15)

While largely similar to the scenario where there is intensive fishing pressure and zoning, there are some differences. The target groups are generally more heavily impacted in this scenario – in particular those groups which are normally robust to fishing pressure – as predators or competitors normally taken as bycatch remain in the system and so release is not to the same level. The exception to this trend is the demersal shark groups which are usually more heavily affected by incidental than direct fishing mortality, thus with the removal of incidental mortality their overall mortality due to fishing pressure is reduced and the resulting biomass levels are higher. The scavenger web is also weak in this scenario as there are fewer dicards. The final difference is that there is often less difference between the different assemblages identified in the model output, particularly in the shallow waters not directly influenced by upwelling (e.g. east coast boxes).

Sequential fisheries with intensive fishing pressure and episodic recruitment pulses once per decade (code: O16)

This scenario produces dynamics intermediate between those from the scenario with intensive fishing and zoning (outlined above) and the scenario with episodic recruitment twice per decade (described below). The effects described below are not as extreme here as the pulses are not frequent enough to allow intensive fishing to be effective for as long a period and effort reduction begins (for most groups) within about five years of when it would commence without episodic recruitment. As a result overall some bycatch groups are marginally more heavily impacted in this scenario. The target groups receiving the pulses are less impacted (i.e. have higher biomass levels) and there is less web restructuring in the various assemblages directly impacted by the pulses and fishing pressure targeting them.

Sequential fisheries with intensive fishing pressure and episodic recruitment pulses twice per decade (code: O17)

The system predicted by this scenario is of the same general form as the scenario with intensive fishing pressure and zoning. However, all the major bycatch groups and some target groups end up at lower biomass levels. The bycatch groups are more heavily impacted as intensive fishing continues beyond the point it tails off due to low CPUE in the other scenario. While some target groups with episodic recruitment are maintained indefinitely by the episodic recruitment others ultimately end up at lower biomass levels, as they become hyperdepleted so to speak – as fishing continues longer than it would if the periodic strong recruitment pulses did not occur (carried primarily by the strong age class) the age classes that are at average levels are more heavily impacted and when stock sizes become too low to allow for sizeable recruit pulses the overall biomass levels have collapsed to very low levels (as little as <1% of virgin biomasses). These effects of fishing allow for web restructuring due to predation release and shifts in dominance, as in the scenario with intensive fishing and zoning, but these shifts are often more pronounced in this recruitment scenario and so there are also stronger differences between assemblages that receive different levels of fishing pressure. The royal blue, light green, light blue and pink areas in Figure 4.8 are heavily depleted with large scale restructuring of their trophic webs (and all become more similar in form). The aqua and yellow areas become subdivided into those areas heavily impacted by fishing (those cells adjoining the other heavily impacted assemblages) and those areas which are only moderately impacted (as they are either far from port or do not receive a large proportion of the episodic pulse and so are not as intensively fished).

Sequential fisheries with intensive fishing pressure and adaptive management based on changing length of fishing season (code: O18)

This scenario is not substantially different from that with intensive fishing pressure and spatial management via zoning (outlined above), except that while the most heavily targeted groups are substantially depleted (dropping to <5% of virgin biomass in some cases) they do not reach the critically low levels seen in the scenario with zoning (ending up as much as 10x more abundant).

Sequential fisheries with intensive fishing pressure and site attachment for all but those groups/species known to be highly migratory (e.g. tunas) (code: O19)

Overall all biomass levels are higher in this scenario than in the scenario with intensive fishing pressure and zoning, except for those groups which benefit the most from predation release as target species are depleted (e.g. small planktivorous fish, cephalopods and gelatinous zooplankton). In contrast, localised effects of fishing are much higher in this scenario, which means differences between assemblages are more pronounced. Interestingly, the interaction of local effects and generally lower levels of predation release in this scenario leads to higher levels of migratory mesopelagics than seen in the other scenarios. Essentially, the biomass of shallow planktivorous fish is more highly constrained than in the scenario with intensive fishing and the use of zoning for management (due to predation in areas not particularly heavily impacted by fishing), but intensive fishing in specific locations still releases the mesopelagics from predation and the migratory forms have the freedom of movement to capitalise on the “surplus” food left by the smaller stock of small pelagics.

Sequential fisheries with intensive fishing pressure and changing coverage (access) of demersal trawl fleets – to represent improvement in targeting due to GPS and acoustic mappers and also switches to gear that can access rougher ground (code: O20)

The end result of this scenario is almost identical to that for the scenario with zoning and intensive fishing pressure (described above), except that many stocks end up at marginally lower biomass levels and the period of depletion and then the collapse of the most effected fisheries happens more quickly as spatial refuges (and required habitats) are removed and stock reduction is accelerated.

Sequential fisheries with intensive fishing pressure and management via Total Allowable Catches (TACs) (code: O21)

Largely similar to the scenario with intensive fishing pressure and zoning, except that the many target groups are more variable through time and ultimately end up at slightly lower biomass levels (at most 20% lower).

Intensive levels of all anthropogenic pressures – fisheries, fivefold increase in nutrient loads, habitat degradation or clearing around human settlements and 10% (about 2 degree) increase in average temperature (code: O22)

Increased productivity in the photic zone (upper water column) close to settlements locally lead to stronger pelagic webs in this scenario compared to the other scenarios with fishing, while epifauna and associated demersal groups decline. The scavenger subweb is also much stronger in this scenario than in many of the other scenarios, though the effects are again fairly localised. As the effects of the other human impacts (and their interaction with fishing pressure) tend to be localised the differences between assemblages also tends to be stronger.

4.4 Sampling Model

A sampling model processes the output of the operating model and generates data with realistic levels of measurement uncertainty (bias and variance), given specifications for the precision and spatiotemporal coverage of data collection for each data type and source. For example, fisheries data are aggregated spatially and temporally (e.g. total catch for an entire area per quarter), whereas field data are only available infrequently (e.g. annual surveys, or diet data is collected once every ten years) from “snap shots” taken at certain “sampling locations” (marked by dots in Figure 4.3). The distribution of these sampling sites mimics stratified sampling schemes used in real ecosystems (Newell and Harris 1997).

An attempt was made to capture the features of data handling and management practices. To this end, the generated data was pooled across “sampling locations” into user-defined zones for calculation and reporting of indices (marked by bold lines in Figure 4.3). Such zones are commonly used in fisheries management both within Australia and worldwide. In this case, the model zones do not exactly match management zones used in the real systems, but are based on the real zones, as well as bioregions defined by the scientific community (IMCRA Technical Group 1998).

To calculate those indicators that are network or model dependent a network model is required. ECOPATH is used here, due to its popularity and the ready availability of its source code (which is incorporated into the sampling model). At each time step of the sampling model an ECOPATH model is created using the sampled data together with simple decision rules for achieving a balanced model (regarding confidence intervals for input parameters and associated orders of parameter adjustment). The most recent form of each type of data is used to create each ECOPATH model. Given the different sampling frequencies this means that the diet data can be many years older then the biomass, catch, discards and mortality estimates used. The standard parameters required by ECOPATH (detritus inputs, and assimilation efficiency and flows to detritus for each group or fishery) were fixed across all runs (values given in Appendix E). Due to sampling variability in data inputs for bother biomass and data model mis-specification can and does occur in the ECOPATH model (e.g. animals with very low biomasses may be omitted from the model altogether or links may be missing from the diet matrix).

A key element of simulation testing indicators is that the true state of the system is known and so the performance of the indicators can be understood by comparing the indicator values against the true values for the main attributes (system characteristics of interest). The attribute values are also recorded by the sampling model directly from the operating model output.

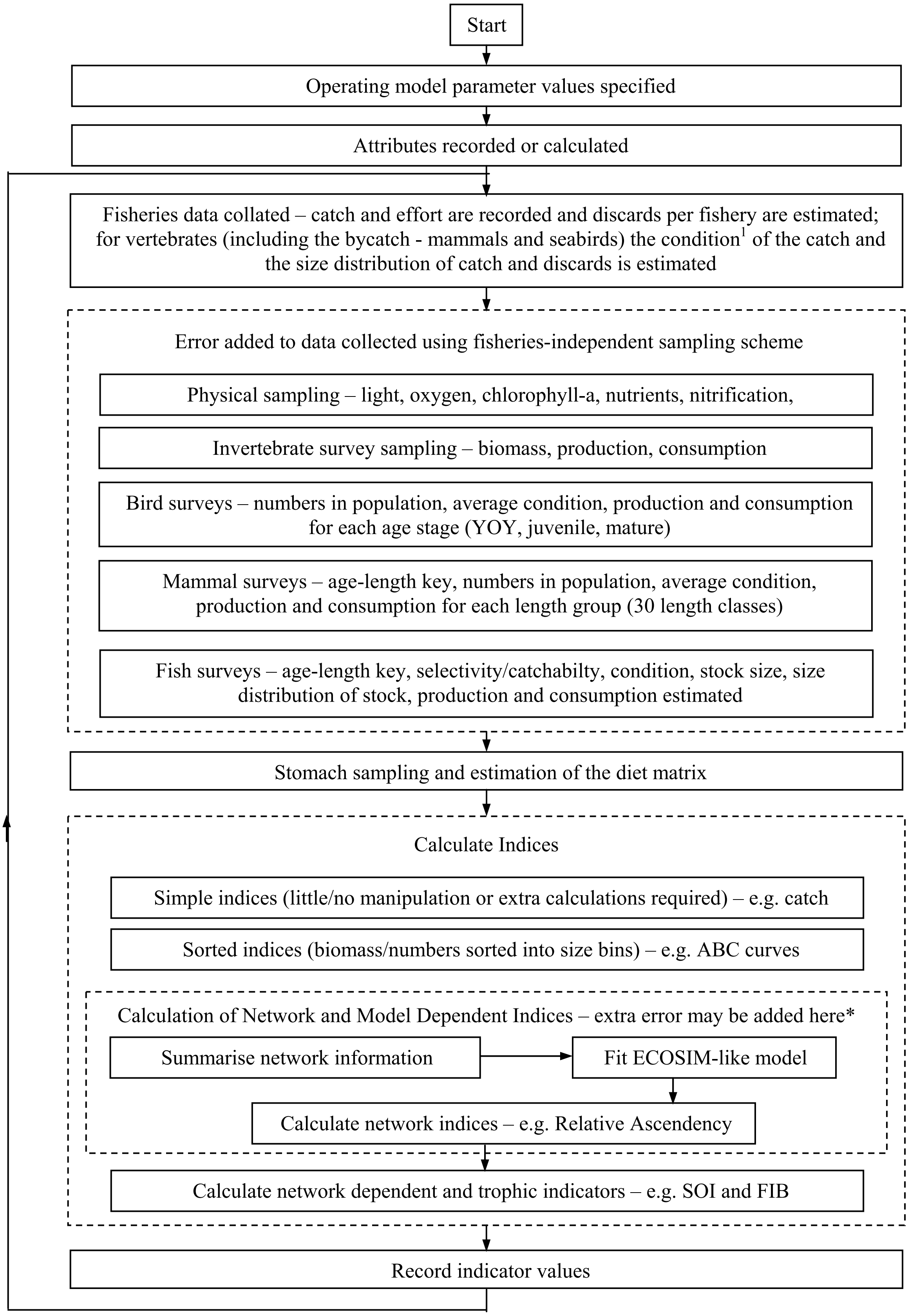
The sampling model is run repeatedly for each fishing scenario (30 times for the coast-to-open ocean system and 100 times for the bay system) to reflect uncertainties in the observation and monitoring process. The coast-to-open ocean system was not repeated as often as the bay system due to the longer iterative fitting time required for the intermediate mass balance (ECOPATH) model for this larger, more complicated system. As the operating model is deterministic the attributes only need to be calculated once (on the first run of the sampling model for each scenario). A flow diagram outlining all the steps taken in the sampling model is given in Figure 4.12. The frequency of sampling of different data types is listed in Table 4.7 and the equations and parameter setting used are given in Appendices E and F.

Table 4.7: Sampling frequency for the different data types

Data Type	Sampling frequency
fisheries (catch, effort, discards)	(cumulative[1]) per month or quarter[2]
size composition of catch and discards	snapshot every 6 months
diet information (stomach contents)	either annually or once per decade[2]
physical and chemical system properties	snapshot every 6 months
invertebrate surveys	annually or every 6 months[2]
bird surveys	annually or every 6 months[2]
mammal surveys	annually or every 6 months[2]
fish surveys	annually or every 6 months[2]

1. Summed over the month or quarter rather than an instantaneous (snapshot) value
2. Alternative parameterisations are used to allow for evaluation of the effects of different sampling designs.

Figure 4.12: Diagram of sampling model steps. Sections in dashed boxes represent submodels with several steps or calculations. Attributes are calculated only once, but the sampling process is repeated (30-100) times.



4.5 Specific Attributes and Indicators

4.5.1 Attributes

A number of attributes, which largely reflect ecosystem- and community-level properties that span the major ecological structures and processes of public and legislative concern, were identified during a workshop that took place during the early stages of this study (Punt *et al.* 2001 attached as Appendix B). Not all of these attributes could be captured within the Atlantis framework. Some of them were dealt with during the GIS analysis, but others will have to be left to future research.

The attributes that are used in the simulation testing are listed in Table 4.8. The attributes biomass, size-structure, spatial-structure and the number of groups to represent 80% of the biomass capture the gross form of an ecosystem and its constituent communities. They also focus on the key or sensitive groups that can be the focus of public and scientific attention. The community- and ecosystem-level attributes diversity, foodweb structure, and trophic level details complete the basic “natural history picture” of the ecosystem. Between them, the biomass, structural and foodweb attributes reflect the ecosystem characteristics that are most visible and readily identified by interested sectors of society (“what’s there?”, “how much is there?” and “what eats what?”). The other attributes listed in Table 4.8 deal with the flows and functioning of the ecosystem. The ecosystem-level attribute nutrient cycling – which reflects the internal state of the ecosystem, its flows and its ability to sustain its current state – is already the focus of water quality monitoring. This focus is due to the widely recognised detrimental effects associated with extreme eutrophication (Rapport 1992, Harris *et al.* 1996). The other community- and ecosystem-level properties “throughput”, “consumption”, and “production” also reflect the system’s ability to support its current state (and any exploitation) in the long-term. While the final community- and ecosystem- level attribute, respiration (particularly relative to production), captures the overall state or maturity of an ecosystem (Christensen 1992). This is important ecologically, but also socially as people tend to find mature systems more aesthetically pleasing (as they usually contain more internalised flows and more specialised and diverse biota).

Table 4.8: A list of the system attributes of potential interest, their attribute-type (Community, Ecosystem, or Population) and the variables used to represent them in the modelling framework. An asterisk denotes that the attribute could be developed for a single species / functional group or for a group of species / functional groups.

Attributes	Attribute type	Model variables used as attributes
Biomass*	C, E, P	Relative biomass of benthic deposit feeders, filter feeders, demersal fish, detritus, large zooplankton, marine mammals, piscivores, planktivores, seagrass and sediment bacteria; Total biomass; Total biomass minus detritus
Consumption	C, E	Total consumption; Consumption by benthic deposit feeders, filter feeders, demersal fish, detritus, large zooplankton, marine mammals, piscivores, planktivores, seagrass and sediment bacteria
Diversity (species, group, flow)	C, E	Presence-absence of each group (and juvenile-adult stage of the higher trophic level groups)[1]
Food web	E	Average trophic level (with individual trophic levels calculated dynamically or fixed at their starting values); System Omnivory Index (SOI)
Number of groups to represent 80% of the biomass	C, E	Relative biomass of each group (and juvenile-adult stages of the higher trophic level groups)
Nutrient cycling	E	Denitrification; Dissolved Inorganic Nitrogen; Ratio of labile:refractory detritus
Production	C, E, P	Chlorophyll-a; Dissolved Inorganic Nitrogen, Total primary production, Total production; Production by benthic deposit feeders, filter feeders, demersal fish, detritus, large zooplankton, marine mammals, piscivores, planktivores, seagrass and sediment bacteria
Position in MDS space	C, E	NA[2]
Respiration	C, E	Respiration; Total respiration; Total primary production / respiration
Size structure*	C, P	Relative biomass and abundance[3] of each group (and juvenile-adult stages of the higher trophic level groups); age structure of vertebrate groups

Table 4.8: Continued.

Attributes	Attribute type	Model variables used as attributes
Spatial structure (of biomass)*	C, P	Relative biomass of benthic deposit feeders, filter feeders, demersal fish, detritus, large zooplankton, marine mammals, piscivores, planktivores, seagrass and sediment bacteria
Trophic level	E	Average trophic level (with individual trophic levels calculated dynamically or fixed at their starting values)
Throughput	E	Total throughput

1. Diversity is only modelled in a crude way. Alternative model frameworks and methods need to be used to consider this attribute fully.

2. Preliminary results showed that this attribute was hard to consistently interpret, and that it was uninformative and difficult to calculate efficiently. As a result it was dropped from the remainder of the analyses.

3. For the invertebrates (modelled only as biomass pools) the abundance is calculated by dividing biomass estimate by fixed “average” size of that kind of invertebrate.

4.5.2 Indicators

There are two major kinds of indicators – those with diagnostic value, and those that simply provide a “warning” about a potential threat to the system (the “canaries in the coal mine”). Given that management needs to determine how to respond to a threat, not just recognise that a threat exists, the majority of the indicators considered here are of the diagnostic type. Those indicators that are primarily an “early warning” are noted individually below.

Close to 100 different types of indicators were identified in the literature review completed as part of objective one of this project (Fulton *et al*. 2004a). It is not possible to test all of these using the simulation approach (due to data requirements and model restrictions). To reduce the set of indicators to a manageable set for testing, criteria used in Fulton *et al*. (2004a) to judge the potential utility of indicators (based on their relative clarity, sensitivity, exclusiveness or measurability) were used to determine the most promising types of indicators to trial. The shortened list was amended slightly to include: (i) a few historically widely-advocated indicators (e.g. diversity indices) that did not rate well, but should be tested given their widespread popularity; and (ii) a small number of indicators that have performed well in recent field work (e.g. maximum length of the catch, Miller and Cury 2003) or were developed since the original list was compiled (e.g. trophic spectra, Chassot and Gascuel 2003). The types of indicators trialled are briefly described below. More complete descriptions of them can be found in Fulton *et al.* (2004a). An explanation regarding robustness ratings and a brief description of those indicators dropped from the main analysis after a preliminary screening are given before the descriptions for the many directly calculated and model-dependent indicators trialled are provided.

4.5.2.1 Robustness Rating and Indicator Selection

Many of the descriptions of the different types of indicators given below refer to a robustness rating. This is a subjective rating (low, medium, high), given to each indicator reviewed by Fulton *et al*. (2004a), that was based on a literature review (and where necessary an expert judgement) on the conceptual foundation, feasibility of implementation, ease of accurate collection, sensitivity, exclusiveness, comprehensiveness and clarity of potential indicators (Jackson *et al*. 2000, ICES 2001, Rochet and Trenkel 2003, Fulton *et al*. 2004a, Niemi *et al.* 2004, Rice and Rochet 2004, Rochet and Rice 2004). While some preliminary indicator assessment frameworks were trialed during the early stages of this project, in collaboration with members of the SCOR working group 9 (WG9) on the quantitative indicators of the ecological effects of fishing, the ratings ultimately subjective, as no tested quantitative selection criteria existed when the review took place. The final form of the SCOR WG9 indicator evaluation framework does appear to show substantial promise, but they have come too late for inclusion in this study.

In general, a poor rating based on this subjective system did mean an indicator was dropped from further consideration by simulation testing. However, if an indicator has been strongly advocated in the past, is already widely used, or if field work had indicated that it may actually have potential, then it was included (if possible given the form of the model) regardless of rating.

4.5.2.2 Indicators dropped during the analyses

Not all of the types of indicator originally selected for trailing on the basis of the literature review were kept for the entire analysis. Given the time involved in calculating and analysing indicators a preliminary screening of the different types of indicators was done to check for their true suitability. Two types of indicators were found to be unsuitable for further analysis (reasons detailed below) and they were dropped from the subsequent full analysis.

4.5.2.2.1 Multi-dimensional Scaling

Multi-dimensional Scaling (MDS) uses similarity (or dissimilarity) matrices to provide a visual representation of the pattern of proximities (i.e., similarities or distances) among a set of observations (Johnson and Wichern 1992). A variety of data types can be combined in this kind of analysis (e.g. temperature, Chlorophyll-a levels, abundance, biomass, % cover), which has seen various MDS plots (e.g. species composition plots) and MDS-based metrics proposed as good community- or ecosystem-level indicators. MDS is also one of the most robust ordination methods for dealing with non-linear monotonic relationships, problematic underlying distributions of abundance, and high sampling variance. In addition, it has been identified in comparative empirical studies as giving the best overall performance (Warwick and Clarke 1991, Warwick and Clarke 1993). As a numerical method that can take such a wide array of data types, it can fairly easily be incorporated in to the simulation testing process. Unfortunately, interpretation of its results is qualitative, potentially misleading, and ICES (2001) argues that it should not be used to derive hypotheses about causality. This type of indicator received a low to medium rating in Fulton *et al.* (2004a), but was initially included in the list of indicators because of its popularity in the field of ecology. Preliminary simulations, performed before the main analysis was commenced, showed that this method had potentially prohibitive computational requirements, could not be consistently interpreted, could be hampered by data collection issues, and was of generally limited utility even in the simulation study setting. As a result, it was dropped from the main analysis.

4.5.2.2.2 Rate of damage

The rate of damage of benthic groups has been identified as potential community- and population-level indicators of the intensity of benthic habitat disturbance caused by mobile fishing gear (Bergman and Hup 1992; Jennings and Kaiser 1999, Prena *et al.*1999, Eleftheriou 2000, Collie *et al.* 2000, Ramsay *et al.* 2001, Stobutzki *et al.* 2001). The successful use of this type of indicator in the role of ecological indicator will require careful selection of indicator species and may require data on natural rates of damage from reference areas. In the sampling model this type of indicator was represented by the rate of damage index (D) is calculated using:

$$D = \frac{\hat{B}_{i,f}}{\hat{B}_i} \tag{4.2}$$

where $\hat{B}_{i,f}$ is the estimated proportion of the biomass of invertebrates in model group i that have interacted with fishing gear but have not been landed or taken on board a fishing vessel and subsequently discarded.

While this type of index can be tested using simulated data, ultimately it was not included in the main body of this study as the preliminary analyses showed it to be of limited utility in this case - as it is sensitive to the period since fishing began in an area (and so is only useful for short periods of time). This may be because while the operating model does include formulations that capture the emergent properties of sub-grid scale benthic processes it does employ functional groups and pools of invertebrate biomass not species and the characteristics of individuals, and so does not produce data with enough signal or resolution to fairly and rigorously test this type of indicator.

4.5.2.3 Directly calculated indicators

Many of the types of indicators considered can be calculated directly from collected data without requiring an intermediate mass balance model to be fitted. Brief descriptions of these types of indicators and the formulations used are given here. Descriptions for model-dependent indicators are given in section 4.5.2.4.

4.5.2.3.1 Biomass or stock size

Various community- and population-level biomass indices are used in fisheries and ecology to judge the health of stocks, communities and ecosystems of interest. They are intuitively simple to understand, but they can also be subject to high levels of natural variation and (depending on the level of aggregation) may be insensitive to human impacts. They were rated as having a potentially high robustness by Fulton *et al*. (2004a), due to their potentially large information content and their ability to track the state of key parts of the system when based on representative assemblages and reliable data. Biomass indices are also readily testable using the simulation approach and the form of operating model employed here. One important practical constraint with biomass indices can be the data and information required to estimate them. If direct measures are not available (and they rarely are in marine systems), then more or less complicated models may be required to obtain estimates of abundance (e.g. stock assessment models used in fisheries). As there is a considerable literature on the problems associated with abundance or biomass estimates for marine populations this problem may be overcome or at least taken into consideration. The formulations used in the sampling model to give biomass estimates are detailed in Appendix F, which deals with data generation by the sampling model. The relative biomass values (relative to un-exploited levels in this case) given by the sampling model are used directly as biomass and stock-size indicators here.

4.5.2.3.2 Total mortality

Mortality indices have a long history of use in fisheries; are reasonably well understood; and potential reference points (e.g. $Z_{0.1}$, Z^* the upper sustainable limit and Z_{mbp} the total mortality leading to maximum biological production) have already been identified (Die and Caddy 1997, Pope *et al.* 2000). Community- and population-level total mortality rates have a high robustness rating (Fulton *et al*. 2004a), despite it being impacted by factors other than fishing. However as with biomass estimates, if mortality estimates can not be directly measured (which is often the case) estimates are usually the product of some simple (e.g. catch curves) or more complicated (e.g. stock assessment) form of modeling or analysis. In the sampling model total mortality (Z) is calculated using for group i using:

$$Z_i = \frac{\hat{P}_i}{\hat{B}_i} \qquad (4.3)$$

(from Christensen *et* al. 2000) where $\hat{P}_i$ is the estimated total production by group *i* and $\hat{B}_i$ is the estimated biomass of group *i*. This was used for all groups as there is no dynamic length information available for the invertebrate groups. The values for the vertebrates were verified in the pre-screening using the length-based formulation given in Die and Caddy (1997):

$$Z_i = \frac{\left(l_{i,\infty} - \bar{l}_i\right) \cdot K_i}{\bar{l}_i - l_{i,c}} \qquad (4.5)$$

where $l_{i,\infty}$ and K_i are the von Bertalanffy growth parameters for group *i*, $\bar{l}_i$ is the average length of the catch and $l_{i,c}$ is the length of first capture. Given the model resolution, the difference between values using method (4.5) and (4.3) were less than sampling variation and so the decision to use (4.3) for all groups seems justified here.

4.5.2.3.3 Potential biological removals (PBR)

These population-level indicators were original defined for use as reference points for marine mammal populations taken as bycatch in fisheries in the United States of America (Wade 1998). It was considered as a potential monitoring indicator here as it may provide information on the relative health of the stocks of some the most vulnerable groups in an ecosystem. The potential biological removals for group *i* ($\hat{N}_{i,PBR}$) as given in Wade (1998) as:

$$\hat{N}_{i,PBR} = \hat{N}_{i,\min} \cdot \frac{1}{2} \cdot r_{\max} \cdot \gamma$$

where $\hat{N}_{i,\min}$ is the minimum population estimate for group *i*; $r_{i,\max}$ is the maximum theoretical (or estimated) net productivity rate for group *i* at a small population size; and γ is a recovery factor between 0.1 and 1 (set to 0.5 here).

4.5.2.3.4 Size-structure

Population-level size data are one of the most commonly and easily collected kinds of fisheries data (Hilborn and Walters 1992) – it is a key part of many fisheries monitoring regimes, and is frequently used in fisheries assessments at the population level. Size data can suffer from a variety of potential sampling problems though. Apart from the usual issues of spatial and temporal coverage, there are problems arising from the (often uncertain) size selectivity of the gear used to sample the population (often commercial fishing gear), and variations in fishing practices, population behaviour, and environmental conditions. All of these issues are explicitly included in the sampling model routines used to generate the survey and monitoring data (detailed in Appendix F, which deals with the generation of size-data by the sampling model) for the vertebrate groups (the form of the operating model means it makes little sense to do it for the invertebrate groups). The values given by the sampling model are then used to give values for average size for vertebrate group *i* ($\bar{l}_i$):

$$\bar{l}_i = \frac{\sum_k \sum_j \hat{N}_{i,j,k} \cdot l_{i,j,k}}{\hat{N}_i} \qquad (4.7)$$

where $l_{i,j,k}$ is the size of animals of group i in size class j and age class k; $\hat{N}_i$ is the estimated total abundance; and $\hat{N}_{i,j,k}$ is the estimated abundance of animals of group i in age class k and size class j. The population-level indicator average size-at-maturity is calculated in the same way with k = age at maturity only, and $\hat{N}_i$ replaced by the estimate of abundance of animals at the age-of-maturity. Condition-at-maturity is calculated in a similar manner, but with l_{ijk} replaced with the weight of animals of group i in size class j at the age of maturity.

4.5.2.3.5 Age-structure

The development of reliable aging techniques for many vertebrate species has seen an increase in age-based indices and assessment methods (Hilborn and Walters 1992). Population-level age structure data are widely collected and used in many fisheries, as well as monitoring programs for some marine birds and mammals – though it is not pertinent in all fisheries (e.g. most invertebrate fisheries). Means and tails of age distribution data can be quite informative about effects of fishing, as it is usually the larger older animals that are targeted by most fishing gears and methods. Mortality estimates are also often derived from age data (see equation 4.5 above). Expanding the species monitored using age-based indices to species (or groups) representative of broader parts of the community may be a good way of using well understood existing methods to address ecosystem questions. In addition to the kinds of sampling issues that arise for size data, the costs of ageing are generally much higher than collecting length data, so keeping the age-based indices as simple as possible is highly recommended (Pauly and Morgan 1987). With these warnings regarding potential problems with complicated age-based indicators in mind, only the simple age-based indicators are tested – age-at-first capture; average-age-of-catch ($\bar{a}_{i,c}$):

$$\bar{a}_{i,c} = \frac{\sum_k \hat{N}_{i,k,c} \cdot a_{i,k,c}}{\hat{N}_{i,c}} \qquad (4.8)$$

proportion of the stock that are juveniles ($p_{\mathrm{i,juv}}$):

$$p_{i,juv} = \frac{\sum_{k=0}^{k=m} \hat{N}_{i,k}}{\hat{N}_i} \qquad (4.9)$$

and the proportion of the catch that are juvenile ($p_{\mathrm{i,juv,c}}$) – calculated in the same way as $p_{\mathrm{i,juv}}$ but using only numbers from the catch not stock abundance estimates. The term $\hat{N}_{i,k,c}$ is the number of animals in the catch of group i in age class k; m is the age of maturity; and $a_{i,k,c}$ is the age of individuals from group i, age class k in the catch. These indicators were only calculated for the vertebrate groups, as there is no dynamic age structure represented for the invertebrate groups. Details of how the age data are generated are given in Appendix F.

4.5.2.3.6 *Catch-rate*

Total catch is a widely reported fisheries statistic. Fishery stock assessment scientists have traditionally made extensive use of total catch and catch-per-unit-effort (CPUE) as indicators of stock health, particularly where fishery independent survey data are unavailable. Population-level catch and CPUE data are readily available for most commercially targeted fish, though community-level catch or CPUE may not be obtained so easily. There are also well-known problems with the assumption that CPUE is linearly proportional to the resource size (Hilborn and Walters, 1992; Jennings *et al.* 2001a). These problems and others to do with effort estimation and standardization of catch rate data are reflected in the sampling model formulations (details in Appendix F), CPUE is then simply the observed catch divided by the effort estimate.

4.5.2.3.7 *Discard rate*

Community- and population-level discard rates (biomass discarded / biomass caught) can be used to determine the true value of CPUE (i.e. what is landed versus what is actually caught in total), but it can also give insight into the pressures and impacts on the entire community exposed to fishing. Discard data can be very important in attempts to assess the degree of vulnerability and the likelihood of recovery of a vulnerable species, and to identify gear types that may have more extensive ecological effects (Jennings *et al.* 2001a). It only received a low to medium robustness rating (Fulton *et al.* 2004a), however, due to its dependence on fisheries data and the resulting potential for confounding by changes in gear, targeting and effort. More importantly, a decline in the discard rate of a species does not necessarily equate to a reduction in the effects of fishing (the population itself may simply be in decline). This can be a problem in using such data to monitor protected or threatened species, where by-catch or discard rates are often used as a measure of fishing pressure (decline in rate is good), whereas they may more accurately be measuring fishing impact (decline in rate is bad). The historical use discard rates in fisheries monitoring and assessment does justify their inclusion in the evaluations performed here – especially as the potential effects of confounding need clarification if indices based on discard rate are to be applied more widely for “ecosystem assessment”.

4.5.2.3.8 *Size-spectra*

Empirical studies have shown that the slope and intercept of biomass size spectra (biomass per logarithmic size class) of marine assemblages are robust measures for detecting trends in the overall impacts of fishing on a fished community (Rijnsdorp *et al.* 1996, Bianchi *et al.* 2000, Rice 2000, Shin *et al. submitted*); although, questions have been raised about their sensitivity, exclusiveness, and whether or not biomass size spectra are actually non-linear (Duplisea and Kerr 1995, Duplisea *et al.* 1997, Fulton *et al*. 2004a, Shin *et al. submitted*). Community- level size spectra are included in the list of indicators to be trialled due to their widely-perceived potential as a useful indicator of the impacts of fishing at the community level. To evaluate the implications of the form of size-spectra for their utility as ecological indicators, two alternative forms of the spectra are considered. The first including all groups and the second only including vertebrates. To include the invertebrates in the size spectra fixed average sizes are assigned to each invertebrate group. While not ideal, the spectra produced using this assumption do match real spectra and so it seems that even using this assumption the simulation testing does allow for a fair assessment of the potential utility as of size-spectra as ecological indicators.

4.5.2.3.9 k-dominance and ABC curves

The steepness and elevation of community-level k-dominance curves (a plot of the cumulative ranked abundance against a log species rank) can be an informative measure of diversity and system perturbation. These statistics have performed well in field trials (Jennings *et al.* 2001b), but can also respond to pressures other than fishing and require comprehensive data sets collected using consistent methods. They are included in the evaluation because of their past performance, relatively simple form, current reputation, and high robustness rating (Fulton *et al.* 2004a). These curves are assessed directly and evaluating the summary statistic κ_A using the method described in section 4.5. κ_N is calculated using:

$$\kappa N = \frac{\left((S+1) - 2\overline{\rho}_N\right)}{(S-1)} \tag{4.10}$$

where $\overline{\rho}_N$ is the average species rank for abundance; and *S* is the number species included in the curve.

ABC curves are combined cumulative k-dominance plots for species biomass and for numbers. They are included in this study because they have no need for reference to a temporal or spatial series of control samples (Warwick 1986). Given that existing data may be poor in some fished systems, and that unimpacted reference areas may no longer exist for most marine ecosystems, evaluation of this potentially highly robust indicator (Fulton *et al.* 2004a) was considered important. The utility of these curves as indicators was assessed directly and also via the summary statistic suggested by Clark (1990):

$$W = 2 \cdot \frac{\left(\overline{\rho}_N - \overline{\rho}_B\right)}{(S-1)} \tag{4.11}$$

where $\overline{\rho}_B$ is the average species rank for biomass.

4.5.2.3.10 Coefficient of variation for total biomass

In stable systems, energy limitations means that low abundance of one species should be compensated for by a higher abundance of another species, so that the sum of their biomasses should remain fairly steady. In contrast, in a fished system the removal of biomass by fisheries may lead the system to a point where the variation of species abundance is less dependent on the abundance of other species, as energy limitations are lifted and interactions among species are not as strong, with the result that the variation in total biomass can increase. The coefficient of variation (CV_t) of total biomass may be a good potential community- or ecosystem-level indicator of this kind of change (particularly in comparison with the *CV* for individual species or groups) (Blanchard and Boucher 2001) – though to be truly robust, it requires fisheries independent data. CV_t is calculated as:

$$CV_t = \sum_i CV_i = \sum_i \frac{100 \cdot \sigma_i}{\overline{x}_i} \tag{4.12}$$

where σ_i is the standard deviation and $\overline{x}_i$ is the mean of the measured variable (in this study, biomass) for species *i*. The coefficient of variation indicators were rated as having a medium to high robustness by Fulton *et al.* (2004a) and are included here for their simplicity and their potential to summarise overall system state.

4.5.2.3.11 Average trophic level

Average trophic level can be a very useful community- or ecosystem-level summary statistic if it identifies a substantial shift in community structure (Jennings *et al.* 2001b). The considerable data requirements and the need for fisheries independent data are potential concerns with this indicator. Many forms of this indicator are included in the analysis. All have the form average trophic level in year *t*:

$$\overline{T}_t = \frac{\sum_i \hat{T}_{i,t} \cdot \hat{B}_{i,t}}{\sum_i \hat{B}_{i,t}} \tag{4.13}$$

where $\hat{T}_{i,t}$ is the estimated trophic level of group *i* in year *t*, and $\hat{B}_{i,t}$ is the estimated biomass. The biomass can refer to the entire stock or only the catch. The groups included can span all groups sampled or only the vertebrate groups (with juvenile and adults treated separately, to take ontogenetic shifts in diet in to account). Each of these variants are calculated using two different forms of the trophic level of group *i* – either calculated dynamically through time, taking into consideration changes in diet in response to changes in the ecological structure of the ecosystem; or held constant (or fixed) at the first estimated value. These two forms allow for consideration of the importance of understanding the current trophic structure of the system (diet data can be expensive to collect and so is not going to be collected as frequently as the biomass data). The lag in data collection means that even with the dynamically calculated trophic levels the trophic level estimate will occasionally be out of date.

4.5.2.3.12 Fishery In Balance (FIB)

This ecosystem-level indicator is calculated using:

$$FIB = \log\left(\frac{\hat{B}_{t,c}}{e_T^{\overline{T}_{t,c}}}\right) - \log\left(\frac{\hat{B}_{0,c}}{e_T^{\overline{T}_{0,c}}}\right) \tag{4.14}$$

where $\hat{B}_{t,c}$ is the estimated catch in year *t*, e_T is the transfer efficiency (set to 0.1 here), and $\overline{T}_{t,c}$ is the average trophic level of the catch in year *t* (which may determined using trophic levels for the individual groups that are be calculated dynamically or fixed at the first estimated value); and year 0 is the reference (baseline) year. This measure should decline only when catches do not increase as much as expected given a shift in targeting to lower trophic groups that have higher rates of production (Pauly *et al*. 2000). There has been a suggestion, however, that it may be confounded by other factors (Caddy and Garibaldi 2000). Given its data requirements, high profile, and the potential misunderstandings as to its use, this indicator is a good candidate for verification with simulation testing. Experience gained with the indicator throughout the study (and discussions with D. Pauly, UBC Canada) suggested the slope of a plot of FIB through time (or at least changes in that slope) may be the more important measure to track, as significant changes may indicate a substantial restructuring of the supporting web (D. Pauly, UBC Canada, *pers. comm.*).

4.5.2.3.13 Diversity index

Diversity indices are commuity- and ecosystem-level indices that are dogged by many potential problems, as they are:

- dependent on richness and evenness means, which means they can have limited application and can be potentially misleading (Hill 1973, Rice and Gislason 1996, Rice 2000)
- susceptible to sampling problems (Soetaert and Heip 1990)
- can be hard to estimate
- have little biological meaning (Rochet and Trenkel 2003)
- may miss some important biological phenomena (e.g. regime shifts or guild replacements)
- be unduly sensitive to routine ecosystem events (such as recruitment pulses) (J. Rice, DFO Canada, *pers. com.*)

As a result of these issues diversity indices received a low to medium ratings in the literature review by Fulton *et al.* (2004a). Nevertheless they are included in the list of indicators tested in this study because of the focus they have received in the ecological literature and because of their apparent wider public acceptance (where catalogues of species have often been presented as simple indicators that society in general can understand). Unfortunately, the use of functional groups in the operating models prevents an intensive consideration of the more sophisticated of the diversity indices (e.g. Réyni index). The simple indices considered in this analysis are a count of the number of communities or groups present in a zone, and a similar count except the juvenile and adult forms of each vertebrate group are counted separately (to account for the large shifts in role the different age stages can have). While diversity indices are also considered in the GIS analysis, the available data has its limitations, which meant that it is not possible to consider the more sophisticated diversity indices in that setting either.

4.5.2.3.14 Benthic habitat complexity

Certain gears (such as bottom trawls) interact with benthic habitats and reduce their complexity – resulting in changes to community structure, shifts in species composition (Auster and Langton 1999, Jennings *et al.* 2001a), and impacts on commercial fish stocks (Lindholm *et al.* 1999). It can be difficult to collect biogenic substrate information, but simple indices based on biomass surveys may be useful. The simple community-level habitat complexity index considered in this study is the ratio of epifauna to infauna. Given the changes in the complexity of biogenic habitat, benthic structure and the ecologifcal state of the ecosystem that accompany high rates of disturbance, this index shows promise as a robust measure of the state of the benthic communities. In addition, it is possible to collect reasonable estimates of biomass at this functional level from field surveys, even when resolving samples to a finer taxonomically resolution is infeasible. Consequently, identifying whether such a simple index has the same promise as other simple ratios (such as ratios of pelagic to demersal fish) could be useful.

4.5.2.3.15 Biomass (or % cover) of cover-defining species or species groups

Habitat heterogeneity (which may be reflected in the biomass or % cover of cover-defining species) is known to be associated with the size and "health" of commercial stocks in at least some fisheries (Sainsbury 1987, Sainsbury 1988, Freese *et al.* 1999, Lindholm *et al.* 1999). While it can be difficult to sample biogenic substrate, new survey methods (e.g. acoustics and underwater video) are alleviating this, at least at the level of higher taxonomic or structural groupings. As some of the most widely used fishing gears are known to cause substantial reductions in the complexity of the epibenthic community, an index summarising the (relative) size of that community may be very useful. As a result the community-level indicator epifaunal biomass is considered in this study. The data available to the GIS analysis allowed for a slightly more detailed consideration of these kinds of indicators. The CSIRO cruise data available for use in the GIS application allows for consideration of the usefulness of the make up of benthic community (with regard to size) and the density of cover.

4.5.2.3.16 Reproductive success

Reproductive success of predatory animals has been identified by many groups (e.g. PICES and CCAMLR; CCAMLR 1992, PNCERS 2000) as a potentially useful population-level indicator of the impacts of harvesting prey species. Although it may have more value as a "warning" indicator, rather than as a diagnostic indicator of population level impacts. Croxall *et al.* (1999) noted that when using indicators based on the reproductive success of predators that it is vital to obtain measures of prey abundance and availability independent of the predator being considered, and such data can be scarce or expensive to obtain. If these problems can be overcome however, this indicator has a medium to high potential robustness (Fulton *et al.* 2004a). This rating, in conjunction with its popularity (particularly among seal and bird researchers), supports its inclusion in the list of indicators. Of particular importance for the study is consideration of how heavily it is impacted by poor data or the openness of the system (the indicator will be of little use if diluted by influx from external pools).

The measures of reproductive success considered here only deal with seabirds, and marine mammals. Two indices are considered for each group and they are the proportion of newborns that survive to weaning (or fledging if seabirds) ($R_{i,w}$):

$$R_{i,w} = \frac{\hat{N}_{i,w}}{\hat{N}_{i,0}} \tag{4.15}$$

and the proportion of newborns that survive to maturity ($R_{i,m}$):

$$R_{i,m} = \frac{\hat{N}_{i,m}}{\hat{N}_{i,0}} \tag{4.17}$$

where $\hat{N}_{i,0}$ is the estimated size of the original cohort for group i, $\hat{N}_{i,w}$ is the estimate of the subsequent abundance of animals at age-of-weaning (or fledging if seabirds) for group i and $\hat{N}_{i,m}$ is the estimated abundance of animals of that cohort that finally reach maturity. These abundance estimates are not projections, but abundance estimates from surveys taken after the appropriate periods (e.g. the number of seabirds surviving t_m years after hatching, where t_m is the time to reach maturity).

4.5.2.3.17 Fish and detrital biomass ratios

Community-level biomass ratios of invertebrates to vertebrates, or planktivorous to piscivorous fish, or pelagic to demersal fish, are conceptually simple trophic indicators. These appear to have good potential for summarising community structure without the need to collect extensive biological data or understand specific trophic linkages, which saw them given a medium to high rating by Fulton *et al.* (2004a). Problems can potentially arise when these indicators are based on fishery-dependent data, where they can be influenced by changes in targeting rather than by changes to the underlying ecological communities (Caddy and Garibaldi 2000). Nonetheless, relative biomass ratios are included in the list of indicators tested here as they may measure changes to high level community or trophic structure, and they are simple with relatively straightforward data requirements. Community- and ecosystem-level detrital biomass ratios should also be simple to calculate, and given the differential roles by the various kinds of detritus can play in provisioning and nutrient cycling it may provide information on broader ecosystem properties.

4.5.2.3.18 Physical properties

An ecosystem perspective on the effects of fishing demands an understanding of the impact of non-fishing related events on the ecosystem. Physical and chemical properties such Chlorophyll-a, Dissolved Inorganic Nitrogen (DIN) and light levels at the sediment surface could be useful in this regard as it is already commonly used in monitoring the effects of eutrophication (Harris *et al.* 1996). These kind of properties, such as Chlorophyll-a, is also increasingly the subject of correlative studies with fisheries production (e.g. Parsons and Chen 1994) and may turn out to be an effective indicators of ecosystem “health”. In addition, by highlighting effects of fishing that impact upon productivity at the lower trophic levels properties, such as Chlorophyll-a, could be directly useful as indicators of the effects of fishing. Given that changes in these kind of properties are not exclusive to fishing related disturbances, by themselves these indicators may not be reliable indicators of the effects of fishing. However, in conjunction with other indicators (e.g. size-spectra), they have the potential to be powerful summary indicators that identify change and the causes underlying it. As a result, these kind of indicators were rated as having a medium robustness by Fulton *et al.* (2004a) and are included in the simulation testing. Chlorophyll-a, DIN and light levels are calculated by the operating model (see Appendix C) and then sampled and reported by the sampling model (see Appendix F).

4.5.2.3.19 Absolute or average maximum length of the catch

Fishing affects the life history traits of fished populations (Jennings and Kaiser 1998, Hall 1999, Rochet *et al.* 2000, Jennings *et al.* 2001a, ICES 2001). Moreover, as community structure changes so does the identity of the target species of the fisheries. Monitoring specific traits across species in the community may be a good way of detecting these effects of fishing. Trait-based indicators, such as maximum size, life span, diet, and distribution, are already used to monitor freshwater (Hellawell 1986, Charvet *et al.* 2000) and marine benthic (Frid *et al.* 2000) communities. Work in the North Sea (ICES 2001) cautions that some traits (though not maximum length of catch) may be relatively insensitive to increased levels of fishing if applied to areas that are already disturbed. The use of traits as indicators was rated as having medium robustness by Fulton *et al.* (2004a) due to the requirement for further development and testing, and in light of the potential existence of thresholds of effectiveness. They are included here because of increasing interest in their application world wide (D. Pauly, UBC Canada, *pers. com.*). In the simulation study three community-level catch traits are calculated, (1) average maximum length of the catch ($\bar{l}_{i,\max,c}$) calculated using fixed $l_{i,\infty}$; (2) average maximum length of the catch calculated using actual dynamic maximum size of each group captured; and (3) the average length of animals caught.

4.5.2.3.20 Rate of denitrification

The rate of denitrification is an ecosystem-level index used in monitoring water quality. It is considered in the simulation study for two reasons. The first is that damage to benthic groups, changes in community structure and functioning, and the release of large amounts of detritus by fishing (as discards or incidentally killed biota) may change the rates of biogeochemical processes in the sediments. This is particularly true for processes like denitrification that may be heavily dependent on the activity levels of benthic fauna. Second, many of the indicators considered in this study respond to changes in the ecological structure of the ecosystem regardless of the cause (whether fishing, eutrophication or general environmental change). As a result, supplementary indicators are required to help identify the ‘true’ cause of ecological change. The use of denitrification rate in water quality monitoring suggests that it may be a good candidate as one of these supplementary indicators (Eyre and Ferguson 2002). Like Chlorophyll-a, the denitrification rate values are calculated by the operating model and then sampled (and reported) by the sampling model (see Appendix F).

4.4.2.3.21 Trophic spectra

One of the most recent indicators developed is the trophic spectra – a plot of the estimated biomass (of stock or catch) per trophic level (Chassot and Gascuel 2003). Non-integer trophic levels are used (recommended bin-size is about 0.1, Colomb *et al* 2004) and in some cases (though not here) a weighted moving average is used to smooth the histogram into a continuous curve. The size and location of any peaks of such a spectra (especially when plotted based on biomass of the catch) can be very informative. It has been shown in field and simulation studies that the size and location of the peaks of trophic spectra shift as fishing impacts a community (Bozec *et al* 2004, Colomb *et al* 2004). While this indicator was developed after the literature review by Fulton *et al*. (2004a) used to pre-screen the indicators for testing the demonstrated potential of trophic spectra as community-level indicators saw them added to this analysis. They are also considered in the GIS analysis. One of the factors considered here was whether they would be equally useful if highly resolved diet data (required in the calculation of non-discrete trophic levels by network models) was not available, are they still informative if you only have “rules of thumb” regarding the trophic level of the groups (or ages) in the catch?

Note that for this indicator, as for average trophic level and FIB (described above), the trophic levels assigned to individual groups can either be calculated dynamically through time or held constant (fixed) at the first estimated value.

4.5.2.4 Model derived indicators retained through the entire analyses

The indicators described in this section are dependent on the fitting of a mass balance model (ECOPATH) as an intermediate step in their calculation.

4.5.2.4.1 Throughput

Throughput (F) is a model-derived ecosystem-level measure calculated as:

$$F = F_Q + F_E + F_X + F_D \qquad (4.18)$$

where F_Q is the flux due to total consumption; F_E is flux due to total emigration minus total immigration; F_X is flux due to total respiration; and F_D is the flux to detritus. That is, it represents the sum of all flows into and out of the groups in the ecosystem. Ulanowicz (1986) suggests it is a better indicator of system size than biomass and it has been used to compare flow networks at different times or locations (e.g. fished versus unfished systems) (Christensen *et al.* 2000). Determination of the throughput of the system is data intensive however, and it is reliant on the assumptions of the model used to calculate it. This simulation approach is ideal for testing such model and data dependent indices, as their calculation can easily be incorporated into the sampling model (see Figure 4.7), and the effects of making incorrect assumptions and of data uncertainty can be assessed directly.

4.5.2.4.2 *Production / Biomass ratio*

It has been suggested that fishing reduces ecosystem maturity, by altering its physical structure and by shifting community composition towards r-selected species. The ratio of total primary production to total biomass (P/B) is believed to be a function of the “maturity” of an ecosystem (Christensen *et al.* 2000), and it may be a useful ecosystem-level indicator of the effects of fishing. P/B can be confounded by other maturity-altering forces on the ecosystem (such as eutrophication or natural disturbance), and so may not identify the effects of fishing alone. It is rarely calculated directly, and is most commonly calculated using ecosystem models rather than directly from empirical data. Consequently, it is reliant on the assumptions used to construct the underlying model(s) and the data used to parameterise and initialise the model(s). All of these issues are readily tested using the simulation approach, and so it is included for testing here.

4.5.2.4.3 *Primary Production Required (PPR)*

Primary Production Required (PPR) is the estimated amount of primary production required to support the harvests taken by fisheries, calculated as:

$$PPR_C = \sum_r \hat{B}_C \cdot \prod_{j,i} \frac{\hat{Q}_j}{\hat{P}_j} \cdot \hat{d}'_{j,i} \tag{4.19}$$

where $\hat{B}_C$ is the estimated catch; $\hat{Q}_j$ is consumption by predator *j*; $\hat{P}_j$ is production by predator *j*; and $\hat{d}'_{j,i}$ is the estimated diet composition of predator *j* on prey *i* with all cycles removed. The PPR required to support each group is also calculated by substituting the production of the group for the $\hat{B}_C$ term.

PPR is an ecosystem-level index of the impact of fishing that provides a measure of the influence of fishing on the lowest trophic levels of the ecosystem (Hall 1999). Estimates of PPR for fisheries have been used to determine pressure on the world’s marine systems (Ryther 1969, Vitousek *et al.* 1986, Pauly and Christensen 1995, Christensen 1995, Beddington 1995). However, more work is needed to determine the true usefulness of this indicator in specific systems rather than at large regional scales. Consideration also needs to be given to how robust the indicator is to data uncertainty and to uncertainty regarding the assumptions used in the models from which estimates of PPR are derived.

4.5.2.4.4 *Ecological footprint of the catch*

Originally the concept of an ecological footprint was defined to account for all flows of energy and matter to and from an economy and converts these into the corresponding area required from nature to support them (Wackernagel and Rees 1996). In the current ecological context, the definition of the ecological footprint given by Christensen *et al.* (2000) is used. That is, the ecological footprint (EF_C) is the area in square kilometers (assuming the unit is) needed to sustain a catch of one ton of the given resource is given by:

$$EF_C = \frac{PPR_C}{(f_p + f_d) \cdot \hat{B}_C} \qquad (4.20)$$

where f_d is the total flow out of detritus; f_p is the total flow out of the primary producers, PPR_C is the primary production required by detritus (see 4.5.2.4.3).

4.5.2.4.5 *System Omnivory Index (SOI)*

The System Omnivory Index (SOI) is an ecosystem-level indicator that measures how feeding relationships are distributed across trophic levels (Christensen *et al.* 2000) using:

$$SOI = \frac{\sum_i \log\left(\frac{\hat{Q}_i}{Q_{\min}}\right) \cdot \left[\sum_j \left(\left(\hat{T}_j - \overline{T}_{prey}\right)^2 \cdot \hat{d}_{ij}\right)\right]}{\sum_i \log\left(\frac{\hat{Q}_i}{Q_{\min}}\right)} \qquad (4.21)$$

where Q_{min} is the minimum rate of estimated consumption amongst all consumers included in the ECOPATH model; $\hat{Q}_i$ is consumption by predator *i*; $\hat{d}_{j,i}$ is the estimated diet composition of predator *i* on prey *j*; and $\overline{T}_{prey}$ is the average trophic level of the prey of species *i*. This indicator was rated as having a medium to high robustness by Fulton *et al.* (2004a). While it can be informative, it requires diet composition, biomass, catch, mortality and consumption rate data of sufficient taxonomic resolution. These data are not always available and so SOI can be heavily influenced by decisions made during the construction of the models used to calculate the index. SOI is included in the list of indicators investigated here (given its apparent potential) to check for the importance of its model dependency.

4.5.2.4.6 *Dominance of detritus*

Detrital dominance is the proportion of total flows that originate from detritus. This ecosystem-level indicator is thought to be an indicator of ecosystem maturity – Odum (1969) proposed that mature ecosystems rely more heavily on detritivory than the more herbivory-dominated immature ecosystems. This index is model-dependent, but it is easily calculated using models such as ECOPATH and so is considered in the simulation testing.

4.5.2.4.7 Relative ascendency

Relative ascendency is a measure of system organisation or development, and has been shown to be correlated with maturity (Christensen 1994). This suggests that it may have potential as an ecosystem-level indicator of large-scale system change resulting from the effects of fishing. Fulton *et al.* (2004a) scored this indicator at medium robustness owing to its model dependence. Nevertheless, it is considered in this study owing to the attention it has received in the ecological literature. Relative ascendency is calculated as:

$$A = \frac{F \cdot \sum_{i,j} g_{ij}\theta_i \log\left(\frac{g_{ij}}{\sum_k g_{kj}\theta_k}\right)}{F \cdot \sum_i \theta_i \log(\theta_i)} \tag{4.22}$$

where F is throughput (see section 4.5.2.4.1 above); g_{ij} fraction of total flow from j is given by

$$g_{ij} = \frac{F_{ij}}{\sum_k F_{kj}} \tag{4.23}$$

with the probability that a unit of energy passes through i (θ_i) is given by

$$\theta_i = \frac{\sum_k F_{ik}}{\sum_{l,m} F_{lm}} \tag{4.24}$$

and F_{ij} is the flow from j to i.

4.5.2.4.8 Residence time

The ecosystem-level indicator residence time is a model-derived measure of how long an average unit of energy stays in the system and is calculated as:

$$t_{res} = \frac{\hat{B}_{tot}}{F_X + F_O + \hat{B}_{c,tot}} \tag{4.25}$$

where $\hat{B}_{tot}$ is estimated total biomass; F_X is flux due to total respiration; F_O is flux due to total exports; and $\hat{B}_{c,tot}$ is estimated biomass of total catches – so the denominator is the sum of all losses from the system. Residence time is thought to increase with system maturity (Christensen 1992). Fulton *et al.* (2004a) gave the index a low to medium robustness rating, as it is another indicator that is only useful in conjunction with other indicators. It is included in the simulation testing as it has some promise and is easily calculated from the values required by the other model-dependent indices.

4.5.2.4.9 Ecotrophic Efficiency (EE)

Ecotrophic efficiency (EE) is the fraction of production of a group that is either used in, or lost, from the system (i.e. everything but mortality due disease and starvation). This population- or group-level indicator shows the proportion of a group's mortality explained by the model used to calculate EE (in this case ECOPATH). While EE is exceptionally difficult to measure directly, trends in EE can be identified if the same type of model is applied repeatedly to the same system through time. These trends have been shown to be good indicators of large scale change in the Black Sea, which is thought to be at least partly due to the effects of fishing (G. Daskalov, CEFAS UK, *pers. com.*). EE is included in the list of indicators considered in the simulation testing due to its performance in the Black Sea and because it is easily calculated while calculating the other model-dependent indices.

4.5.2.4.10 Total values of network characteristics

Ecological network analysis has been used in ecology for close to twenty years (Ulanowicz, 1986, Kay *et al*. 1996). It is based on input-output models of energy or material flow through a trophic network (e.g. ECOPATH). An ecosystem's ecological state can be summarised using ecosystem-level network characteristics derived from these models. Network indices do suffer from the same concerns as all model-dependent indices, but they are included in this analysis as they are by-products of the calculation of other indices, and they have been considered to be useful within the fields of ecology and sociology (Johnson *et al*. 2001).

4.5.2.4.11 Trophic efficiency

Trophic efficiency is the fraction of production transferred from one trophic level to the next. Fishing has the potential to modify trophic efficiency by altering the ecological structure of the ecosystem and resulting levels of production. Trophic efficiency is an ecosystem-level indicator that has been used in the field of water quality monitoring to consider changes in production and the ecological structure of the ecosystem (Kemp *et al*. 2001). Unfortunately, it is not free of the typical concerns associated with network indices and the methods used to calculate them. Given the focus given this indicator in the water quality literature, and the ease of calculating trophic efficiency from the data values produced by the sampling model, it is included in the study.

4.6 Analysis

The ability of the indicators to correctly mimic the attributes was assessed in a number of ways, depending on the dimensionality of the indicator.

4.6.1 Attribute correlations

Correlations among the attributes are evaluated using Pearson and Spearman – i.e. parametric and non-parametric – statistics. This analysis is useful in two ways. First it sheds light on the real value of a high number of correlations for some indicators (are they correlated with attributes that are all actually correlated with each other). Secondly, and more importantly, it identifies those attributes that are not correlated with any others. This is information is critical because while it is permissible to have an indicator (or more likely a suite of indicators) that tracks only one of a correlated set of attributes (as you are effectively tracking them all), each of the attributes that are not correlated with others must be tracked individually (which will almost undoubtedly expand the size of the suite of indicators required to effectively monitor an ecosystem).

4.6.2 Univariate indicators

The effectiveness of the various one-dimensional indicators is evaluated using bi-plot, correlation and regression analyses, conducted using the R statistical package (Venables *et al.* 2002).

The correlation analysis forms the largest part of the assessment of the univariate indicators. In this analysis the ability of the indicators to correctly mimic the attributes is evaluated by plotting each indicator against each attribute and examining the extent of (Pearson and Spearman) correlation. These analyses are repeated for each scenario and at two levels of spatial aggregation (when the sample data are aggregated to the level of management zones and when they are aggregated over the entire modelled area; see Figure 4.3). This means that for each indicator, 864 correlations are performed for the bay system (2 degrees of data aggregation by 16 scenarios by 27 attributes), and 1656 are performed for the coast-to-open ocean system (2 degrees of data aggregation by 23 fishing scenarios by 36 attributes).

To aid in interpretation, the results are normalised so that an indicator that is correlated with all the attributes has a score of 1.0 and indicators that are correlated with fewer attributes have a score of less than one. Then, to try and determine if specific pressures or conditions within a system govern the performance of the indicators, the normalised scores are compared on a two-dimensional non-metric Multidimensional Scaling (MDS) plot derived from a Bray Curtis similarity matrix to identify groups of scenarios with similar indicator-attribute correlation structure. Finally, the performance of the indicators is categorised based on: (i) the absolute value of the correlation coefficients; (ii) the percentage of the attributes represented (i.e percentage of attributes that an indicator was correlated with); and (iii) consistency of the correlations across scales of data aggregation, scenario and ecosystem-type (definitions for these terms are given in Table 4.9).

4.6.3 Multivariate indicators

Evaluating multivariate indicators is much more difficult. Analysis of these indicators is done in two ways in this study. First univariate indicators are derived from these multi-variate indicators (e.g. slope of size spectra) and these are then included in the univariate analysis described above. Second the multivariate indicators are evaluated directly by simply comparing their shapes through time to check for substantial changes at points coinciding with major changes observed in the attributes.

Table 4.9: Categorisation scheme used to grade the performance of the univariate indicators based on the results of the correlation analyses.

Category	**Definition**
Consistency	
Consistent	Similar results in all circumstances
Inconsistent	Results differ between scenarios, scales or ecosystem-types
Percentage of attributes represented	
Broad	correlated ($\lvert r\rvert > 0.5$) with 40% or more of the attributes
Restricted	correlated ($\lvert r\rvert > 0.5$) with less than 40% of the attributes
Strength of correlation[1]	
Very Clear	$\lvert r\rvert > 0.9$
Strong	$0.9 \geq \lvert r\rvert > 0.7$
Recognisable	$0.7 \geq \lvert r\rvert > 0.5$
Uninformative	Non-linear or $\lvert r\rvert \leq 0.5$

1. The correlation coefficients for the indicator had to have a value in the defined range for 85% (or more) of the correlations to be categorised as having the associated strength (e.g. if $\lvert r\rvert > 0.9$ in 85% of the correlations it is classed as Very Clear).

4.7 RESULTS

The two correlation methods (Pearson and Spearman) varied in the exact number of linear correlations for every indicator in every scenario. However, the differences between the methods were minor and generally results were consistent across the two methods. Comparisons of the two methods across indicators, scenarios, system-types and degree of data aggregation, showed that the frequency with which one method identified a recognisable linear indicator-attribute relationship when the other method did not was less than 10%. In all cases where the methods disagreed, the performance was rated as recognisable only (i.e. $0.7 > \lvert r\rvert > 0.5$). In addition, there was no systematic form to these minor differences in results between the methods – i.e the difference was for a single scenario in one system type in each case. Given this match in the pattern of results of the Spearman and Pearson correlation analysis, only the results from the Pearson analysis are presented here.

For ease of understanding, the results are presented in two steps. First, the results for univariate indicators (i.e. every thing except the k-dominance curves and size and trophic spectra) are presented on a scenario-by-scenario basis, with a description of the overall pattern of results. Second, a summary of the findings from the multivariate indicators (k-dominance curves and size and trophic spectra) are presented. This form of presentation highlights the basic and most useful results, and hopefully minimises the confusion that would ensue from the full presentation of (often very similar) results from hundreds of analyses. Details of how the results have been summarised and what has been omitted are given in each section.

4.7.1 Preliminary Analysis - Attribute correlations

The correlation analyses of the various attributes listed in Table 4.8 shows that there are five fairly independent groups of attributes (membership of the groups is given in Table 4.10). Correlation with other attributes within a group is strong, while connections with attributes located in other groups is weak at best. In keeping with the pattern of strong correlations within the group and weaker connections to other groups, it was necessary to split Group A into A1 and A2 in Table 4.10. This is done because, while attributes in A2 are correlated with those in A1 in a majority of the scenarios, and so there is a single identifiable group, the larger Group A does break into the two smaller sub-groups in a number of scenarios – with those attributes dealing with internal flow characteristics only correlated weakly with those concerning average trophic level, biomass and size- or age-structure.

The core relationships between attributes within each group are largely maintained from scenario-to scenario, at both scales of data aggregation and in both ecosystem types (shallow bay and coast-to-open-ocean). However, the same is not true for the weaker connections between groups. While the weaker connections may be lost scenario-to-scenario within an ecosystem type when data are aggregated to the level of the entire ecosystem, they are more often lost when data are only aggregated to the level of the zone. In addition, the weaker correlations are less likely to be found in the shallow bay ecosystem than in the coast-to-open ocean ecosystem.

In contrast to the variability of the weaker connections between groups the attributes within each group are more reliably correlated with each other. Aside from the division of Group A into A1 and A2 mentioned above, the only core member of any group of attributes that is not reliably correlated with the other attributes in its group is SOI. Even in those scenarios where the attributes in sub-groups A1 and A2 are strongly correlated with each other and the need for distinguishing the sub-groups is not apparent, SOI is only sporadically correlated with the biomass attributes, the detritus biomass ratio (labile: refractory detritus), the age structure attributes and some of the size structure attributes. It is important to note though that while the correlation relationships defining the Groups A-E identified in Table 4.10 are generally reliable, the absolute magnitude of the correlations did vary from scenario-to-scenario, and across scales of data aggregation and system type. Therefore monitoring only one attribute per group and assuming its behaviour captures that of all other attributes in the group would probably be ill advised in reality.

Table 4.10: Groups of attributes identified by the correlation analysis. Solid lines indicate core membership of a group (i.e. strongly correlated with the other attributes in the group), while dash lines signify only weak correlations which are not necessarily present in all scenarios. “Non-leavers” refers to those groups which do not actively migrate in/out of the modelled area, “leavers” refers to groups such as the highly migratory piscivorous groups (tunas), seabirds and the several kinds of marine mammal groups that do leave the modelled area at some point in their life cycle.

Attribute	Group A1	Group A2	Group B	Group C	Group D	Group E
Age structure (of vertebrates)	———	– – – –		———	– – – –	
Average trophic level	———	– – – –				
Biomass – leavers			———			
Biomass – non-leavers	———	– – – –	– – – –	– – – –	– – – –	– – – –
Chlorophyll-a			———			
Consumption	– – – –	———			– – – –	
Denitrification				———		– – – –
Dissolve Inorganic Nitrogen			– – – –			———
Diversity					———	
Nitrification				———		– – – –
Production	– – – –	———			– – – –	
Ratio of labile:refractory detritus	———	– – – –				
Respiration	– – – –	———				
Size structure	———	– – – –	———	———		
SOI	– – – –	– – – –				
Total biomass	———	– – – –				
Total biomass minus detritus	———	– – – –				
Total consumption	– – – –	———				
Total primary production			———			
Total primary production /respiration	– – – –	———	– – – –			
Total production	– – – –	———	– – – –			
Total respiration	– – – –	———				
Total throughput	———	———	– – – –			

4.7.2 Analysis - Univariate Indicators

4.7.2.1 Correlation Analyses

Tables 4.11 to 4.14 give the normalised performance of the various indicators (the number of correlations with $|r| > 0.5$, rescaled such that if the indicator is correlated with all the attributes it has a score of 1.0). For the majority of univariate indicators considered the results are reported per indicator. However, where an indicator applied to individual groups performed similarly across the groups a general result is given rather than exhaustively listing each indicators/group combination (e.g. “Ecotrophic Efficiency – individual groups”). The indicators based on these groups are also referred to generally (e.g. “Primary Production Required – indicator groups”) rather than specifically per group.

Before considering the results in more detail, it is important to note that the lack of correlations for many indicators in the scenarios where there is no commercial fishing is not due to a lack of appropriate data, as sufficient data to calculate all indices are generated from research surveys performed in these scenarios. The low scores in Tables 4.11 - 4.14 for some indicators in these scenarios is actually due to the generally lower levels of contrast in these unfished states. Without pressures on the system, natural levels of variation can swamp any other signals in the model output.

The presence of strong contrast in a scenario was not enough to guarantee strong indicator performance, however. No indicator in any scenario was correlated ($|r| > 0.5$) with all the attributes used in the analyses, which would actually be impossible due to the correlation structure among the attributes (see Table 4.10). Moreover, there was no scenario (or indicator) where all the correlations where rated as strong or better ($|r| > 0.7$). On average, across all indicator-scenario-ecosystem-scale combinations, an indicator was correlated with about 14.5% of the attributes. The distribution of the results is very skewed though (Figure 4.13), as the highest percentage of attributes an indicator is correlated with is about 85% for the “condition at maturity of top predators” in the coast-to-open ocean ecosystem across a range of scenarios and both degrees of data aggregation. Nevertheless, an interesting pattern of indicator-level (or type) can be overlayed on this skewed overall distribution. Across the scenarios in both systems, community- and ecosystem-level indicators (e.g. fish biomass ratios, or the coefficient of variation of total biomass (CV_t)) are much more likely than population-level indicators to be broadly correlated with the attributes. Comparing the solid and dashed lines in Figure 4.13 it is obvious that nearly all indicators correlated with more than 40% of the attributes are community or ecosystem-level indicators.

The skewed distribution of attribute-indicator correlations also means that some attributes were not indexed well. Over two thirds of the linear correlations between indicators and attributes involved biomass-related attributes (e.g. biomass, the number of groups representing 80% of the total biomass and the spatial structure of the biomass), or attributes to do with size structure, trophic level or food web structure. In contrast, the attributes dealing with the internal ecological system structure and flow (respiration, production, nutrient cycling and throughput) in the ecosystem are relatively poorly indexed, with only about 10% of the indicator-attribute correlations involving any of these attributes. Notably, community and ecosystem-level indicators (such as the w-statistic from the k-dominance curves, the slope of the biomass size spectra, biomass ratios like the pelagic:demersal fish biomass ratio, and the average trophic level indicators and many of the model dependent indicators) are correlated with these system-level attributes. However, there are many potential problems with the diet data and model dependent indicators if data quality is poor.

Inspection of outliers on the bi-plots (discussed further below), shows that a number of (mostly model or fisheries-dependent) indicators (listed in Table 4.15) performed quite differently if: data is sparse; sampling or estimation errors are higher than average; if there was a mis-specification between any assumed network structure or fishing history used to calculate the indicators (e.g. in the ECOPATH model), and the actual ecosystem structure, or fishing history. Specific scenarios performed to examine this facet of indicator performance were not run explicitly. Instead during investigation of outliers during preliminary examination of the indicator performance it became clear that when the random error was above average then these changes in performance occur. In all cases performance was worse when data was poor, but for the majority of indicators effected in this way the drop in performance was quite severe and they were rated as “uninformative” (see Table 4.9 for definitions) when applied to poor data.

A further complication in the indicator performance is that there is an interaction between the number of correlations per indicator, the level of fishing pressure, and the scale of data aggregation. When the systems were unfished, 66% of the indicators are correlated with more attributes when data are aggregated to zones rather than across the entire system. This is because the Atlantis framework includes processes and mechanisms which can produce considerable spatial and temporal variation in system attributes even in unexploited systems. As a result, analyses using data aggregated to zones maximizes the spatial contrast in the system attributes and results in the greatest potential for correlations to be observed. In contrast, in many parts of the fished model ecosystems there have been great reductions in the spatial contrasts. In turn, this results in a switch in the most useful scale of data aggregation, with nearly 70% of the indicators correlated with more attributes when using data aggregated to the system-level rather than the zone in the intensively fished scenarios.

Irrespective of these patterns concerning the level of data aggregation and the level of fishing pressure, broad examination of the results in Tables 4.11 – 4.14 shows that the same types of indicators perform well at both levels of aggregation and across most scenarios. Indicators that consistently have a recognisable (or better) linear correlation with a broad range of attributes are often simple indicators – e.g. biomass per group, biomass ratios, size (or condition) at maturity and the slope of the biomass size spectrum. While more complicated indicators, such as model dependent indicators, did perform well in some circumstances (e.g. the Average trophic level indicators, or Ecotrophic Efficiency when considered for the indicator groups as a whole) their sensitivity to data quality ultimately makes them much less useful.

Further details of the overall patterns of results across scenarios, system-type and levels of data aggregation will be given below in sections 4.1.7.3 and 4.1.7.4, which deal with the multidimensional scaling plots (based on the normalised performance scores of the various indicators in the different scenarios) and the overall categorisation of the indicators.

Figure 4.13: Plot of the frequency of indicators against the percentage of attributes they are correlated ($|r| > 0.5$) with.

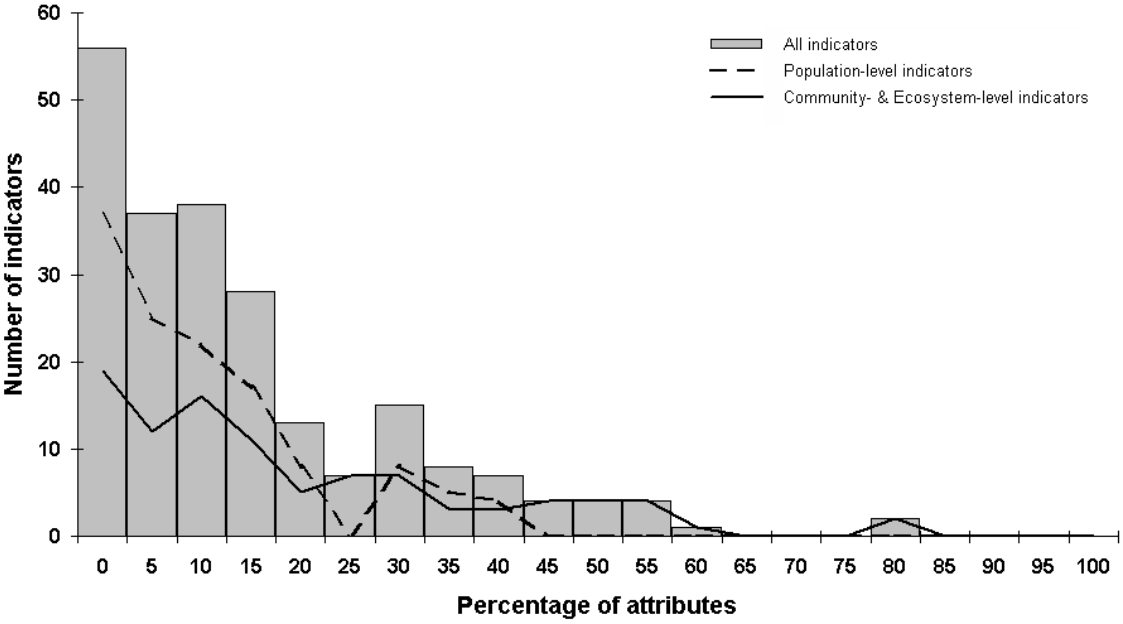


Table 4.11: Normalised linear correlation results for the shallow bay ecosystem analyses, using data aggregated to the level of the management zones identified in Figure 4.3. Codes for the scenario names come from Table 4.6, grey colour indicators unfished scenarios. Mean results across unfished (SCN) and fished (SC/SO) also given. (F) represents "fixed".

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Average trophic level	0.22	0.22	0.22	0.15	0.11	0.07	0.07	0.07	0.11	0.11	0.00	0.07	0.19	0.15	0.07	0.07	0.15	0.09
Average trophic level (F)	0.22	0.22	0.22	0.11	0.11	0.07	0.11	0.07	0.11	0.11	0.00	0.19	0.22	0.07	0.07	0.07	0.11	0.12
Average trophic level of vertebrate catch (F)	0.22	0.11	0.17	0.11	0.11	0.04	0.11	0.04	0.04	0.04	0.00	0.04	0.07	0.04	0.04	0.04	0.04	0.05
Average trophic level of vertebrates (F)	0.00	0.26	0.13	0.00	0.00	0.00	0.04	0.00	0.00	0.04	0.04	0.19	0.22	0.00	0.00	0.00	0.00	0.07
Average trophic level of catch (F)	0.30	0.26	0.28	0.04	0.04	0.07	0.00	0.04	0.04	0.07	0.04	0.07	0.04	0.04	0.04	0.04	0.04	0.04
Average trophic level of the catch	0.00	0.00	0.00	0.04	0.00	0.07	0.04	0.04	0.04	0.04	0.04	0.00	0.00	0.04	0.04	0.04	0.04	0.03
Average trophic level of vertebrate catch	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.01
Average trophic level of vertebrates	0.00	0.00	0.00	0.11	0.07	0.00	0.00	0.11	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03
Biomass – bacteria	0.15	0.33	0.24	0.19	0.15	0.07	0.22	0.07	0.07	0.15	0.04	0.33	0.41	0.15	0.07	0.07	0.07	0.19
Biomass – cephalopods	0.15	0.19	0.17	0.15	0.11	0.19	0.19	0.19	0.19	0.19	0.04	0.22	0.19	0.11	0.19	0.04	0.15	0.17
Biomass – demersal fish	0.19	0.11	0.15	0.11	0.11	0.00	0.00	0.00	0.00	0.11	0.04	0.00	0.3	0.07	0.00	0.00	0.00	0.07
Biomass – detritus	0.11	0.11	0.11	0.22	0.19	0.15	0.22	0.19	0.19	0.19	0.04	0.33	0.37	0.19	0.19	0.11	0.15	0.22
Biomass – epifauna	0.30	0.22	0.26	0.04	0.04	0.00	0.33	0.07	0.07	0.15	0.00	0.22	0.11	0.07	0.07	0.00	0.04	0.14
Biomass – gelatinous zooplankton	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Biomass – individual microfauna groups	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.30	0.37	0.33	0.22	0.11	0.26	0.26	0.19	0.26	0.22	0.04	0.19	0.19	0.22	0.19	0.11	0.19	0.19
Biomass – macroalgae	0.26	0.22	0.24	0.15	0.07	0.07	0.15	0.11	0.11	0.11	0.00	0.19	0.19	0.11	0.07	0.07	0.04	0.12
Biomass – mammals	0.33	0.22	0.28	0.07	0.07	0.04	0.11	0.07	0.07	0.22	0.04	0.22	0.15	0.11	0.11	0.04	0.00	0.13
Biomass – piscivores	0.26	0.22	0.24	0.11	0.07	0.00	0.00	0.00	0.00	0.11	0.00	0.15	0.15	0.07	0.00	0.00	0.00	0.06
Biomass – planktivores	0.26	0.19	0.22	0.11	0.11	0.00	0.07	0.04	0.00	0.15	0.04	0.22	0.26	0.07	0.00	0.00	0.00	0.11
Biomass – seabirds	0.22	0.22	0.22	0.11	0.04	0.07	0.11	0.07	0.07	0.22	0.04	0.22	0.26	0.11	0.07	0.04	0.11	0.14
Biomass – seagrass	0.07	0.00	0.04	0.04	0.04	0.00	0.04	0.04	0.04	0.04	0.04	0.11	0.04	0.04	0.04	0.00	0.00	0.05
Biomass – sharks	0.19	0.19	0.19	0.11	0.07	0.00	0.00	0.00	0.00	0.11	0.00	0.22	0.15	0.04	0.00	0.00	0.00	0.07
BSS - slope	0.26	0.19	0.22	0.15	0.15	0.15	0.19	0.19	0.3	0.04	0.26	0.22	0.22	0.07	0.15	0.07	0.07	0.20

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Capacity	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.02
Chlorophyll a	0.04	0.07	0.06	0.11	0.04	0.04	0.00	0.00	0.00	0.04	0.00	0.04	0.04	0.04	0.00	0.00	0.00	0.02
Condition at maturity – seabird	0.26	0.19	0.22	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.00	0.30	0.22	0.19	0.15	0.15	0.11	0.16
Condition at maturity – top predators	0.33	0.11	0.22	0.07	0.04	0.07	0.22	0.00	0.00	0.26	0.00	0.00	0.22	0.04	0.00	0.00	0.00	0.10
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – cephalopods	0.07	0.07	0.07	0.04	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.07	0.07	0.04	0.00	0.00	0.00	0.03
CPUE – demersal fish	0.11	0.07	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.02
CPUE – epifauna	0.07	0.07	0.07	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.04	0.07	0.00	0.00	0.00	0.00	0.02
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	<0.01
CPUE – piscivores	0.07	0.07	0.07	0.04	0.04	0.07	0.00	0.00	0.00	0.00	0.04	0.11	0.04	0.00	0.00	0.00	0.00	0.03
CPUE – planktivores	0.07	0.04	0.06	0.04	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.02
CPUE – sharks	0.11	0.07	0.09	0.00	0.00	0.00	0.00	0.22	0.00	0.04	0.04	0.04	0.26	0.04	0.00	0.00	0.00	0.08
CVt	0.22	0.3	0.26	0.11	0.11	0.11	0.19	0.11	0.11	0.11	0.04	0.30	0.19	0.11	0.11	0.11	0.11	0.15
Denitrification	0.26	0.26	0.26	0.19	0.07	0.11	0.19	0.07	0.07	0.11	0.04	0.00	0.11	0.11	0.07	0.04	0.11	0.08
Detrital dominance	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Discards – demersal fish	0.22	0.22	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.22	0.04	0.00	0.00	0.00	0.04
Discards – individual groups	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Discards – invertebrates	0.26	0.26	0.26	0.15	0.04	0.11	0.07	0.04	0.07	0.19	0.04	0.11	0.30	0.22	0.04	0.00	0.07	0.12
Discards – piscivores	0.22	0.22	0.22	0.15	0.00	0.07	0.00	0.00	0.04	0.15	0.04	0.00	0.00	0.04	0.00	0.00	0.00	0.03
Discards – planktivores	0.19	0.19	0.19	0.15	0.11	0.04	0.04	0.04	0.04	0.11	0.04	0.07	0.19	0.07	0.04	0.04	0.04	0.07
Dissolved Inorganic Nitrogen	0.07	0.11	0.09	0.04	0.04	0.04	0.00	0.00	0.00	0.04	0.00	0.04	0.22	0.04	0.04	0.00	0.00	0.04
Diversity – count of groups present	0.30	0.19	0.24	0.11	0.04	0.00	0.11	0.11	0.11	0.11	0.04	0.26	0.22	0.07	0.07	0.00	0.07	0.14
Diversity – number of communities	0.22	0.30	0.26	0.22	0.19	0.44	0.22	0.41	0.37	0.19	0.44	0.37	0.22	0.44	0.30	0.22	0.37	0.31
Ecotrophic Efficiency – indicator groups	0.56	0.48	0.52	0.56	0.41	0.30	0.56	0.37	0.41	0.41	0.04	0.52	0.44	0.48	0.33	0.22	0.33	0.39
Ecotrophic Efficiency – individual groups	0.11	0.07	0.09	0.04	0.04	0.04	0.07	0.04	0.04	0.04	0.00	0.07	0.04	0.04	0.04	0.04	0.04	0.04
Ecological footprint – demersal fish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Ecological footprint – piscivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FIB (F)	0.00	0.00	0.00	0.11	0.07	0.00	0.00	0.00	0.00	0.11	0.00	0.19	0.15	0.04	0.00	0.00	0.00	0.06
FIB slope	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FIB	0.00	0.00	0.00	0.11	0.07	0.00	0.00	0.00	0.00	0.11	0.00	0.15	0.15	0.04	0.00	0.00	0.00	0.06

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Infauna/Epifauna (biomass ratio)	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Invertebrate:Finfish biomass ratio	0.22	0.30	0.26	0.28	0.31	0.08	0.18	0.21	0.11	0.19	0.15	0.23	0.19	0.11	0.23	0.23	0.11	0.19
Invertebrate:Finfish catch ratio	0.01	0.01	0.01	0.02	0.00	0.02	0.04	0.04	0.00	0.00	0.02	0.04	0.04	0.02	0.01	0.00	0.00	0.02
labile / refract detritus (biomass ratio)	0.11	0.11	0.11	0.04	0.04	0.00	0.07	0.00	0.04	0.04	0.00	0.11	0.07	0.04	0.00	0.00	0.00	0.05
Light levels on the sediment surface	0.15	0.04	0.09	0.07	0.00	0.07	0.07	0.07	0.07	0.04	0.04	0.15	0.07	0.07	0.00	0.07	0.07	0.07
Maximum length of catch	0.22	0.22	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.04	0.04	0.00	0.00	0.00	0.00	0.02
Net primary production / biomass	0.00	0.00	0.00	0.07	0.07	0.00	0.00	0.00	0.00	0.07	0.00	0.07	0.11	0.07	0.00	0.00	0.00	0.04
Nitrification	0.26	0.26	0.26	0.15	0.07	0.11	0.19	0.07	0.07	0.11	0.00	0.19	0.11	0.11	0.07	0.04	0.00	0.11
Number of mammals caught during fishing	0.22	0.22	0.22	0.22	0.22	0.19	0.22	0.19	0.22	0.19	0.00	0.11	0.19	0.19	0.15	0.15	0.00	0.16
Number of seabirds caught during fishing	0.19	0.11	0.15	0.15	0.04	0.00	0.11	0.04	0.07	0.00	0.04	0.22	0.15	0.15	0.00	0.00	0.00	0.09
Overhead	0.15	0.11	0.13	0.00	0.00	0.00	0.07	0.04	0.04	0.07	0.04	0.19	0.07	0.00	0.00	0.07	0.00	0.07
PBR – mammals	0.22	0.15	0.19	0.11	0.07	0.11	0.15	0.11	0.11	0.11	0.04	0.19	0.15	0.07	0.07	0.07	0.00	0.12
PBR – seabirds	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.04	0.00	0.00	0.00	0.03
Pelagic: Demersal (catch biomass ratio)	0.22	0.22	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.19	0.15	0.04	0.00	0.00	0.00	0.06
Pelagic: Demersal (fish biomass ratio)	0.22	0.22	0.22	0.04	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.15	0.00	0.00	0.00	0.00	0.03
Primary Production Required – catch	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Primary Production Required – indicator grps	0.44	0.44	0.44	0.41	0.19	0.26	0.44	0.37	0.37	0.44	0.00	0.63	0.56	0.48	0.22	0.19	0.30	0.4
Primary Production Required – individ. grps	0.07	0.07	0.07	0.04	0.00	0.04	0.04	0.04	0.04	0.07	0.00	0.11	0.04	0.04	0.04	0.00	0.04	0.05
Proportion of stock juvenile – demersal fish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Proportion of stock juvenile – piscivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Proportion of stock juvenile – planktivores	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.02
Proportion of the catch juvenile	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PS / ZP (catch biomass ratio)	0.15	0.11	0.13	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	<0.01
PS / ZP (fish biomass ratio)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.07	0.04	0.00	0.00	0.00	0.02
Relative Ascendency	0.15	0.11	0.13	0.15	0.00	0.00	0.07	0.07	0.00	0.07	0.04	0.22	0.07	0.04	0.07	0.00	0.00	0.08

Indicator	SCN1	SCN2	SCN (mean)	SC1	SC2	SC3	SC4	SC5	SC6	SC7	SC8	SC9	SC10	SO1	SO2	SO3	SO4	SC/SO (mean)
Reproductive success of mammals	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Reproductive success of seabirds	0.00	0.22	0.11	0.11	0.00	0.00	0.07	0.00	0.07	0.07	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.04
Size at maturity – demersal fish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Size at maturity – piscivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SOI	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total benthic consumption	0.37	0.52	0.44	0.00	0.19	0.26	0.26	0.26	0.41	0.33	0.00	0.44	0.33	0.22	0.26	0.11	0.26	0.29
Total benthic production	0.48	0.44	0.46	0.37	0.30	0.22	0.41	0.30	0.37	0.41	0.04	0.52	0.41	0.33	0.22	0.11	0.22	0.35
Total biomass	0.15	0.19	0.17	0.15	0.11	0.19	0.22	0.19	0.19	0.15	0.00	0.19	0.44	0.15	0.19	0.11	0.19	0.20
Total biomass – invertebrate (macrofauna)	0.33	0.45	0.39	0.07	0.11	0.11	0.07	0.04	0.07	0.07	0.11	0.07	0.09	0.04	0.07	0.07	0.04	0.07
Total biomass – vertebrate	0.42	0.38	0.40	0.30	0.26	0.22	0.33	0.30	0.30	0.26	0.30	0.26	0.30	0.33	0.33	0.26	0.26	0.29
Total catch – invertebrate (macrofauna)	0.00	0.01	<0.01	0.07	0.11	0.07	0.04	0.04	0.11	0.11	0.11	0.07	0.04	0.11	0.11	0.07	0.04	0.08
Total catch – vertebrate	0.01	0.02	0.02	0.04	0.04	0.04	0.07	0.02	0.04	0.04	0.00	0.04	0.07	0.04	0.04	0.04	0.04	0.04
Total consumption	0.00	0.00	0.00	0.19	0.11	0.00	0.00	0.00	0.00	0.19	0.00	0.07	0.00	0.11	0.00	0.00	0.00	0.04
Total consumption - indicator groups	0.18	0.12	0.15	0.15	0.04	0.01	0.11	0.04	0.07	0.00	0.04	0.22	0.11	0.11	0.04	0.00	0.04	0.09
Total consumption - individual groups	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	<0.01
Total discard rate	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total flow to detritus	0.26	0.22	0.24	0.15	0.00	0.11	0.19	0.11	0.07	0.07	0.04	0.07	0.00	0.15	0.07	0.04	0.15	0.08
Total mortality – demersal fish	0.04	0.04	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.04	0.04	0.00	0.00	0.00	0.00	0.02
Total mortality – filter feeders	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – macrozoobenthos	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – mammals	0.04	0.07	0.06	0.07	0.00	0.00	0.04	0.04	0.04	0.04	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.03
Total mortality – piscivores	0.11	0.15	0.13	0.04	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.02
Total mortality – planktivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – seabirds	0.00	0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.04	0.00	0.00	0.00	0.00	0.02
Total mortality – sharks	0.04	0.04	0.04	0.04	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.00	0.00	0.00	0.01
Total planktonic production	0.56	0.44	0.5	0.26	0.15	0.22	0.33	0.22	0.22	0.22	0.04	0.19	0.22	0.26	0.19	0.11	0.22	0.21
Total primary production	0.00	0.00	0.00	0.15	0.07	0.00	0.00	0.00	0.00	0.07	0.04	0.11	0.00	0.11	0.00	0.00	0.00	0.03
Total production	0.00	0.00	0.00	0.22	0.11	0.00	0.00	0.00	0.00	0.07	0.04	0.07	0.00	0.07	0.00	0.00	0.00	0.03
Total production - indicator groups	0.18	0.12	0.15	0.15	0.04	0.01	0.11	0.04	0.07	0.00	0.04	0.22	0.11	0.11	0.04	0.00	0.04	0.09
Total production - individual groups	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	<0.01

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Total residence time	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total respiration	0.07	0.04	0.06	0.07	0.07	0.07	0.07	0.04	0.07	0.07	0.00	0.07	0.00	0.07	0.04	0.04	0.07	0.05
Total system exports	0.22	0.22	0.22	0.07	0.04	0.04	0.00	0.04	0.04	0.11	0.00	0.22	0.00	0.04	0.00	0.00	0.04	0.06
Total throughput	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.02
Trophic efficiency – detritus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trophic efficiency – production	0.15	0.11	0.13	0.00	0.00	0.00	0.04	0.04	0.00	0.00	0.00	0.11	0.04	0.00	0.00	0.00	0.00	0.03
Trophic spectra – peak	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Trophic spectra – slope (F)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Trophic spectra – slope	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
w-abc	0.22	0.07	0.15	0.07	0.04	0.00	0.04	0.04	0.04	0.07	0.19	0.07	0.11	0.07	0.04	0.07	0.07	0.08

Table 4.12: Normalised linear correlation results for the shallow bay ecosystem analyses, using data aggregated across the entire system. Codes for the scenario names come from Table 4.6, grey colour indicators unfished scenarios. Mean results across unfished (SCN) and fished (SC/SO) also given. (F) represents "fixed".

Indicator	SCN1	SCN2	SCN (mean)	SC1	SC2	SC3	SC4	SC5	SC6	SC7	SC8	SC9	SC10	SO1	SO2	SO3	SO4	SC/SO (mean)
Average trophic level	0.00	0.00	0.00	0.00	0.15	0.04	0.30	0.07	0.11	0.11	0.04	0.04	0.00	0.19	0.04	0.26	0.11	0.04
Average trophic level (F)	0.00	0.33	0.17	0.04	0.04	0.07	0.15	0.07	0.04	0.15	0.00	0.00	0.41	0.04	0.04	0.15	0.04	0.09
Average trophic level of vertebrate catch (F)	0.00	0.00	0.00	0.19	0.19	0.30	0.26	0.22	0.22	0.26	0.00	0.37	0.67	0.19	0.26	0.19	0.15	0.25
Average trophic level of vertebrates (F)	0.07	0.04	0.06	0.15	0.19	0.22	0.37	0.19	0.11	0.37	0.07	0.00	0.52	0.07	0.15	0.11	0.15	0.19
Average trophic level of catch (F)	0.11	0.41	0.26	0.11	0.15	0.37	0.37	0.37	0.41	0.37	0.04	0.33	0.04	0.07	0.41	0.07	0.11	0.23
Average trophic level of the catch	0.00	0.00	0.00	0.15	0.19	0.44	0.30	0.07	0.26	0.22	0.00	0.00	0.00	0.15	0.37	0.07	0.15	0.17
Average trophic level of vertebrate catch	0.00	0.00	0.00	0.15	0.04	0.04	0.00	0.00	0.00	0.00	0.07	0.00	0.11	0.04	0.04	0.04	0.04	0.04
Average trophic level of vertebrates	0.00	0.00	0.00	0.11	0.07	0.15	0.07	0.00	0.11	0.07	0.07	0.00	0.04	0.11	0.07	0.11	0.11	0.08
Biomass – bacteria	0.04	0.37	0.20	0.37	0.33	0.52	0.41	0.30	0.44	0.37	0.00	0.15	0.19	0.33	0.19	0.33	0.33	0.30
Biomass – cephalopods	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Biomass – demersal fish	0.04	0.04	0.04	0.22	0.30	0.41	0.41	0.33	0.41	0.41	0.00	0.33	0.63	0.11	0.19	0.15	0.07	0.28
Biomass – detritus	0.07	0.37	0.22	0.30	0.30	0.52	0.44	0.33	0.41	0.33	0.15	0.15	0.19	0.30	0.11	0.30	0.30	0.29
Biomass – epifauna	0.00	0.00	0.00	0.48	0.48	0.48	0.44	0.37	0.48	0.30	0.00	0.22	0.00	0.44	0.41	0.52	0.52	0.37
Biomass – gelatinous zooplankton	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Biomass – individual microfauna groups	0.00	0.01	<0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.11	0.07	0.09	0.04	0.00	0.07	0.19	0.04	0.04	0.04	0.04	0.11	0.07	0.04	0.00	0.04	0.04	0.05
Biomass – macroalgae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Biomass – mammals	0.07	0.44	0.26	0.15	0.19	0.07	0.11	0.00	0.04	0.11	0.41	0.37	0.52	0.15	0.33	0.11	0.11	0.19
Biomass – piscivores	0.00	0.04	0.02	0.19	0.22	0.30	0.30	0.22	0.07	0.30	0.11	0.52	0.56	0.19	0.15	0.11	0.15	0.24
Biomass – planktivores	0.04	0.00	0.02	0.26	0.30	0.41	0.48	0.41	0.48	0.44	0.00	0.37	0.44	0.22	0.41	0.22	0.22	0.33
Biomass – seabirds	0.11	0.30	0.20	0.15	0.19	0.30	0.04	0.00	0.07	0.04	0.26	0.44	0.41	0.11	0.07	0.11	0.11	0.16
Biomass – seagrass	0.11	0.07	0.09	0.26	0.22	0.30	0.11	0.04	0.04	0.11	0.00	0.26	0.00	0.19	0.26	0.30	0.26	0.17
Biomass – sharks	0.15	0.15	0.15	0.15	0.19	0.30	0.26	0.15	0.22	0.11	0.30	0.30	0.52	0.26	0.30	0.15	0.30	0.25
BSS - slope	0.04	0.04	0.04	0.41	0.30	0.41	0.44	0.48	0.48	0.44	0.00	0.37	0.85	0.26	0.22	0.41	0.22	0.38

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Capacity	0.04	0.48	0.26	0.07	0.07	0.00	0.07	0.11	0.07	0.04	0.04	0.04	0.52	0.07	0.00	0.07	0.11	0.09
Chlorophyll a	0.15	0.44	0.30	0.19	0.15	0.15	0.04	0.00	0.00	0.00	0.00	0.07	0.04	0.15	0.04	0.04	0.07	0.07
Condition at maturity – seabird	0.11	0.30	0.20	0.19	0.26	0.30	0.37	0.30	0.44	0.37	0.33	0.41	0.48	0.19	0.33	0.19	0.19	0.31
Condition at maturity – top predators	0.22	0.19	0.20	0.11	0.15	0.15	0.41	0.15	0.30	0.19	0.11	0.15	0.48	0.07	0.26	0.07	0.11	0.19
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – cephalopods	0.00	0.00	0.00	0.07	0.07	0.15	0.15	0.11	0.15	0.11	0.00	0.00	0.00	0.07	0.11	0.07	0.07	0.08
CPUE – demersal fish	0.00	0.00	0.00	0.04	0.04	0.07	0.11	0.07	0.11	0.07	0.00	0.00	0.22	0.04	0.07	0.04	0.04	0.07
CPUE – epifauna	0.15	0.11	0.13	0.04	0.04	0.19	0.11	0.07	0.07	0.11	0.11	0.07	0.00	0.07	0.15	0.07	0.04	0.08
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	0.01	<0.01	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	<0.01
CPUE – piscivores	0.00	0.00	0.00	0.07	0.07	0.15	0.11	0.11	0.15	0.11	0.00	0.00	0.19	0.07	0.11	0.11	0.07	0.10
CPUE – planktivores	0.04	0.04	0.04	0.04	0.11	0.11	0.11	0.07	0.11	0.11	0.00	0.00	0.15	0.07	0.11	0.07	0.07	0.08
CPUE – sharks	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	<0.01
CVt	0.00	0.00	0.00	0.00	0.00	0.04	0.11	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.04	0.00	0.11	0.03
Denitrification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	<0.01
Detrital dominance	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Discards – demersal fish	0.00	0.33	0.17	0.15	0.07	0.22	0.30	0.15	0.33	0.22	0.00	0.37	0.63	0.15	0.19	0.11	0.15	0.22
Discards – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.00	<0.01	<0.01	<0.01	<0.01	0.00	<0.01
Discards – invertebrates	0.04	0.52	0.28	0.37	0.15	0.26	0.48	0.15	0.33	0.30	0.41	0.44	0.48	0.33	0.52	0.37	0.15	0.34
Discards – piscivores	0.00	0.00	0.00	0.30	0.15	0.48	0.41	0.26	0.48	0.41	0.00	0.30	0.04	0.30	0.37	0.30	0.26	0.29
Discards – planktivores	0.00	0.00	0.00	0.26	0.30	0.41	0.44	0.44	0.44	0.44	0.00	0.52	0.56	0.30	0.44	0.30	0.30	0.37
Dissolved Inorganic Nitrogen	0.15	0.67	0.41	0.04	0.04	0.04	0.11	0.04	0.04	0.11	0.19	0.19	0.44	0.07	0.04	0.04	0.04	0.10
Diversity – count of groups present	0.11	0.00	0.06	0.00	0.44	0.22	0.15	0.19	0.15	0.37	0.26	0.07	0.44	0.00	0.30	0.22	0.41	0.23
Diversity – number of communities	0.15	0.27	0.21	0.37	0.33	0.52	0.41	0.30	0.44	0.37	0.37	0.15	0.19	0.33	0.26	0.33	0.33	0.34
Ecotrophic Efficiency – indicator groups	0.26	0.56	0.41	0.7	0.59	0.67	0.56	0.41	0.52	0.44	0.56	0.56	0.19	0.52	0.56	0.59	0.56	0.53
Ecotrophic Efficiency – individual groups	0.04	0.11	0.07	0.11	0.11	0.19	0.15	0.11	0.15	0.11	0.07	0.11	0.04	0.07	0.15	0.11	0.11	0.11
Ecological footprint – demersal fish	0.00	0.00	0.00	0.04	0.11	0.04	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
Ecological footprint – piscivores	0.00	0.00	0.00	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
FIB (F)	0.04	0.22	0.13	0.22	0.30	0.44	0.37	0.30	0.41	0.37	0.04	0.52	0.56	0.22	0.33	0.22	0.22	0.32
FIB slope	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.04	0.00	0.00	<0.01
FIB	0.00	0.00	0.00	0.30	0.30	0.48	0.33	0.22	0.44	0.41	0.00	0.44	0.44	0.19	0.52	0.19	0.15	0.31

Indicator	SCN1	SCN2	SCN (mean)	SC1	SC2	SC3	SC4	SC5	SC6	SC7	SC8	SC9	SC10	SO1	SO2	SO3	SO4	SC/SO (mean)
Infauna/Epifauna (biomass ratio)	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.07	0.15	0.03
Invertebrate:Finfish biomass ratio	0.40	0.40	0.40	0.40	0.37	0.46	0.37	0.49	0.37	0.37	0.40	0.37	0.44	0.44	0.37	0.37	0.44	0.40
Invertebrate:Finfish catch ratio	0.00	0.01	<0.01	0.00	0.00	<0.01	0.00	0.01	0.00	0.01	<0.01	0.00	0.00	0.00	0.00	0.01	0.00	<0.01
labile / refract detritus (biomass ratio)	0.11	0.07	0.09	0.30	0.30	0.33	0.33	0.26	0.33	0.37	0.11	0.15	0.00	0.22	0.00	0.30	0.30	0.24
Light levels on the sediment surface	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Maximum length of catch	0.00	0.33	0.17	0.15	0.11	0.30	0.33	0.19	0.37	0.33	0.04	0.19	0.59	0.07	0.26	0.07	0.11	0.22
Net primary production / biomass	0.15	0.15	0.15	0.19	0.19	0.07	0.11	0.07	0.07	0.07	0.11	0.07	0.19	0.11	0.07	0.07	0.11	0.11
Nitrification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	<0.01
Number of mammals caught during fishing	0.00	0.00	0.00	0.00	0.00	0.04	0.22	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.22	0.06
Number of seabirds caught during fishing	0.07	0.15	0.11	0.07	0.11	0.04	0.00	0.00	0.07	0.00	0.26	0.48	0.19	0.07	0.22	0.07	0.07	0.12
Number of communities	0.15	0.27	0.21	0.37	0.33	0.52	0.41	0.30	0.44	0.37	0.37	0.15	0.19	0.33	0.26	0.33	0.33	0.34
Overhead	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
PBR – mammals	0.04	0.04	0.04	0.11	0.15	0.15	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.00	0.07	0.06
PBR – seabirds	0.11	0.37	0.24	0.19	0.22	0.00	0.07	0.00	0.04	0.07	0.30	0.48	0.33	0.19	0.04	0.19	0.00	0.15
Pelagic: Demersal (catch biomass ratio)	0.00	0.00	0.00	0.22	0.19	0.44	0.30	0.15	0.30	0.00	0.00	0.11	0.11	0.22	0.00	0.19	0.30	0.18
Pelagic: Demersal (fish biomass ratio)	0.04	0.04	0.04	0.41	0.52	0.22	0.44	0.41	0.26	0.19	0.00	0.07	0.59	0.07	0.15	0.07	0.07	0.25
Primary Production Required – catch	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Primary Production Required – indicator grps	0.19	0.48	0.33	0.63	0.63	0.59	0.59	0.48	0.59	0.44	0.33	0.56	0.63	0.56	0.56	0.59	0.63	0.56
Primary Production Required – individ. grps	0.04	0.07	0.06	0.15	0.15	0.15	0.11	0.11	0.11	0.11	0.04	0.07	0.15	0.07	0.07	0.07	0.07	0.10
Proportion of stock juvenile – demersal fish	0.00	0.00	0.00	0.00	0.00	0.04	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
Proportion of stock juvenile – piscivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Proportion of stock juvenile – planktivores	0.11	0.22	0.17	0.19	0.26	0.48	0.41	0.26	0.41	0.37	0.26	0.33	0.11	0.19	0.37	0.19	0.19	0.29
Proportion of the catch juvenile	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	<0.01
PS / ZP (catch biomass ratio)	0.04	0.04	0.04	0.41	0.52	0.41	0.41	0.15	0.26	0.04	0.00	0.41	0.04	0.22	0.22	0.30	0.22	0.26
PS / ZP (fish biomass ratio)	0.00	0.00	0.00	0.44	0.44	0.33	0.26	0.11	0.07	0.00	0.00	0.37	0.37	0.30	0.33	0.44	0.33	0.27
Relative Ascendency	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Reproductive success of mammals	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
Reproductive success of seabirds	0.00	0.00	0.00	0.00	0.00	0.15	0.19	0.11	0.15	0.19	0.00	0.00	0.00	0.11	0.11	0.11	0.11	0.09
Size at maturity – demersal fish	0.00	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
Size at maturity – piscivores	0.15	0.11	0.13	0.15	0.22	0.19	0.19	0.11	0.30	0.15	0.00	0.00	0.04	0.07	0.11	0.04	0.00	0.11
SOI	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.07	0.00	0.04	0.26	0.00	0.00	0.03
Total benthic consumption	0.11	0.59	0.35	0.19	0.26	0.44	0.41	0.33	0.41	0.41	0.33	0.37	0.30	0.19	0.37	0.22	0.19	0.31
Total benthic production	0.19	0.52	0.35	0.52	0.48	0.56	0.56	0.41	0.52	0.48	0.30	0.37	0.26	0.48	0.48	0.52	0.52	0.46
Total biomass	0.00	0.33	0.17	0.00	0.00	0.07	0.11	0.04	0.07	0.00	0.00	0.00	0.04	0.00	0.07	0.00	0.00	0.03
Total biomass – invertebrate (macrofauna)	0.54	0.49	0.52	0.48	0.37	0.37	0.26	0.59	0.48	0.37	0.41	0.37	0.37	0.41	0.41	0.37	0.26	0.39
Total biomass – vertebrate	0.11	0.11	0.11	0.15	0.19	0.07	0.15	0.15	0.07	0.07	0.07	0.22	0.19	0.19	0.07	0.11	0.15	0.13
Total catch – invertebrate (macrofauna)	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	<0.01	<0.01	<0.01	<0.01	0.00	0.00	0.00	0.01	<0.01
Total catch – vertebrate	0.00	<0.01	<0.01	0.00	0.00	<0.01	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.01	<0.01
Total consumption	0.04	0.41	0.22	0.07	0.07	0.07	0.07	0.00	0.04	0.00	0.04	0.11	0.04	0.11	0.00	0.07	0.11	0.06
Total consumption - indicator groups	0.07	0.15	0.11	0.07	0.11	0.11	0.11	0.07	0.07	0.07	0.15	0.18	0.19	0.15	0.22	0.07	0.07	0.12
Total consumption - individual groups	0.00	0.01	<0.01	<0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	<0.01	<0.01	0.00	0.00	0.00	0.00	<0.01
Total discard rate	0.00	0.00	0.00	0.15	0.19	0.19	0.22	0.22	0.15	0.19	0.33	0.26	0.00	0.11	0.15	0.11	0.07	0.17
Total flow to detritus	0.04	0.44	0.24	0.19	0.22	0.15	0.15	0.07	0.11	0.04	0.07	0.07	0.22	0.07	0.07	0.11	0.19	0.12
Total mortality – demersal fish	0.00	0.00	0.00	0.11	0.15	0.22	0.33	0.11	0.30	0.22	0.04	0.00	0.48	0.15	0.04	0.26	0.19	0.19
Total mortality – filter feeders	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – macrozoobenthos	0.00	0.00	0.00	0.00	0.00	0.07	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.03
Total mortality – mammals	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	<0.01
Total mortality – piscivores	0.04	0.19	0.11	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.07	0.19	0.04	0.00	0.00	0.00	0.00	0.03
Total mortality – planktivores	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.00	<0.01
Total mortality – seabirds	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	<0.01
Total mortality – sharks	0.07	0.26	0.17	0.00	0.00	0.07	0.04	0.00	0.11	0.00	0.19	0.00	0.04	0.04	0.04	0.04	0.04	0.04
Total planktonic production	0.15	0.11	0.13	0.19	0.15	0.07	0.15	0.04	0.04	0.04	0.07	0.22	0.04	0.04	0.00	0.11	0.11	0.09
Total primary production	0.00	0.00	0.00	0.22	0.19	0.19	0.26	0.11	0.26	0.11	0.15	0.15	0.04	0.07	0.07	0.04	0.07	0.14
Total production	0.15	0.11	0.13	0.19	0.22	0.19	0.15	0.11	0.11	0.11	0.15	0.11	0.04	0.15	0.07	0.11	0.15	0.13
Total production - indicator groups	0.07	0.15	0.11	0.07	0.11	0.11	0.11	0.07	0.07	0.07	0.15	0.18	0.19	0.15	0.22	0.07	0.07	0.12
Total production - individual groups	0.00	0.01	<0.01	<0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	<0.01	<0.01	0.00	0.00	0.00	0.00	<0.01

Indicator	**SCN1**	**SCN2**	**SCN (mean)**	**SC1**	**SC2**	**SC3**	**SC4**	**SC5**	**SC6**	**SC7**	**SC8**	**SC9**	**SC10**	**SO1**	**SO2**	**SO3**	**SO4**	**SC/SO (mean)**
Total residence time	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total respiration	0.00	0.37	0.19	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Total system exports	0.00	0.41	0.20	0.33	0.41	0.44	0.33	0.07	0.07	0.33	0.19	0.11	0.37	0.33	0.11	0.41	0.37	0.28
Total throughput	0.41	0.00	0.20	0.11	0.04	0.07	0.07	0.04	0.04	0.04	0.04	0.04	0.59	0.04	0.07	0.04	0.11	0.10
Trophic efficiency – detritus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trophic efficiency – production	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trophic spectra – peak	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Trophic spectra – slope (F)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Trophic spectra – slope	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
w-abc	0.07	0.04	0.06	0.26	0.22	0.11	0.11	0.11	0.04	0.04	0.11	0.07	0.00	0.33	0.3	0.33	0.37	0.17

Table 4.13a: Normalised linear correlation results for the scenarios ON1, ON2, and O1 to O14 for the coast-to-open-ocean ecosystem analyses, using data aggregated to the level of the management zones identified in Figure 4.4. ON (mean) given here is identical to that in table 4.13b (table split due to space limitations). (F) represents “fixed”.

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Average trophic level	0.00	0.00	0.00	0.39	0.31	0.00	0.53	0.06	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.33	0.00	0.19
Average trophic level (F)	0.00	0.00	0.00	0.36	0.33	0.39	0.53	0.28	0.42	0.42	0.42	0.28	0.50	0.42	0.39	0.31	0.42	0.36
Average trophic level of vertebrate catch (F)	0.00	0.00	0.00	0.03	0.22	0.17	0.17	0.19	0.17	0.22	0.19	0.22	0.31	0.22	0.22	0.22	0.19	0.19
Average trophic level of vertebrates (F)	0.03	0.03	0.03	0.03	0.03	0.22	0.03	0.06	0.19	0.17	0.19	0.03	0.00	0.22	0.17	0.03	0.19	0.08
Average trophic level of catch (F)	0.53	0.50	0.51	0.50	0.39	0.28	0.50	0.47	0.28	0.33	0.28	0.39	0.39	0.28	0.33	0.39	0.33	0.36
Average trophic level of the catch	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Average trophic level of vertebrate catch	0.36	0.36	0.36	0.33	0.17	0.28	0.42	0.47	0.28	0.31	0.31	0.22	0.33	0.28	0.31	0.22	0.33	0.27
Average trophic level of vertebrates	0.00	0.00	0.00	0.00	0.17	0.31	0.22	0.03	0.31	0.22	0.22	0.19	0.08	0.19	0.19	0.19	0.22	0.16
Biomass – bacteria	0.50	0.53	0.51	0.53	0.36	0.69	0.56	0.90	0.72	0.75	0.72	0.36	0.90	0.75	0.75	0.36	0.72	0.57
Biomass – cephalopods	0.50	0.50	0.50	0.44	0.36	0.42	0.53	0.50	0.42	0.42	0.39	0.39	0.53	0.44	0.42	0.33	0.39	0.40
Biomass – demersal fish	0.61	0.56	0.58	0.53	0.03	0.44	0.58	0.67	0.47	0.44	0.44	0.42	0.53	0.47	0.42	0.44	0.42	0.44
Biomass – detritus	0.90	0.86	0.88	0.90	0.83	0.86	0.90	0.90	0.83	0.86	0.81	0.75	0.90	0.83	0.75	0.72	0.78	0.79
Biomass – epifauna	0.36	0.36	0.36	0.31	0.19	0.25	0.39	0.39	0.25	0.22	0.22	0.25	0.39	0.25	0.22	0.22	0.28	0.24
Biomass – gelatinous zooplankton	0.53	0.50	0.51	0.50	0.33	0.25	0.53	0.39	0.25	0.28	0.28	0.39	0.39	0.28	0.22	0.39	0.28	0.35
Biomass – individual microfauna groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.67	0.44	0.56	0.47	0.28	0.00	0.72	0.39	0.00	0.11	0.11	0.25	0.06	0.11	0.03	0.28	0.03	0.24
Biomass – macroalgae	0.03	0.03	0.03	0.03	0.06	0.22	0.03	0.31	0.22	0.22	0.22	0.03	0.22	0.22	0.22	0.06	0.22	0.23
Biomass – mammals	0.56	0.58	0.57	0.53	0.25	0.28	0.25	0.56	0.47	0.47	0.47	0.44	0.58	0.47	0.50	0.44	0.50	0.12
Biomass – piscivores	0.64	0.69	0.67	0.56	0.14	0.33	0.67	0.61	0.28	0.33	0.33	0.33	0.61	0.28	0.36	0.33	0.33	0.44
Biomass – planktivores	0.47	0.44	0.46	0.31	0.03	0.28	0.44	0.56	0.47	0.44	0.44	0.31	0.67	0.50	0.56	0.25	0.44	0.37
Biomass – seabirds	0.42	0.42	0.42	0.36	0.00	0.31	0.42	0.58	0.31	0.33	0.33	0.19	0.36	0.33	0.50	0.19	0.33	0.35
Biomass – seagrass	0.14	0.03	0.08	0.17	0.14	0.03	0.06	0.08	0.03	0.06	0.03	0.11	0.03	0.03	0.03	0.14	0.03	0.28
Biomass – sharks	0.56	0.56	0.56	0.56	0.03	0.42	0.53	0.58	0.47	0.44	0.44	0.31	0.67	0.44	0.47	0.33	0.47	0.08
BSS - slope	0.56	0.53	0.54	0.50	0.39	0.42	0.53	0.00	0.44	0.44	0.42	0.42	0.53	0.42	0.44	0.39	0.42	0.41

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Capacity	0.19	0.47	0.33	0.19	0.00	0.36	0.50	0.47	0.39	0.36	0.36	0.33	0.42	0.39	0.42	0.36	0.36	0.34
Chlorophyll a	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Condition at maturity – seabird	0.31	0.31	0.31	0.25	0.00	0.39	0.36	0.50	0.39	0.39	0.36	0.25	0.47	0.39	0.36	0.22	0.39	0.31
Condition at maturity – top predators	0.90	0.90	0.90	0.90	0.00	0.78	0.90	0.90	0.90	0.90	0.90	0.90	0.90	0.90	0.69	0.90	0.90	0.82
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
CPUE – cephalopods	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05
CPUE – demersal fish	0.00	0.00	0.00	0.03	0.00	0.03	0.11	0.00	0.06	0.11	0.06	0.00	0.11	0.08	0.14	0.03	0.06	0.01
CPUE – epifauna	0.01	0.01	0.01	0.03	0.03	0.00	0.06	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.06
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – piscivores	0.00	0.01	<0.01	0.08	0.00	0.08	0.14	0.00	0.06	0.08	0.06	0.00	0.08	0.08	0.08	0.08	0.06	0.06
CPUE – planktivores	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.06	0.00	0.03
CPUE – sharks	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
CVt	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.02
Denitrification	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.03	0.00	0.18
Detrital dominance	0.06	0.06	0.06	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.06
Discards – demersal fish	0.00	<0.01	<0.01	0.08	0.00	0.11	0.28	0.08	0.11	0.11	0.08	0.08	0.11	0.14	0.11	0.17	0.11	0.20
Discards – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	<0.01	<0.01	0.00	<0.01
Discards – invertebrates	0.00	0.00	0.00	0.17	0.17	0.19	0.33	0.19	0.28	0.19	0.19	0.17	0.19	0.22	0.19	0.19	0.19	0.19
Discards – piscivores	0.00	0.00	0.00	0.00	0.00	0.22	0.50	0.08	0.19	0.22	0.19	0.25	0.00	0.22	0.22	0.19	0.17	0.16
Discards – planktivores	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.06
Dissolved Inorganic Nitrogen	0.03	0.03	0.03	0.00	0.00	0.42	0.00	0.53	0.44	0.44	0.42	0.00	0.56	0.42	0.42	0.00	0.44	0.21
Diversity – count of groups present	0.61	0.58	0.60	0.58	0.50	0.00	0.58	0.06	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.50	0.00	0.28
Diversity – number of communities	0.47	0.47	0.47	0.53	0.42	0.44	0.47	0.47	0.47	0.47	0.44	0.44	0.33	0.44	0.33	0.44	0.47	0.44
Ecotrophic Efficiency – indicator groups	0.14	0.61	0.38	0.28	0.25	0.47	0.56	0.67	0.47	0.53	0.50	0.50	0.58	0.50	0.50	0.50	0.53	0.48
Ecotrophic Efficiency – individual groups	0.02	0.15	0.09	0.02	0.03	0.11	0.15	0.17	0.12	0.12	0.12	0.12	0.13	0.12	0.12	0.11	0.12	0.11
Ecological footprint – demersal fish	0.00	0.00	0.00	0.00	0.00	0.19	0.19	0.17	0.14	0.14	0.22	0.22	0.08	0.14	0.22	0.17	0.22	0.14
Ecological footprint – piscivores	0.00	0.00	0.00	0.00	0.00	0.06	0.08	0.17	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.06	0.05
FIB (F)	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
FIB slope	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
FIB	0.00	0.00	0.00	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24

Indicator	ON1	ON2	ON (mean)	O1	O2	O3	O4	O5	O6	O7	O8	O9	O10	O11	O12	O13	O14	O (mean)
Infauna/Epifauna (biomass ratio)	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Invertebrate:Finfish biomass ratio	0.36	0.50	0.43	0.47	0.33	0.30	0.31	0.29	0.30	0.35	0.34	0.19	0.45	0.36	0.31	0.38	0.29	0.33
Invertebrate:Finfish catch ratio	0.01	0.01	0.01	0.08	0.05	0.08	0.08	0.06	0.05	0.06	0.06	0.05	0.04	0.06	0.05	0.03	0.06	0.06
labile / refract detritus (biomass ratio)	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Light levels on the sediment surface	0.03	0.03	0.03	0.03	0.03	0.42	0.03	0.58	0.44	0.44	0.42	0.03	0.56	0.44	0.47	0.03	0.44	0.23
Maximum length of catch	0.06	0.06	0.06	0.58	0.42	0.44	0.58	0.56	0.47	0.47	0.44	0.44	0.67	0.50	0.56	0.42	0.47	0.49
Net primary production / biomass	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Nitrification	0.00	0.00	0.00	0.00	0.00	0.42	0.00	0.56	0.44	0.44	0.42	0.00	0.56	0.47	0.44	0.00	0.44	0.21
Number of mammals caught during fishing	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.03	0.06	0.03	0.03	0.00	0.06	0.00	0.03	0.00	0.03	0.01
Number of seabirds caught during fishing	0.00	0.00	0.00	0.06	0.00	0.25	0.03	0.19	0.22	0.28	0.31	0.03	0.33	0.33	0.31	0.03	0.28	0.14
Overhead	0.11	0.47	0.29	0.11	0.00	0.00	0.50	0.06	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.36	0.00	0.16
PBR – mammals	0.00	0.00	0.00	0.22	0.25	0.03	0.00	0.08	0.25	0.25	0.28	0.28	0.28	0.33	0.31	0.28	0.31	0.22
PBR – seabirds	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.19	0.25	0.31	0.28	0.00	0.31	0.31	0.28	0.00	0.31	0.13
Pelagic: Demersal (catch biomass ratio)	0.56	0.56	0.56	0.53	0.06	0.25	0.53	0.17	0.22	0.25	0.25	0.06	0.00	0.25	0.28	0.06	0.25	0.18
Pelagic: Demersal (fish biomass ratio)	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Primary Production Required – catch	0.00	0.00	0.00	0.33	0.00	0.31	0.28	0.19	0.28	0.28	0.28	0.25	0.33	0.31	0.25	0.25	0.28	0.23
Primary Production Required – indicator grps	0.47	0.44	0.46	0.42	0.56	0.39	0.53	0.61	0.39	0.39	0.39	0.39	0.47	0.42	0.42	0.42	0.39	0.42
Primary Production Required – individ. grps	0.18	0.16	0.17	0.11	0.10	0.09	0.18	0.21	0.09	0.09	0.09	0.08	0.15	0.12	0.11	0.08	0.09	0.10
Proportion of stock juvenile – demersal fish	0.56	0.56	0.56	0.53	0.42	0.44	0.53	0.64	0.47	0.47	0.44	0.44	0.56	0.44	0.47	0.44	0.47	0.46
Proportion of stock juvenile – piscivores	0.50	0.47	0.49	0.44	0.36	0.39	0.53	0.47	0.39	0.39	0.36	0.36	0.33	0.39	0.33	0.36	0.39	0.37
Proportion of stock juvenile – planktivores	0.44	0.42	0.43	0.36	0.25	0.28	0.44	0.56	0.47	0.44	0.44	0.25	0.67	0.50	0.56	0.25	0.44	0.37
Proportion of the catch juvenile	0.42	0.50	0.46	0.42	0.44	0.08	0.42	0.28	0.08	0.08	0.08	0.39	0.03	0.14	0.14	0.44	0.08	0.27
PS / ZP (catch biomass ratio)	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.69	0.06	0.06	0.06	0.00	0.44	0.06	0.03	0.00	0.06	0.07
PS / ZP (fish biomass ratio)	0.47	0.44	0.46	0.33	0.08	0.00	0.50	0.06	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.08	0.00	0.08
Relative Ascendency	0.06	0.06	0.06	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24

Indicator	ON1	ON2	ON (mean)	O1	O2	O3	O4	O5	O6	O7	O8	O9	O10	O11	O12	O13	O14	O (mean)
Reproductive success of mammals	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Reproductive success of seabirds	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Size at maturity – demersal fish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	<0.01
Size at maturity – piscivores	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.03	0.00	0.01
SOI	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Total benthic consumption	0.50	0.36	0.43	0.44	0.33	0.36	0.47	0.50	0.36	0.33	0.33	0.33	0.44	0.39	0.39	0.33	0.36	0.36
Total benthic production	0.50	0.36	0.43	0.47	0.33	0.36	0.47	0.50	0.39	0.39	0.33	0.33	0.42	0.39	0.39	0.33	0.36	0.37
Total biomass	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.08	0.03	0.03	0.00	0.00	0.00	0.06	0.03	0.00	0.03	0.01
Total biomass – invertebrate (macrofauna)	0.22	0.12	0.17	0.06	0.03	0.14	0.17	0.09	0.12	0.19	0.13	0.08	0.07	0.11	0.15	0.25	0.35	0.14
Total biomass – vertebrate	0.33	0.42	0.38	0.31	0.32	0.39	0.33	0.35	0.33	0.29	0.40	0.33	0.32	0.36	0.32	0.33	0.36	0.34
Total catch – invertebrate (macrofauna)	0.00	0.02	0.01	0.10	0.08	0.06	0.10	0.08	0.05	0.13	0.05	0.09	0.08	0.06	0.07	0.05	0.10	0.08
Total catch – vertebrate	0.00	<0.01	<0.01	0.04	0.06	0.03	0.07	0.07	0.03	0.10	0.03	0.04	0.04	0.05	0.08	0.10	0.09	0.06
Total consumption	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Total consumption - indicator groups	0.03	0.03	0.03	0.19	0.11	0.11	0.06	0.14	0.22	0.28	0.24	0.19	0.21	0.31	0.11	0.23	0.28	0.14
Total consumption - individual groups	<0.01	0.00	<0.01	0.00	<0.01	0.00	0.00	0.01	0.01	0.00	0.01	0.01	0.00	0.01	<0.01	0.00	0.00	<0.01
Total discard rate	0.00	0.00	0.00	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Total flow to detritus	0.00	0.36	0.18	0.08	0.14	0.25	0.42	0.31	0.25	0.22	0.22	0.31	0.28	0.22	0.25	0.33	0.19	0.26
Total mortality – demersal fish	0.00	0.31	0.15	0.25	0.00	0.31	0.25	0.19	0.25	0.31	0.33	0.33	0.31	0.36	0.33	0.31	0.31	0.25
Total mortality – filter feeders	0.44	0.25	0.35	0.44	0.39	0.06	0.42	0.00	0.03	0.03	0.06	0.39	0.03	0.06	0.03	0.39	0.00	0.22
Total mortality – macrozoobenthos	0.50	0.50	0.50	0.44	0.36	0.36	0.47	0.56	0.36	0.36	0.33	0.36	0.42	0.36	0.33	0.33	0.36	0.37
Total mortality – mammals	0.00	0.31	0.15	0.22	0.00	0.00	0.17	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.00	0.14	0.00	0.06
Total mortality – piscivores	0.00	0.08	0.04	0.06	0.00	0.08	0.03	0.19	0.00	0.03	0.00	0.06	0.08	0.08	0.06	0.03	0.08	0.05
Total mortality – planktivores	0.50	0.50	0.50	0.44	0.36	0.36	0.47	0.61	0.39	0.39	0.36	0.36	0.50	0.39	0.42	0.36	0.39	0.39
Total mortality – seabirds	0.00	0.06	0.03	0.03	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.06	0.00	0.02
Total mortality – sharks	0.00	0.08	0.04	0.03	0.00	0.00	0.03	0.06	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	<0.01
Total planktonic production	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.19	0.11	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.03
Total primary production	0.03	0.25	0.14	0.03	0.03	0.33	0.17	0.39	0.31	0.31	0.28	0.25	0.33	0.31	0.31	0.25	0.28	0.23
Total production	0.33	0.25	0.29	0.42	0.56	0.00	0.17	0.06	0.00	0.00	0.00	0.25	0.00	0.00	0.00	0.25	0.00	0.16
Total production - indicator groups	0.03	0.03	0.03	0.19	0.11	0.11	0.06	0.14	0.22	0.28	0.24	0.19	0.21	0.31	0.11	0.23	0.28	0.14
Total production - individual groups	<0.01	0.00	<0.01	0.00	<0.01	0.00	0.00	0.01	0.01	0.00	0.01	0.01	0.00	0.01	<0.01	0.00	0.00	<0.01

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Total residence time	0.06	0.06	0.06	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Total respiration	0.00	0.56	0.28	0.00	0.14	0.19	0.50	0.47	0.19	0.19	0.19	0.39	0.31	0.19	0.19	0.39	0.19	0.29
Total system exports	0.28	0.33	0.31	0.28	0.25	0.22	0.39	0.36	0.19	0.22	0.22	0.19	0.17	0.22	0.25	0.19	0.22	0.22
Total throughput	0.03	0.06	0.04	0.03	0.03	0.28	0.06	0.56	0.47	0.44	0.44	0.03	0.67	0.50	0.56	0.03	0.44	0.24
Trophic efficiency – detritus	0.06	0.53	0.29	0.03	0.22	0.58	0.50	0.28	0.56	0.28	0.22	0.56	0.28	0.22	0.28	0.56	0.22	0.36
Trophic efficiency – production	0.33	0.28	0.31	0.42	0.56	0.00	0.11	0.03	0.00	0.28	0.17	0.31	0.28	0.03	0.25	0.33	0.19	0.21
Trophic spectra – peak	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trophic spectra – slope (F)	0.00	0.00	0.00	0.14	0.06	0.00	0.08	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.00	0.14	0.00	0.07
Trophic spectra – slope	0.00	0.00	0.00	0.06	0.25	0.00	0.14	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.00	0.25	0.00	0.12
w-abc	0.03	0.06	0.04	0.03	0.03	0.00	0.06	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.03	0.00	0.02

Table 4.13b: Normalised linear correlation results for the scenarios O15 to O22 for the coast-to-open-ocean ecosystem analyses, using data aggregated to the level of the management zones identified in Figure 4.4. ON (mean) given here is identical to that in table 4.13a (table split due to space limitations). (F) represents “fixed”.

Indicator	O15	O16	O17	O18	O19	O20	O21	O22	O (mean)
Average trophic level	0.31	0.03	0.31	0.31	0.42	0.31	0.25	0.31	0.19
Average trophic level (F)	0.28	0.42	0.31	0.28	0.39	0.31	0.25	0.31	0.36
Average trophic level of vertebrate catch (F)	0.22	0.22	0.11	0.11	0.08	0.22	0.19	0.22	0.19
Average trophic level of vertebrates (F)	0.03	0.00	0.03	0.03	0.03	0.03	0.03	0.03	0.08
Average trophic level of catch (F)	0.36	0.36	0.39	0.33	0.33	0.39	0.22	0.39	0.36
Average trophic level of the catch	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Average trophic level of vertebrate catch	0.17	0.33	0.22	0.19	0.19	0.22	0.14	0.22	0.27
Average trophic level of vertebrates	0.19	0.22	0.00	0.08	0.00	0.17	0.17	0.14	0.16
Biomass – bacteria	0.36	0.81	0.44	0.31	0.47	0.36	0.28	0.36	0.57
Biomass – cephalopods	0.33	0.44	0.39	0.28	0.33	0.36	0.25	0.33	0.40
Biomass – demersal fish	0.44	0.47	0.44	0.42	0.42	0.47	0.33	0.47	0.44
Biomass – detritus	0.83	0.81	0.67	0.72	0.69	0.81	0.53	0.69	0.80
Biomass – epifauna	0.25	0.17	0.08	0.25	0.17	0.25	0.17	0.22	0.24
Biomass – gelatinous zooplankton	0.42	0.31	0.39	0.36	0.42	0.42	0.22	0.42	0.35
Biomass – individual microfauna groups	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.36	0.00	0.33	0.28	0.42	0.31	0.19	0.03	0.24
Biomass – macroalgae	0.06	0.19	0.00	0.06	0.00	0.03	0.03	0.25	0.23
Biomass – mammals	0.44	0.56	0.44	0.39	0.50	0.44	0.31	0.06	0.12
Biomass – piscivores	0.33	0.44	0.28	0.28	0.39	0.33	0.28	0.47	0.44
Biomass – planktivores	0.28	0.50	0.22	0.14	0.11	0.28	0.22	0.33	0.37
Biomass – seabirds	0.19	0.36	0.17	0.19	0.17	0.19	0.17	0.31	0.35
Biomass – seagrass	0.08	0.00	0.11	0.31	0.11	0.14	0.11	0.25	0.28
Biomass – sharks	0.33	0.47	0.42	0.31	0.33	0.33	0.28	0.06	0.08
BSS - slope	0.42	0.44	0.42	0.36	0.44	0.42	0.25	0.33	0.41
Capacity	0.36	0.42	0.36	0.31	0.42	0.33	0.25	0.11	0.34
Chlorophyll a	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Condition at maturity – seabird	0.25	0.42	0.28	0.22	0.22	0.25	0.22	0.25	0.31
Condition at maturity – top predators	0.90	0.90	0.90	0.53	0.89	0.90	0.69	0.90	0.82
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
CPUE – cephalopods	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.05
CPUE – demersal fish	0.06	0.06	0.03	0.06	0.00	0.08	0.00	0.03	0.01
CPUE – epifauna	0.00	0.06	0.06	0.00	0.00	0.00	0.00	0.08	0.06
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – piscivores	0.06	0.08	0.06	0.03	0.00	0.06	0.06	0.08	0.06
CPUE – planktivores	0.06	0.03	0.03	0.03	0.00	0.06	0.06	0.06	0.03
CPUE – sharks	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
CVt	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.02
Denitrification	0.00	0.00	0.03	0.00	0.03	0.03	0.00	0.25	0.18
Detrital dominance	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.08	0.06
Discards – demersal fish	0.08	0.11	0.08	0.06	0.14	0.08	0.00	0.25	0.20
Discards – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Discards – invertebrates	0.17	0.19	0.17	0.14	0.19	0.17	0.06	0.14	0.19
Discards – piscivores	0.22	0.14	0.22	0.08	0.00	0.17	0.14	0.11	0.16
Discards – planktivores	0.19	0.00	0.03	0.08	0.00	0.17	0.14	0.03	0.06
Dissolved Inorganic Nitrogen	0.00	0.47	0.00	0.00	0.03	0.03	0.03	0.03	0.21

Indicator	O15	O16	O17	O18	O19	O20	O21	O22	O (mean)
Diversity – count of groups present	0.50	0.03	0.50	0.47	0.56	0.50	0.31	0.47	0.28
Diversity – number of communities	0.49	0.36	0.44	0.44	0.50	0.46	0.49	0.33	0.44
Ecotrophic Efficiency – indicator groups	0.50	0.53	0.44	0.42	0.50	0.50	0.36	0.47	0.48
Ecotrophic Efficiency – individual groups	0.11	0.12	0.11	0.10	0.11	0.11	0.08	0.11	0.11
Ecological footprint – demersal fish	0.19	0.06	0.08	0.08	0.19	0.22	0.14	0.00	0.14
Ecological footprint – piscivores	0.06	0.06	0.03	0.03	0.08	0.06	0.06	0.00	0.05
FIB (F)	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.01
FIB slope	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
FIB	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Infauna/Epifauna (biomass ratio)	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Invertebrate:Finfish biomass ratio	0.33	0.33	0.36	0.36	0.29	0.38	0.33	0.28	0.33
Invertebrate:Finfish catch ratio	0.06	0.10	0.05	0.03	0.06	0.09	0.02	0.07	0.06
labile / refract detritus (biomass ratio)	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Light levels on the sediment surface	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.23
Maximum length of catch	0.47	0.50	0.56	0.44	0.50	0.47	0.28	0.44	0.49
Net primary production / biomass	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Nitrification	0.00	0.47	0.03	0.00	0.00	0.00	0.00	0.03	0.21
Number of mammals caught during fishing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
Number of seabirds caught during fishing	0.03	0.31	0.00	0.03	0.03	0.03	0.03	0.03	0.14
Overhead	0.36	0.03	0.33	0.31	0.42	0.33	0.25	0.06	0.16
PBR – mammals	0.28	0.31	0.00	0.14	0.28	0.28	0.14	0.19	0.22
PBR – seabirds	0.00	0.28	0.00	0.00	0.03	0.00	0.00	0.00	0.13
Pelagic: Demersal (catch biomass ratio)	0.06	0.03	0.00	0.06	0.44	0.06	0.03	0.06	0.18
Pelagic: Demersal (fish biomass ratio)	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Primary Production Required – catch	0.25	0.25	0.00	0.17	0.19	0.25	0.19	0.25	0.23
Primary Production Required – indicator groups	0.42	0.39	0.33	0.39	0.39	0.39	0.33	0.39	0.42
Primary Production Required – individual groups	0.08	0.08	0.02	0.08	0.15	0.08	0.06	0.08	0.10
Proportion of stock juvenile – demersal fish	0.44	0.50	0.42	0.39	0.42	0.44	0.33	0.47	0.46
Proportion of stock juvenile – piscivores	0.36	0.39	0.36	0.31	0.33	0.36	0.25	0.33	0.37
Proportion of stock juvenile – planktivores	0.25	0.50	0.25	0.19	0.25	0.25	0.19	0.25	0.37
Proportion of the catch juvenile	0.42	0.06	0.25	0.31	0.56	0.44	0.33	0.44	0.27
PS / ZP (catch biomass ratio)	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.07
PS / ZP (fish biomass ratio)	0.08	0.03	0.08	0.08	0.11	0.08	0.06	0.08	0.08
Relative Ascendency	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Reproductive success of mammals	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Reproductive success of seabirds	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Size at maturity – demersal fish	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Size at maturity – piscivores	0.03	0.00	0.00	0.03	0.06	0.03	0.03	0.03	0.01
SOI	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Total benthic consumption	0.36	0.36	0.39	0.39	0.31	0.33	0.22	0.28	0.36
Total benthic production	0.36	0.36	0.36	0.39	0.33	0.33	0.22	0.28	0.37
Total biomass	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
Total biomass – invertebrate (macrofauna)	0.12	0.16	0.20	0.09	0.12	0.19	0.16	0.08	0.14
Total biomass – vertebrate	0.33	0.35	0.43	0.33	0.37	0.35	0.25	0.31	0.34
Total catch – invertebrate (macrofauna)	0.11	0.11	0.12	0.04	0.08	0.03	0.05	0.05	0.08
Total catch – vertebrate	0.04	0.10	0.07	0.05	0.03	0.02	0.06	0.08	0.06
Total consumption	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Total consumption - indicator groups	0.11	0.11	0.00	0.05	0.08	0.03	0.03	0.08	0.14

Indicator	O15	O16	O17	O18	O19	O20	O21	O22	O (mean)
Total consumption - individual groups	0.00	0.01	0.00	0.00	0.00	0.01	<0.01	0.00	<0.01
Total discard rate	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Total flow to detritus	0.33	0.25	0.22	0.28	0.22	0.31	0.28	0.39	0.26
Total mortality – demersal fish	0.31	0.28	0.25	0.22	0.19	0.00	0.19	0.25	0.25
Total mortality – filter feeders	0.39	0.06	0.33	0.36	0.36	0.36	0.25	0.33	0.22
Total mortality – macrozoobenthos	0.36	0.42	0.39	0.31	0.39	0.36	0.22	0.36	0.37
Total mortality – mammals	0.14	0.00	0.17	0.11	0.14	0.00	0.06	0.11	0.06
Total mortality – piscivores	0.00	0.00	0.03	0.11	0.11	0.00	0.00	0.00	0.05
Total mortality – planktivores	0.36	0.42	0.39	0.31	0.44	0.36	0.22	0.36	0.39
Total mortality – seabirds	0.06	0.00	0.08	0.03	0.03	0.00	0.00	0.06	0.02
Total mortality – sharks	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	<0.01
Total planktonic production	0.00	0.00	0.00	0.14	0.00	0.03	0.00	0.03	0.03
Total primary production	0.28	0.39	0.00	0.17	0.22	0.31	0.22	0.03	0.23
Total production	0.28	0.03	0.17	0.17	0.22	0.31	0.22	0.36	0.16
Total production - indicator groups	0.11	0.11	0.00	0.05	0.08	0.03	0.03	0.08	0.14
Total production - individual groups	0.00	0.01	0.00	0.00	0.00	0.01	<0.01	0.00	<0.01
Total residence time	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Total respiration	0.39	0.22	0.36	0.33	0.47	0.39	0.28	0.39	0.29
Total system exports	0.19	0.22	0.19	0.17	0.22	0.19	0.17	0.19	0.22
Total throughput	0.03	0.50	0.03	0.03	0.03	0.03	0.03	0.03	0.24
Trophic efficiency – detritus	0.56	0.22	0.00	0.44	0.50	0.56	0.39	0.53	0.36
Trophic efficiency – production	0.28	0.22	0.00	0.17	0.22	0.31	0.22	0.36	0.21
Trophic spectra – peak	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trophic spectra – slope (F)	0.14	0.00	0.14	0.19	0.17	0.14	0.11	0.08	0.07
Trophic spectra – slope	0.25	0.00	0.25	0.28	0.31	0.25	0.19	0.11	0.12
w-abc	0.03	0.00	0.03	0.00	0.06	0.03	0.03	0.06	0.02

Table 4.14a: Normalised linear correlation results for the scenarios ON1, ON2, and O1 to O14 for the coast-to-open-ocean ecosystem analyses, using data aggregated across the entire system. ON (mean) given here is identical to that in table 4.14b (table split due to space limitations). (F) represents "fixed".

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Average trophic level	0.61	0.56	0.58	0.61	0.61	0.00	0.64	0.03	0.00	0.00	0.00	0.58	0.00	0.03	0.00	0.58	0.00	0.31
Average trophic level (F)	0.58	0.61	0.60	0.47	0.61	0.00	0.53	0.00	0.00	0.00	0.00	0.53	0.00	0.03	0.00	0.61	0.00	0.28
Average trophic level of vertebrate catch (F)	0.00	0.00	0.00	0.00	0.36	0.58	0.19	0.06	0.61	0.58	0.56	0.31	0.58	0.47	0.31	0.36	0.58	0.35
Average trophic level of vertebrates (F)	0.08	0.08	0.08	0.11	0.08	0.08	0.08	0.03	0.06	0.06	0.06	0.08	0.03	0.08	0.08	0.08	0.06	0.07
Average trophic level of catch (F)	0.00	0.00	0.00	0.00	0.00	0.56	0.56	0.86	0.58	0.58	0.53	0.00	0.58	0.58	0.56	0.00	0.58	0.30
Average trophic level of the catch	0.00	0.00	0.00	0.00	0.00	0.03	0.47	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.08
Average trophic level of vertebrate catch	0.00	0.00	0.00	0.06	0.53	0.42	0.03	0.00	0.44	0.42	0.42	0.58	0.50	0.53	0.47	0.53	0.58	0.44
Average trophic level of vertebrates	0.06	0.06	0.06	0.06	0.08	0.47	0.06	0.00	0.44	0.39	0.39	0.06	0.03	0.39	0.50	0.08	0.36	0.17
Biomass – bacteria	0.33	0.33	0.33	0.33	0.22	0.31	0.31	0.56	0.31	0.31	0.31	0.31	0.33	0.00	0.31	0.31	0.31	0.30
Biomass – cephalopods	0.00	0.00	0.00	0.00	0.00	0.72	0.00	0.53	0.72	0.69	0.72	0.00	0.69	0.00	0.67	0.00	0.75	0.28
Biomass – demersal fish	0.67	0.67	0.67	0.61	0.75	0.75	0.61	0.75	0.75	0.75	0.78	0.72	0.67	0.75	0.75	0.72	0.75	0.70
Biomass – detritus	0.90	0.90	0.90	0.90	0.31	0.90	0.90	0.50	0.90	0.90	0.90	0.86	0.90	0.00	0.90	0.90	0.90	0.80
Biomass – epifauna	0.64	0.64	0.64	0.58	0.72	0.00	0.61	0.17	0.00	0.00	0.00	0.75	0.00	0.03	0.00	0.72	0.00	0.37
Biomass – gelatinous zooplankton	0.72	0.69	0.71	0.64	0.28	0.14	0.67	0.00	0.11	0.14	0.06	0.75	0.14	0.00	0.11	0.69	0.14	0.39
Biomass – individual microfauna groups	0.00	0.01	<0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.06	0.22	0.14	0.22	0.00	0.06	0.17	0.03	0.03	0.03	0.03	0.25	0.03	0.00	0.03	0.19	0.06	0.10
Biomass – macroalgae	0.00	0.44	0.22	0.00	0.03	0.75	0.00	0.72	0.78	0.72	0.78	0.00	0.72	0.00	0.69	0.00	0.75	0.32
Biomass – mammals	0.14	0.14	0.14	0.11	0.22	0.64	0.56	0.75	0.19	0.19	0.19	0.25	0.17	0.19	0.11	0.22	0.19	0.29
Biomass – piscivores	0.53	0.53	0.53	0.47	0.69	0.75	0.56	0.69	0.75	0.72	0.78	0.72	0.61	0.75	0.67	0.72	0.75	0.68
Biomass – planktivores	0.03	0.03	0.03	0.50	0.64	0.67	0.06	0.78	0.61	0.64	0.61	0.64	0.53	0.67	0.67	0.61	0.64	0.57
Biomass – seabirds	0.08	0.08	0.08	0.11	0.08	0.69	0.08	0.81	0.67	0.64	0.64	0.08	0.17	0.69	0.67	0.08	0.67	0.33
Biomass – seagrass	0.03	0.67	0.35	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.06
Biomass – sharks	0.64	0.67	0.65	0.61	0.72	0.75	0.64	0.81	0.75	0.75	0.78	0.72	0.67	0.75	0.75	0.72	0.75	0.70
BSS – slope	0.69	0.69	0.69	0.58	0.72	0.78	0.64	0.28	0.75	0.75	0.72	0.75	0.64	0.72	0.72	0.72	0.75	0.65

Indicator	ON1	ON2	ON (mean)	O1	O2	O3	O4	O5	O6	O7	O8	O9	O10	O11	O12	O13	O14	O (mean)
Capacity	0.69	0.69	0.69	0.58	0.00	0.17	0.64	0.69	0.25	0.00	0.00	0.78	0.19	0.00	0.00	0.72	0.00	0.41
Chlorophyll a	0.00	0.00	0.00	0.03	0.00	0.03	0.00	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.06
Condition at maturity – seabird	0.11	0.06	0.08	0.11	0.08	0.14	0.03	0.67	0.00	0.11	0.14	0.06	0.75	0.19	0.17	0.06	0.08	0.14
Condition at maturity – top predators	0.89	0.78	0.83	0.90	0.90	0.90	0.78	0.90	0.06	0.90	0.90	0.90	0.90	0.90	0.90	0.90	0.90	0.85
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – cephalopods	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – demersal fish	0.00	0.00	0.00	0.11	0.17	0.00	0.31	0.33	0.33	0.19	0.33	0.00	0.19	0.33	0.33	0.06	0.19	0.15
CPUE – epifauna	0.01	0.00	<0.01	0.03	0.14	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – piscivores	0.07	0.07	0.07	0.14	0.14	0.14	0.00	0.11	0.19	0.17	0.19	0.19	0.11	0.19	0.19	0.17	0.19	0.15
CPUE – planktivores	0.00	0.00	0.00	0.00	0.17	0.14	0.19	0.17	0.17	0.19	0.17	0.17	0.00	0.17	0.17	0.17	0.17	0.14
CPUE – sharks	0.01	0.01	0.01	0.14	0.11	0.00	0.11	0.03	0.14	0.00	0.08	0.00	0.03	0.00	0.03	0.00	0.00	0.03
CVt	0.08	0.08	0.08	0.22	0.72	0.03	0.64	0.78	0.06	0.06	0.03	0.75	0.08	0.08	0.06	0.75	0.03	0.40
Denitrification	0.28	0.28	0.28	0.28	0.28	0.22	0.25	0.72	0.22	0.22	0.22	0.28	0.14	0.22	0.22	0.25	0.22	0.27
Detrital dominance	0.53	0.00	0.26	0.75	0.83	0.03	0.00	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.17
Discards – demersal fish	0.00	0.00	0.00	0.03	0.03	0.03	0.11	0.56	0.03	0.03	0.03	0.03	0.06	0.00	0.03	0.03	0.03	0.06
Discards – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	<0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Discards – invertebrates	0.00	0.01	<0.01	0.11	0.08	0.08	0.67	0.25	0.11	0.11	0.14	0.08	0.08	0.06	0.08	0.11	0.11	0.13
Discards – piscivores	0.00	0.00	0.00	0.08	0.08	0.42	0.67	0.58	0.44	0.39	0.39	0.08	0.03	0.44	0.53	0.11	0.39	0.24
Discards – planktivores	0.00	0.00	0.00	0.06	0.39	0.42	0.11	0.19	0.36	0.39	0.39	0.39	0.03	0.42	0.47	0.33	0.25	0.28
Dissolved Inorganic Nitrogen	0.36	0.00	0.18	0.25	0.28	0.28	0.31	0.72	0.28	0.28	0.28	0.03	0.31	0.33	0.31	0.00	0.28	0.23
Diversity – count of groups present	0.08	0.08	0.08	0.11	0.08	0.06	0.08	0.69	0.06	0.06	0.06	0.08	0.08	0.11	0.06	0.08	0.06	0.11
Diversity – number of communities	0.47	0.42	0.45	0.44	0.51	0.22	0.47	0.47	0.44	0.28	0.47	0.22	0.22	0.30	0.28	0.42	0.31	0.36
Ecotrophic Efficiency – indicator groups	0.75	0.72	0.74	0.75	0.83	0.83	0.72	0.78	0.83	0.83	0.78	0.86	0.78	0.72	0.78	0.83	0.83	0.78
Ecotrophic Efficiency – individual groups	0.17	0.17	0.17	0.09	0.12	0.12	0.17	0.20	0.12	0.12	0.12	0.12	0.10	0.11	0.23	0.12	0.11	0.12
Ecological footprint – demersal fish	0.00	0.00	0.00	0.00	0.08	0.53	0.56	0.64	0.53	0.44	0.61	0.58	0.03	0.56	0.58	0.44	0.53	0.46
Ecological footprint – piscivores	0.00	0.00	0.00	0.00	0.00	0.39	0.42	0.69	0.42	0.69	0.67	0.64	0.17	0.14	0.28	0.67	0.67	0.38
FIB (F)	0.00	0.00	0.00	0.00	0.00	0.00	0.58	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.03
FIB slope	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	<0.01
FIB	0.00	0.00	0.00	0.00	0.00	0.03	0.58	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.08

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Infauna/Epifauna (biomass ratio)	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.06
Invertebrate:Finfish biomass ratio	0.56	0.58	0.57	0.56	0.40	0.53	0.50	0.49	0.53	0.47	0.56	0.55	0.53	0.44	0.39	0.56	0.52	0.50
Invertebrate:Finfish catch ratio	0.01	0.01	0.01	0.01	<0.01	0.00	0.01	0.02	0.00	0.01	0.00	<0.01	0.01	0.00	0.01	0.00	<0.01	0.01
labile / refract detritus (biomass ratio)	0.39	0.31	0.35	0.11	0.00	0.03	0.50	0.78	0.06	0.06	0.03	0.28	0.08	0.06	0.06	0.00	0.03	0.14
Light levels on the sediment surface	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Maximum length of catch	0.00	0.00	0.00	0.00	0.00	0.03	0.61	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.09
Net primary production / biomass	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.06
Nitrification	0.14	0.11	0.13	0.17	0.22	0.00	0.14	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.00	0.22	0.00	0.11
Number of mammals caught during fishing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Number of seabirds caught during fishing	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.03	0.17	0.36	0.11	0.00	0.53	0.08	0.14	0.00	0.31	0.10
Overhead	0.00	0.28	0.14	0.53	0.00	0.25	0.19	0.69	0.25	0.00	0.00	0.00	0.22	0.03	0.00	0.25	0.00	0.14
PBR – mammals	0.00	0.00	0.00	0.03	0.03	0.08	0.03	0.06	0.06	0.06	0.06	0.03	0.06	0.03	0.06	0.03	0.06	0.04
PBR – seabirds	0.00	0.00	0.00	0.06	0.06	0.58	0.06	0.03	0.56	0.56	0.56	0.06	0.03	0.50	0.53	0.06	0.56	0.23
Pelagic: Demersal (catch biomass ratio)	0.00	0.00	0.00	0.44	0.42	0.67	0.61	0.72	0.64	0.64	0.61	0.56	0.64	0.69	0.67	0.42	0.64	0.53
Pelagic: Demersal (fish biomass ratio)	0.58	0.61	0.60	0.61	0.61	0.03	0.61	0.78	0.06	0.06	0.03	0.61	0.08	0.08	0.06	0.61	0.03	0.37
Primary Production Required – catch	0.00	0.00	0.00	0.61	0.61	0.67	0.33	0.56	0.64	0.56	0.61	0.61	0.53	0.69	0.61	0.61	0.56	0.56
Primary Production Required – indicator grps	0.69	0.69	0.69	0.67	0.75	0.78	0.67	0.78	0.75	0.75	0.78	0.75	0.69	0.75	0.75	0.75	0.78	0.73
Primary Production Required – individ. grps	0.28	0.28	0.28	0.19	0.27	0.29	0.26	0.25	0.30	0.30	0.29	0.27	0.21	0.33	0.29	0.27	0.30	0.25
Proportion of stock juvenile – demersal fish	0.00	0.03	0.01	0.50	0.72	0.75	0.03	0.69	0.75	0.75	0.47	0.72	0.58	0.75	0.64	0.72	0.75	0.64
Proportion of stock juvenile – piscivores	0.19	0.08	0.14	0.17	0.47	0.61	0.06	0.75	0.61	0.53	0.03	0.08	0.03	0.61	0.64	0.08	0.53	0.28
Proportion of stock juvenile – planktivores	0.53	0.08	0.31	0.53	0.58	0.50	0.03	0.78	0.53	0.14	0.03	0.53	0.50	0.47	0.06	0.50	0.08	0.36
Proportion of the catch juvenile	0.69	0.67	0.68	0.64	0.75	0.81	0.58	0.83	0.78	0.72	0.19	0.75	0.61	0.75	0.69	0.67	0.75	0.67
PS / ZP (catch biomass ratio)	0.00	0.00	0.00	0.00	0.00	0.36	0.42	0.00	0.50	0.50	0.44	0.00	0.47	0.28	0.47	0.00	0.47	0.20
PS / ZP (fish biomass ratio)	0.28	0.33	0.31	0.08	0.61	0.67	0.11	0.67	0.64	0.64	0.61	0.58	0.67	0.67	0.67	0.58	0.64	0.55
Relative Ascendency	0.00	0.28	0.14	0.50	0.00	0.03	0.25	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.28	0.03	0.12

Indicator	ON1	ON2	ON (mean)	O1	O2	O3	O4	O5	O6	O7	O8	O9	O10	O11	O12	O13	O14	O (mean)
Reproductive success of mammals	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.08	0.00	0.03	0.00	0.00	0.00	<0.01
Reproductive success of seabirds	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.78	0.06	0.06	0.03	0.00	0.08	0.08	0.06	0.00	0.03	0.06
Size at maturity – demersal fish	0.53	0.06	0.29	0.00	0.19	0.03	0.39	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.09
Size at maturity – piscivores	0.08	0.08	0.08	0.08	0.72	0.75	0.08	0.33	0.00	0.69	0.69	0.72	0.08	0.69	0.69	0.67	0.72	0.46
SOI	0.08	0.08	0.08	0.11	0.75	0.06	0.08	0.78	0.06	0.06	0.03	0.75	0.08	0.08	0.06	0.75	0.03	0.36
Total benthic consumption	0.69	0.69	0.69	0.67	0.78	0.81	0.67	0.78	0.78	0.81	0.78	0.81	0.75	0.81	0.78	0.78	0.81	0.75
Total benthic production	0.69	0.69	0.69	0.67	0.78	0.81	0.64	0.78	0.78	0.78	0.78	0.78	0.72	0.81	0.78	0.75	0.81	0.74
Total biomass	0.50	0.50	0.50	0.50	0.58	0.00	0.58	0.00	0.00	0.00	0.00	0.56	0.00	0.00	0.00	0.56	0.00	0.27
Total biomass – invertebrate (macrofauna)	0.53	0.58	0.56	0.56	0.31	0.53	0.30	0.44	0.29	0.47	0.33	0.32	0.29	0.31	0.41	0.38	0.39	0.38
Total biomass – vertebrate	0.11	0.06	0.08	0.50	0.36	0.56	0.60	0.44	0.59	0.55	0.48	0.51	0.46	0.44	0.49	0.50	0.59	0.51
Total catch – invertebrate (macrofauna)	0.00	<0.01	<0.01	0.00	0.00	0.00	0.00	0.01	<0.01	0.00	0.00	<0.01	0.01	0.00	0.00	0.00	<0.01	<0.01
Total catch – vertebrate	0.01	<0.01	<0.01	0.00	0.00	0.00	<0.01	0.00	0.01	0.01	0.00	0.00	<0.01	0.00	0.00	0.01	0.01	<0.01
Total consumption	0.67	0.69	0.68	0.61	0.72	0.03	0.64	0.78	0.06	0.06	0.03	0.78	0.08	0.08	0.06	0.72	0.03	0.43
Total consumption - indicator groups	0.11	0.03	0.07	0.04	0.04	0.09	0.03	0.03	0.11	0.36	0.11	0.04	0.25	0.18	0.14	0.18	0.31	0.10
Total consumption - individual groups	0.00	0.01	<0.01	0.00	<0.01	0.00	0.01	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Total discard rate	0.00	0.00	0.00	0.00	0.06	0.03	0.42	0.78	0.06	0.06	0.03	0.03	0.08	0.08	0.06	0.03	0.03	0.10
Total flow to detritus	0.03	0.69	0.36	0.03	0.03	0.78	0.64	0.75	0.75	0.78	0.72	0.78	0.69	0.81	0.75	0.72	0.78	0.59
Total mortality – demersal fish	0.03	0.03	0.03	0.39	0.53	0.58	0.03	0.36	0.50	0.50	0.61	0.58	0.53	0.50	0.50	0.56	0.58	0.45
Total mortality – filter feeders	0.64	0.67	0.65	0.64	0.67	0.64	0.58	0.69	0.58	0.61	0.61	0.56	0.61	0.67	0.67	0.64	0.64	0.60
Total mortality – macrozoobenthos	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – mammals	0.03	0.03	0.03	0.03	0.03	0.39	0.39	0.03	0.03	0.03	0.03	0.03	0.03	0.00	0.03	0.03	0.03	0.06
Total mortality – piscivores	0.03	0.03	0.03	0.14	0.36	0.42	0.03	0.31	0.36	0.39	0.42	0.33	0.14	0.28	0.47	0.25	0.39	0.28
Total mortality – planktivores	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.08	0.03	0.03	0.06	0.03	0.25	0.06	0.03	0.03	0.04
Total mortality – seabirds	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.00	0.03	0.03	0.03	0.03
Total mortality – sharks	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.39	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.05
Total planktonic production	0.08	0.19	0.14	0.08	0.08	0.69	0.08	0.03	0.03	0.03	0.03	0.06	0.11	0.06	0.08	0.06	0.03	0.10
Total primary production	0.67	0.69	0.68	0.67	0.78	0.78	0.64	0.72	0.75	0.78	0.72	0.78	0.69	0.81	0.72	0.78	0.78	0.70
Total production	0.67	0.69	0.68	0.58	0.72	0.78	0.64	0.72	0.78	0.78	0.72	0.75	0.69	0.81	0.72	0.72	0.78	0.71
Total production - indicator groups	0.11	0.03	0.07	0.04	0.04	0.09	0.03	0.03	0.11	0.36	0.11	0.04	0.25	0.18	0.14	0.18	0.31	0.10
Total production - individual groups	0.00	0.01	<0.01	0.00	<0.01	0.00	0.01	0.00	0.00	0.00	<0.01	0.00	0.00	0.00	0.00	0.00	0.00	<0.01

Indicator	**ON1**	**ON2**	**ON (mean)**	**O1**	**O2**	**O3**	**O4**	**O5**	**O6**	**O7**	**O8**	**O9**	**O10**	**O11**	**O12**	**O13**	**O14**	**O (mean)**
Total residence time	0.72	0.69	0.71	0.00	0.06	0.03	0.69	0.78	0.06	0.06	0.03	0.83	0.08	0.08	0.06	0.81	0.03	0.37
Total respiration	0.67	0.69	0.68	0.61	0.72	0.78	0.64	0.03	0.78	0.75	0.72	0.78	0.69	0.00	0.72	0.72	0.75	0.65
Total system exports	0.00	0.69	0.35	0.03	0.03	0.72	0.64	0.17	0.67	0.69	0.64	0.75	0.56	0.69	0.67	0.72	0.67	0.56
Total throughput	0.64	0.69	0.67	0.58	0.72	0.03	0.64	0.78	0.06	0.06	0.03	0.78	0.08	0.08	0.06	0.72	0.03	0.40
Trophic efficiency – detritus	0.72	0.00	0.36	0.69	0.75	0.19	0.00	0.83	0.53	0.53	0.47	0.00	0.33	0.56	0.64	0.00	0.53	0.34
Trophic efficiency – production	0.19	0.14	0.17	0.33	0.61	0.75	0.25	0.69	0.72	0.78	0.72	0.53	0.69	0.81	0.72	0.56	0.78	0.55
Trophic spectra – peak	0.00	0.00	0.00	0.28	0.50	0.00	0.61	0.00	0.00	0.00	0.00	0.44	0.00	0.00	0.00	0.50	0.00	0.24
Trophic spectra – slope (F)	0.00	0.00	0.00	0.31	0.28	0.25	0.61	0.69	0.25	0.25	0.25	0.28	0.28	0.25	0.28	0.28	0.28	0.30
Trophic spectra – slope	0.00	0.00	0.00	0.53	0.50	0.17	0.61	0.72	0.19	0.5	0.42	0.50	0.44	0.47	0.31	0.50	0.50	0.46
w-abc	0.39	0.36	0.38	0.03	0.03	0.53	0.00	0.33	0.44	0.42	0.42	0.00	0.39	0.50	0.47	0.00	0.42	0.22

Table 4.14b: Normalised linear correlation results for the scenarios O15 to O22 for the coast-to-open-ocean ecosystem analyses, using data aggregated across the entire system. ON (mean) given here is identical to that in table 4.14a (table split due to space limitations). (F) represents "fixed".

Indicator	**O15**	**O16**	**O17**	**O18**	**O19**	**O20**	**O21**	**O22**	**O (mean)**
Average trophic level	0.58	0.00	0.58	0.42	0.67	0.58	0.42	0.50	0.31
Average trophic level (F)	0.39	0.00	0.39	0.33	0.58	0.64	0.47	0.58	0.28
Average trophic level of vertebrate catch (F)	0.33	0.61	0.03	0.22	0.00	0.39	0.25	0.36	0.35
Average trophic level of vertebrates (F)	0.08	0.03	0.08	0.06	0.11	0.08	0.11	0.08	0.07
Average trophic level of catch (F)	0.00	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.30
Average trophic level of the catch	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.08
Average trophic level of vertebrate catch	0.58	0.58	0.53	0.42	0.58	0.58	0.47	0.50	0.44
Average trophic level of vertebrates	0.08	0.00	0.03	0.06	0.03	0.08	0.08	0.03	0.17
Biomass – bacteria	0.31	0.31	0.33	0.28	0.39	0.31	0.19	0.31	0.30
Biomass – cephalopods	0.00	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.28
Biomass – demersal fish	0.72	0.69	0.56	0.53	0.67	0.72	0.61	0.75	0.70
Biomass – detritus	0.90	0.90	0.90	0.81	0.90	0.90	0.78	0.90	0.80
Biomass – epifauna	0.72	0.00	0.58	0.56	0.64	0.72	0.61	0.67	0.37
Biomass – gelatinous zooplankton	0.69	0.06	0.64	0.53	0.78	0.69	0.58	0.72	0.39
Biomass – individual microfauna groups	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	<0.01
Biomass – large zooplankton	0.19	0.06	0.14	0.17	0.14	0.14	0.19	0.14	0.10
Biomass – macroalgae	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.53	0.32
Biomass – mammals	0.22	0.33	0.31	0.19	0.56	0.22	0.22	0.28	0.29
Biomass – piscivores	0.72	0.69	0.58	0.53	0.69	0.72	0.61	0.75	0.68
Biomass – planktivores	0.61	0.36	0.36	0.47	0.75	0.61	0.56	0.61	0.57
Biomass – seabirds	0.06	0.58	0.06	0.08	0.08	0.08	0.06	0.08	0.33
Biomass – seagrass	0.00	0.00	0.44	0.00	0.03	0.03	0.00	0.69	0.06
Biomass – sharks	0.72	0.69	0.61	0.53	0.67	0.72	0.61	0.75	0.70
BSS – slope	0.72	0.69	0.64	0.00	0.67	0.72	0.61	0.69	0.65
Capacity	0.72	0.28	0.67	0.58	0.69	0.75	0.61	0.69	0.41
Chlorophyll a	0.00	0.06	0.03	0.00	0.03	0.00	0.00	0.06	0.06
Condition at maturity – seabird	0.06	0.00	0.03	0.06	0.06	0.06	0.06	0.08	0.14
Condition at maturity – top predators	0.90	0.69	0.90	0.90	0.90	0.90	0.90	0.90	0.85
Condition at maturity – small bodied groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – cephalopods	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – demersal fish	0.00	0.14	0.00	0.00	0.06	0.00	0.00	0.11	0.15
CPUE – epifauna	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.03	0.01
CPUE – individual groups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPUE – piscivores	0.14	0.14	0.03	0.14	0.19	0.14	0.14	0.19	0.15
CPUE – planktivores	0.17	0.14	0.11	0.11	0.00	0.17	0.14	0.17	0.14
CPUE – sharks	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.03
CPUE – demersal fish	0.44	0.06	0.61	0.58	0.72	0.72	0.58	0.75	0.40
CPUE – epifauna	0.28	0.25	0.28	0.25	0.28	0.28	0.19	0.31	0.27
CPUE – piscivores	0.00	0.06	0.03	0.00	0.00	0.00	0.00	0.81	0.17
CPUE – planktivores	0.03	0.03	0.03	0.03	0.06	0.03	0.03	0.03	0.06
CPUE – sharks	0.58	0.00	0.58	0.42	0.67	0.58	0.42	0.50	0.31
Discards – demersal fish	0.39	0.00	0.39	0.33	0.58	0.64	0.47	0.58	0.28

Indicator	**O15**	**O16**	**O17**	**O18**	**O19**	**O20**	**O21**	**O22**	**O (mean)**
Discards – invertebrates	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Discards – piscivores	0.11	0.08	0.17	0.08	0.14	0.11	0.06	0.11	0.13
Discards – individual groups	0.08	0.14	0.08	0.03	0.14	0.08	0.06	0.11	0.24
Discards – planktivores	0.36	0.11	0.36	0.22	0.00	0.33	0.25	0.42	0.28
Dissolved Inorganic Nitrogen	0.00	0.28	0.25	0.00	0.00	0.00	0.19	0.39	0.23
Diversity – count of groups present	0.08	0.06	0.08	0.06	0.11	0.08	0.08	0.11	0.11
Diversity – number of communities	0.28	0.44	0.47	0.44	0.30	0.44	0.44	0.28	0.36
Ecotrophic Efficiency – indicator groups	0.83	0.75	0.72	0.64	0.81	0.83	0.67	0.75	0.78
Ecotrophic Efficiency – individual groups	0.11	0.10	0.10	0.09	0.10	0.11	0.09	0.14	0.12
Ecological footprint – demersal fish	0.42	0.56	0.64	0.39	0.78	0.53	0.61	0.03	0.46
Ecological footprint – piscivores	0.61	0.08	0.31	0.19	0.17	0.64	0.56	0.06	0.38
FIB (F)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.03
FIB slope	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
FIB	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.03	0.08
Infauna/Epifauna (biomass ratio)	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.03	0.06
Invertebrate:Finfish biomass ratio	0.54	0.55	0.52	0.57	0.53	0.50	0.32	0.47	0.50
Invertebrate:Finfish catch ratio	<0.01	<0.01	0.00	<0.01	0.00	0.03	0.04	0.02	0.01
labile / refract detritus (biomass ratio)	0.33	0.06	0.25	0.06	0.03	0.08	0.14	0.08	0.14
Light levels on the sediment surface	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Maximum length of catch	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.03	0.09
Net primary production / biomass	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.03	0.06
Nitrification	0.22	0.00	0.31	0.19	0.28	0.22	0.14	0.08	0.11
Number of mammals caught during fishing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Number of seabirds caught during fishing	0.00	0.42	0.00	0.00	0.00	0.00	0.00	0.00	0.10
Overhead	0.25	0.28	0.06	0.00	0.00	0.00	0.00	0.17	0.14
PBR – mammals	0.03	0.06	0.00	0.03	0.03	0.03	0.03	0.03	0.04
PBR – seabirds	0.06	0.50	0.00	0.06	0.06	0.06	0.06	0.06	0.23
Pelagic: Demersal (catch biomass ratio)	0.47	0.56	0.44	0.31	0.56	0.53	0.00	0.39	0.53
Pelagic: Demersal (fish biomass ratio)	0.61	0.06	0.31	0.44	0.69	0.61	0.53	0.64	0.37
Primary Production Required – catch	0.61	0.53	0.00	0.44	0.69	0.61	0.56	0.64	0.56
Primary Production Required – indicator groups	0.75	0.72	0.67	0.61	0.72	0.78	0.61	0.75	0.73
Primary Production Required – individual groups	0.27	0.22	0.05	0.17	0.27	0.27	0.22	0.31	0.25
Proportion of stock juvenile – demersal fish	0.72	0.67	0.56	0.53	0.61	0.72	0.61	0.75	0.64
Proportion of stock juvenile – piscivores	0.06	0.00	0.11	0.06	0.03	0.08	0.08	0.50	0.28
Proportion of stock juvenile – planktivores	0.50	0.06	0.11	0.36	0.03	0.50	0.42	0.64	0.36
Proportion of the catch juvenile	0.67	0.72	0.61	0.50	0.61	0.69	0.58	0.75	0.67
PS / ZP (catch biomass ratio)	0.00	0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.20
PS / ZP (fish biomass ratio)	0.58	0.56	0.31	0.39	0.64	0.58	0.53	0.58	0.55
Relative Ascendency	0.28	0.06	0.08	0.00	0.00	0.00	0.00	0.03	0.12
Reproductive success of mammals	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Reproductive success of seabirds	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.03	0.06
Size at maturity – demersal fish	0.00	0.03	0.00	0.00	0.53	0.00	0.00	0.00	0.09
Size at maturity – piscivores	0.67	0.00	0.00	0.50	0.06	0.67	0.58	0.67	0.46
Size at maturity – small bodied groups	0.72	0.06	0.03	0.58	0.69	0.75	0.61	0.72	0.36
SOI	0.78	0.72	0.67	0.58	0.72	0.78	0.64	0.75	0.75
Total benthic consumption	0.78	0.72	0.67	0.56	0.72	0.75	0.64	0.75	0.74
Total benthic production	0.56	0.00	0.00	0.42	0.53	0.56	0.50	0.58	0.27

Indicator	**O15**	**O16**	**O17**	**O18**	**O19**	**O20**	**O21**	**O22**	**O (mean)**
Total biomass	0.72	0.06	0.67	0.58	0.69	0.75	0.61	0.78	0.43
Total biomass – invertebrate (macrofauna)	0.38	0.38	0.40	0.36	0.28	0.33	0.48	0.43	0.38
Total biomass – vertebrate	0.52	0.50	0.49	0.58	0.56	0.57	0.40	0.46	0.51
Total catch – invertebrate (macrofauna)	<0.01	0.01	0.00	0.01	<0.01	0.00	0.00	0.01	<0.01
Total catch – vertebrate	0.00	0.01	<0.01	0.00	0.00	<0.01	<0.01	0.00	<0.01
Total consumption	0.00	0.06	0.25	0.00	0.08	0.00	0.00	0.03	0.10
Total consumption - indicator groups	0.03	0.25	0.00	0.11	0.00	0.03	0.00	0.00	0.10
Total consumption - individual groups	0.00	0.01	0.00	<0.01	0.01	0.00	0.00	0.00	<0.01
Total discard rate	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	<0.01
Total flow to detritus	0.72	0.69	0.00	0.58	0.69	0.75	0.61	0.03	0.59
Total mortality – demersal fish	0.56	0.47	0.39	0.36	0.31	0.00	0.47	0.53	0.45
Total mortality – filter feeders	0.56	0.58	0.47	0.44	0.56	0.56	0.44	0.72	0.60
Total mortality – macrozoobenthos	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total mortality – mammals	0.03	0.03	0.03	0.03	0.03	0.00	0.03	0.03	0.06
Total mortality – piscivores	0.33	0.22	0.39	0.19	0.03	0.17	0.31	0.31	0.28
Total mortality – planktivores	0.03	0.03	0.00	0.03	0.03	0.03	0.03	0.06	0.04
Total mortality – seabirds	0.03	0.08	0.17	0.03	0.03	0.00	0.03	0.03	0.03
Total mortality – sharks	0.03	0.08	0.08	0.03	0.00	0.00	0.03	0.03	0.05
Total planktonic production	0.06	0.03	0.03	0.06	0.06	0.06	0.00	0.42	0.10
Total primary production	0.78	0.69	0.00	0.56	0.72	0.75	0.64	0.78	0.70
Total production	0.72	0.69	0.67	0.56	0.69	0.72	0.61	0.78	0.71
Total production - indicator groups	0.03	0.25	0.00	0.11	0.00	0.03	0.00	0.00	0.10
Total production - individual groups	0.00	0.01	0.00	<0.01	0.01	0.00	0.00	0.00	<0.01
Total residence time	0.81	0.06	0.69	0.61	0.81	0.81	0.67	0.08	0.37
Total respiration	0.72	0.69	0.67	0.58	0.69	0.75	0.61	0.78	0.65
Total system exports	0.72	0.58	0.67	0.56	0.69	0.72	0.61	0.03	0.56
Total throughput	0.72	0.06	0.67	0.58	0.69	0.75	0.61	0.06	0.40
Trophic efficiency – detritus	0.00	0.53	0.00	0.00	0.00	0.00	0.00	0.81	0.34
Trophic efficiency – production	0.53	0.69	0.00	0.36	0.17	0.53	0.42	0.50	0.55
Trophic spectra – peak	0.44	0.00	0.53	0.33	0.61	0.50	0.39	0.25	0.24
Trophic spectra – slope (F)	0.25	0.25	0.25	0.25	0.28	0.28	0.19	0.28	0.30
Trophic spectra – slope	0.50	0.47	0.53	0.33	0.64	0.50	0.42	0.44	0.46
w-abc	0.00	0.39	0.00	0.00	0.00	0.00	0.00	0.39	0.22

Table 4.15: List of indicators that are substantially impacted by data quality.

Indicator	Degree of drop in performance
Average trophic level	The strength of correlation for Average trophic level (F), Average trophic level of vertebrates (F) and Average trophic level of vertebrate catch (F) weaken to recognisable; all other Average trophic level indices drop to uninformative.
Capacity	Drops to uninformative
CPUE	Drops to uninformative
Detrital dominance	Drops to uninformative
Discards	Drops to uninformative
Ecotrophic Efficiency	Drops to uninformative
Efficiency of catch	Drops to uninformative
FIB	Drops to uninformative
Invertebrate:Finfish catch ratio	Drops to uninformative
Net primary production / biomass	Drops to uninformative
Overhead	Drops to uninformative
Primary Production Required (PPR)	Strength of correlation weakens to recognisable for PPR – catch and PPR – indicator groups
Relative Ascendency	Drops to uninformative
SOI	Drops to uninformative
Total catch	Drops to uninformative
Total consumption	Strength of correlation weakens to recognisable
Total discard rate	Drops to uninformative
Total flow to detritus	Drops to uninformative
Total production	Strength of correlation weakens to recognisable
Total residence time	Drops to uninformative
Total respiration	Drops to uninformative
Total throughput	Drops to uninformative
Trophic efficiency	Drops to uninformative

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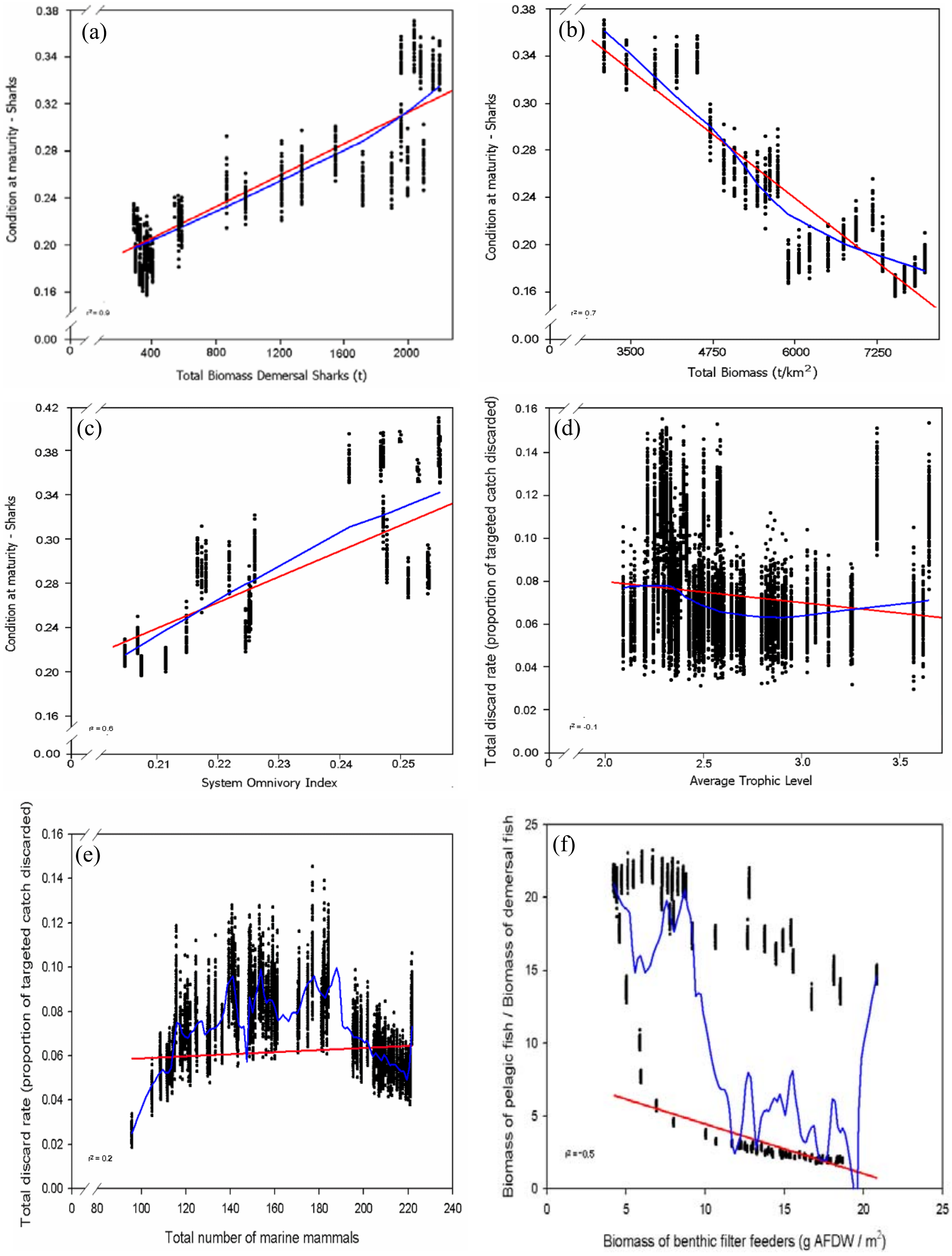
4.7.2.2 Bi-plots

Unsurprisingly, the bi-plot results show that indicator-attribute relationships with relatively high *r* values have a clear linear relationship, while those with lower *r* values have more noise about the curve or lack a linear relationship at all. Indicator-attribute relationships rated as very clear ($|r| > 0.9$) have clear linear fits in the bi-plots (e.g. Figure 4.14a), while those with strong or recognisable relationships have linear fits, but with increasing levels of noise about the line (e.g. Figure 4.13b). Indicators with uninformative correlations ($|r| \leq 0.5$) have a flat, non-linear or multi-armed shape to the bi-plot (e.g. Figure 4.14c – 4.14e). The majority of the individual indicator-attribute bi-plots fall into this latter case (and thus the relatively low rating of many indicators in many cases in Tables 4.11 – 4.14). As the bi-plots do not provide information not already supplied by the results of the correlation analyses, only a brief summary of the results is given (below) and the hundreds of individual plots are not presented here.

The majority of the uninformative indicator-attribute relationships were flat, with no clear form

Figure 4.14: Examples of the various forms of indicator-attribute relationships found in the bi-plots (indicator on vertical axis, attribute on horizontal axis in each case): (a) very clear linear relationship (from scenario O2); (b) strong linear relationship (from scenario O2); (c) recognisable linear relationship (from scenario O2); (d) flat (uninformative) relationship (from scenario SC5); (e) non-linear (uninformative) relationship (from scenario SC5); and (f) multi-armed (uninformative) relationship (from scenario SC5). In each case the red line is the best linear fit and the blue line is the loess curve.



of any type to the bi-plot (e.g. Figure 4.13c). Nearly one third of the uninformative indicator-attribute relationships have clear and relatively simple but non-linear forms (Figure 4.13d), and the remainder are multi-armed or gave some indication of bifurcation (Figure 4.13e). While there is academic interest in indicators with non-linear indicator-attribute relationships, in the role of indicators for use in management only indicators with a clear linear relationship to attributes are of real practical utility.

4.7.2.3 Multidimensional scaling (MDS)

This multivariate analysis of indicator performance clearly demonstrated that the performance of indicators was not uniform and that certain sets of conditions (scenarios) lead to similar patterns in the proportion of the attributes the indicators are correlated with. The grouping of scenarios on the MDS plots remain the same whether the MDS is done using the results from any of the Tables 4.11 – 4.14 (or even the entire set). For purposes of clarity in the plots only the MDS plots that are produced using the results in Tables 4.12 and 4.14 will be discussed further here.

Figure 4.15 is the MDS plot produced when using the results per scenario from Table 4.12 (which deals with the results for the shallow bay operating model and data aggregated over the entire system). There appears to be four main groupings here: (I) contains the unimpacted scenario and the scenario where there was a no trawl MPA; (II) is the unfished scenario with a five fold in increase in nutrient loading; (III) is the scenario with sequential fish down and a five fold increase in nutrients; and (IV) is a large loose set that contains all the other scenarios, though there is some nesting within this cluster. – the open system scenarios tending to cluster together to the left hand end of this set (dashed oval with the solid oval marked III in Figure 4.15). This pattern of groups suggests that indicators may perform differently when ecosystems on the scale of large shallow bays are put under different forms of anthropogenic pressure. Reassuringly their performance is largely the same across all manner of fishing scenarios.

The patterns are not as straight forward in Figure 4.16, which is produced using the results per scenario from Table 4.14 (which deals with the results for the coast-to-open-ocean operating model and data aggregated over the entire system). In this case there appear to be three main groupings: (I) contains both the unfished scenarios, the lightly fished scenario, the scenario with all anthropogenic pressures in place and the intensively fished scenarios with zoning, mammal recovery from past harvesting, site attached fish stocks, episodic recruitment once per decade, gear creep, no trawl MPAs, no incidental habitat destruction, changing TACS, fishing season length and fishing access; (II) the scenario where there is light fishing pressure and the mammals are recovering from past large scale harvesting; and (III) contains the scenarios with intensive fishing pressure and gear limitation, no take MPAs (fixed and rotating), rotating no trawl MPAs, coastal habitat degradation, management based on the stock size of vulnerable groups, changing discarding practices, episodic recruitment pulses twice per decade, and lastly the scenario with light fishing pressure and episodic recruitment pulses. Once again there may be some clustering within the larger looser groups I and III. For instance, the unfished scenarios both fall within the one region of group I. There is less of a suggestion of a simple relationship between indicator performance and the degree of human impacts in this case. The much larger regional scale of the operating model in this case allows for many more assemblages and site-to-site differences between different areas in the modelled area. In turn,

Figure 4.15: MDS plot considering the effect of scenario definition on indicator performance for the shallow bay operating model (created using the results from Table 4.12). The clusters marked I to IV are discussed in the text.

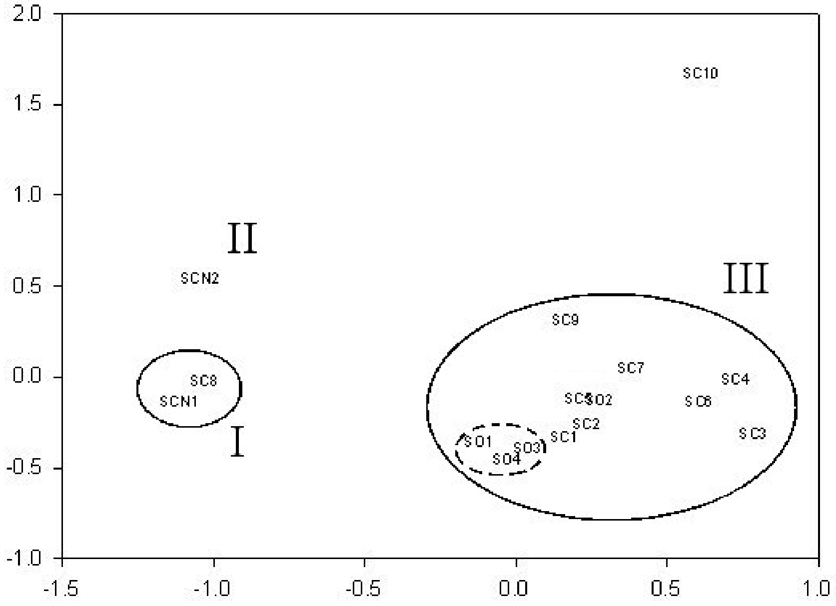
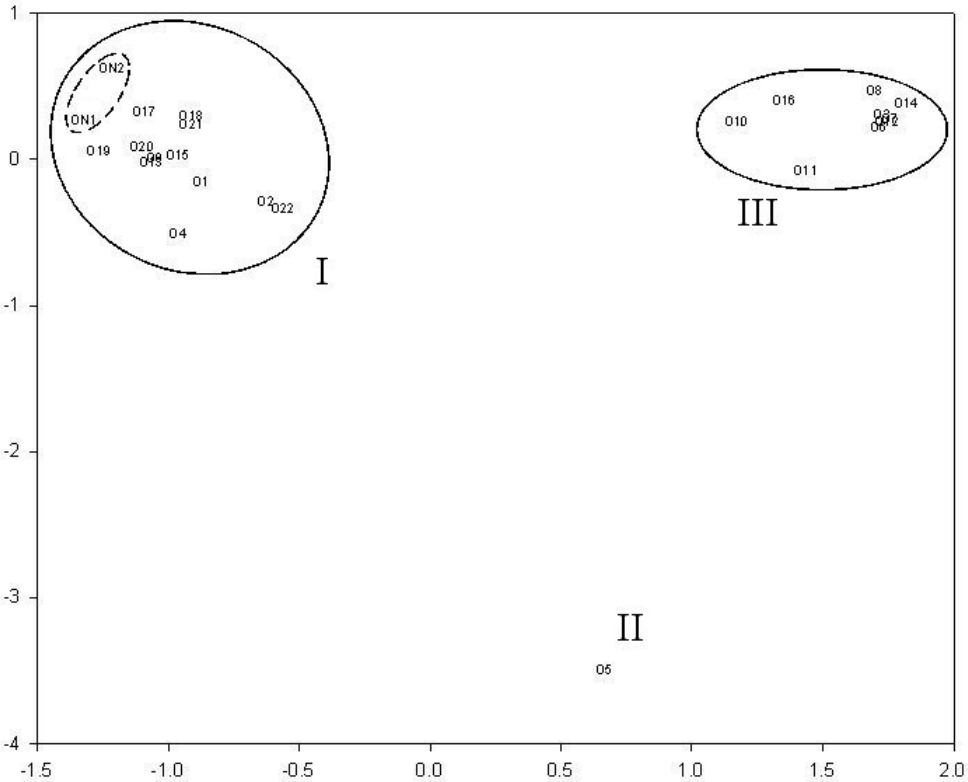


Figure 4.16: MDS plot considering the effect of scenario definition on indicator performance for the coast-to-open-ocean operating model (created using the results from Table 4.14). The clusters marked I to III are discussed in the text.



this means that the performance of indicators applied across such large and variable areas will be impacted by the strength of the various subsystems (or assemblages). As a result, if fishing (or any other human activity) has a differential effect across a regional ecosystem then attention must be given to any subsystems within that area. Given that the management zones used in this simulation testing framework often spanned multiple subsystems (as they do in reality), it is not surprising that similar MDS results are found when considering the results when the data is aggregated to the zone rather than over the entire system. Preliminary consideration of much smaller zones (Fulton *unpublished*) suggests that while simpler relationships between indicator performance and levels of human impact may be transiently evident when the level of data aggregation matches subsystems, shifts in the extent of assemblages with changes in the ecosystem means even these zones ultimately face the same problems as over time as they will cease to match the subsystems. The one case where data from small zones may be beneficial in the long-term is if results from large areas are contrasted with the results from within MPAs. Again this finding is based on preliminary work and would need to be more rigorously considered in future work.

Strikingly, consideration of the contribution of specific indicators the existing MDS results indicates that for most indicators there is no clear separation of performance based solely on the absolute magnitude or type of the pressure on the ecosystem. This is particularly the case when considering large regional scale ecosystems. The indicators respond to change in general, rather than change only due to fishing, eutrophication, coastal degradation or environmental shifts. Consequently, it appears that multiple indicators (including indicators such as Chlorophyll a, which perform poorly in a fisheries context but are proven indicators of the effects of eutrophication due to nutrient pollution) will be needed so that the pattern across the suite can be used to try and discern the true underlying cause for the observed changes.

4.7.2.4 Categorisation

Drawing together the correlations and plotted results and the definitions given in Table 4.9, the indicators can be categorised based on their overall performance (see Table 4.16). Recalling that the entries marked “individual groups” summarise the behaviour of tens of indicators applied biological-group-by-biological-group, a number of clear patterns emerge from the categorisation. First, the categorisation

Whether considering the indicators based on consistency, percentage of attributes represented or average strength of the correlations, nearly two thirds of the indicators with the most desirable characteristics (i.e. those rates as “consistent”, “broad” or with $|r| > 0.7$) are community- or ecosystem-level indicators. In contrast, all of the “uninformative”, and the majority of the “restricted”, indicators are population-level or sensitive to data quality. While the indicators rated as “inconsistent” were not all population-level indicators, over half were sensitive to data quality. While 70% of the population-level “inconsistent” indicators are not particularly sensitive to data quality, 90% the “inconsistent” ecosystem-level indicators are impacted strongly by data quality. Irrespective of indicator sensitivity to data quality however, over 90% of the indicators rated as “inconsistent” are sensitive to: (i) whether the system is open or closed, and (ii) whether they have been calculated using data aggregated at the ecosystem level. For instance, the vertebrate reproductive success and stock structure indicators (e.g. reproductive success of mammals and proportion (by number) of a stock that is juvenile) are only useful when dealing with closed populations. Consequently, these indicators may not be particularly informative in situations where there is a non-negligible flow of recruits into the study area from external stocks. Moreover, if ecosystem-level effects of fishing are of interest then system-level data must be used for these indicators to be effective. Work in real ecosystems (e.g. the Antarctic, Watanuki *et al*. 2004) shows that these indicators can be informative from a population perspective if considered at local scales, but from an ecosystem perspective data from across the system must be used.

This kind of verification of indicator properties is exactly the kind of test simulation analyses such as those in this study are best suited for. To that end the categorisation can be used to identify those indicators which are unlikely to be useful across the board and also a core set that are likely to perform well in most situations. The categorisation in Table 4.16 shows that a fifth of the most promising indicators from the literature are only weakly correlated with a restricted number of attributes, and then only in certain circumstances. Of those that do perform consistently at different scales and across ecosystems of different types, 70% are correlated with only a restricted set of attributes and a further 10% are broadly, but only fairly weakly, correlated with the system attributes of most interest. This leaves only a limited number of indicators that consistently track changes in a broad set of attributes, while also being robust to ecosystem-type, pattern of exploitation, and degree of data aggregation. These indicators include the relative biomass of indicator groups (such as top predators, cephalopods, seagrass, planktivores and gelatinous zooplankton), biomass ratios, vertebrate size at maturity, and the slope of the biomass size spectrum.

While indicators rated as “consistent”, “broad” and “strong” or “very clear” are particularly desirable, if a set of indicators of this form together do not span all the attributes of interest then they will have to be supplemented with indicators which do not rate as highly overall, but which perform well for those attributes missed by the rest of the set. Similarly, in practice, indicators that may perform poorly for fisheries induced changes in the ecological properties of the ecosystem, but which react strongly to other forces on the ecosystem are needed to help and discern the underlying cause behind any observed shifts in ecosystem structure and dynamics. Therefore indicators such as Chlorophyll a, Dissolved Inorganic Nitrogen, total production, total consumption and average trophic level must be added to a list of potentially useful indicators despite their “restricted” rating.

4.7.3 Analysis - Multivariate Indicators

The multivariate indicators - biomass size spectra, trophic spectra and dominance curves - appear to provide information on many different attributes simultaneously (e.g. biomass, diversity, size structure, trophic level and production), though they are not necessarily strongly related to any individual attributes. This means that they appear to provide good measures of overall system state, though by themselves it would be hard to discern which attribute was being most heavily impacted by changes in an ecosystem’s ecological properties.

Table 4.16a: Categorisation of informative indicators (|r| > 0.5) based on consistency across scales and system-types, percentage of attributes represented and average strength of the correlations. Definition of terms is given in Table 4.9 and the main text (for data quality), (F) stands for fixed (trophic values of each group are fixed at starting values and are not recalculated dynamically) and the letters in parentheses for the inconsistent indicators specify under what circumstances they did correlate with the ecosystem attributes (z – using data aggregated to the management-zone level; s – using data aggregated to the system-level; b – in bay ecosystem only; o – in coast-to-open ocean ecosystem only; c – in relatively closed systems only; and m – mixed (e.g. bz and os)).

	Consistent		Inconsistent	
	Broad	Restricted	Broad	Restricted
Very clear	Biomass – cephalopods Biomass – demersal fish Biomass – detritus Biomass – piscivores Biomass – planktivores Biomass – sharks Condition at maturity – top predators Size at maturity – demersal fish Total biomass	Average trophic level (good data) Average trophic level (F) (good data) Biomass – planktivores Condition at maturity – seabirds Labile / Refractory detritus (biomass ratio) Maximum length of catch PS / ZP (biomass ratio) Size at maturity – piscivores Total primary production	Total discard rate (good data) (s)	
Strong	Biomass – epifauna Biomass – gelatinous zooplankton Biomass – mammals BSS – slope FIB (good data) Invertebrate:Finfish (biomass ratio) Pelagic: Demersal (biomass ratio) PS / ZP (catch biomass ratio) SOI (good data) Total biomass – vertebrate Total consumption (good data) Total production (good data)	Average trophic level of catch (good data) Average trophic level of catch (F) (good data) Average trophic level of vertebrates (F)(good data) Average trophic level of vertebrate catch (F)(good data) Biomass – macroalgae Biomass – seabirds Biomass – seagrass[1] Chlorophyll a[1] CPUE – cephalopods (good data) CPUE – demersal fish (good data) CPUE – epifauna (good data) CPUE – piscivores (good data)		Average trophic level of vertebrates (good data)(c) Average trophic level of vertebrate catch (good data) (c) Biomass – macroalgae (z) Ecological footprint – piscivores (good data) (o) Proportion of stock juvenile – demersal fish (o) Proportion of stock juvenile – piscivores (o) Proportion of stock juvenile – planktivores (c) Trophic efficiency – detritus (good data)(o)

	Consistent		Inconsistent	
	Broad	Restricted	Broad	Restricted
Strong (cont.)		CPUE – planktivores (good data)		Trophic efficiency – production (good data) (z)
		CPUE – sharks (good data)		
		CVt		Total residence time (o) (good data)
		Detrital dominance (good data)		
		Discards – demersal fish (good data)		
		Discards – invertebrates (good data)		
		Discards – piscivores (good data)		
		Discards – planktivores (good data)		
		Diversity – simple count		
		Dissolved Inorganic Nitrogen[1]		
		Ecotrophic Efficiency – indicator groups (good data)		
		Ecological footprint – demersal fish (good data)		
		FIB (F) (good data)		
		Infauna/Epifauna (biomass ratio)		
		PBR – mammals		
		PBR – seabirds		
		Pelagic: Demersal (catch biomass ratio)		
		PPR – catch (good data)		
		Total biomass – invertebrate (macrofauna)		
		Total benthic consumption		
		Total respiration (good data)		
		Total throughput (good data)		
		Trophic spectra – peak		
		w–abc		

	Consistent		Inconsistent	
	Broad	Restricted	Broad	Restricted
Recognizable	Biomass – bacteria Diversity – number of communities PPR – indicator groups (good data) Proportion of the catch juvenile Relative Ascendency (good data) Total consumption (poor data) Total planktonic production Total production (poor data) Total mortality – demersal fish Total system exports (good data) Trophic spectra – slope	Average trophic level (F)(poor data) Average trophic level of vertebrates (F) (poor data) Biomass – large zooplankton Capacity (good data) Ecotrophic Efficiency – individual groups (good data) Invertebrate:Finfish (catch ratio)(good data) Nitrification Number of mammals caught Number of seabirds caught PPR – catch (poor data) PPR – indicator groups (poor data) PPR – individual groups (good data) Total benthic production Total catch – invertebrate (macrofauna)(good data) Total catch – vertebrate (macrofauna)(good data) Total consumption – indicator groups Total flow to detritus (good data) Total mortality – mammals Total mortality – piscivores Total mortality – sharks Total production – indicator groups Trophic spectra – slope (F)	Reproductive success of seabirds (c)	Average trophic level of vertebrate catch (F)(poor data) (c) Denitrification (m) FIB – slope (o) (good data) Light at sediment surface (z) Net primary production / biomass (good data) (s) Overhead (good data) (z) Reproductive success of mammals (c) Total mortality – filter feeders (o) Total mortality – macrozoobenthos (oz) Total mortality – planktivores (o) Total mortality – seabirds (o)

1. These indicators are very good ancillary indicators for checking for the additional effects of eutrophication.

Table 4.16b: List of all indicators rated as uninformative

Rating	Indicators
Uninformative	Average trophic level (poor data)
	Average trophic level of catch (poor data)
	Average trophic level of catch (F) (poor data)
	Average trophic level of vertebrates (poor data)
	Average trophic level of vertebrate catch (poor data)
	Biomass – individual microfauna groups
	Capacity (poor data)
	CPUE – cephalopods (poor data)
	CPUE – demersal fish (poor data)
	CPUE – epifauna (poor data)
	CPUE – individual groups (good or poor data)
	CPUE – piscivores (poor data)
	CPUE – planktivores (poor data)
	CPUE – sharks (poor data)
	Condition at maturity – non-target small-bodied groups
	Detrital dominance (poor data)
	Discards – demersal fish (poor data)
	Discards – individual groups (good or poor data)
	Discards – invertebrates (poor data)
	Discards – piscivores (poor data)
	Discards – planktivores (poor data)
	Ecotrophic Efficiency – indicator groups as a whole (poor data)
	Ecotrophic Efficiency – individual groups (poor data)
	Ecological footprint – demersal fish (poor data)
	Ecological footprint – piscivores (poor data)
	FIB (poor data)
	FIB (F) (poor data)
	FIB – slope (poor data)
	Invertebrate:Finfish (catch ratio)(poor data)
	Net primary production / biomass (poor data)
	Overhead (poor data)
	PPR – individual groups (poor data)
	Relative Ascendency (poor data)
	SOI (poor data)
	Total catch – invertebrate (macrofauna)(poor data)
	Total catch – vertebrate (macrofauna)(poor data)
	Total consumption – individual groups
	Total discard rate (poor data)
	Total flow to detritus (poor data)
	Total production – individual groups
	Total residence time (poor data)
	Total respiration (poor data)
	Total system exports (poor data)
	Total throughput (poor data)
	Trophic efficiency – detritus (poor data)
	Trophic efficiency – production (poor data)

4.7.3.1 Biomass Size Spectra (BSS)

Visual inspection of biomass size spectra indicates that they appear to provide clear signals of ecological changes in an ecosystem due to the impacts of fishing. For instance in Figure 4.17a shows a time series of the benthic biomass spectrum in the shallow bay model when there is no commercial fishing, while Figure 4.17b is the spectrum when there is intensive fishing, and Figure 4.17c is the spectrum when there is fishing and a fivefold increase in nutrient loading. The impacts of human activities on the ecosystem are quite marked in these figures.

In Figure 4.17a, while there is some variation through time, the biomass contribution from each major invertebrate size groups sampled remains relatively stable. Admittedly the signal is clearest for the most easily sampled size classes (epibenthic macrozoobenthos, benthic grazer and benthic filter feeder groups), while the contributions due to the microfauna and flora are underestimated.

In Figure 4.17b the effects of fishing are clear as the green band quickly contracts as direct and indirect effects of fishing reduce the epibenthic biomass. Moreover, despite sampling selectivity problems and the resulting under representation of the microfauna and flora in the spectra, an increase in the biomass of these groups is also clearly visible (which indicates a shift in community composition due to fishing) where the patch of bright blue is present beginning around year 60. Once fishing has ceased, recovery of the epibenthos can also be tracked in these spectra, as shown by the return of the green band towards the end of Figure 4.17b.

In reality, biomass size spectra are more commonly done for pelagic groups. The calculation of biomass spectra in the MSE framework used here encounters many of the same issues noted in the real world application pelagic BSS. If all of the pelagic groups are included in the pelagic biomass size spectrum, then it has a strong non-linear shape (the blue-to-red-to-blue pattern in Figure 4.18a) and shows little overall impact of fishing. However, if the spectrum for the pelagic system is restricted to the descending arm of the non-linear curve (Figure 4.18b), it shows a clearer impact of fishing (on the larger, more vulnerable groups). In addition the spectrum also becomes effectively linear (orange-to-blue within any one time slice) and changes through time are much easier to identify and interpret, as reinforced by the high overall performance of the slope of the BSS in the univariate correlation analyses above.

For all the scenarios examined, biomass spectra appear to capture changes in the ecological state of the ecosystem, without the requirement of collecting costly diet or rate information. This is not to say this indicator is not without its own problems. Quite apart from the issues of sampling selectivity and defining the lower endpoint of the spectrum, it is as sensitive to changes due to eutrophication as to those produced by fishing (as seen in Figure 4.17c). As with the many univariate indicators, the BSS responds to change regardless of the cause.

Figure 4.17: Timeseries of benthic biomass size spectra (using stock sizes for the entire ecosystem) for the shallow bay scenarios in which there is: (a) no commercial fishing; (b) a sequential fish down with patchy incidental mortality; and (c) commercial fishing and a fivefold increase in nutrients

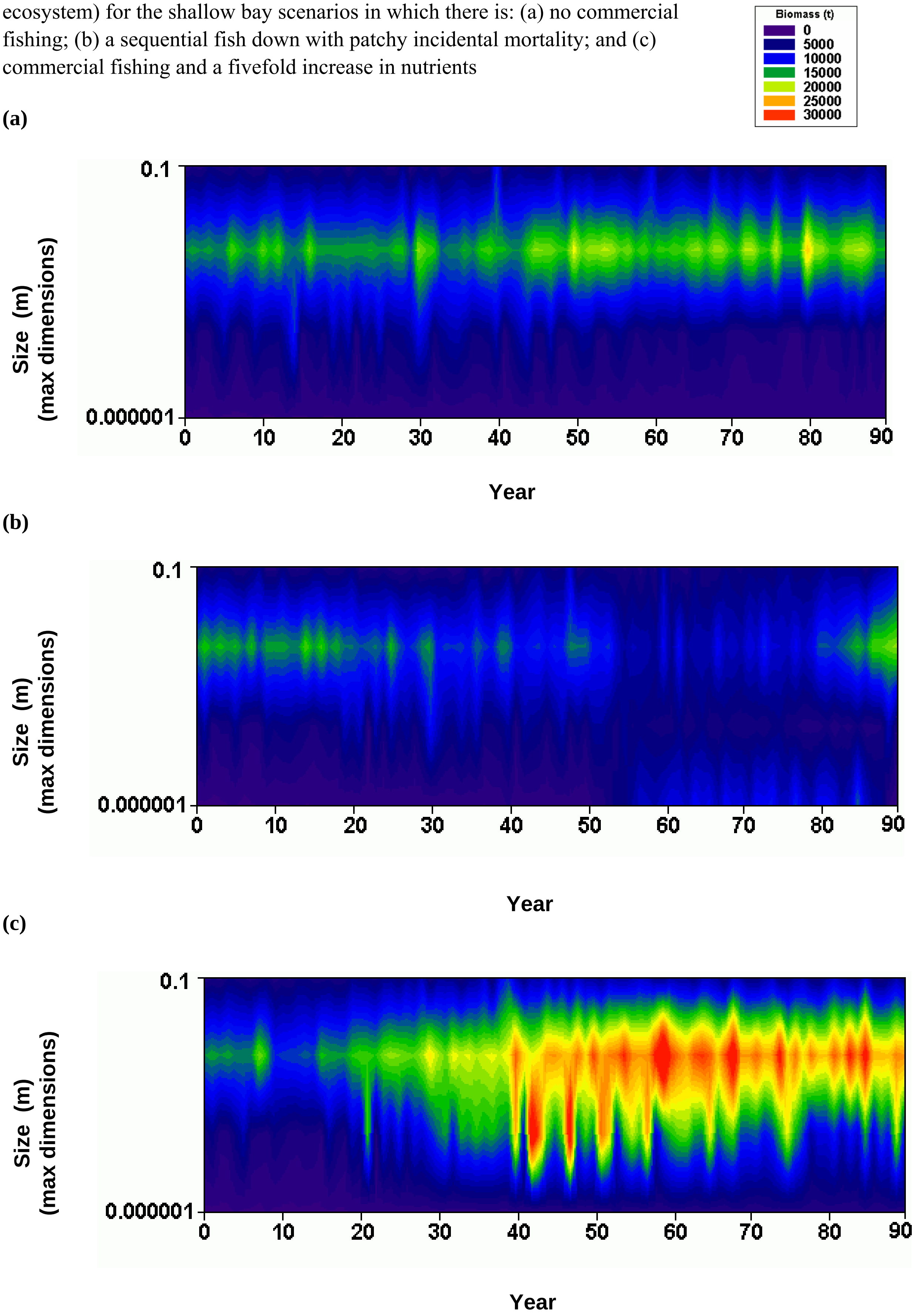
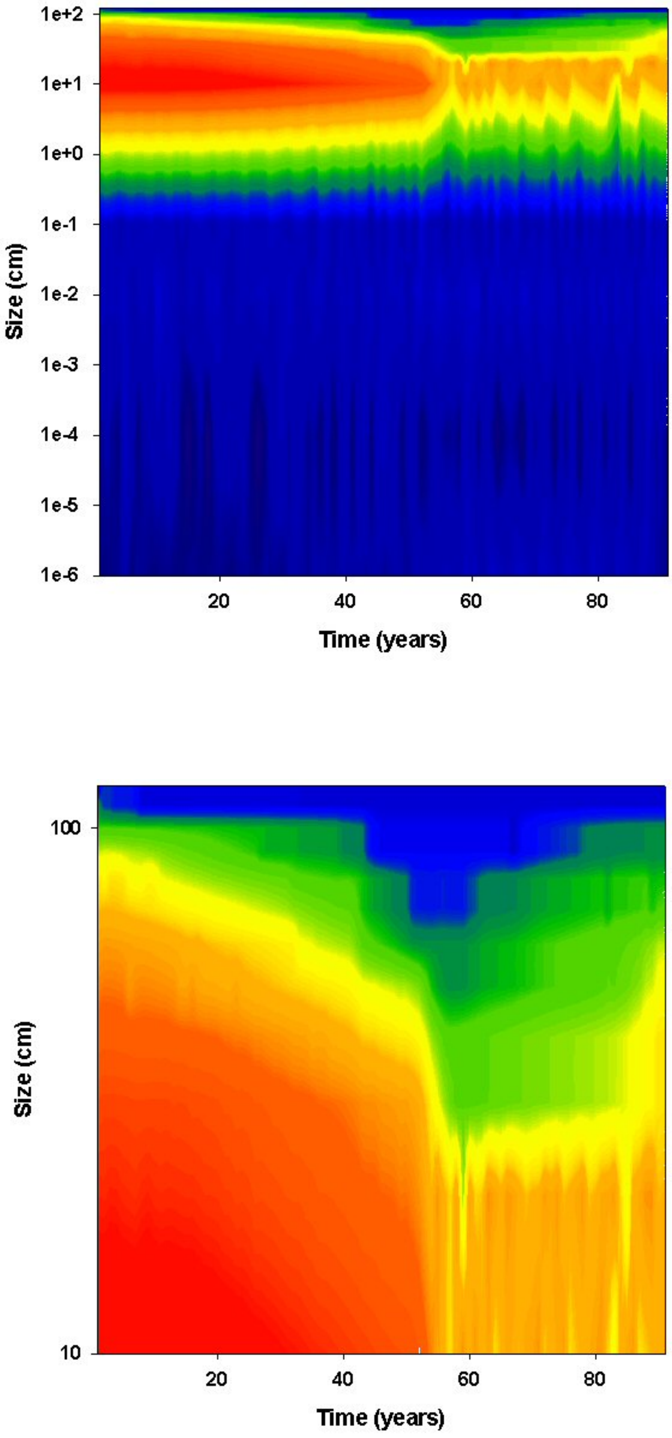


Figure 4.18: Time series of pelagic biomass size spectra (using total stock size for the ecosystem) for the shallow bay scenario in which there is a sequential fish down with patchy incidental mortality: (a) includes all pelagic, and (b) is restricted to the groups (and age classes) measuring over 10cm in size.

Biomass (t)
🟪 0
🟦 1
🟩 10
🟨 100
🟧 1000
🟥 10000


(a)

(b)

4.7.3.2 Trophic Spectra

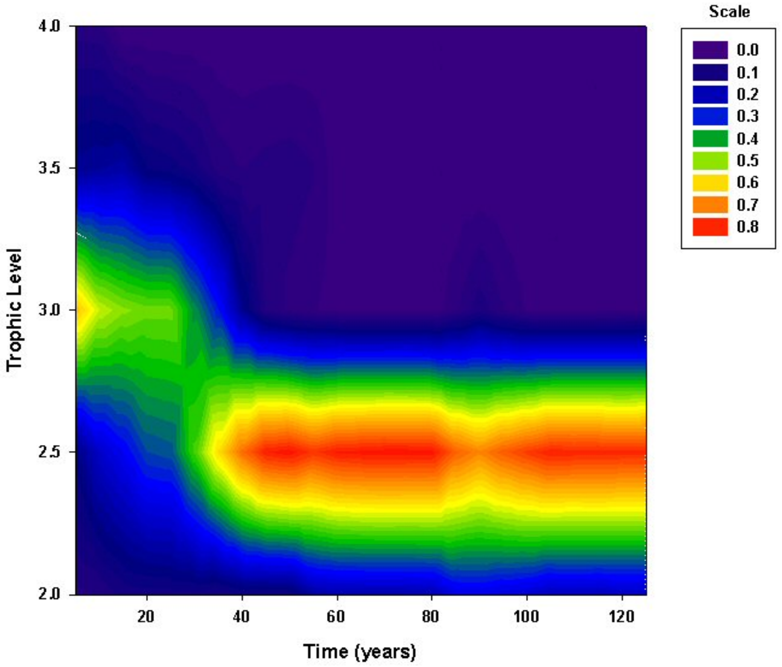
The univariate indicators derived from the trophic spectra all performed reasonably well (they are consistent, with recognisable to strong correlation strength, but are often restricted in the number of attributes they are correlated with). This clear relationship with attributes is obvious once trophic spectra are plotted through time. As higher order stocks are exhausted and the fish communities (and catch) are increasingly dominated by stocks (and age classes) with lower and lower trophic levels, the peak of the trophic spectra captures this shift (Figure 4.20). When plotted on a linear scale the shift is clear (Figure 4.20a), though details of what is happening for the top-most trophic levels is hard to discern unless a log scale is used (Figure 4.20b). This is particularly important if these indicators are to be used to monitor stock recovery after collapses, or if they are to be part of a suite of indicators used to consider the status of vulnerable groups at the top of the trophic web. For instance, in Figure 4.20a there is a slight suggestion of a minor increase in the contribution of animals with trophic level 3 about year 90, when a number of fisheries reopen after a period of closure (or very reduced effort levels). In contrast, the increase is quite clear in the log plot (Figure 4.20b). More importantly, the log scale clearly identifies the final collapse of the high trophic level target groups in around year 110 (when fishing and bycatch mortality from the reopened fisheries have not only negated any stock regrowth that occurred when the fisheries were closed, but have gone further to cause stock collapse), while the linear scale gives no notice of this event at all.

4.7.3.3 Dominance Curves

Dominance curves (especially ABC curves) plotted through time appear to pick up changes in the ecological state of the ecosystem (Figure 4.21). A view supported by the consistent and strong correlation performance of the w-abs statistic (which is derived from these curves) in the univariate analyses discussed above. These curves seem to preform consistently across scenarios and scales of data aggregation. In addition, they also appear to respond quite quickly to change. As with the other multivariate indicators however, ABC curves simply register a change in the system and respond similarly to system perturbation due to fishing and that due to eutrophication. It is important to note that while this method is robust to noise due to sampling error here (where it was calculate using data from functional groups), this method will be more susceptible to the effects of noise in real systems as it will more commonly be applied using species not functional groups in that case.

Figure 4.20: Trophic spectra of the relative contribution of the different trophic levels to the total catch through time from the coast-to-open-ocean scenario with intensive fishing pressure and zoning: (a) with a linear scale, and (b) with a $\log_{10}$ scale.

(a)



(b)

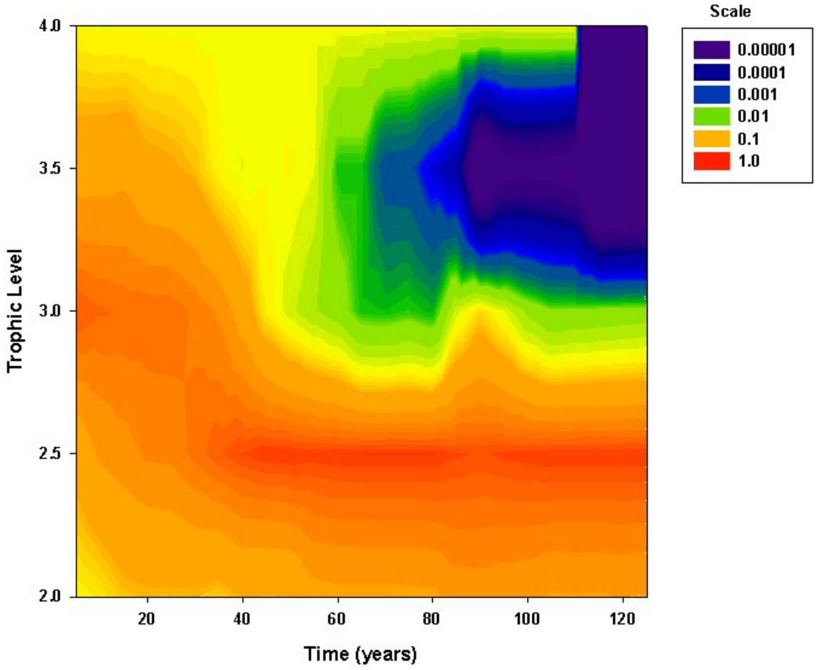
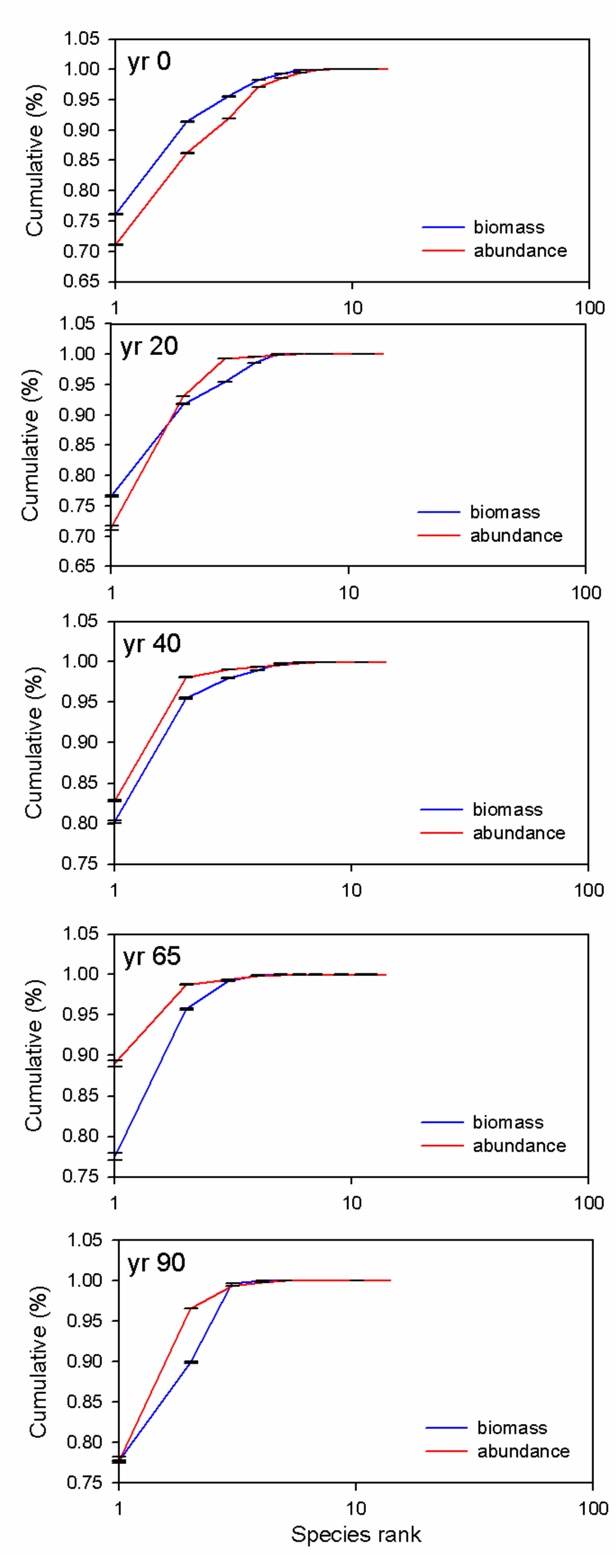


Figure 4.21: ABC curves over time based on biomass and abundance data for the groups > 1cm in size aggregated over the entire system for shallow bay scenario with a sequential fish down and patchy incidental mortality. The horizontal black lines are error bars.



No commercial fishing as yet

Fisheries developing

Fishing pressure is increasing, particularly on the higher trophic levels, but is not yet extreme

Fishing pressure is extremely intense

Fisheries on the higher trophic levels have been closed for a decade and have just reopened with low levels of permissible effort

4.8 Major Findings from Simulation Testing

The clearest message conveyed by the results of the simulation testing of the indicators is that the simpler the basis of the indicator, the more robust it is likely to be to the vagaries of sample design, any sampling or modelling errors, the spatial resolution of the sample data, natural variation, and an ecosystem's ecological structure. Essentially, indicators that

- involve multiple steps in their calculation (e.g. relative ascendency)
- require data that are difficult to collect reliably (e.g. diet data), or
- are reliant on intermediate models (e.g. network indices or relative ascendency)

are less robust due to their sensitivity to both the quality of the original data, and also to the number of steps involved in their calculation. This is not a criticism of the theory underlying these indicators, or the models used to calculate them, but is simply a reflection of the cumulative effects of additional errors potentially introduced at each step in the process. This is also not to say these indicators and models are of no use in any role – they have other specific uses in their own right, they can potentially provide useful insight into a system's dynamics and state, and they are essential for policy analysis and MSE. It is simply being suggested that, based on the results from the simulation testing presented here, it would be unwise to construct a monitoring system that relied solely, or even heavily, on these kinds of data or computationally intensive indicators. This finding is unlikely to take researchers and planners working with indicators by surprise, given that ease of data collection and simplicity of calculation are already widely recognised screening criteria for indicator selection (Jackson *et al*. 2000, ICES 2001).

The second major finding from these results is that the population-level indices preferred by management bodies in the past may be of limited utility in the monitoring of the ecosystem effects of fishing, at least in isolation (i.e. without supporting information from higher-level indicators). It appears more likely that consideration of information at the level of the community, or even the entire ecosystem, will be necessary. This will be discussed further in the next chapter, but given the slow responsiveness of ecosystem-level indicators, community-level indicators may be most informative (e.g. consider the biomass of the epifauna in general rather than specific subgroups). While some of the good performance of community-level indicators in this simulation exercise may be a reflection of the use of functional groups in the operating model, the same conclusion has been drawn empirically in other fields (e.g. water quality monitoring, Frost *et al*. 1992). This agreement between model and real world findings gives credence to the suggestion that the community-level may actually be the most informative level of aggregation with regard to discerning the impacts of human activities on ecosystems.

Lastly, the variability in the performance of individual indicators from scenario-to-scenario in combination with the patchiness in some of the attribute correlations quite clearly shows that no single indicator will reliably capture the ecological impacts of fishing. A suite of indicators is far more likely to successfully convey the changing state of an ecosystem under pressure across the enormous range of potential behaviours that could be manifest in a real world ecosystem. A recommended basis set, with associated reference directions (similar to reference points from the fisheries and conservation literature), is given in the final chapter of this report.

5 GIS METHODS

5.1 Basic Approach

The development of ecological indicators using real-world data often requires the integration and analysis of a wide variety of data types, formats and large volumes of information. The use of Geographic Information Systems (GIS) provides a readily available suite of tools that allows for this integration without the loss of spatial information. The use of GIS for exploratory data analysis is well documented (Kemp and Meaden, 2002; Hamilton and Cocks 1995, Eastwood *et al*. 2003) and provides resources for understanding the data at hand. Additionally, the spatial analysis functionality often available in most desktop GIS packages provides tools for more complex data analysis and visualisation.

The spatial processing capabilities of GIS provides a powerful tool for analysing data at different scales, and within different analysis units. The capacity to vary analysis units spatially is especially useful for marine data, which often varies considerably in both its spatial scale and extent. It also allows analysis of particular datasets to occur at a range of scales, which can be chosen to best fit the data available, or the questions being asked.

The data integration capabilities of GIS also provides a useful framework for capturing environmental information. With the development of satellite imagery, aerial photography and hyperspectral imagery, broad scale identification of a range of different biological and physical parameters has become much easier than in the past. For example the identification of seagrass change for Cockburn Sound in Western Australia (Kendrick et. al, 2002). The ability to automate many of these processes to some degree has also improved the temporal resolution of datasets and statistical methods have improved allowing determination of spatial interannual variability of parameters. The development of image processing has been done in the terrestrial environment, however applications are becoming more common for aquatic and marine systems (Fréon *et al.* 2004).

The evolution of GIS from primarily a tool for terrestrial ecology, to one useful for applications to marine science has taken place largely since the 1990's (Wright and Bartlett, 2001). There are now a number of excellent references available outlining applications of GIS to the marine and coastal sciences (Wright and Bartlett 2001; Valavanis 2002), and these show the maturation of the tools available for the field. Applications of GIS to fisheries science vary in complexity, from resource information or "digital atlas" type products, to simple spatial analysis and mapping of first order fisheries spatial statistics, to more advanced analysis, geostatistics and modelling applications (e.g. Nishida 2001).

The strength of GIS has traditionally been in the presentation of spatial data, with the temporal component often being difficult to visualise. Fisheries data generally has a strong time component and the ability to display time and space in an effective manner is often difficult. A number of methods were explored within the GIS to attempt to solve this problem. One approach involved the generation of movie loops of a number of parameters over a range of time and space scales. This allows for data exploration and can provide some insight into underlying patterns, before a thorough statistical analysis is undertaken.

5.2 GIS Applications

Exploratory Data Analysis and Visualisation

Exploratory Data analyses facilitates the identification of any issues with the data and helps to identify appropriate analyses. GIS tools allow an understanding of the spatial extent, density and precision of the data. They also provide a good understanding of the spatial overlaps between datasets. The amount of overlap is a major determining factor in deciding which datasets can be used for spatial overlays and subsequent analysis.

The development of integrated fisheries data management systems suggests that the use of GIS for fisheries management is becoming more sophisticated and common place. Kemp and Meaden (2002) describe the FishCAM2000 application which provides an integrated framework for data analysis and visualisation. FishCAM has been developed for all levels of the industry, from on-board data capture, data analysis for fishermen, regional and national fishery managers, plus modelling and analysis toolkits for scientists to assist in stock prediction.

Dedicated software, such as FishCAM, are not the only means by which GIS is used in fisheries management. In the study and characterisation of the Chatham Rise off New Zealand, Wood (2001) describes the use of GIS to support and enhance existing software packages. In this case, a range of GIS packages are used for data visualisation and generation of quality maps for reports, geo-referencing Relational Database Management System (RDBMS) data (spatial overlay with other features for attribution), modelling bathymetric data, and error checking of RDBMS data. Additional statistics and modelling packages (Matlab, S-Plus) are used for further data analysis.

Stanbury and Starr (1999) describe the establishment of a GIS for the Monterey Bay National Marine Sanctuary, which provides a good example of a digital atlas providing ready access to information resources for managers. GIS allows the integration of a range of data types (formats) and themes (biological, physical, socio-political, imagery) to support a range of applications, including: evaluation of natural processes; permitting and monitoring coastal development, and assessing environmental impacts.

With the increasing role GIS plays in areas from basic data visualisation to coordination of multiple data types the approach easily lends itself to consideration of ecological indicators in a fisheries context. In particular, Garcia and Staples (2000) have recognised that graphical representations will be useful for communication with the stakeholders and public in general. For example, the juxtaposition (or overlay) of simple indicators on a map allows the emergence of conclusions otherwise not available from simple tables and time series of numbers. GIS will play an important role in this respect, but other types of graphical representation will still be required (Garcia and Staples 2000).

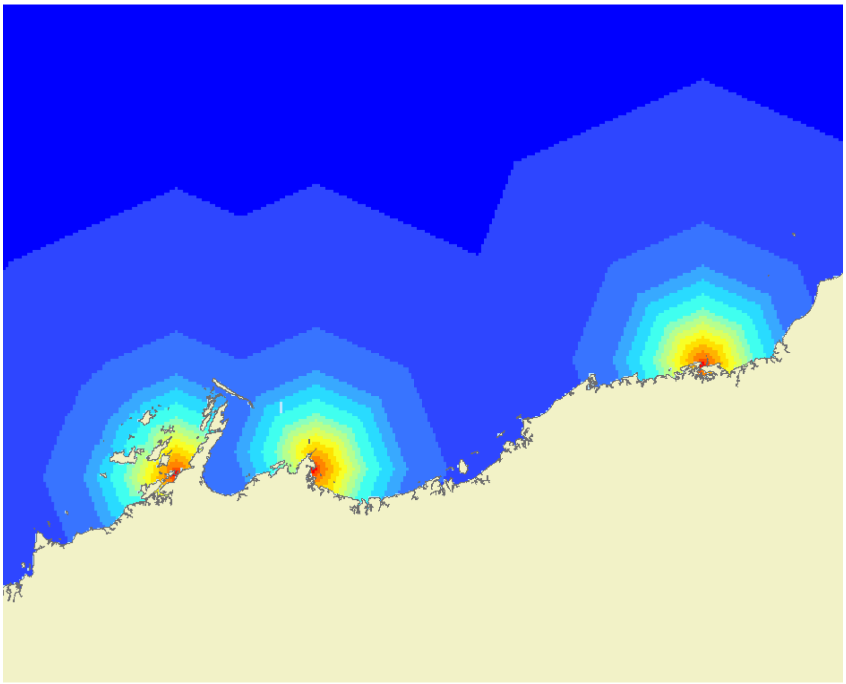
Ecological Indicators

The application of spatial analyses to ecological indicators and fisheries science has evolved from existing terrestrial and aquatic applications. There are a range of examples of varying complexity from both types of environment. Variation occurs in the spatial scale employed and in the range and type of indicators measured. Even the role of GIS can vary: it can be used to supply the base, spatially resolved, information for additional data analysis techniques; at the other extreme it can form part of integrated modelling exercises aimed at filling information gaps regarding anthropogenic activities, such as fishing. It is beyond the scope of this report to exhaustively list all examples of the application of GIS to the marine environment. Therefore, a brief overview is given below for a representative set of examples, which spans the range of approaches that have been described in the literature, from the development of environmental and ecological indicators, to biogeographic classifications, and integration with fishery science and assessments.

He et. al. (2000) outline the use of GIS for integrating a range of biological, physical and human environmental characteristics. This work was done in the context of the development of integrated ecological indicators for assessment of watershed condition and to support resource management. In a similar manner, Xu et. al. (2001) used GIS for the integration of six environmental characteristics including biological and physical factors to create an index of trophic states for lake eutrophication assessment. Paul et. al. (2001) developed an index of estuarine condition for a biogeographic province in Virginia. In the study, 7 of the 48 metrics evaluated were chosen, with the final index was based on the ecological interpretation and relevance of the individual metrics and the ability to meet the calibration and validation targets.

Caddy and Carocci (1999) describe the use of GIS for modelling fishing intensity from port based fleets using a Gaussian Effort Allocation Model (GEAM). The approach attempts to model how the impacts of port based fleets attenuate with distance using a “friction of distance” concept. GIS provides the spatial analysis tools to generate analysis layers determining the distances required to travel to areas fished, and includes barriers such as islands in increasing the distance travelled, rather than a simple linear distance. The approach models fishing effort based on distance to port, and the time since the opening of the fishery, in order to identify the areas of maximum effort. Application of the first analysis step (distance allocation with impedance from barriers to travel) to the NW Shelf study area is shown in Figure 5.1.

Figure 5.1: Initial analysis performed for Gaussian Effort Allocation for three ports on the NW Shelf: Dampier; Port Samson; and Port Hedland. The first analysis step involved establishing an “accumulated distance” surface for selected ports.



Other fisheries approaches have included the integration of fishery information on catch and effort with the economic return from the fishery. Rueda and Defeo (2003) studied the spatial population structure and economic potential of an estuarine lagoon in Columbia. Using Monte Carlo analysis, they assessed the status of the fishery, and used model based estimates of the fishery performance (from geostatistics and kriging) to quantify the uncertainty in abundance, individual price by size and variable costs per haul of the fishery. They found that values of abundance, fish size and biovalue (economic potential) were spatially structured, and varied between species and season. The spatial risk analysis showed that 30% of the lagoon had profitable levels of fish abundance.

In a similar study, Larcombe et.al (2001) used GIS for analysis of the spatial distribution and intensity of trawl operations for the South East Fishery in Australia. The analysis was performed to assess possible marine disturbance as part of a regional marine planning process; and involved the production of maps, and quantitative analysis of fishing effort over various spatial scales, depths and intensities.

Abella *et al*. (2001) used GIS for data exploration of demersal fisheries off the coast of Viareggio, Northern Tyrrhenian and Ligurian Seas, Italy. Maps of effort distribution by gear type, and catch rates for the main commercial species were generated. Additional analyses were done using multivariate techniques to identify target species for each fishery. Multivariate analysis allowed better understanding of the dynamics of fleet effort over time, as many factors affect fishing behaviour – including: different fishing traditions; changes due to season, market or economical constraints; and vessel modification, introduction and retirement. A better understanding of multi-species and multi-gear fishery provides a better foundation for management of the fishery. For example different gear types operating in the same region do not necessarily capture the same species mix. Maps of seasonal changes in catch and effort can assist in the management of the fishery and additional information may be obtained from mapping species size distributions. Garibaldi and Caddy (1998) used GIS in conjunction with available species distribution data to generate species richness maps from presence/absence data for the Mediterranean and Black Seas. The study involved the integration of a range of fish distribution data and bathymetry to generate presence absence information for 536 species over the region.

The application of spatial analysis to support processes such as the identification of candidate areas for Marine Protected Areas (MPAs), and generating contextual information to support the MPA process is another useful application of GIS that contributes to fisheries management. Manson and Die (2001) found that spatial analysis incorporating fisheries information gives an important understanding of the human uses of areas being considered for MPAs and may improve management outcomes (e.g. more effective biodiversity conservation), and provide an understanding of impacts of management measures such as area closures on catch rates. The analysis by Manson and Die (2001) uses biogeographic regionalisation (IMCRA), fishing data and bathymetry as a surrogate for ecosystems (in the absence of adequate habitat or ecosystem data) to identify the impact of fisheries closures for MPA planning on the fishery. The actual development of biogeographic classifications may also utilise GIS in conjunction with available species information.

Booth (2000) describes the use of spatial analyses to improve management advice provided from stock assessment models. By using a spatial approach, the biomass of the stock can be relatively easily disaggregated by region or life history stage, in turn facilitating and improving age-structured modelling. Additional information from a fishery data may be obtained through multivariate analysis of the fishery to provide information on the interactions between different gear types and the species assemblages caught by the fishery.

5.3 Application to NW Shelf

Approach

It was decided that it would be informative to test some of the indicators considered in the modelling exercise using real world data. With a mind to the future ease of application of potential indicators it was also decided that the testing would largely be done using a GIS approach (a few system-wide indices are calculated without recall to GIS). In this way an evaluation would be made to decide if the indicators could readily be generated and assessed in the GIS environment. If they could this could facilitate the real world use of ecological indicators in the future. The Northwest Shelf (NWS) of Western Australia was identified as a useful region in which to base this testing, as there was a ready availability of data for the area – due to the Northwest Shelf Joint Environmental Management Study (NWSJEMS) and other sources of historical data (e.g. from Fisheries Western Australia)

In order to determine the analysis process, indicators identified by Fulton *et al*. (2004a) are assessed for their applicability (Table 5.1). Considerations included the availability of data to calculate the indicators, and the suitability for use within the GIS processing and analysis environments. The primary data sources identified were: fishery logbook records, with shot level spatial information; and catch and effort data at a coarser spatial resolution (Fisheries WA statistical areas – one degree squares), but with more detailed species information. Additional habitat information was obtained from CSIRO cruises from the 1980's and 1990's. Other limited data from NWSJEMS was omitted from the work discussed here due to a lack of adequate spatial extent in comparison to the datasets which were used. As a result a number of indicators that were heavily reliant on the availability of suitable habitat or fish distribution data could not be considered in the final analysis. It may be possible in the future to use surrogates for the distributions of species (e.g. attempt to use depth and latitude to define species distributions), but this was not attempted in this study.

Figure 5.2: Location Map showing NWSJEMS study area and analysis units used.

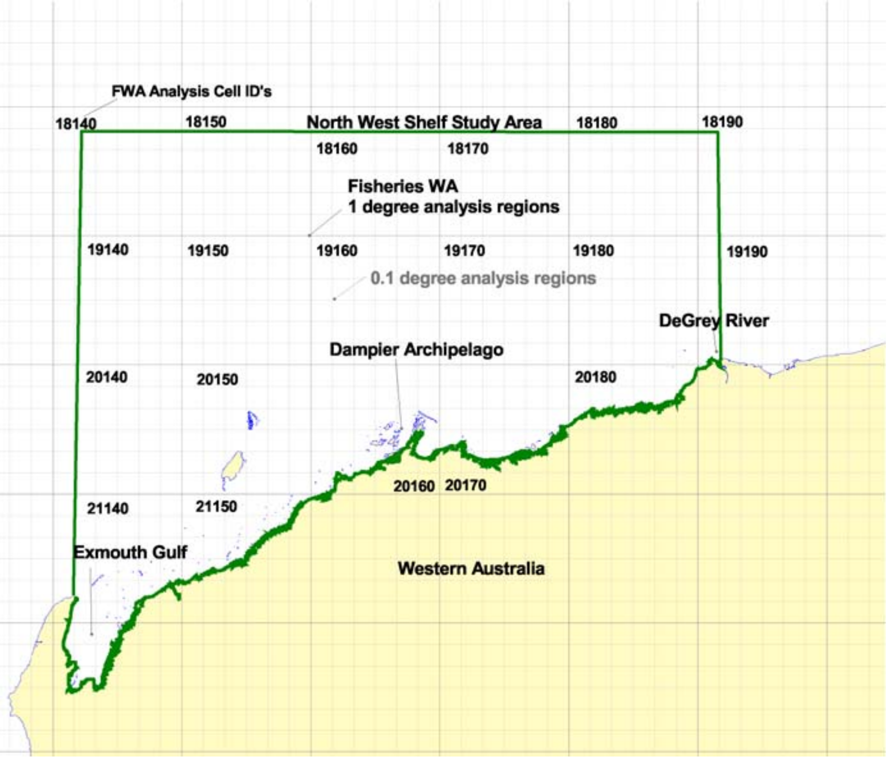


Table 5.1: Ecological Indicators (and methods) identified as being suitable for GIS application.

Spatial Component (*and method*)	**Indicator(s)**	**Comments**	**Applied**
Spatial patterns and relationships	Connectance, Fragmentation	The analysis of spatial relationships between ecosystem units (habitats) has been well developed for terrestrial systems (e.g. Fragstats, McGarigal and Marks, 1995). Spatial pattern analysis requires habitat mapping at a consistent scale across the whole study area. This level of habitat mapping is not available for the NW Shelf study area.	N
GIS Analysis *Spatial statistics* *Geostatistics*	Ratio catchment area to semi - enclosed area	This indicator would be relatively simple to apply in most situations where catchment boundaries are available. As the study area is an open one, it was not implemented here.	N
	Diversity – number communities	Can be calculated with available habitat/community data	Y
	Ratio of disturbed to undisturbed habitats	Can be calculated with available habitat/community data and suitable disturbance information (e.g. fishing grounds, trawl tracks, human activities). In this case, due to the depth preferences of the habitat defining groups the area that is actually suitable habitat that is open to fishing has almost all been fished in the last decade. Thus in this case this indicator was effectively identical to the ratio of protected to unprotected habitats listed below (and thus the analysis was not repeated).	N
	Ratio of protected to unprotected habitats	Can be calculated with available habitat/community data and information on protected areas (e.g. Marine protected areas, Marine parks, conservation areas, areas closed to fishing).	Y
	Area of available habitat occupied	Can be calculated with available habitat/community data, but at the taxonomic and spatial resolution of the data relative to the area of interest the rates are close to 100% and as such not a fair test of the indicator	N
Environmental Attributes *Mapping environmental characteristics.*	Chlorophyll a, Turbidity	Ocean colour satellite sensors such as SeaWIFS and MODIS provide a ready source of information at regional scales including chlorophyll, colour and turbidity, though these sensors are often limited in their application in shallow waters (around <30m) due to interaction with the ocean bottom. The existence of highly turbid inshore waters and a deep chlorophyll layer offshore across the study area made this kind of analysis less useful in this case.	N
Community Structure	Biomass ratios	Can be calculated from available fisheries data	Y
	Small-bodied:Large-bodied spp.	Calculated from CAES data and general species life history information	Y
Existing analysis methods which can be applied in a spatial context	Pelagic:Demersal species	Calculated from CAES data and general species ecological information	Y
	Invertebrates:Finfish	Calculated from CAES data and general taxonomic information	Y
	Average trophic Levels Trophic spectra	Calculated from CAES data and general diet information	Y
	Diversity – number species	Can be calculated from available fisheries data. Total species was used as a simple diversity indicator.	Y
	Maximum length of catch	Can be calculated from available fisheries data and general life history information. Length data was poor here (for many only “small”, “large” classification is possible) so this indicator was dropped in this case (note that the small:large bodied biomass ratio can be used as a proxy for this measure).	N

Spatial Component (*and method*)	**Indicator(s)**	**Comments**	**Applied**
Population Structure *Existing analysis methods which can be applied in a spatial context*	Abundance (or density) Cover of habitat defining taxa	Requires suitable habitat data to be calculated. In this analysis cruise small and large benthos data are used to define the mean occurrences of each size class for analysis regions.	Y
	Protection of populations (% of distribution of species overlap with fisheries, ratio of protected areas overlap with species)	This is a subset of the more general analysis dealing with 'Ratio of protected to unprotected habitats'. Can be calculated if suitable species distribution data and areas under protection are available.	Y
	Range and distribution of species.	Can be calculated if suitable species distribution data exists. Some data of this kind is available for the study area, but as it is not complete this indicator was not pursued further here.	N
	CPUE trends	Calculated from available fisheries data	Y
	Total Catch	Calculated from available fisheries data	Y

5.3 Methods

5.3.1 Source Data

Two grids are used during the data analysis: a one sixth of a degree polygon grid was generated over the region for this data using ArcInfo; and the Western Australia Fisheries statistical boundaries, which define a 60 nautical mile (one degree) grid.

Data used in the analysis consisted of

i) Western Australian fisheries vessel logbooks (also referred to as FWA vessel logbooks) containing trawl information including date, vessel ID, trawl start and end coordinates, trawl time, and catch weight by species (excluding bycatch)
ii) Western Australian fisheries broad scale catch and effort data (also referred to as FWA CAES data) – excluding bycatch
iii) Habitat information consisting of CSIRO research cruise data containing occurrences of small and large benthos identified from trawl photos.
iv) Trophic, ecological and size information for each species, which was obtained from the literature (see list of references in Appendix A2) and FISHBASE (http://www.fishbase.org/trophic/T.html).
v) Marine park boundary study areas from the Western Australian Department of Conservation and Land Management.
vi) Fishery License areas obtained from Fisheries Western Australia.

Fisheries independent datasets which were assessed for potential utility were identified from the CSIRO metadata repository (MarLIN) using a search for NWSJEMS datasets.

5.3.2 Data Processing

Reserved or Protected Areas within Fisheries

Spatial analysis for calculation of areas which are not fished, along with protected areas, are performed in ArcView 3.3 GIS. Fisheries License boundaries are overlaid with the boundaries of the proposed marine protected areas to give the area of the fishery within the MPA's. Summary statistics are then calculated from the resulting attribute tables.

FWA Vessel Logbooks

The logbook data is aggregated using the one sixth of a degree analysis units. The individual trawl data is map to the one sixth of a degree cells by performing a spatial intersection between an arc created from the start and end coordinates of each trawl and the analysis cells. The proportion of the total length of each resulting arc is added to the attributes of the trawl tracks to create a relative length of total trawl length within each analysis cell. Calculations are then performed using the relative catch, which was calculated by multiplying catch values by the relative length to give a proportional catch value. Note that while some snapshots are given below for this kind of data and this data was used to draw inferences about the interconnection between % cover of benthic habitat and fishing pressure much of the work undertaken at this scale can not be presented here due to issues of commercial confidentiality at the spatial and temporal scales this data set allows.

FWA CAES Data

Additional levels of detail of species information are available in the CAES data from FWA, but it is at a coarser spatial resolution – available in FWA 60 Nautical Mile statistical units. No additional spatial data processing is performed on this data, but it was necessary to marry the species level catch and effort from this dataset with general information on each species (e.g. length, basic diet and life history information (from Bulman and Althaus *in prep* and literature given in Appendix A2)) from other sources to calculate some of the indicators considered. The following are the equations used to calculate each of the indicators evaluated using CAES data, note that tildes are used here to distinguish these values calculated using real data from that calculated using simulated data (elsewhere in the report). Note that the data used was truncated at the end of 1999 as data was not available for any full year (and so annual analyses are not possible) after that point.

Average trophic level

The trophic analysis using the CAES data was performed using normalised trophic values for each cell with the estimated trophic level of the catch ($T'_{i,C}$) calculated as:

$$T'_{i,C} = \frac{\sum_i B'_{i,C} \cdot T'_i}{\sum_i B'_{i,C}} \tag{5.1}$$

where $B'_{i,C}$ is the catch recorded for species *i*, and T'_i is the trophic level assigned to species *i* based on available trophic information.

Biomass ratio of pelagic : demersal fish species

The ratio of pelagic to demersal species in the catch in each cell in each year ($B'_{PD,C}$) is calculated using:

$$B'_{PD,C} = \frac{\sum_i B'_{i,C}}{\sum_j B'_{j,C}} \tag{5.2}$$

where $B'_{i,C}$ is the total biomass of catch of pelagic species *i* for the year from the cell; $B'_{j,C}$ is the total biomass of catch of demersal species *j* for the year from the cell.

Biomass ratio of small- : large-bodied fish species

The ratio of small bodied to large bodied finfish species in the catch in each cell in each year ($B'_{SL,C}$) is calculated using:

$$B'_{SL,C} = \frac{\sum_i B'_{i,C}}{\sum_j B'_{j,C}} \tag{5.3}$$

where $B^{'}_{i,C}$ is the total biomass of catch of small-bodied (maximum standard length <30cm) species i for the year from the cell; $B^{'}_{j,C}$ is the total biomass of catch of large-bodied (maximum standard length $\geq$ 25cm) species j for the year from the cell.

Biomass ratio of invertebrates : finfish species

The ratio of invertebrates to finfish in the catch in each cell in each year ($B'_{IF,C}$) is calculated using:

$$B'_{IF,C} = \frac{\sum_i B'_{i,C}}{\sum_j B'_{j,C}} \tag{5.4}$$

where $B^{'}_{i,C}$ is the total biomass of catch of invertebrate species i for the year from the cell; $B^{'}_{j,C}$ is the total biomass of catch of finfish species j for the year from the cell.

CSIRO Cruise data – Benthos Classification

Survey trawl start and end positions are converted to line features and the relative length of each trawl calculated using the same technique as for the FWA logbook data discussed above. The mean number of occurrences of benthos in size class i (H'_j) for each spatial cell is calculated using:

$$H'_i = \frac{\sum_k \frac{\sum_j H'_{j,i}}{n_k}}{n} \tag{5.5}$$

where the two size classes are small (<20cm tall) and large ($\geq$ 20cm tall); H'_j is the number of benthos occurrences in observation j at station k; n_k is the total number of observations at station k; and n is the total number of stations in the spatial cell under consideration.

Calculations are done once for the whole time series for the one sixth of a degree analysis units, as the spatial resolution of the available data was not adequate to generate annual statistics. Further, due to the lack of an adequate time series it was not possible to do a direct comparison to fisheries data to identify any relationships at this spatial scale. Instead inferences had to be drawn based on the spatial overlap of overall fishing effort and mean habitat cover per cell (e.g. was mean cover lower where fishing pressure was higher?).

Bioregionalisation of the NWS

The final indicator tested on the NWS using GIS was the number of communities present. Aggregation of the available information to the one degree grid made little sense in this case, and so the map of Hierarchical classification level 3 - Biogeomorphological units from the NWSJEMS bioregionalisation exercise (CMR and DEP 2002) was used instead. The habitat map generated in this way is a bioregionalisation based on information from an analysis of fish distributions for the offshore region (deeper than 20m), and the best available topographic and bathymetry data for the coastal zone. Further details of the methods used are available in CMR and DEP (2002).

5.3.3 Visualisation of Indicators

Once datasets are transformed into a format suitable for use within the GIS, the indicators are calculated. Where time series data is not available (or where simple summary statistics are required) the data are mapped and overlaid to provide information on their spatial relationships. Indicators considered in this way are: (i) the proportion of the potentially fishable area (based on fishery licence areas) that is currently closed to fishing, and (ii) would be closed to fishing under the proposed additional marine protected area plan.

Where timeseries data are available, a range of visualisations (maps) are generated from different temporal breakdowns of the data – annual, monthly, weekly, and daily. From the fisheries data maps are generated for each time and data type. In general only the annual figures are displayed in map form here.

The NWS Vessel logbook data was used to generate maps for the total trawl length (effort), total catch (kg) and CPUE. While similar indices (number of operations (effort), total catch (kg) and CPUE) are considered using the CAES data, it was more usefully used to consider annual time series per block of: the total number of species (which is used as an approximation of diversity); the ratio of catch for pelagic and demersal species; the ratio of the catch of small- and large-bodied finfish; the ratio of invertebrate and finfish catch; mean trophic level and trophic spectra. The trophic indicators were repeatedly evaluated – using all species, top 10 species by weight of catch for each year, top 10 species by weight of catch for the whole time series, and only those species available for the whole series – but as these were all largely similar in result only the evaluation using all species is included here.

Animation of Time Series

Additional data exploration was undertaken through the generation of movie loops for some of the data types available in the logbook data. This was done for time series with a temporal resolution greater than one year (monthly, weekly, daily). Loops created at these temporal scales improve the ability to visualise time series with finer time steps. This would otherwise have been difficult with a large number static images or maps. The loops of spatial visualisations of the data were created for a number of time periods and then these were exported from the GIS and merged into Quick-Time movie streams using an encoder. Figure 5.5 shows samples of the still used in the generation of the movie loops.

5.4 Results

5.4.1 Simple spatial statistics

The fishery license areas, MPA boundaries and reserved areas are shown in Figure 5.2, for finfish trap and trawl fisheries, and Figure 5.3 for prawn fisheries (the three prawn fisheries managed in the region are Exmouth Gulf in the west, Onslow in the centre and Nickol Bay to the east of Dampier Archipelago). The total areas of each license type within each fishery and the total areas under the proposed MPA's are shown in Table 5.2. Approximately half of the area licenced for the finfish trawl fishery, and just over one fifth of that for the finfish trap fishery, is closed to fishing. For the prawn fishery, just over 1% is closed and a further 4.5% is identified as nursery areas and closed seasonally. As these values are calculated from the license areas, the actual areas fished are likely to be much smaller, due to the natural distribution of the target species. This is apparent in Figure 5.4, which shows the distribution of trawl shots for the full time series, with a significant proportion of shots being concentrated in the western to central-southern portions of the fishery.

In addition to the Ningaloo Reef Marine Park MPA that already exists on the NWS, a set of new MPAs have been proposed for Dampier Archipelago and the Barrow Island-Monte Bellos group. The analysis of the additional area that would become closed to fishing under these new indicates that it would only be a small percentage of the existing fishing grounds. The area covered by these MPAs is considerably smaller area than that closed under existing fisheries agreements. Only 2.4% of the finfish trap area and 3.5% of the prawn fishery area would be closed due to the institution of the MPAs and the trawl fishery would not be impacted at all. The size of these areas means that they are unlikely to impact upon total fisheries landings at all (though obviously there would be a great potential for localised effects). While these areas are small, from a non-industries perspective they may still be an effective measure for those species impacted by fishing if they cover larval source locations or habitats or species that either have limited distributions, or are threatened or endangered. Additional analyses of the full range and distribution of habitats and fish species present in the area would be required to identify and quantify these issues.

Table 5.2: Spatial Statistics for Trap, Trawl and Prawn Fisheries. Total area, closed area and percentage of each in proposed MPA's.

	Area			
Fisheries Details	**Total**	**Open**	**Closed**	**Nursery**
Finfish Trap Fishery				
Trap - Area (km^2)	115147	90104	25043	
Trap - % of Total		78.25	21.75	
Trap - % in MPA	3.78	2.40	8.72	
Finfish Trawl Fishery				
Trawl - Area (km^2)	113618	57390	56228	
Trawl - % of Total		50.51	49.49	
Trawl - % in MPA	3.83	0.00	7.74	
Prawn Trawl Fishery				
Prawn - Area (km^2)	116580	111352	1296	3932
Prawn - % of Total		95.52	1.11	3.37
Prawn - % in MPA	3.73	3.53	0.00	0.20

Figure 5.2: Map showing FWA Trap and Trawl Fishery boundaries, the proposed Marine Protected Areas's and the NWSJEMS study area.

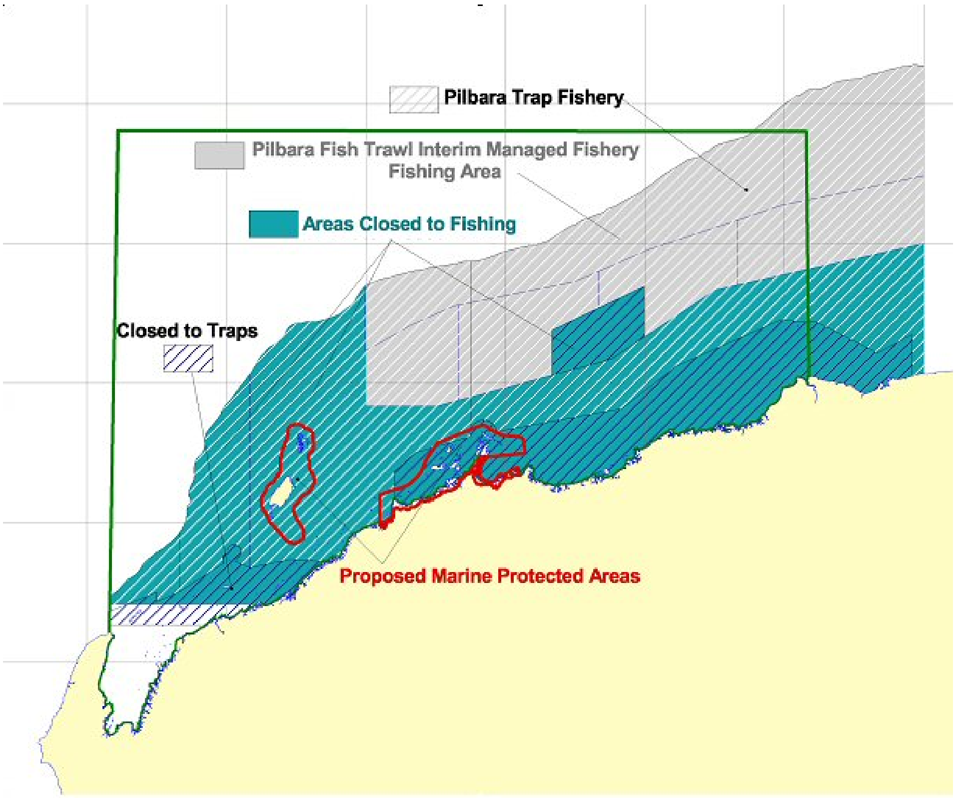


Figure 5.3: Map showing FWA Prawn Fishery boundaries and the NWSJEMS study area.

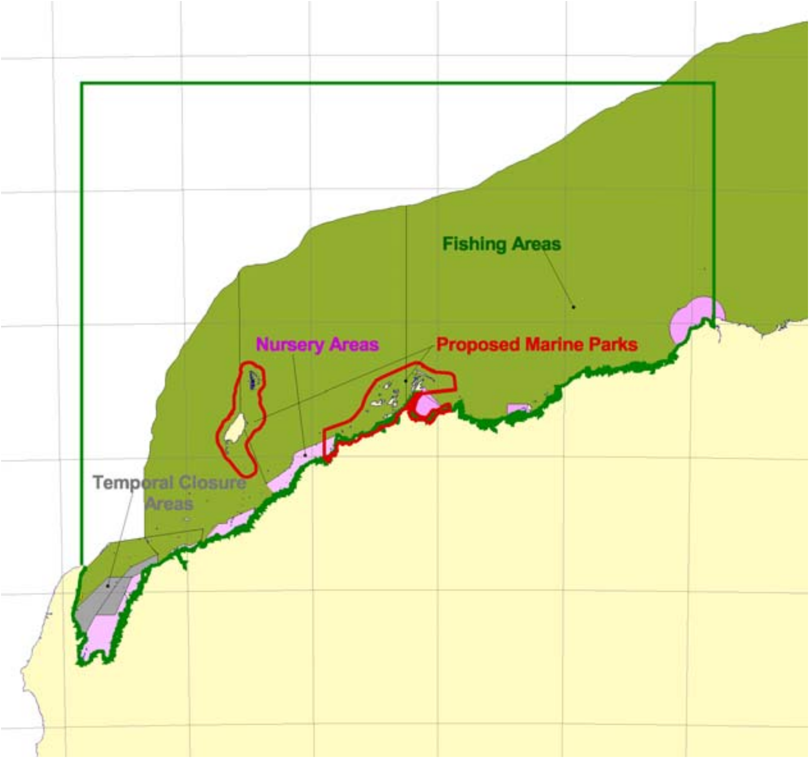
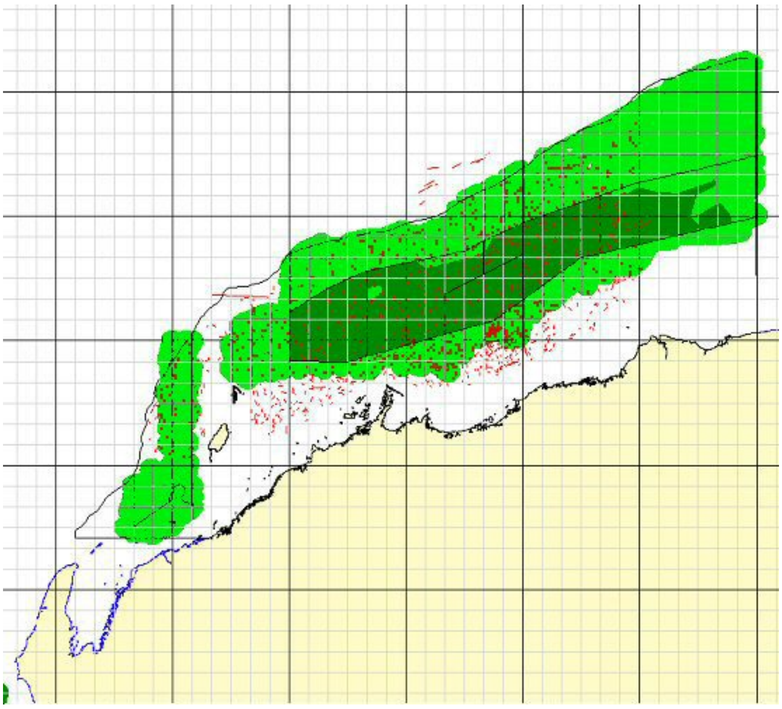


Figure 5.4: Map of FWA finfish fishery boundaries, and the location of logbook (green area) and research cruise (red tracks) effort. This gives a good indication of the spatial distribution and intensity of data available for analysis (light green indicates patchy cover and dark green solid cover by logbook tracks). Note that the edge of the light green areas is buffered by 10km to comply with commercial confidentiality requirements.



5.4.2 FWA Vessel Logbooks

Logbook catch and effort data is at the one sixth of a degree spatial scale for the range of time scales defined in section 5.3.3, In addition, animations for each type of data was undertaken for the daily time scale. Visualisation of indices calculated using fisheries data at this level of time and spatial detail is strongly recommended as it provides useful insights into the patterns of fleet behaviour that are not available when considering longer time periods or static maps. Unfortunately, while example snapshots are given in Figure 5.5 commercial confidentiality prevents the publication of further details of the analysis of the data on this scale for any data other than total annual catch. The time series of total annual vertebrate catch on the fine grid (Figure 5.7) shows that there has been an extension to the east of the area over which notable (>1000 kg) total catches have been taken. This is particularly true once there was effort displacement with the closure of the central zone 3 area. Perhaps more significantly, there appears to been a distinct fall and then partial recovery in the absolute magnitude of the total catches taken in each area. This time series would be more useful if a matching one for invertebrate total catch could be considered in parallel. Another useful variant would be a time series of overall (vertebrate and invertebrate) catch.

5.4.3 FWA CAES Data

CPUE

To overcome the issues of commercial confidentiality in the FWA logbook effort data it is aggregated annually and projected onto the one degree grid used with the CAES data. Combining this effort data with the CAES catch data produces the time series of annual CPUE per block given in Figure 5.6. There does not appear to be any strong trends in this time series, beyond the relatively consistent presence of high end CPUEs in those blocks covering the most westerly end of the licenced trawl fishery area (light grey area in Figure 5.2). While there is some suggestion of the dip in returns seen in the total catch time series (Figure 5.7) it is not very clear.

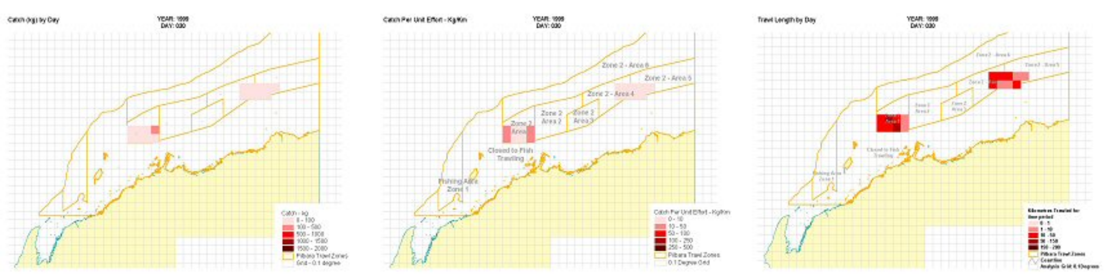
Diversity (total number of species)

The total number of species caught in each block (Figure 5.8) shows a general increase through time. In particular, there is a substantial increase in the numbers of species caught inshore beginning in the late 1980s. The increase may be due to a combination of better reporting of species over the period, a broadening of the range of species being caught in the fishery, or a shift in the fish community structure. While there is insufficient data to rule out any of these hypotheses the first one is most likely. The static nature of the number of fish caught in the most offshore boxes is probably due to fairly light exploitation levels at those locations (Figure 5.4).

Pelagic:Demersal Biomass ratio

The ratio of pelagic to demersal species for the region over the full time series (Figure 5.9) shows that across the entire area the catch is generally dominated by demersal catches (which are between two- to five-fold larger than catches of pelagic groups). This is perhaps not surprising given that catches from demersal trawls are under consideration here. What is noticeable though is that with the reduction in the Taiwanese trawl fishery and the establishment of the domestic trawl fishery in the early 1990s there was a marked switch in the patterns of catch generated overall by the NWS fisheries. In the decade prior to 1993 the ratio in each cell is usually between 0.2 and 1, though more extreme values are occasionally present, particularly inshore or around the Dampier archipelago. After 1993 extremes are much more common and the fishery seems to have an almost bi-modal pattern across the entire area.

Figure 5.5: Example frames from three animations of different daily time series generated from FWA logbook data (snapshots all from day 30 of 1999): (a) daily total catch, (b) daily CPUE, and (c) daily trawl-length (effort)



A: Daily Total Catch – 1999, Day 30

B: Daily CPUE – 1999, Day 30

C: Daily Trawl-Length – 1999, Day 30

Figure 5.6: Time series of maps of the CPUE of species per block. Effort data from FAW logbooks and catch from FWA CAES data.

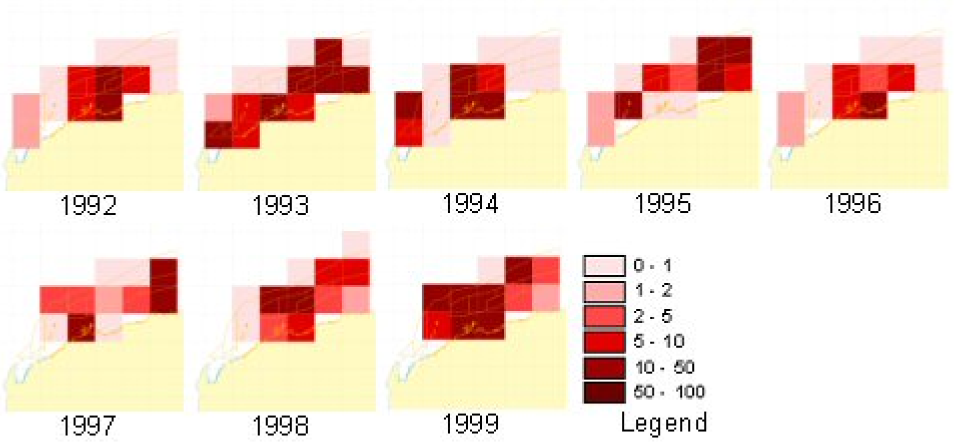


Figure 5.7: Time series of maps of the total catch per fine grid cell (from FWA logbooks).

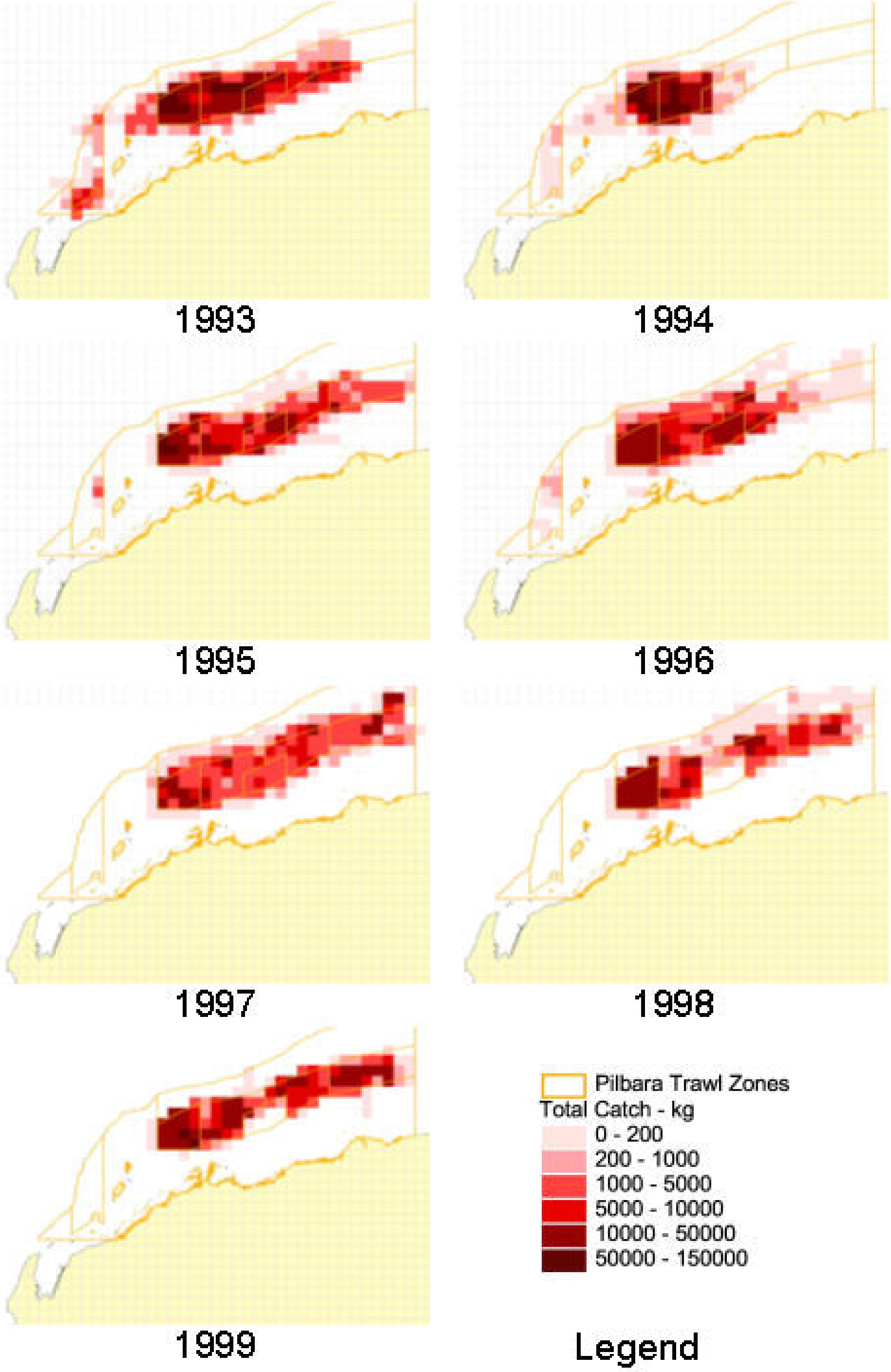


Figure 5.8: Time series of maps of the total number of species per block.

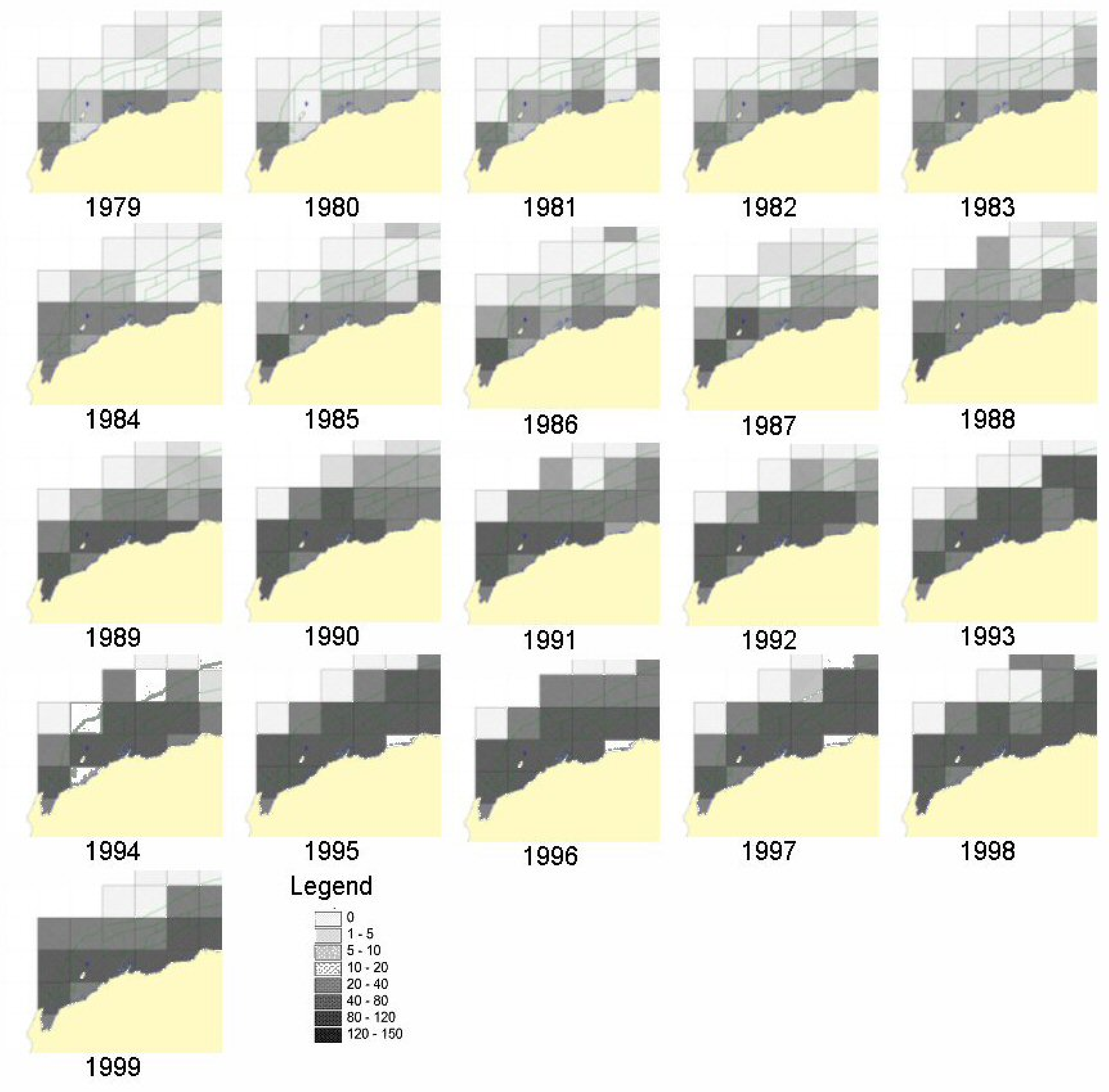


Figure 5.9: Time series of maps of the ratio of pelagic to demersal catch by catch weight per block.

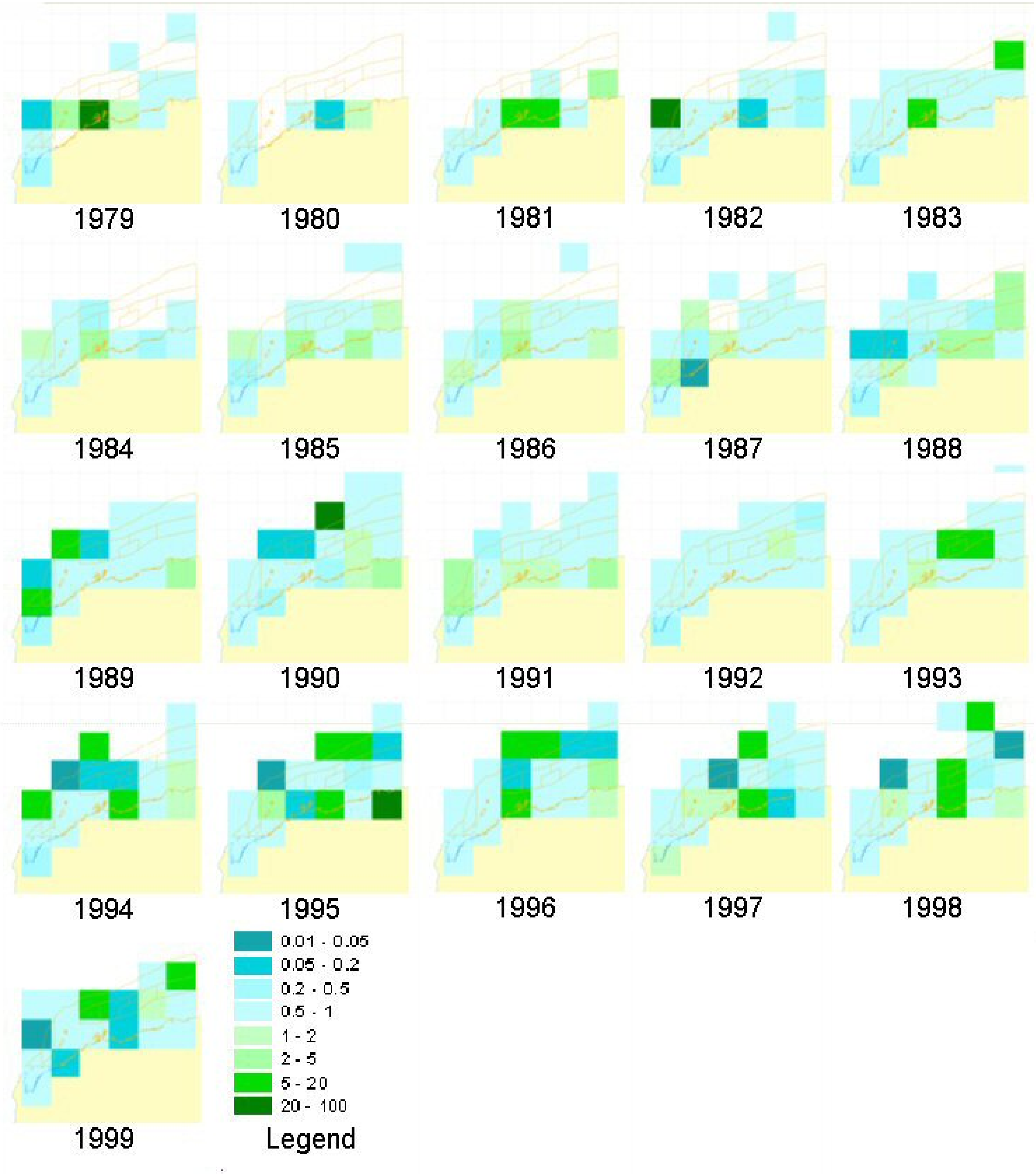
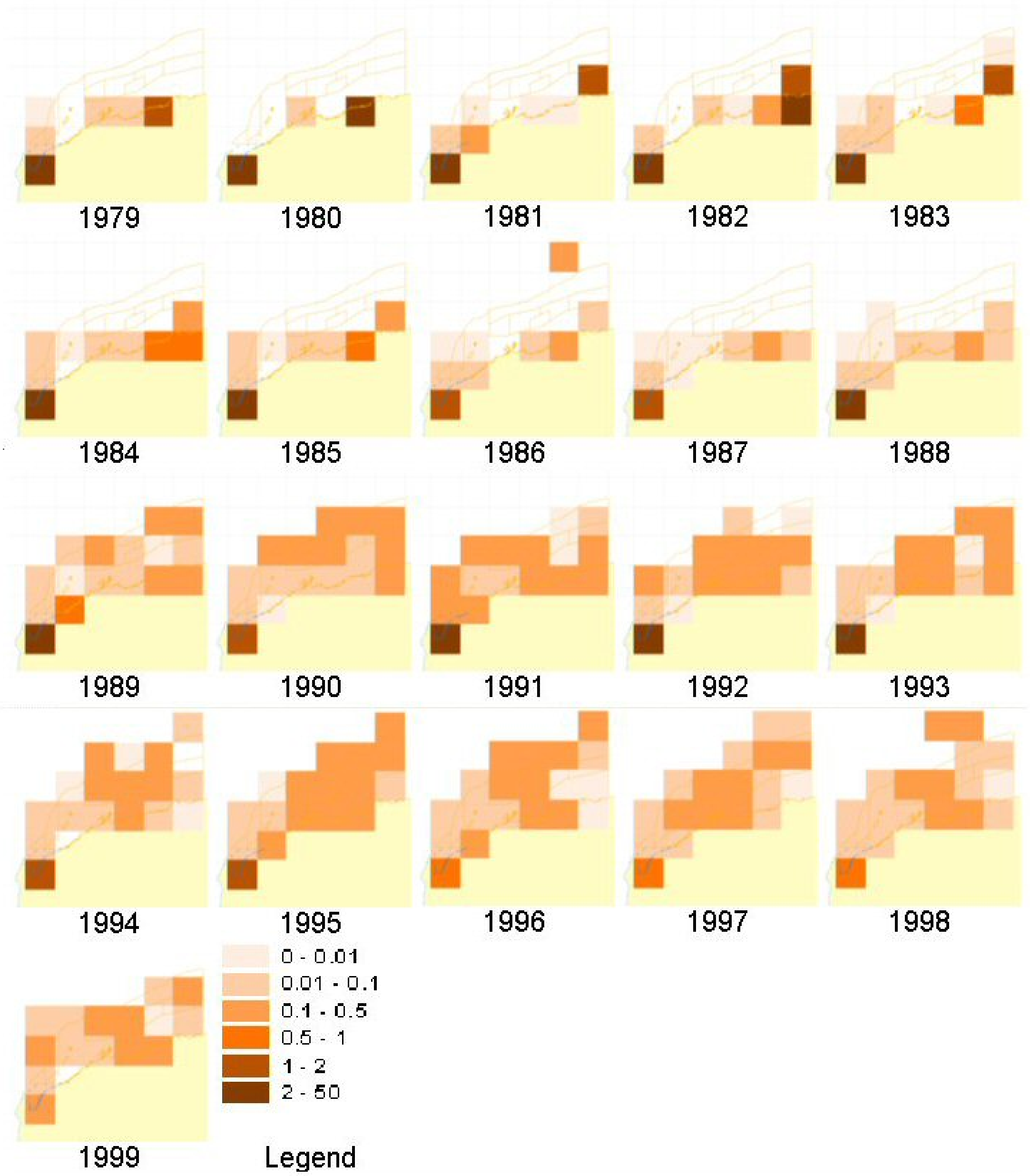


Figure 5.10: Time series of maps of the ratio of small-bodied to large-bodied species by catch weight per block.



Small-:Large-bodied Finfish Biomass Ratio

The ratio of small bodied to large bodied species (Figure 5.10) shows a general trend towards smaller bodied fish through time for the central portion of the study area, with some degree of stabilisation in the mid 1990s. What this pattern of results is highlighting is the shift in finfish community structure associated with the targeting of large benthic habitat associated finfish by the demersal trawl fisheries (also observed by Sainsbury 1987, 1988).

For the eastern and western ends of the area (De Grey River and Exmouth Gulf respectively) there is a reversal of the trend seen in the centre, with initially high rations of small bodied species, tending towards large bodied species (after 1984 in the east and after 1993 in the west). This suggests that the balance between prawn and finfish fisheries in these two sites have shifted through time, as the total catch by the Exmouth prawn fishery declined, particularly between 1986 and 1999 (FWA, 2000)

Invertebrate: Finfish Biomass Ratio

The ratio of biomass of catch of invertebrates to finfish (Figure 5.11) shows that there is a consistent preference for invertebrates inshore – particularly around Exmouth Gulf and Nickol Bay, and to a lesser extent the DeGrey River – and finfish offshore. A temporal trend is also evident in the inshore blocks, with the high ratios along most of the coastal zone for the period 1988 to 1991 and retracting to the west from 1992 onwards. With regard to the use of this indicator to track wider-scale shifts in overall community composition it is interesting to note that there is no shift to higher proportions of invertebrates in the central offshore cells in parallel with a shift to smaller-sized finfish offshore seen in Figure 5.10. This suggests that any shift in the community due to fishing either does not extend to the invertebrate species, or that any such shift is not reflected in the landed catch of fisheries that have traditionally targeted finfish and may be reluctant (or incapable for economic or licencing reasons) to switch to invertebrates instead. Without further information on historical drivers of targeting in the fisheries and fisheries independent data that includes invertebrate and finfish species it will be hard to verify the extent of community shifts due to fishing in the different areas.

Average Trophic Level

Figure 5.12 shows the mean trophic level for each region in each year. Values are generally higher offshore and in the northern region for the entire period. The influence of the prawn fisheries is evident in the consistently low average trophic levels around Exmouth Gulf and other inshore areas. More importantly there appears to be a two-part pattern of decline through the period. The first part starts with higher average trophic levels in 1979 and declining through time to 1991-1992, there is a sudden increase before a second decline between 1997 and 2000. The two periods of decline may indicate some degree of "fishing down the foodweb" (*sensu* Pauly *et al.* 1998) interrupted by a change in targeting due to the cessation of the foreign trawl fishery and the development and establishment of a domestic trawl fishery at that time. Unfortunately, consideration of detailed catch composition information and fisheries independent diet and community composition data (unfortunately neither of which is available in the detail required for the entire period) would be necessary to verify or contradict this explanation for the observed pattern in average trophic level.

Figure 5.11: Time series of maps of the ratio of invertebrates to finfish by catch weight per block.

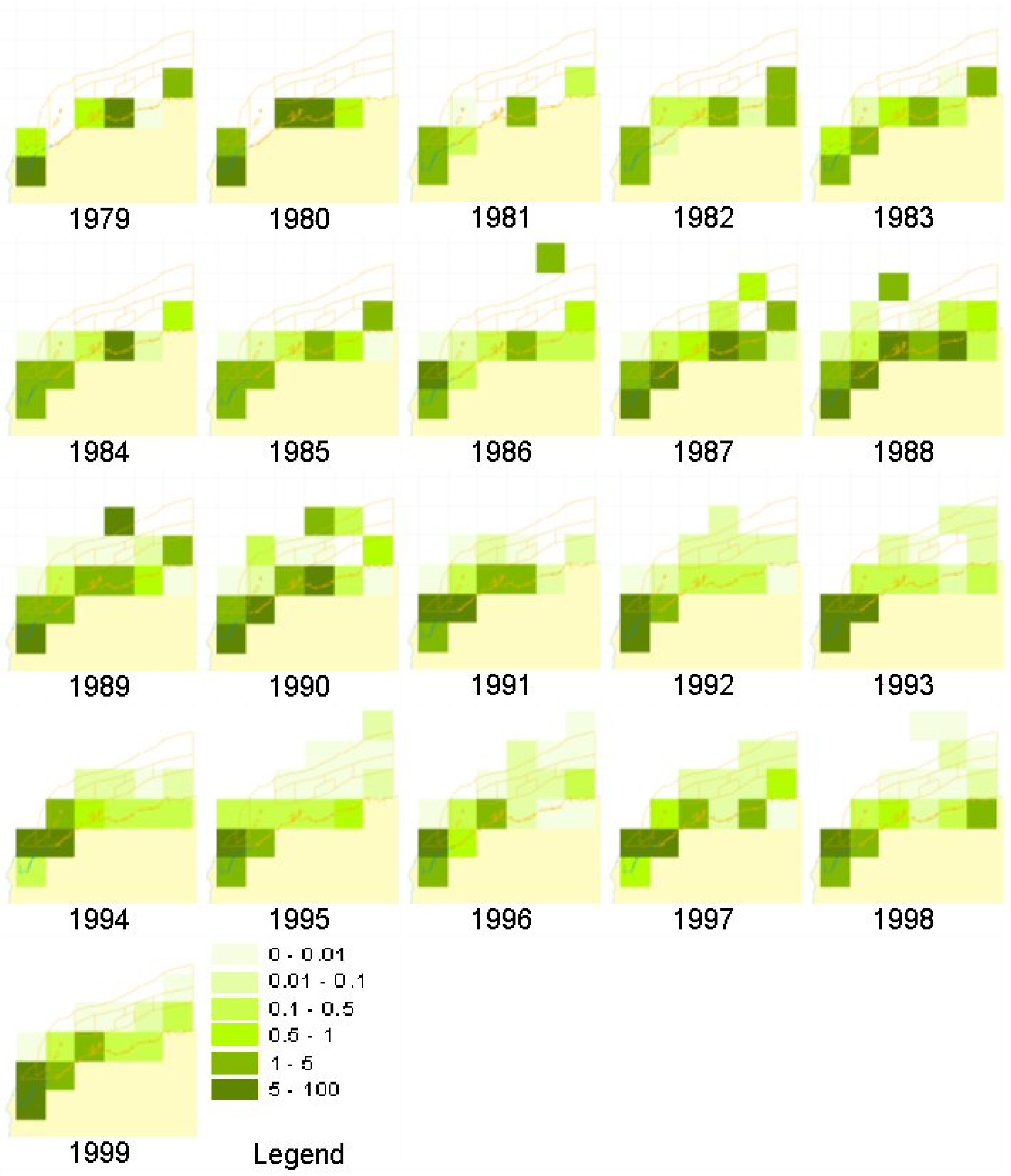
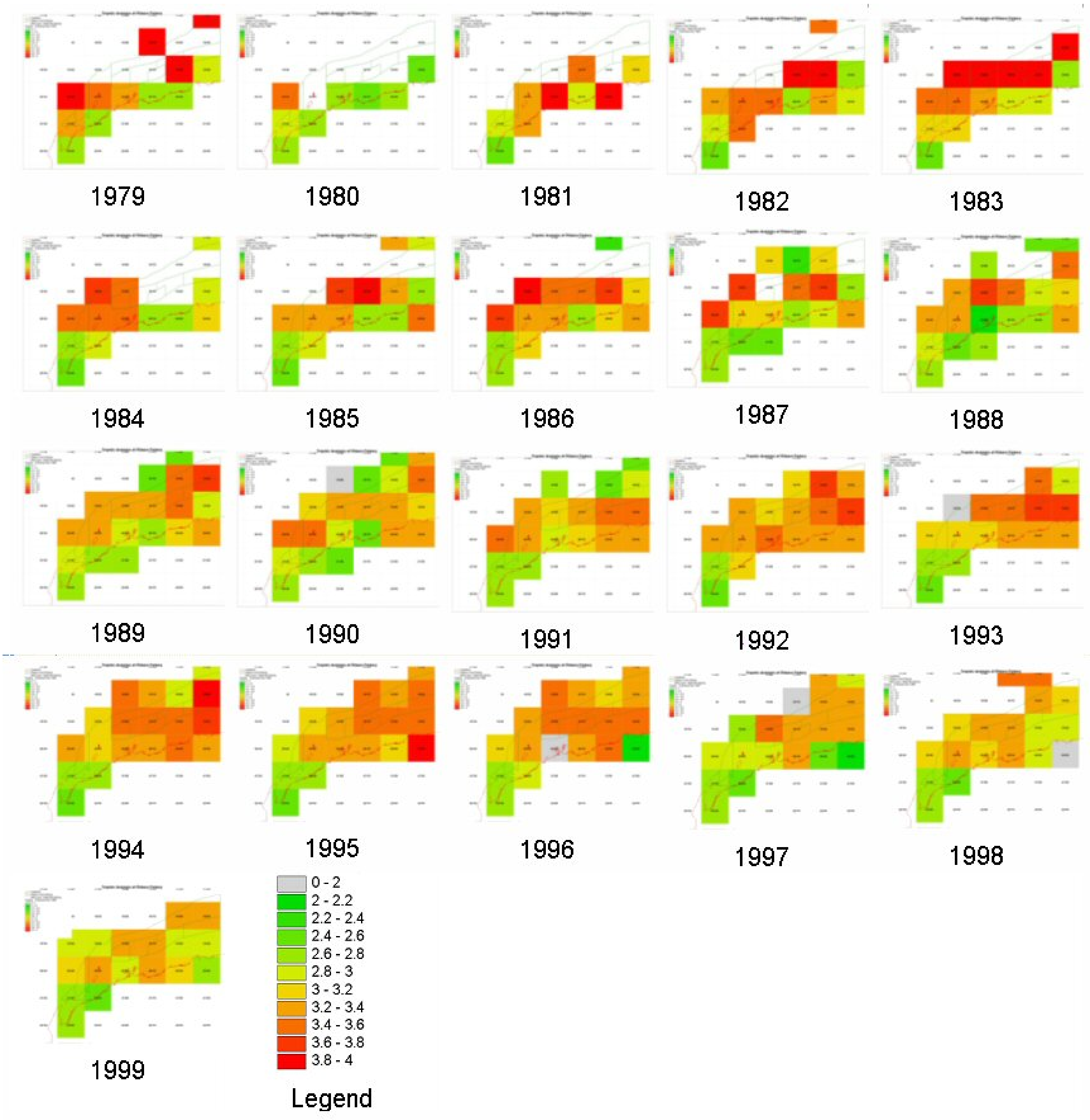


Figure 5.12: Time series of maps of average trophic level per block.



Trophic Spectra

The trophic spectra displayed in Figure 5.13 detail the changes in the trophic structure of the catch for specific regions through time. There is no consistent pattern in the change in these spectra through time across the entire area.

Considering the offshore blocks there does seem to be a clear shift to peaks at lower trophic levels in blocks 20140, 19160 and 18190. There is also some degree of shift of the spectra toward lower trophic values in blocks 20150 and 19170, but the stronger signal there is the overall flattening of the spectra. In contrast, the last two offshore blocks (19180 and 19190) both display shifts of the peaks in the spectra to higher trophic levels.

Turning attention to the inshore blocks, area 21140 (which covers a large area of Exmouth Gulf) stands out as different to the other inshore blocks. It has a clear trend towards much strong peaks at lower trophic values through time. The other four inshore blocks all have bimodal spectra that appear to fluctuate between (i) a strong peak in the lower trophic levels with a secondary peak at higher trophic levels, and (ii) a strong peak in the higher levels and a secondary peak at lower levels. While the exact form of these curves is not identical across the inshore blocks, the fluctuation between the two broad types does seem to be coincident at the different locations and may reflect some form of environmental forcing (e.g. cyclonic influence on prawn recruitment).

While these spectra give a community level perspective to the changes in fisheries catch through time their shape alone is not enough information to determine what is occurring in the ecosystem (particularly when based solely on fisheries data). A more detailed understanding of the drivers behind the changes in the trophic spectra requires information on catch make-up and fisheries targeting behaviours. While this extra step was not taken here, it is strongly recommended that any future application of these indices should perform such an analysis.

5.4.4 CSIRO Cruise Data - Benthos

The maps in Figure 5.14) show the mean number of occurrences of small and large benthos per trawl. There is a definite difference in the numbers and distribution between the two size classes. Large benthos have a relatively limited distribution with three areas of higher density. The main area is north-east of Nickol Bay, and another offshore in a line tangential to the shoreline north-east of Barrow Island. The third area is more dispersed, and is offshore of the DeGrey River. The small benthos have a broader distribution with higher relative numbers across the whole study area. Notably, there are substantially higher numbers at similar points to the peaks for the large benthos. This may be an indication of general habitat suitability or health over the region, but is likely to be at least partly due to an association between the benthos types.

Additional information may be obtained by performing a comparison between mean numbers of benthos, and the trawl density for the region (Figure 5.15). This kind of comparison may provide insight into a possible between trawling and benthos height, with trawling causing habitat damage and removal of large benthos and so favouring the occurrence of small benthos. While it is noteworthy that the peaks in the occurrence of large and small benthos largely lie outside the most intensively fished areas, it must be cautioned that depth and substrate preferences by the benthos species may also be contributing (at least in part) to their pattern of distribution.

Figure 5.13: Trophic spectra (normalised weight per trophic level) from 1979 to 1999 for the FWA statistical regions.

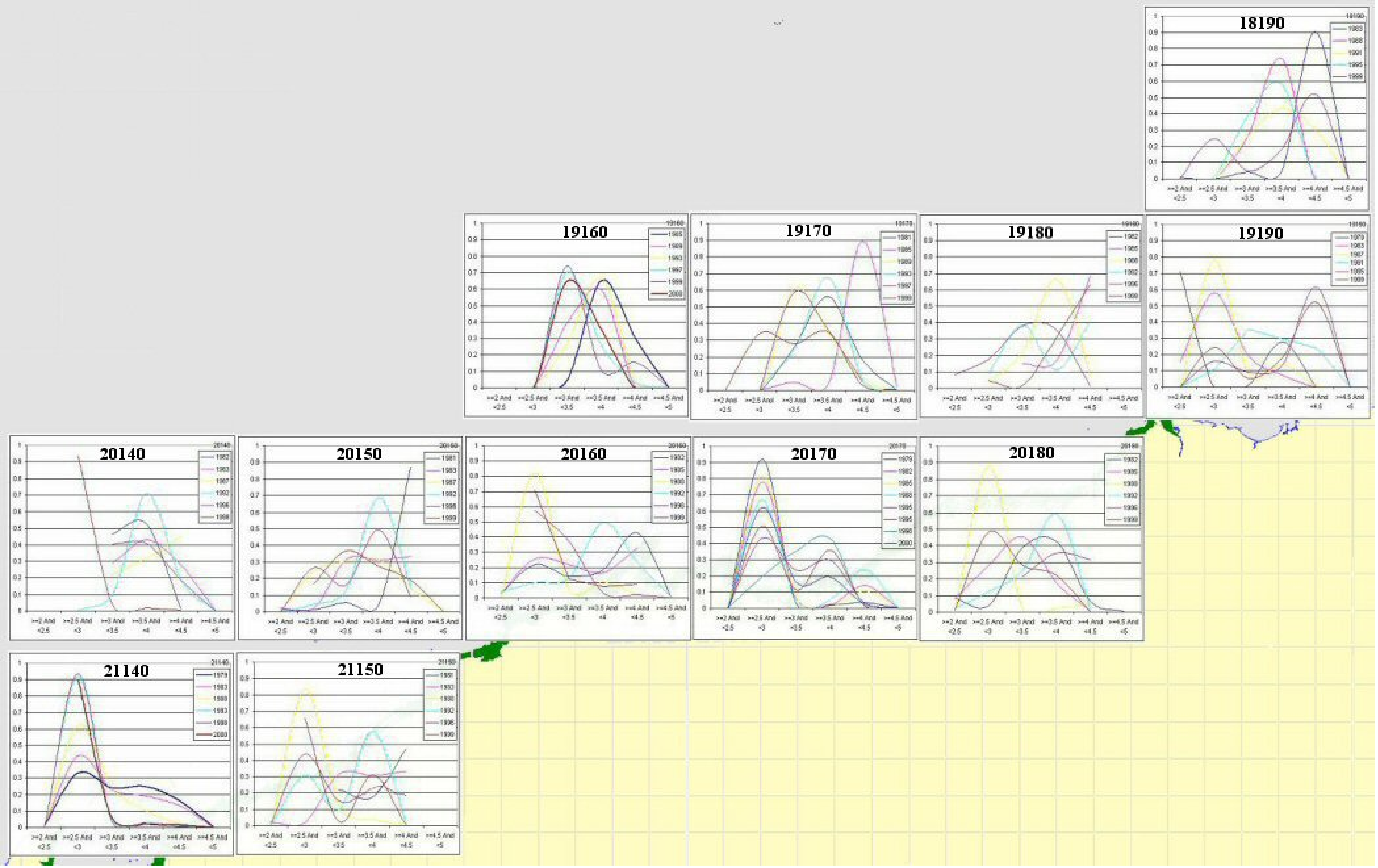
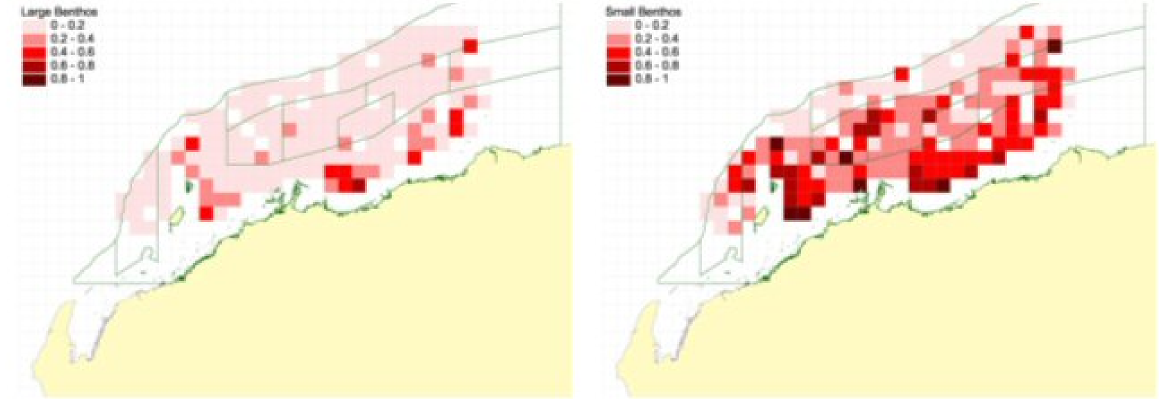


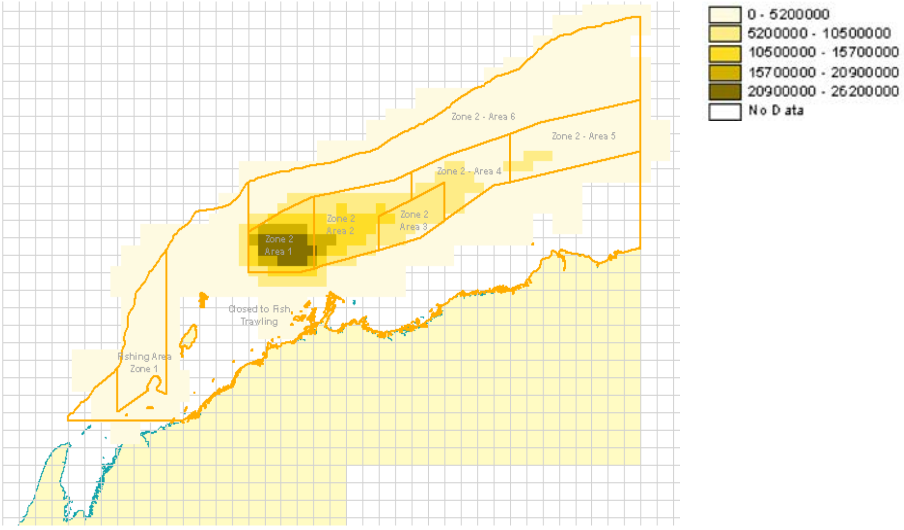
Figure 5.14: Mean presence and absence of large and small benthos from CSIRO research cruises.



A: Mean of large benthos presence/absence per trawl

B: Mean of small benthos presence/absence per trawl

Figure 5.15: Total trawl length (m) for whole time series.

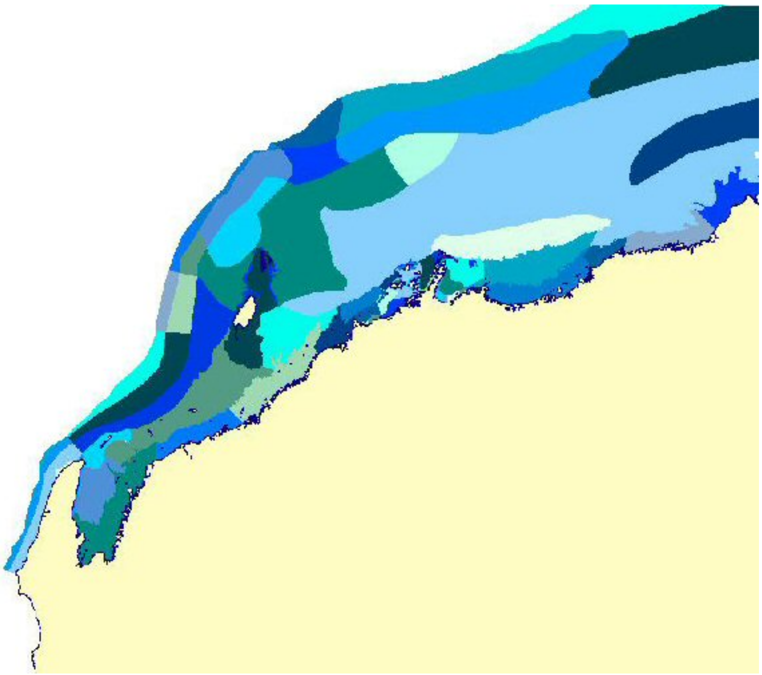


An initial time series was developed for the benthos classes, however the sparseness of the trawl track coverage compared to the analysis cell size adopted meant the coverage was too small to map effectively. A potential means of addressing this problem in the future would be to repeat the analysis at a broader spatial scale, or at a coarser time resolution, so that adequate numbers of trawl data points lie within each analysis unit.

5.4.5 Bioregionalisation of the NWS

Figure 5.16 is the community-habitat-bathymetry map generated by the bioregionalisation exercise for NWSJEMS (CMR and DEP 2002). There are 80 different habitat units for the study region at this level of biogeomorphological units. The most interesting feature of this map is the coincidence of the large homogeneous units across the areas most heavily fished during the last few decades and much smaller scale units elsewhere. This suggests that fishing may be homogenising the system, though without additional analysis it must be cautioned that (as for the distribution of the benthic habitats) physical characteristics (e.g. sediment type) may also be contributing to this distribution of units.

Figure 5.16: Map of the level 3 biogeomorphological (community-habitat-bathymetry) units for the NWS study area.



5.5 Discussion of GIS Analyses

While data limitations have ultimately constrained the list of indicators that can be evaluated, GIS has been successfully (and effectively) applied here in the calculation and consideration of a number of key ecological indicators of the effects of fishing. While there are limitations to the simple indicators considered here, the same would be true of more complicated indicators too – which would also have the added issue of actually calculating them given the data available. It is reassuring that the general patterns of performance seen in the simulation studies are supported by the GIS results, with simple community level indicators providing potentially useful ecological insights, while CPUE indicators were not particularly informative. Admittedly, total CPUE was used here, rather than group or species level CPUE. It remains to be seen, for the NWS at least, if the usefulness of CPUE indices as ecological indicators improves when it is disaggregated in that way.

Importantly, existing, “off the shelf” software has been used for all the data manipulation, analysis and visualisation. With a small amount of customisation, through the native application development language, the software is able to generate a range of ecological and fisheries statistics at various spatial and temporal scales from the data available. Although no further analyses were considered here, additional detailed spatial analysis could be performed via an interface between GIS and statistical packages such as R or S-Plus. The ability to perform more robust statistical analyses on almost any spatial unit the user requests, should provide a very powerful tool for the application of ecological indicators. As most GIS packages have some form of application development language (VB, ArcView-Avenue) they can be provided with the ability to create powerful tools for data analysis for general desktop use, within the reach of fisheries managers, as well as scientists (FishCAM would appear to be a good applied example of this).

The application of ecological indicators to the NWS, relying on existing real data sets, has not been as comprehensive or thorough as would have been preferred. It has however highlighted the potential drawbacks facing the application of ecological indicators of any degree of complexity to large spatial extents or remote areas. First, there is the issue of commercial confidentiality, particularly in fisheries with only a few operators; while this may mean that publishing indicator values may not be possible (as was the case here) it should not be used as a reason not to investigate such indicators in the first place. A far more important issue that the GIS analysis has highlighted is the issue of fisheries independent data.

The Northwest Shelf of Australia has had relatively low human impact from the terrestrial environment, and has a low local human population level. As a result, obtaining adequate ancillary data to support the application of ecological indicators across a range of indicator types has proven difficult and it was necessary to rely almost solely on fisheries data for consideration of the indicators. The lack of suitable fishery independent indicators makes the outcome inadequate for fully evaluating fishing impacts. This problem will be true of any system in which fisheries independent data are scarce. The analyses presented here do show that simple indicators based on fisheries data can go some way to showing whether shifts in communities and ecosystems may have occurred, but without fisheries independent data (or detailed data on fisher behaviour) it is hard to verify whether the shifts seen are a result of ecological changes in the ecosystem or to changes in fleet dynamics and targeting preferences.

The analysis of a range of datasets has also shown some shortcomings in the approach of using GIS. A lack of adequate spatial or temporal coverage is difficult to overcome, for example in the use of the benthos data. One alternative would be to aggregate either spatially or temporally to provide enough coverage to compare to the other datasets with better coverage. The decision about which scale to aggregate over will depend upon the questions being asked, and the types of data available. Techniques such as kriging (a statistical technique using variograms to interpolate between points in a heterogeneous grid) may be used to fill data poor areas, but an understanding of the nature of the environment being studied is required in order to appropriately apply these techniques. While spatially resolved information can be useful, a single overall value for the chosen indicators may be the more judicious choice in many data poor cases.

One way in which more fisheries independent data could be incorporated into similar studies in the future would be to call upon remotely sensed data. Many terrestrial applications of ecological indicators have taken advantage of this approach – as a range of attributes can be derived from remotely sensed data and often for extended time periods (e.g. Landsat from 1972 onwards). Additional use of aerial photography can also extend the time series over which indicators can be evaluated. While similar applications of remotely sensed ancillary data are not as common in the marine realm, remote sensing techniques are becoming more appropriate for applications to the marine environment (Fréon *et al.* 2004). The kinds of data which could be used in this way include ocean colour, turbidity, Chlorophyll *a*, altitude and sea surface roughness, and aerial photographs of bottom habitat (though this last type of data tends to be limited to shallow waters). Other remote sensing techniques under development for the marine environment include sonar and acoustic measurements, which are beginning to approach the levels of detail required for this sort of analysis (Davies *et al.* 1997).

Additional approaches and indicators, such as spatial structure, which have been applied terrestrially, and have been identified as possibly suitable for fishery application, often require data at a spatial resolution not currently available for many marine and coastal waters. Many of the spatial structure indices described by Fragstats (McGarigal and Marks, 1995) require a level of detail for habitat mapping that is generally not available for marine regions; and associated time series data that can provide an indication of change or impacts are even rarer. Fragstats was designed for use in studies of terrestrial ecology and as such provides a number of indices on the spatial structure and diversity of a region, as well as on the spatial relationships and characteristics of habitat "patches" at the local, regional and study-system scale. There may be the potential to apply the analysis methods in Fragstats to the narrow coastal band where remote sensing techniques can be applied across the spatial scales required, so that adequate time series at those scales can be constructed.

Once suitable data have been obtained and entered into GIS data structures, the range of spatial analyses and overlays possible is quite large. While the type of results obtained from the sort of first order spatial analysis applied here tends to be fairly simplistic, the ability to generate the statistics relatively easily and quickly is very beneficial. Spatial summaries such as relative area calculations for combinations of data layers (e.g. areas of prawn fisheries in proposed MPA's) provide informative background for additional, more detailed analyses. For example, the simple spatial analysis giving the areas currently closed to fishing as a percentage of the licenced area indicates that a high proportion of the area is closed to fishing at present. However, while interesting in itself, further consideration of these statistics is required, as this information may be misleading. Without suitable information on fish distributions, or habitats within the fishery, an incomplete picture is portrayed by such simple indices. Additional information on fish distributions would provide the relative area for species within the license areas, and give a better understanding of the potential protection provided to a species by the system of MPAs. Likewise, information on habitats would also give an indication of the degree of protection of habitat types. These analyses would provide a better understanding of the status of habitats within the study area. For example information on the relative areas of each habitat type would allow better planning of license area closures or MPA locations and design.

This message of the need for simple indicators backed up by a more detailed understanding of the system is perhaps the most important message to come out of this real world application of ecological indicators. Simple indices are attractive as they tend to be robust, are easy to calculate, and usually convey important information about the ecosystem (or at least components of the ecosystem). However it is vital that their values be put in context. This can be done by employing a suite of indicators using different data and targeting different parts of the ecosystem. Such an approach was not used to its full potential here, but even the simple picture that could be constructed (and in this case verified by people with decades of experience in the ecosystem) gave useful insights into the changes observed in the system over time and how they may be related to fishing (and other) pressures.

6 DISCUSSION

6.1 Performance of indicators - in the GIS and simulation studies

To garner a full appreciation of the potential performance of various ecological indicators when applied in reality (given current constraints on data collection in Australia), it is important to compare the performance of indicators analysed in the simulation and GIS studies. Table 6.1 lists the consistency in performance of the indicators in the two types of analysis. Reassuringly, performance was similar under the two methods – those that performed well in one performed well in both, those that performed poorly in one performed poorly in both, and those sensitive to data quality in one proved to be sensitive to data quality in both. Across the two methods of analysis, indicators do have a slightly stronger performance in simulation modelling than in the GIS study; this is due to the longer time series and additional fisheries independent data available in the simulation study. If the simulation data are subdivided to consider time series and levels of fisheries independent data of the order available to the GIS analysis, then the absolute performance of the indicators in both studies is very similar. This suggests that the results of the simulation are a good reflection of the likely performance of the various indicators in reality. It also reinforces the general findings common to the two studies – that simple community-level indicators will probably be most effective and that management involving the use of ecological indicators will benefit from the establishment of unfished reference areas and the collection of fisheries independent data.

Table 6.1: Consistency of performance of indicators considered in the simulation and GIS analyses; where a star ranking is used to reflect the relative consistency of the indicators ('0' = no consistency through to '****' for very similar results in both analyses)

Indicator	Overall performance	Consistency of performance across GIS/simulation
Average trophic level of catch	Recognizable	*** for signal strength **** for data quality sensitivity[1]
Diversity	Recognizable (potentially)	*** signal strength is recognizable or better in both, but in GIS this may actually be due to better reporting rather than measured ecological change
CPUE	Recognizable (potentially)	** for signal strength **** for data quality sensitivity[1]
Cover (or biomass) of epibenthos	Strong	****
Invertebrate:Finfish catch ratio	Recognizable	**** for signal strength **** for data quality sensitivity[1]
Maximum length of catch[2]	Strong	****
Number of communities	Recognizable	****
Pelagic:Demersal (catch biomass ratio)	Strong	****
Total catch	Recognizable	*** for signal strength **** for data quality sensitivity[1]
Trophic spectra of catch	Recognizable	*** for signal strength **** for data quality sensitivity[1]

1. Without fisheries independent data this indicator may be uninformative/weakly recognizable.
2. In the GIS study the small:large bodied species biomass ratio was used as a proxy for maximum length of catch

6.2 Guidelines for selection of suites of indicators

The many studies and reports dealing with the frameworks and criteria used to judge the utility of ecological indicators of the impacts of any monitored human activity (be it fishing, eutrophication or anything else) have all come to the same conclusions – indicators must be evaluated objectively and ultimately a suite of indicators will be required (Landres 1992, Rapport 1992, Jackson *et al*. 2000, ICES 2001, Tegler *et al*. 2001, Rochet and Trenkel 2003, Rice and Rochet 2004, Rochet and Rice 2004, Cury and Christensen *in prep*). The simulation testing of indicators in this study is an ideal means of evaluating the potential diagnostic utility of indicators and (perhaps unsurprisingly) it has clearly demonstrated that a suite of indicators is not only preferable, but is fundamentally required for the detection of the ecological impacts of the effects of fishing. Relationships between attributes of interest in an ecosystem mean that no single indicator (or type of indicator) can provide a complete picture of the ecological state of the ecosystem. Rather, a suite of indicators is required, each focusing on different attributes, using different kinds of data, and spanning groups and processes with fast and slow dynamics. If such a suite is employed then there is the greatest chance of successfully detecting shifts in the ecosystem's ecological state, whether from desirable-to-undesirable (from society or industry's perspective) or vice versa. The performance, in any one scenario, of the entire set of indicators considered here shows why the use of large numbers of indicators will be problematic not only from a cost standpoint but also because they can be incredibly hard to interpret – giving contra-indications and being of varying real use. This issue has previous been acknowledged by researchers attempting to develop frameworks for indicator evaluation (Rice and Rochet 2004). Thus, the need for efficiency in the suite will mean that the number of indicators included will have to be constrained, but the twin need for reliability means that indicators tracking only a limited number of attributes that are supposedly correlated with other attributes of interest may be ill advised, and that a small degree of potential redundancy may be needed to ensure that a complete picture is obtained of the system in all situations. Moreover, as indicators respond to change in the ecosystem's ecological properties, and are not tied solely to any one cause of change, the chosen suite will need to include indicators that may seem to be superfluous at first inspection – for example the inclusion of Chlorophyll a in a suite constructed to monitor for fishing effects. This is because these indicators will be needed to help identify the true cause behind any observed shifts in the ecological properties of an ecosystem – no manner of fisheries management will solve a problem being caused by coastal pollution, for instance.

While suites of indicators are necessary because no indicator can be guaranteed to cover everything and to be informative in all situations, it is possible to put forward the minimum requirements of an effective suite of indicators and to identify a list of the most effective and robust indicators to use for each attribute, and their associated reference directions (similar in concept to reference points for population-level indices already used in fisheries; Link *in press*, Shin *et al. submitted*). As animals or physical properties of ecosystems are directly measured, it may be most informative to identify the main biological groups (these may be species, but are preferably functional groups or guilds) that need to be included in an indicator suite. These groups are:

a) groups with fast turnover rates (e.g. phytoplankton, zooplankton and bacteria) – these respond quickly to any ecological changes in the ecosystem and react sufficiently fast to be potentially useful as early warning indicators (unfortunately the non-selective responsiveness of these groups means they can give "false positive" results);

b) groups that are the targets of fisheries – these can usefully summarise the current state and structure of that part of the foodweb of most interest to humans and fisheries;

c) habitat defining groups (particularly in coastal ecosystems); and

d) charismatic (or sensitive) groups (often found at the highest trophic levels in the system – these groups (which may be individual species in this case) usually have slow dynamics and can convey information about the underlying ecological state of the ecosystem, and provide a long-term measure of how heavily the ecosystem has been impacted by fishing.

Using a suite that deals with carefully selected indicator groups from these categories will achieve the best combination of early signal detection and system characterisation. By combining information from groups across these categories, a suite should also guard against false signals, as it should capture a wider range of attributes and be less likely to suffer from data error and system dependencies. With the results of the simulation testing, GIS analyses and the categories of biological groups listed above in mind, guidelines regarding the most effective and robust indicators are given in Table 6.2. As with all guidelines there are a few caveats.

First, these guidelines are general in nature and are there to provide a starting point for the development of system specific suites of indicators. They are not intended to be a list of indicators that are slotted into field programs without further thought. Furthermore, the suggested indicators are not an exhaustive set. There are indicators within the list tested here that performed well but have been omitted from this recommended set. This was done on the basis that the recommended set should contain the minimum set of indicators needed to span the ecosystem attributes and that the ease of sampling must be taken into consideration (given the experiences we had with attempting to apply a short list of indicators in a GIS framework). Thus size based indicators were preferred to indicators reliant on good quality diet data. Similarly size at maturity is put before condition at maturity (especially as there is some question as to whether the strong performance of condition at maturity may be partly a model artefact (Carl Walters, UBC, *pers. com.*), though it has been observed to hold in some real world systems (Jae Choi, Fisheries & Oceans Canada, *pers. com.*)). While the performance of the recommended indicators in the simulation testing recommends them as potentially useful in real world systems, it is strongly advised (as might be anticipated) that indicators yet to be developed, or ones specific to the ecosystem of interest, be added to this list based on system-specific knowledge.

Second, the reference directions given are the directions of change for the indicator that would be considered undesirable in the majority of systems, but there is still some degree of system (or definition) dependency for some of the indicators. For instance, the undesirable reference direction for the biomass of demersal fish is given as a drop in Table 6.2, as this is the case for demersal fish in total (including site attached and long lived species), but if the biomass in question only refers to flatfish (say) then the undesirable reference direction would actually be an increase (Hall 1999, Link *in press*). Similarly, many of the reference directions for the network indices are actually more important relative to each other rather than in an absolute sense – as the ratios between total production, consumption, respiration and through put have ecosystem maturity implications (Odum 1971). Thus, these guidelines should be used as a

Table 6.2: Guidelines for indicators for each attribute of interest – where applicable, undesirable reference directions (arrows) are given for each recommended indicator, this is discussed further in the text. An asterisk denotes that there is some degree of system dependency in the reference direction.

Attribute	Indicator	Reference Direction (Comments on changes requiring further consideration)
Biomass (species, group, community)	Biomass of indicator groups or communities	
	Biomass of demersal fish	↓*
	Biomass of gelatinous zooplankton	↑
	Biomass of cephalopods	↑
	Biomass of habitat forming epifauna	↓
	Biomass of piscivores	↓
	Biomass of planktivores	↑
	Biomass of scavengers (e.g. scavenging epifauna)	↑
	Biomass of seagrass	↓
	Biomass of top predators	↓
Consumption (species, group, community)	Estimated consumption by indicator groups or communities	
Diversity (species, group, flow)	Simple (transparent) diversity indices	↓
Food web structure	Diet data (if good quality) – if good quality data are lacking, the indicators listed below for size structure and trophic level also give insight on the gross food web structure	Substantial shifts in web structure (membership or pathway strength)
Number of groups to represent 80% of the biomass (group, community)	Biomass of indicator groups or communities	↓*
Nutrient cycling	Estimated denitrification rate (particularly for shallow water ecosystems)	↓
	Dissolved inorganic nitrogen levels	↑

Attribute	Indicator	Reference Direction (Comments)
Nutrient cycling (continued)	Network totals – if good quality data are available	
	Total consumption	↑[1]
	Total net production (total primary production – total respiration)	↑
	Total production	↓[1]
	Total biomass	↓[1]
Production (species, group, community)	Estimated production by indicator groups or communities	
	Total primary production may be more useful if network structure information is lacking (total primary production is also often easier to determine directly)	↑*
Respiration (species, group, community)	Estimated respiration (from network models) – if good quality diet data and knowledge of the ecological structure of the ecosystem are available	↑[2]
	Indicator group totals (e.g. respiration by a dominant indicator group) – if good quality data are not consistently available	↑
	Estimated denitrification (if in shallow water systems)	↓
Size structure (species, group, community)	Biomass ratios	
	PS:ZP	↓
	pelagic:demersal fish biomass ratios	↑
	large:small plankton	↓
	infauna:epifauna	↑
	Maximum length of the catch	↓
	Size at maturity	↓
	Slope of biomass size spectra	↓
	W (from ABC curves)	↓
Spatial structure of biomass (species, group, community)	Best derived by mapping the indicators for biomass	Large-scale shifts (either expansion or contraction)

Attribute	Indicator	Reference Direction (Comments)
Trophic level (group, community)	Average trophic level (if good quality diet data are available).	↓
	Size-structure indicators are also useful (given the relationship between size and trophic level) – also more easily collected and less error prone	See size structure indicators
	Trophic value at peak of trophic spectra	↓
Throughput	Estimated throughput (using network models) – if good quality data are available	↓[3]
	Network totals – if good quality data are available	
	Total consumption	↑[1]
	Total production	↓[1]
	Total biomass	↓[1]

1. Reference directions for these values may be more important relative to each other than in an absolute sense.
2. Reference direction for Respiration may be more important in relation to total production as together they can be an indication of ecosystem maturity *sensu* Odum (1971).
3. Reference direction for Throughput is more important in relation to total biomass as together they can be an indication of ecosystem maturity *sensu* Odum (1971).

guide only, with careful consideration given to the identity of indicators used and any responses observed in each ecosystem monitored. The quite different patterns of indicator change observed offshore and in Exmouth Gulf in the GIS analysis should be a clear warning of how individual fishing histories and ecosystem characteristics (even at sub-regional scales) can have a strong impact on what can be considered an “undesirable” reference direction for an indicator (e.g. offshore an increase in the invertebrate:finfish catch biomass ratio is undesirable, while in Exmouth a decrease is undesirable).

6.3 Performance of indicators - comparisons with previous studies

The relative performance of the various indicators in the simulation tests and GIS analysis generally supports previous conclusions about the potential usefulness of the indicators based on field and small-scale experimental trials. Indicators that have received positive attention in the past, as they use easily sampled data and require few calculation steps (e.g. biomass ratios), performed well in this study. Of particular note, biomass of indicator groups, biomass size spectra and size at maturity have both been previously proposed as preferred indicators (Jennings *et al*., 1999; Murawski, 2000; ICES, 2001; Rochet and Trenkel, 2003, Niemi *et al.* 2004, Shin *et al submitted*), as they require relatively few data and they perform consistently well regardless of the level of exploitation intensity or the structure of the underlying ecosystem. Similarly, the simple diet-based indicators that showed potential here (e.g. average trophic level) have been recognised in the literature as attractive candidate indicators (Jennings *et al*., 2001b). Moreover, just as these indicators have been found to be sensitive to data quality in this study, debate continues over the best means of calculating them empirically so that data requirements are minimised (Rochet and Trenkel, 2003).

Potentially the largest discrepancies between previous results and the findings presented here concern the network and model-dependent indicators. Indicators such as Relative Ascendancy have often been proposed in the (mainly theoretical) literature as useful summaries of system state or system dynamics, though they are increasingly being put forward as real world indicators in fields such as eutrophication monitoring (Patricio *et al* 2004). It has been stated that the strength of these indicators is that they are commensurate with ecosystem properties such as trends in species richness, resource internalisation, trophic specialisation and succession (Christensen, 1994; Ulanowicz and Abarca-Arenas, 1997). However, their dependence on a) data that are difficult to collect reliably (e.g. diet data), b) models, and c) a good knowledge of the ecological structure of the ecosystem, means that they are very sensitive to data quality and their performance can often be poor. It may well be that these types of indicators and models have value in well studied and data rich systems, but based on how they performed in the simulation study and how hard they were to attempt to implement on the Northwest Shelf of Australia it would clearly be unwise to construct a monitoring system that relied solely, or even heavily, on such indicators.

6.4 The role of indicators in management

Indicator-type and tactical versus strategic management

Based on overall performance and robustness in the simulation testing and GIS applications, community- and ecosystem-level indicators appear to (generally) be the most informative indicators; they are often easier to calculate (e.g. simple biomass ratios); and they dominate the list of indicators given in the guidelines above. This is perhaps not too surprising given that most attributes of interest are also community- and ecosystem-level properties (Table 4.8). A few population-level indicators appear to perform quite well, but these are often system specific or refer to specific indicator groups rather than to any group at random (e.g. biomass of indicator groups like gelatinous zooplankton or planktivores as an indicator for biomass attributes). In general, population-level indicators evaluated for individual groups appear to be too sensitive to short-term fluctuations or other species-specific factors to be effective indicators for integrated community- and ecosystem-level attributes.

Nevertheless, given the temporal scale on which management and economic investment in fisheries occurs, indicator “responsiveness” is extremely important. The faster a signal of change can be identified the better, so that management actions can be put in place before the change has progressed to the point where it may take a very long time for the ecological properties of the ecosystem to return to their previous state. Once this need for responsiveness is also taken into account, community-level indicators appear to be the best compromise among data requirements, signal strength, responsiveness, and sensitivity to natural variability, in all ecosystem types. Community-level indicators respond more quickly than ecosystem-level indicators and so are of more use to management, but are still sufficiently aggregated measures that they adequately encapsulate fundamental information about ecosystem state and are not particularly sensitive to short-term variations. These qualities of community-level indicators have also been recognised in empirical water quality monitoring studies (Frost *et al.*, 1992). Unfortunately, in contrast to the field of water quality monitoring, communities potentially impacted by fishing can have much longer “generation times” and can be harder or more expensive to sample, which impacts negatively upon the actual time required to detect a signal. Recent field studies indicate that careful thought needs to given to the power of the monitoring schemes employed to detect any time series trends (especially in already heavily impacted areas) when using community-level indicators (Nicholson and Jennings, 2004). Simulation studies such as the one employed in this study can aid in evaluating such power. Another potential solution lies within the suite of indicators chosen, and how they are used to inform the management process.

While species level catch composition indicators were not considered in this study, community-level indices (e.g. pelagic:demersal, invertebrate:finfish and small-:large-bodied catch ratios) were and they performed well. Given this level of performance and the use of species-level catch composition in traditional fisheries management such indices should be given further attention in future work. In general though, the failure of population-level indicators to capture community- and ecosystem-level attributes partly explains the failure of indicators traditionally used in fisheries management to perform well in this study. The failure is also due in part to their potential sensitivity to data quality and their underlying assumptions. For instance, irrespective of ecosystem type or scale of interest, catch-per-unit-effort (CPUE) at the individual species level is unlikely to be a useful indicator in most situations, as it is often non-linearly related to abundance or is sensitive to data quality. At its best it is strongly related to ecosystem attributes, and the CPUE of species groups (e.g. planktivores or demersal sharks) rather than individual species can be informative. Unfortunately even then the usefulness of CPUE indices is heavily dependent on the history of the fishery (spatial coverage, levels and patterns of effort through time) and on data quality. Thus, while CPUE is likely to continue to be used for population level assessments of individual species (particularly where fishery independent data are lacking), the results of this study suggest that it should not be relied on as a primary indicator for use in evaluating broader impacts of fishing on by-catch species, habitats and food chains. However, if CPUE is used as a tactical indicator – in much the same way it is today and along with other responsive indicators –it should be supplemented (potentially only periodically) with strategic advice based on community-level indicators. Given the problems with data collection (expense, spatial and temporal scales), quality and availability encountered in the GIS analysis of some of the simplest and most robust indicators, this two-pronged approach seems like the only workable compromise between the demands of day-to-day management of marine natural resources and the data requirements and responsiveness of the most effective indicators of the ecological impacts of fishing. Other researchers in the fields of ecosystem-based fisheries management have also reached this conclusion (Link *in press;* Doug Butterworth, UCT; Simon Jennings, CEFAS, *pers. com.*).

Indicators and the issue of biodiversity

As has been stressed repeatedly here, no single indicator (or even type of indicator) is without problems or guaranteed to capture all pertinent details of ecological change. Quite apart from the issue of responsiveness, community level indicators are incapable of identifying species replacement (by invading or competing species) unless such events directly impact upon community level processes and structures. In the context of overall ecosystem functioning, such events may not be important. As long as some species is present to fulfil the required role in the ecosystem or community, its exact identity may be unimportant. However, the identity of the species may be critically important from a human perspective – for harvesting, aesthetic or conservation reasons. Consequently, biodiversity or other lower level indicators focusing on species of specific interest will need to be included in the suite of indicators chosen, so that they can supplement integrated system-level information provided by the community level indicators with specific details of critical importance from the human point-of-view.

System specific signal strength

Experience gained during the simulation and GIS components of this study highlights the potential for challenges involved in monitoring to differ between ecosystems types. Aspects include the degree of closure of a system, the number of sub-systems it may span, and also the number and strength of connections within the ecosystem. Open systems may be harder to monitor simply because not all the forces on the system components lie within the boundaries of the monitored area. In contrast, monitoring regional scale ecosystems can potentially become highly complicated due to different subsystems within the area providing opposing (or at least differing) signals. This is underlined by the finding that pelagic and demersal systems respond in different ways and so present different monitoring challenges. Due to the degree and strength of interconnectedness in benthic systems, they are harder to characterise simply (and so require more indicators to represent all the potential nuances), but signal strength is often high and on balance the impacts of fishing are clear and detectable. In contrast, pelagic systems are simpler to characterise (and so require fewer indicators), but the signal strength can be quite poor, meaning that there can be difficulty in detecting important change until long after it has occurred.

In attempting to verify whether the patterns of system specific signal strength observed in this study held generally in the real world, it was discovered that the focus of the majority of studies considering indicators and the impacts of fishing has been on fisheries affecting benthic habitats (e.g. trawl fisheries). Hopefully, this pattern of research is a by-product of the value of these fisheries, and the length of time (centuries) that they have been intensively exploited and studied, and not because the study of pelagic ecosystems has proven to be less fruitful

7 CONCLUSIONS

7.1 Broad Recommendations

The many reviews of indicators for ecosystem management (be it in a marine, freshwater or terrestrial setting) have acknowledged that ecosystems are large, complex, with open boundaries and that on the whole they are still relatively poorly understood (Landres 1992). While there is clearly a growing knowledge base to draw upon, the field of ecosystem based management is in its infancy and guidelines are still quite general. Simulation testing potential indicators, and applying a suite of the most promising candidate indicators to ecosystems that have been studied for many years, are necessary steps in the advancement of this field. Work to that end in this study has resulted in the list of recommended biological groups and robust indicators discussed in chapter 6 and listed in Table 6.2 above, but more generally it has resulted in the following set of broader recommendations:

1) Ideally indicators must be easily measured, cost effective (i.e. the indicator summarises or characterises many of the system attributes without requiring infeasible amounts of data), precise, and be easily understood and interpreted (i.e. the indicator must be measuring what it is thought to be measuring). In reality, monitoring agencies will have to take a pragmatic approach to indicators selection, employing those that are as close to this optimum as possible in their ecosystem of interest.

2) There is unlikely to ever be a definitive set of indicators that will work in every situation. To varying degrees, the specific indicators appropriate for any particular ecosystem will depend on the characteristics of that ecosystem and on the types of fisheries (and other human activities) in place. Nevertheless, indicators that require large amounts of data, or data that is difficult to collect, or employ intermediate models (or multiple steps) in their calculation, are unlikely to be consistently reliable in any system, as they are prone to error that can obscure the underlying signal. While the calculation of these kinds of indices (and the construction of the models) may be informative in its own right, they should not be the sole focus of on-going long-term monitoring schemes designed to measure and manage the impacts of fishing on marine ecosystems.

3) A suite of indicators (which comply with the criteria in (1) as much as possible) will be required to ensure the reliability of the overall picture of the ecological state of the ecosystem provided by the indicators, to allow for efficient detection of change and to aid in identifying of the underlying causes of change. This suite will have to span a wide range of processes (with different associated rates), biological groups, and indicator types (“tactical” and “strategic”, “early warning” and “integrated system state”) and will almost undoubtedly require data that span many temporal and spatial scales.

4) In the longer term, indicators will have a role within the broad-scale adaptive management processes associated with a shift to EBFM. They will be used both for performance reporting against management objectives (once those are fully operationalized) and as part of feedback decision rules that determine and set

management arrangements to mitigate negative ecological impacts of fishing. Indicators of various types. Indicators of various types and levels, particularly population- and community-level indicators, will be required. Population-level indicators will be used tactically (such as those currently used by fisheries management, e.g. estimated stock sizes) and community-level indicators may play a more strategic role (that can put signals from the population-level indicators into the broader ecosystem perspective). It is also likely that the use of indicators in management will benefit from the establishment of unfished reference areas, and the collection of fisheries-independent information on other parts of the ecosystem, as signals are easier to detect and interpret if there is a natural baselines or reference values available, and if there is a broader understanding of the overall system dynamics.

7.2 The practical application of indicators (and GIS)

Applications of GIS to logbook data with the aim of understanding exploited resources are increasing. It seems a simple extension to go on to include fisheries independent data (if available), and to use GIS packages to aid in the calculation of ecological indicators of the ecological effects of fishing and ultimately management. Development of the FishCAM application is a good example of an approach providing a range of visualisation and analysis tools for fishery management.

In this study data exploration and application of GIS to simple ecological indicators have been successfully performed at a range of spatial scales. These scales have been dictated by the scales of the available data, but working within such constraints will be necessary for any effective application of ecological indicators in real world systems. Furthermore, at present the types of data available around the world means that the analysis of logbook data undertaken at a one sixth of a degree resolution here would probably be at the more detailed end of the spectrum of possible analyses. For many systems around the world data will only be available at coarser scales. Given the relative performance of the different indicators at the different scales of data aggregation in the simulation part of this study, using data at coarser scales may be more informative anyway, at least for some indicators.

Problems with the older data present in the dataset used in the GIS analysis described here highlight other data quality issues that will no doubt dog some real world attempts to employ ecological indicators to consider the effects of fishing on ecosystems. Older data may have poorer spatial resolution and precision than more recent data and may well have been collected using different gear or alternative sampling regimes (e.g. at a different spatial or temporal resolution). Unfortunately, as understanding ecosystems requires long-term data sets, attention will have to be focused on how to reconcile older and more recent datasets.

Looking to the future, as computing technology advances the ability to process spatial data at finer resolutions will increase. This will hopefully assist in the management of not only the resource, but the surrounding ecosystems too. This will be especially important when the ecosystems are being fished harder, when new stocks are discovered, or when more stringent management requirements are introduced. Given the crucial role that spatial processes and structure play in ecology and ecosystem functioning as a whole, GIS is well placed to assist in the ongoing analysis of fishery data – providing the ability to process information at almost any spatial scale (data and time permitting).

7.3 Specific recommendations on indicators for use in management

Best candidate indicators

There is no avoiding the fact that there will be some degree of system dependency when drawing up lists of ecological indicators for use in ecosystem monitoring. Even amongst those indicators with the strongest performances in the analyses presented here, there will be variability in their actual signal strength and sensitivity across the range of marine ecosystems within Australian waters. Nonetheless, drawing on the fuller list in Table 6.2, the following is a preliminary set that can be used as a starting point for more intensive and system-specific consideration of appropriate indicators for each of Australia's regional marine ecosystems:

- relative biomass of demersal fish, small pelagic fish, marine mammals, piscivorous fish, scavengers, cephalopods, gelatinous zooplankton - trends in these may be followed directly, but when used to calculate biomass ratios (e.g. PS:ZP, pelagic:demersal fish) they will also give insight to web structure
- proportional cover of habitat defining epifauna and macrophytes
- simple diversity indices (number of communities in bioregionalisations or simple taxonomic counts)
- size spectra – the slope of this spectra can be a particularly strong indicator of community structure
- maximum (or mean) length of the catch
- average trophic level or trophic spectra – "rules-of thumb" categorisations of animals can be informative and less logistically taxing while still providing useful levels of meaning to these indicators (obviously a basis on good quality diet data remains preferable, however)
- size at maturity of main target, bycatch and top predators
- physical system characteristics (e.g. temperature, turbidity and chlorophyll a)

Further work defining which of these is most appropriate in temperate versus tropical, deep versus shallow, and pelagic versus demersal systems would be a very useful next step in ecosystem based fisheries management in Australia.

Data demands associated with ecological indicators in fisheries management

Even without further research, it is possible to identify the kinds of data demands that will be associated with the application of different kinds of ecological indicators. For the most effective use of indicators it will be necessary to collect fisheries independent data. It is possible to calculate fisheries dependent versions of all of the biological indicators listed above, but, as shown by the GIS analysis, without fisheries independent information they can be very difficult to interpret. More importantly, in systems such as the east coast of the US where long-term fisheries dependent and independent time series exist, a comparison of the two shows that signal strength in the fisheries dependent data can be weak or non-existent (Link and Sporcic *pers. com.*). This is particularly a problem when dealing with data for non-target species.

Turning to data needs in Australia. Fisheries dependent data of the form already collected by AFMA is a useful start. It is most easily expanded by increased observer coverage and the retention of the kind of size-based information typically collected when considering age-structure and discard issues.

Beyond the kinds of data typical of good commercial fisheries and bycatch monitoring programs, the most effective data extensions would be through fisheries independent data. The most easily collected independent data are climate and ocean colour data, which are now fairly widely available (with regard to spatial coverage). These kinds of data are an important part of the set of indicators outlined above, but may not be a sufficient supplement to fisheries dependent data if physical forcing is not a consistently dominant part of ecosystem forcing. For best use of ecological indicators, fisheries independent surveys are needed (so there is a better coverage of the ecosystem as a whole). These surveys will need to collect wet weight, size, taxonomic (and if possible diet) data for as wide a range of ecosystem components as possible.

Fisheries independent surveys are costly, so it is paramount that these are designed carefully to be as cost-effective as possible and so that they are not terminated in short order due to logistical constraints and apparent lack of useful returns. It will take time for independent surveys to build up sufficient data points for comparative temporal analyses. This problem can be dealt with to some degree by having spatial replication across areas with differential fishing pressure. Nevertheless it is important to use cost effective designs so that if sampling regimes are initiated they are not dropped after only one or two iterations. Given the many examples of multi-year or even multi-decade surveys around the globe, the value of long-term time series can not be overstated.

Many state fisheries agencies, universities and research bodies already undertake fisheries independent, threatened species, and habitat monitoring surveys and it is likely that many useful lessons can be gained from careful considerations of their successes, failures and actual degree of effectiveness or power. The first steps in such a process have already been taken in places like the Gulf of Carpentaria (Hart 2002). In addition, a lot can be learnt from consideration of the design and implementation of such highly successful survey programs as that found on the northeast coast of the United States. By having dedicated fisheries independent surveys with on board electronic data entry stations and protocols regarding the collection of weight, length, age and diet data, they collect and enter all the requisite data in the time it takes to move station-to-station and there is little fall back to shore-based laboratory work. This kind of sample design can lead to a great deal of value adding to any samples that are taken while avoiding the potentially crippling labour, cost and time bottlenecks associated with long periods of collection post-processing. While a lot can be learnt from US methods, the sheer density of stations monitored almost seasonally in the US is beyond the means of Australian resources – it would take 800 – 3000 stations for the area covered by the southeast fishery alone if Australia adopted the northeast US scheme without modification. A more reasonable and cost effective approach may be the combination of a rotational set of independent surveys and collaboration with the fishing industry to collect a wider range of biological and environmental data. Rotational surveys would mean that no one area is sampled every year, but it would be sampled every 5-6 years (say) and through time a wide coverage could be built up. In combination with the kind of extra biological and environmental data collection by fishers that is seen in places like Western Australia (Penn 2003) a sufficient amount of data may be available to use ecological indicators in a strategic way.

Fisheries and ease of monitoring using ecological indicators

Not all marine ecosystems currently exploited by fisheries are equally suited to monitoring by ecological indicators. As pelagic and demersal system components respond to pressures, such as fishing, in dissimilar ways they present distinct monitoring challenges. The spatial scales that typify demersal assemblages are relatively small (in contrast to regional ecosystem scales) while the degree of network connectedness is high. As a result, more indicators will be needed to characterise demersal systems and any data used to calculate these indicators will require more density of coverage than for pelagic communities. The extra effort required for demersal systems will pay off however, as the associated signal strength will probably be much higher and clearer than for pelagic systems. The state of pelagic systems is easier (in general) to summarise, but the associated signal strength is often poor, making timely detection of trends difficult. In shelf and slope waters for integrated ecosystem-based fisheries management (EBFM) it will almost undoubtedly prove necessary to combine information from demersal and pelagic subsystems in order to obtain a full ecosystem-scale picture. Some indicators even hinge on such combinations (e.g. pelagic:demersal fish biomass ratios).

What this differential (and combined) response of ecosystem components to fishing means is that indicators can probably be applied more easily in Bass Strait; the inshore waters of Norfolk Island, Christmas Island and Cocos (Keeling) Islands; the shelf-slope areas of the southeast, northwest, west coast, Coral Sea, Great Australian Bight, Gulf of Carpentaria, South Tasman rise, and Torres Strait. The distances involved in reaching the Antarctic islands will make data collection more difficult in those cases, but this may be offset by higher observer coverage. The fisheries where the application of ecological indicators may be most limited are the deep water pelagic fisheries – in particular the tuna and billfish fisheries. Midwater pelagic fisheries (e.g. small pelagic and squid fisheries) may be easier as information on the communities supplying those fisheries would be a useful part of the data collated for consideration of entire shelf-slope ecosystems. The fact that ecosystems often span fisheries (or vice versa) can not be forgotten when identifying areas where ecological indicators are to be put into use. While management is easiest at the fisheries level, piecemeal estimation of indicators is unlikely to lead to successful ecosystem-based management. To achieve EBFM it is important to keep an ecosystem focus even if that spans multiple fisheries.

7.3 Future work

This study is far from a definitive statement on the topic of ecological indicators of the ecosystem effects of fishing. There are many aspects of this topic that would benefit from further consideration. These include: verification in real world systems of the real utility of the indicators which were found to be robust in this simulation study; specific consideration of the suitability of the indicators recommended above for each of Australia's regional ecosystems; the use of alternative models to check those attributes that are not completely amenable to the simulation models used here (e.g. rates of damage, and the more effective representation of biodiversity and how it is impacted by fishing); determination of the most cost-effective use of information from reference areas; and determination of reference levels for use with specific ecological indicators.

Real world verification of the potential utility of the indicators identified as robust in this study and the final selection of ecological indicators for use in waters fished by commonwealth fisheries are clearly the most important areas for future research in this area in the short term. The GIS analysis presented here is a first step in such verification, but it should be repeated in more systems with differing properties and anthropogenic pressures. While it may be true that further modelling studies evaluating alternative indicators may highlight further candidate indicators, it is important that real world verification of those indicators already identified should not be postponed waiting for the results of any future modelling studies. A useful exercise in this vein would be for each major Australian fishery to allocate resources to attempt a compilation of the candidate indicators listed in section 7.3. Such an exercise would simultaneously give some indication of the real usefulness of indicators, synthesis existing data and give an indication of future monitoring needs.

Other aspects of the findings of this study that could be usefully extended is the identification of reference points (or directions) for the indicators and the best use of information from marine protected areas as reference data for indicator evaluation. Both of these issues could be investigated using real world information or the simulation based management strategy framework employed here. Any work on the best use of MPA data would also shed light on the scale such unfished reference areas would need to be to provide useful baselines.

The issue of reference points/levels/directions for ecological indicators used in ecosystem-based management is a topic that will benefit from additional attention. Ultimately, indicators need to be linked to reference levels (of some kind) and associated with management actions. To date, it has proven very difficult to specify reference levels for any ecosystem indicator due, for example, to difficulties in generalizing across ecosystem (ICES, 2001). It is likely that this will remain a difficult task given that specifying reference points for single-species fisheries management has also been difficult and controversial (e.g. Hilborn, 1997). The two most promising approaches to this issue are: (i) the definition of reference directions, as used here (Table 6.2) and in work such as that by Link (*in press*) and *Shin et al* (*submitted*); and the extension of the reference point concept to ecosystems via the use of principal components analysis (PCA) (Link *et al*. 2002). The first approach is fairly simple to implement in the short-term, although it still deals with changes in indicators individually and leaves open the topic of how to reconcile indicators that are providing orthogonal or contradictory information. The second approach is still in the preliminary stages of development, but in the long-term it (or something like it) may be the best means of integrating the information from a wide range of indicators in a simple way. Link's PCA approach involves plotting the scores for the first two principal components through time. The result is a map of system-state through time, which immediately lends itself to defining reference limits (or areas) and targets. Only future development and testing will show whether this approach (or others like it) are well suited to all ecosystem types. If the approach does prove to be of widespread utility, then development of reference points for ecological indicators will be greatly simplified, as the first step to understanding ecological changes in an ecosystem is having some means of summarising time-trends in the suite of indicators used to monitor it. Once a method of integrating the information from ecological indicators has been found, the greater issue of the development and testing of an adaptive management framework that incorporates ecological indicators in decision rules aimed at satisfying the demand for ecosystem-based management, remains to be completed.

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APPENDIX A1 – GENERAL REFERENCES

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APPENDIX B – PUNT *ET AL* 2001

This is the final workshop report that came from the workshop on ecological modelling and what approach to take in the simulation testing of the ecological indicators, which was held in Hobart in August 2001. It has previously been cited as :

Punt, A. E., Fulton, E. A., and Smith, A. D. M. 2001. Report on Hobart Ecological Modelling Workshop, CSIRO Marine Laboratories, 28–30 August 2001. Internal Report. CSIRO, Hobart. 23 pp.

HOBART ECOLOGICAL MODELLING WORKSHOP

CSIRO Marine Laboratories, 28-30 August 2001

B1. Preliminaries

The Chair, Tony Smith, opened the meeting and reminded participants that the workshop forms part of the project “Ecological indicators for fisheries management: non-target species, habitats and food chains”, funded by Environment Australia, Agriculture Fisheries and Forestry Australia, and the Australian Fisheries Management Authority[1]. The project has two objectives:

- to review current use of ecological indicators; and
- to identify robust indicators for the effects of fishing on species, communities and ecosystems.

The purpose of the second objective (the focus of this workshop) is to evaluate “ecological” indicators (quantities which can be calculated from the data collected from a fishery or from fishery-independent research) in terms of how well they are able to mimic changes in the quantities that are characteristics of the ecological system, and that are of interest to managers / the public (these characteristics are called attributes in this report).

Together with suitably chosen reference points, ecological indicators can serve two purposes in managing impacts of fishing. First, they can be used to define performance measures that are used to determine how well management objectives are being achieved. Second, they can be used as part of decision rules to determine adaptive management strategies. Both of these uses are common in single species fisheries management, but are yet to be widely adopted in managing the broader ecological impacts of fishing (Sainsbury *et al.* 2000).

The approach to be used for the evaluation of indicators in this study is the Management Strategy Evaluation (MSE) approach. In the context of the evaluation of ecological indicators, the MSE approach involves:

1. Selection of a set of attributes to quantify the management objectives (the issues people are interested in).
2. Development of a set of models (the operating models) that represent the “real world” for the purposes of the calculations, and that can be used to generate the types of data

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[1] This project will be referred to as “the project” in the remainder of the report.

on which potential indicators could be based. The operating model therefore needs to represent the ecosystem of interest, the impact of fishing and other anthropogenic effects on ecosystems, and the "monitoring" of ecosystems (through fisheries and fishery-independent research).

3. Simulation of the performance of each indicator for each (ecological) operating model. This involves:
    a. generating a large number of data sets;
    b. calculating the indicators for each data set (this may involve the application of some sort of assessment approach such as Ecopath); and
    c. comparing the time-trends in the indicators with the time-trends in the attributes.

The use of the MSE approach to evaluate the utility of indicators has a long history for target species of fisheries (e.g. Kirkwood 1981, Punt *et al*. 2001) and the properties of indicators for such species are relatively well understood. However, this is not the case for ecological indicators because these indicators are not well developed, and because ecological models have not been designed with a view to being used as operating models. Tony Smith highlighted that the focus of the workshop was to provide guidelines for the development of operating and "monitoring" / observation models, the selection of attributes and indicators, and the metrics to evaluate indicators.

The workshop recognized that there are several additional questions related to ecological indicators that require consideration but which are not part of the project.

1. The use of ecological indicators in a full feedback management loop.
2. Attribution of an identified impact to a specific cause (fishing versus other anthropogenic uses versus natural fluctuations).
3. The performance of indicators for specific ecosystems; instead the focus in the project is on the generic properties of indicators.
4. The ability of an indicator to quantify whether a specific event (represented by a reference value) has occurred and the value of the indicator relative to some 'pristine' level.

In reviewing the objectives for the workshop, it was agreed that it was not necessary to tailor the operating models to any specific ecosystem to any great extent. Rather it was agreed that the operating models would be "caricatures" for the purposes of simulation and testing. It was noted that the workshop participants did not include many "discipline-specific" specialists. Instead, most of participants were modelers because the aims of the workshop were model-focused and the participants involved had a background in several ecological disciplines.

B2. Review of current modeling approaches

B2.1 Trophic models

Villy Christensen introduced the Ecopath with Ecosim (EwE) software suite that has been under development for more than a decade, with numerous applications throughout the world. The software has more than 2500 registered users representing 120 countries, and more than a hundred ecosystem models applying the software have been published (www.ecopath.org). EwE has three main components: Ecopath – a time-invariant, mass-balanced snapshot of the system; Ecosim – a time dynamic simulation module for policy exploration; and Ecospace – a spatial and temporal dynamic module designed primarily for exploring the impact and placement of protected areas. EwE can be used to describe ecosystem resources and their interactions, to evaluate the ecosystem effects of fishing (including indirect effects, for example, through habitat modification), to evaluate the effects of environmental change, to predict bioaccumulation of persistent pollutants, to evaluate the impact and placement of marine protected areas, to evaluate uncertainty in the management process, and to explore management policy options incorporating economic, social, and ecological considerations, and legal constraints.

The foundation of the EwE suite is an Ecopath model (Christensen and Pauly 1992, Pauly *et al.* 2000), which creates a static mass-balanced snapshot of the resources in an ecosystem and their interactions, represented by trophically linked biomass ‘pools’. The biomass pools consist of a single species, or species groups representing ecological guilds. Pools may be further split into ontogenetic (juvenile/adult) groups that can then be linked together in Ecosim. Ecopath data requirements are relatively simple, and generally already available from stock assessments, ecological studies, or the literature: biomass estimates, total mortality estimates, consumption estimates, diet compositions, and fishery catches.

Ecosim provides a dynamic simulation capability at the ecosystem level, with key initial parameters derived from the base Ecopath model. The following represent the key computational aspects.

- Parameter estimation uses the mass-balance results from Ecopath.
- Variable speed splitting enables efficient modeling of the dynamics of both ‘fast’ (phytoplankton) and ‘slow’ groups (whales);
- Effects of micro-scale behaviors on macro-scale rates (top-down vs. bottom-up control) are incorporated explicitly.
- Biomass and size structure dynamics for key ecosystem groups are included using a mix of differential and difference equations. As part of this, EwE incorporates:
    - juvenile size/age structure by monthly cohorts, density- and risk-dependent growth;
    - adult numbers, biomass, and mean size accounting via delay-difference equations; and
    - the stock-recruitment relationship is an ‘emergent’ property of competition / predation interactions of juveniles rather than being pre-specified.

Ecosim uses a system of differential equations that express biomass flux rates among pools as a function of time-varying biomass and harvest rates (for equations see Walters *et al.* 1997, 2000). Predator-prey interactions are moderated by prey behavior to limit exposure to predation, such that biomass flux patterns can show either bottom-up or top down (trophic cascade) control (Walters 2000). Ecosim allows for the fitting of predicted biomasses to time series data through repeated simulations. In addition to a series of nonlinear optimization routines, the model can also be tuned to data in a feedback-process by examining some of the crucial ecological parameters in the EwE model (notably total mortality rates and the settings for top-down/bottom-up control).

The inclusion of time series data in EwE facilitates its use for exploring policy options for ecosystem-based management of fisheries. The time series fitting has so far been done only for a few ecosystems, as the facility is very recent. However, the results from these studies have been very encouraging. Villy Christensen noted that an important preliminary conclusion is that the model is capable of producing a reasonable fit to time series data (including fisheries and ecosystem components) compared to those obtained using single species models. This indicates a capability, or at least the potential, to replicate the known history of the ecosystems.

The Ecosim module provides two ways to explore the impacts of alternative fishing policies.

- Fishing rates can be 'sketched' over time and results (catches, economic performance indicators, biomass changes) examined for each sketch. This is using Ecosim in a 'gaming' mode, where the aim is to encourage rapid exploration of options.
- Formal optimization methods can be used to search for fishing policies that would maximize a particular policy goal or 'objective function' for management.

The second approach provides an 'open loop' policy exploration simulation, and acknowledges that policy may be defined as a way to reach a broadly defined goal, that fisheries policies are often implemented via *TAC*s that are recalculated annually, and through regulation that affects fleet structure and deployment. Most fisheries research has up to now been on policy implementation only, and the intention with this tool is to enable fisheries scientists to advise both on policy formulation and on its implementation.

The goal function for policy optimization is defined by the user in Ecosim, based on maximizing a weighted policy objective consisting of four features: (a) fisheries rent, (b) social benefits, (c) mandated rebuilding of species, and (d) ecosystem structure and 'health'. The objective function can therefore be thought of as a 'multi-criterion objective', representing a weighted sum of the four objectives: economic, social, legal, and ecological. Ecosim uses a nonlinear optimization procedure known as the Davidson-Fletcher-Powell (DFP) method to iteratively improve the objective function by changing relative fishing rates.

Ecosim also includes 'closed loop policy simulations' to evaluate monitoring alternatives in terms of their implications for temporal variation in biomasses and also the objective function value components used in searches for optimum long-term fishing rate plans. The idea in the closed loop simulation is to model not only the ecological dynamics over time, but also the dynamics of the stock assessment and regulatory process. A closed loop simulation includes 'submodels' for the dynamics of assessment (data gathering, random and systematic errors in biomass and fishing rate estimates) and for the implementation of assessment results through limitation of annual fishing effort.

The Ecospace model is a spatial, meso-scale version of Ecosim that is designed to quantify the effects of protected areas (seasonal or full-time closures), to predict spatial distributions, and to evaluate spatial effects, e.g., feeding related or due to habitat changes. The features of Ecospace are that it:

- replicates Ecosim dynamics over a spatial grid of 'homogeneous' cells;
- links cells through dispersal of organisms and fishing effort movement/allocation;
- incorporates a 2-D advection model;
- accounts for spatial variation in productivity and the costs of fishing; and
- represents habitat 'preferences' by differential dispersal, feeding and predation rates.

Ecospace also links to GIS databases for primary production, depth, temperature, habitat structures, etc. and to FishBase (www.fishbase.org) for depth and species distributions.

In discussion, it was noted that the EwE suite could form the basis for operating models (Ecosim / Ecospace), or be used as an assessment model to generate indicators (Ecopath). The use of Ecopath as an assessment model is discussed further in Section 4.

B2.2 Habitat models

Francis Pantus introduced the discussion on habitat models. A model has been developed to estimate the relative impact of trawling on classes/types of benthic organisms within the GBRMP/WHA, under various marine park management scenarios. The module of most relevance to this workshop is the impact estimation model. This sub-model aims to convert overall trawl effort (e.g. boat days) into relative removed biomass. The model simulates trawling on large areas (in comparison to the per-shot area covered) by sub-dividing the area into pixels that are of the size of the gear used. The negative binomial distribution was found to describe the distribution of trawl “hits” over various trawl aggregation patterns and total amounts of effort expended. Combining this with benthic biota biomass depletion curves for various ‘taxonomic units’ (from repeated trawling field experiments) and trawl logbook data resulted in spatially explicit estimates of relative biomass removed. Other impacts like ‘landscape’ changes were not considered.

In discussion, it was noted that the approach used to assess habitat usage at the sub-6x6 grid-level was of potential value in modeling the impact of gear damage on habitat / benthic communities in operating models. It was also agreed that this approach would be of value when developing an observation model to capture the process of sampling habitats. The approach used to identify vulnerable species in the Northern Prawn Fishery (Stobutzki *et al.* 2001) could be used to identify the ‘indicator species’ for some ecological indicators. Simulation of the process of constructing tables of features related to the ‘vulnerableness’ of species could provide a guide to the utility of this approach.

B2.3 Multi-species / size model

Kevin Stokes introduced the discussion of these models. He started by noting that the most common current fisheries management framework is to use highly idealized single species stock assessments to derive parameters of direct use in management decision making. These parameters are most frequently estimates of biomass or fishing rate relative to some reference level. The reference levels used for many fish stocks derive from simple theoretical work, and are often driven through adoption of international agreements and national legislation that usually only define high-level objectives. The detailed specification of indicators and reference points, and methodological choices, is commonly the preserve of scientists.

Multi-species (biological and technical interaction) models range from simple two species (e.g. cod and shrimp) interaction models, to highly complex models of many species and many fleets interacting in space and time. They have been developed in many regions, for a variety of purposes (e.g. to provide management advice, to undertake assessments, or as a basis for ecological enquiry). To a lesser extent, they have also been used to evaluate management strategies.

MSVPA (Stokes 1992, Magnusson 1995) is a multi-stock virtual population analysis “tuned” using single species VPA outputs, and linked by estimating the “suitability” of each prey and predator type (by species and age). It has been applied in the North Sea, Baltic Sea, Alaska and NE USA. The suitabilities are estimated to correspond with observed stomach samples. MSFOR is the same structural model used in forecasting mode. Variants of this have also been used in spatially disaggregated forms, or with length based forecasting. Although data hungry and apparently complex, the MSVPA/FOR models are in essence quite simple. They better capture inter-annual variability in natural mortality than single species counterparts, and provide a potentially useful tool for management scenario evaluation. At the heart of an MSVPA/FOR, however, is a set of strong structural assumptions about the multi-species system and how it is linked and operates. Sensitivity to structural uncertainty has been little investigated. Nevertheless, MSFOR structure, parameterized through MSVPA, has potential as an operating model as robustness to structural uncertainty could readily be investigated. The biggest problem with such an approach is the acquisition of data needed for credible parameterization. Such models, and the large-scale data collection to underpin them, are, therefore, likely to remain the preserve only of highly industrialized and exploited fishery systems.

More complex multi-species systems have been developed for the Barents Sea (MULTSPEC) (e.g. Tjelmeland and Bogstad, 1998) and Iceland (Bormicon) (e.g. Stefansson and Palsson, 1997). These models have spatial structure and include a range of species. In common with MSVPA, they require large amounts of data for parameter estimation. Unlike most MSVPA/FOR applications (the main exception being the Baltic Sea), MULTSPEC and Bormicon have been used to provide advice for fisheries management. Multi-species models have also been developed within CCAMLR to address ecological interactions between species in the Southern Ocean, and the possible impacts of fishing for krill on the rest of the food chain.

Relatively little MSE work has been carried using multi-species operating models. Scenario Barents Sea has been developed to evaluate management strategies for the Barents Sea based on a similar model structure to that used for MULTSPEC. A multi-species model was used to evaluate the impact of trends in the South African fur seal population on those for the hake fishery off South Africa. In this case, the originally envisaged “minimally realistic model” for hake, other predatory fish, seals (and various other components) initially suggested that a cull strategy could be beneficial to the hake fishery. Incorporation of a slightly more “realistic” two hake model, including cannibalism, reversed the initial suggestion (Punt and Butterworth 1995).

The issue of structural uncertainty and the ability to predict most bedevils multi-species models in fisheries. As part of larger ecosystem models, this feature is potentially of the greatest significance for answering specific questions pertaining to individual species. Yodzis (1998, 2000) used foodweb models to investigate the effects of reducing the abundance of a top predator on fisheries yields, and to ask whether it was possible to identify “modules” within these systems that could be modelled separately. The results generally showed that when all links in the system were accounted for, the median return to fisheries following a reduction in predator numbers was less than that predicted by a simple surplus yield calculation. In many cases, it was more likely that fisheries yields would decrease following a cull of predators than that they would increase.

There is a long history of using aggregate models based on size-structure to characterize systems and to investigate the effects of fishing (Silvert and Platt 1981). For example, Pope *et al.* (1988) show that the log slope of the aggregate size spectrum in the North Sea and Georges Bank is clearly related to fishing pressure. This is not surprising – fishing tends to concentrate on larger species, thus reducing the aggregate biomass in larger size classes, and possibly also reducing predation on smaller fish.

Recent work (Duplisea and Bravington 1999) to consider optimal fishing strategies using both size-aggregated and multiple-species models demonstrated that the strategies were effectively the same regardless of the structural model used. The optimal strategy in both cases was to fish the biggest sizes hardest, and the smallest the least. Aggregate models, however, whilst simpler, easier to set up and explore, cannot display behaviors at an individual species level that might be of interest in fisheries management or ecosystem management.

Lack of predictive power needs to be considered when selecting multi-species operating models. The issues that affect predictability are many and varied, but include the inherent indeterminacy of complex, non-linear systems, the desire to predict at spatial and temporal scales finer than those at which data are collected, and the various types of uncertainty (process, observation, implementation and structural). The implications of structural uncertainty and the lack of predictability make predictive approaches to ecosystem-based management less than appealing. The only recourse is to a systems analysis and the MSE approach. What levels of model complexity should be adopted depends upon the questions to be asked.

B2.4 Single species models

Andre Punt introduced the PBR (Potential Biological Removals) approach (Wade 1998) that is used to set limits on the removals from marine mammal populations managed under the U.S. Marine Mammal Protection Act (MMPA). The PBR formula and its parameters were chosen to satisfy goals inferred from the MMPA. The 'MNPL goal' involves maintaining populations above the maximum net productivity level (MNPL), the 'carrying capacity goal' involves allowing populations to recover to close to their carrying capacities, and the 'recovery goal' involves allowing highly depleted populations to recover to MNPL at close to the maximum possible rate. Although originally designed to satisfy the goals of the U.S. MMPA, this approach has also been used to manage by-catch of marine mammals in jurisdictions outside of the U.S. (e.g. Maunder *et al*. 2000).

The MMPA goals were operationalized and various PBR formulations evaluated in terms of their ability to satisfy the management goals. This evaluation was conducted using variants of the simulation protocol developed by the International Whaling Commission (Donovan 1989, Taylor 1993). Sensitivity was examined *inter alia* to bias in the realized catch, the abundance estimates, and the survey coefficients of variance. The final selection was:

$$PBR = N_{\min} \tfrac{1}{2} R_{\max} F_R$$

where $N_{\min}$ is a "minimum population estimate" (the lower 20th percentile of the point estimate of abundance),

$R_{\max}$ is the "Maximum theoretical (or estimated) net productivity rate" (0.04 for cetaceans and 0.12 for pinnipeds), and

F_R is a "recovery factor" (0.5 for the MNPL goal and 0.15 for the carrying capacity and recovery goals).

In discussion, it was noted that the PBR approach had been developed for managing marine mammal populations using feedback control but that the ratio of the catch to the PBR was a potential indicator of the impact of the magnitude of by-catch. Some aspects of the operating models used in development of the PBR approach could be included in the operating models.

B3. Selection of attributes

Ideally, the attributes arise from the high-level management goals such as “maintaining ecosystem health and integrity” and “biodiversity conservation”. Table B1 lists the set of potential attributes identified during the workshop. This list was constructed integrating attributes at the species, community, habitat, and ecosystem level. All of the attributes can potentially be computed using the Atlantis and Ecosim / Ecospace modeling frameworks. Table B1 ignores potential attributes (such as age- and genetic-structure) which cannot be computed straightforwardly using BM2 and Ecosim / Ecospace.

It was agreed that there was need to examine the extent to which attributes are correlated. In principle, this provides an indication of the value of some indicators assuming “perfect information”. This might allow the use of a more easily measured attribute as a proxy for one less easily obtained. A key aspect of the design of the operating model is to be able to represent a realistic level of spatial and temporal correlation among the attributes, for example, by appropriate choice of spatial structure and processes. It was noted that it would be difficult to explicitly represent spatial fragmentation of habitat, given the likely spatial structure of the operating model. Similarly, Table B1 does not include quantities such as the “resilience” and “stability” of the ecosystem.

Table B1: The potential attributes. An asterisk denotes that the attribute could be developed for a single species / functional group or a group of species / functional groups.

No	Attributes
1	Biomass*
2	Diversity (species, group, flow)
3	Size structure*
4	Position in MDS space
5	Trophic level
6	Consumption
7	Spatial structure (of biomass)*
8	Throughput
9	Food web structure
10	Respiration
11	Nutrient cycling
12	Production
13	# groups to represent 80% of the biomass

B4. Review of indicators

There are a very large number of potential indicators. Some are direct measures of attributes, while others are “proxies”. Table B2 lists the set of indicators identified during the workshop for initial investigation. This list is a small sub-set of the complete list of potential indicators identified by the review part of the project. The list given in Table B2 was selected primarily because the indicators concerned are (or can be) well defined and can be measured or estimated using the data collected during fishing operations or by means of scientific sampling. The indicators are often measured as the extent of difference between a reference and a present state. Several indicators could be based on the difference between the present and the "pristine" state. However, what determines "pristine" for dynamic ecological systems is not clear / defined. In principle, indicators could be based on the difference in the values of some measurable quantity between fished and no-take areas.

Indicators 20–25 in Table B2 require the use of an "ecological assessment model". It was agreed that an attempt should be made to use Ecopath for this purpose and Villy Christensen thought this should be possible (able to be "automated"). Indicators could be based on Ecopath snapshots conducted at various points in time. It was noted that numerical routines (e.g. the inverse method) are under development to fit Ecopath when the input data are unbalanced and there was a need to collaborate with the developers of Ecopath to ensure that the most efficient numerical routines are used. Several of the indicators in Table B2 (e.g. indicator 1) apply to a sub-set of species. These could be harvest, by-catch, threatened, responsive, or sensitive species. The approach described in Section 2.2 could potentially be applied to identify indicator species.

Table B2: The indicators for initial consideration.

No	Indicator
1	Biomass / stock size
2	Total mortality (catch divided by PBR-type catch limit)
3	Size / age-structure
4	Catch-rate
5	Discard rate
6	Size-spectra (using log size-classes)[a]
7	k-dominance curves
8	Coefficient of variance for total biomass
9	FIB
10	Average trophic level
11	Diversity index (e.g. Reyni or Shannon-Weiner)
12	Species composition (MDS plots)
13	Rate of damage[b]
14	Benthic habitat complexity[c]
15	Biomass of cover-defining species / species groups
16	Reproductive success
17	Ratios of piscivores : planktivores and / or demersal fishes
18	Chlorophyll-a
19	Redfield ratio
20	Throughput
21	Production / biomass
22	System ominvory index
23	Dominance of detritus
24	Relative ascendancy
25	Residence time (= biomass/(respiration+export))
26	Index of Biological Integrity (IBI)

Notes: a – may require giving species without size/age structure "average" sizes.
b – computed as the percentage of the population affected for invertebrates.
c – indexed, perhaps, by the mean fractal value of sediments

B.5 Metrics to evaluate indicators

There are several ways in which to compare the distributions for the change in an indicator with the change in an attribute. Figure 1 shows one graphical approach to illustrate the performance of an indicator. The x-axis shows the change in the value of the attribute and the y-axis shows the change in the indicator. Ideally, there should be a 1-1 relationship between the attribute and the indicator and the indicator should be precise (left panel). The right panel of Figure 1 shows an undesirable situation. The indicator does not change in a well-behaved manner with the attribute, and it is imprecise. Note the need for the operating model to generate contrast in the values of the attribute.

In principle, there may be value in regressing the value of an indicator on more than one attribute because some indicators may quantify more than one attribute. How to evaluate the performance of a multi-dimensional indicator (e.g. MDS scaling) using the approach of Figure 1 is unclear. Note that Figure 1 could be computed for any choice of time-period (e.g. years 1-10 to capture performance for a "developing fishery", years 50-70 to capture performance for a "mature" / "stable" fishery, and years 70-100 to capture a "recovering" / "depleted" fishery).

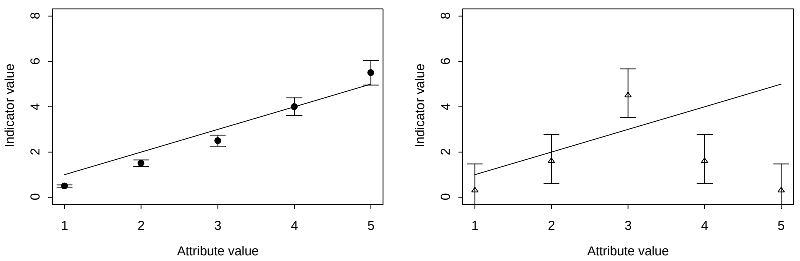


Figure B1: One graphical approach to contrast the performances of two indicators.

B6. Specifications for ecological operating models

B6.1 Overview of the BM2 model

Beth Fulton introduced BM2, a biogeochemical process model that tracks the nitrogen and silicon pools of a generic foodweb and associated gaseous, detrital and mineral compartments. It shares features with EwE, but the latter is biomass-pool rather than nutrient-pool based. The spatial geometry of BM2 is made up of polygons (boxes) that correspond to the geographical form of the modelled system (the current configuration is a 59 box representation of Port Phillip Bay - see Figure 2). The area and shape of the polygons reflect the speed with which physical variables (depth, flow, sediment type) change within particular parts of the bay. The standard form of BM2 also uses a 3-layer (water column, epibenthic, sediment) vertical resolution and a daily time-step, but it is possible to increase the vertical and temporal resolution of the model. BM2 is driven by seasonal variation in solar irradiance and temperature, as well as nutrient inputs from point sources, atmospheric deposition of dissolved inorganic nitrogen (DIN), and exchanges with the Bass Strait boundary box. The physical submodel also includes bioirrigation, bioturbation, sediment burial, nitrification and denitrification (all of which are affected by the degree of biological activity in a cell).

The foodweb submodel of BM2 was constructed by extending and replicating (and modifying where necessary) the general form of the process equations implemented for the biological components in the Port Phillip Bay Integrated Model (Murray and Parslow 1997). As a result, the invertebrate groups (primary producers and consumers) are treated as simple biomass pools and all groups use relatively simple equations for processes such as growth and assimilation, excretion, mortality, and waste production. The physiological detail explicit in other nutient-pool ecosystem models (e.g. the European Regional Seas Ecosystem Model - Baretta *et al.* (1995)) is not included in BM2, as Fulton *et al.* 2004b has shown that this is not usually necessary.

The vertebrate (usually fish) groups in BM2 use an age-structured “average individual” formulation, in which the numbers of individuals and the condition of the “average individual” in each age-class is tracked by spatial cell. The grazing and excretion processes are slightly more complex than for the invertebrates, as vertebrates are assumed to have a “structural” and a “reserve” biomass pool. The “structural” pool represents the skeleton and other body parts that cannot be reabsorbed if conditions become poor, while the “reserve” pool represents fat stores, gonads and other body parts that may be reabsorbed or depleted if metabolic maintenance requirements cannot be met. Despite these complications, the general resolution of the equations (growth and assimilation, excretion, mortality and waste production) for the vertebrates is similar to that for the other biological components of BM2.

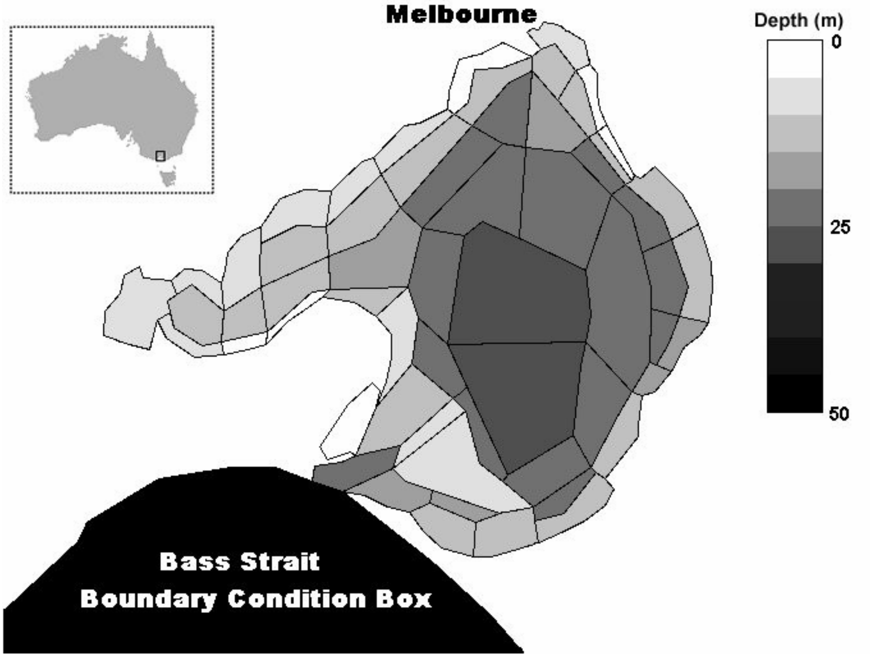


Figure B2: The current configuration of the BM2 model for Port Phillip Bay.

BM2 contains a few processes and groups not considered in other ecosystem models. Dinoflagellates are frequently represented explicitly in ecological models of the water column, but mixotrophy is not. In the past, this reflects that little was known about mixotrophy and because it was considered to have negligible impacts. However, there is now clear evidence that dinoflagellates can have significant impacts (via predation and competition) on phytoplankton and zooplankton, despite their relatively low densities and growth rates (Hall *et al.* 1993, Jacobson 1999). Moreover, the behaviour and persistence of dinoflagellates observed in natural systems had previously proved difficult to reproduce in simulations, suggesting that some mechanism, crucial in nature, was lacking and mixotrophy seemed a prime candidate for this missing process.

The other part of the system treated unconventionally is bacteria and their associated effects on sediment chemistry and remineralisation. Ecosystem and water quality models have traditionally treated bacteria in much the same way as all other invertebrates, using the same formulations and making only minor modifications to linkages, resource utilisation terms and parameter values. This approach is adopted for the free-floating pelagic bacteria in BM2, but a different approach is used for the three groups of attached bacteria (pelagic attached bacteria, aerobic and anaerobic sediment bacteria). The growth rates of attached bacterial populations (water column and sediment) in BM2 are equated to the availability of colonisable substrata (the detrital groups) rather than to more grazer-like consumption of prey resources. The activities of these bacteria are then directly linked to the nitrification-denitrification model.

The optional submodels in BM2 include forage- and density-dependent fish movement, a fisheries effort dynamics model, fishing-induced mortality on groups that are not fishery targets, functional group invasions, and alternative functional responses and mortality schemes. Some of these submodels will need to be used to correct for shortfalls in the basic model structure. For example, top predators (sharks, birds and marine mammals) are not included explicitly in the basic structure of BM2, but they will need to be explicit in at least some “ecological indicator” scenarios, as they will be the groups of interest. Similarly, alternative fish recruitment and movement schemes may need to be considered because the default assumption is constant recruitment.

Comparisons of BM2 with 276 real shallow marine systems (Fulton *et al.* 2004b) indicates that the performance of BM2 was good overall. As a generic bay system, it reproduces realistic levels of biomass, production and consumption. It also reproduces realistic community composition and spatio-temporal dynamics - such as the formation of assemblages in different parts of the bay and seasonal, annual and long-term variation (e.g. macrophyte-barrens). The model’s output is also consistent with empirical relationships observed in the field (such as Sheldon size spectra and the ratio of chlorophyll-a to dissolved inorganic nitrogen in the water column).

The model can produce some anomalous behavior, such as the almost exponential growth of the deposit feeders under certain parameterizations and nutrient conditions. It does not currently perform well under oligotrophic conditions. However, modifications to the model to remove these problems should be possible.

B6.2 Fishery models

David McDonald introduced this section. In general, fleet dynamics involves both spatial and temporal aspects of fishery fleet behaviour. Spatial aspects are often addressed by modelling fishing location choices using decision rules that take account of expected costs and catches among alternative fishing sites, subject to management constraints. Temporal aspects include investment decisions and inter-temporal allocation of effort within and across fishing seasons.

When decision process models are used to track or forecast spatial and temporal fleet dynamics, one might select from a suite of *ad hoc* rules or one might draw on formal analytical tools such as optimal control theory to derive decision rules. The approach chosen will depend on the purpose of the modelling task. The rules used, however, would preferably incorporate fishers’ responses to different management strategies, as well as to changing environmental conditions.

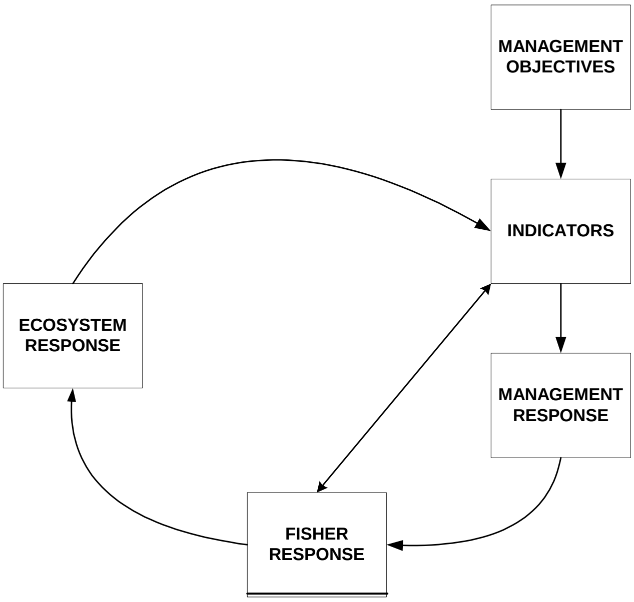
B6.3 Observation models

The observation models need to generate all of the data required by the indicators. This does not mean that individual observations or measurements need to be modeled explicitly, but rather that "summary observations" with realistic statistical properties (including bias and variance) are required. Two types of "data" are required:

1. Fishery-dependent data: landed catches, discarded catches, catch-rates, size / age-structure.
2. Scientific sampling: indices of density, age / size structure, habitat usage, diet fractions (and other data needed to apply Ecopath), breeding success

The catches (landed and discarded), the catch-rates and the indices of density can be assumed to be log-normally distributed while the age- and size-composition data can be generated by sampling multinomially from the actual catch (or population) age- and size-composition. Habitat impact and monitoring at the sub-grid level may need to be generated using the approach outlined in Section 2.2. Villy Christensen has default distributions to represent the uncertainty associated with the inputs to Ecopath, and the generation of the diet fractions, consumption estimates, etc. can be based on these distributions.

Figure B4: Schematic diagram of MSE framework.



B6.4 Forcing models

Scott Condie gave an introduction to oceanographic forcing models. There are a variety of ways to include forcing in spatially resolved models:

1. Ecospace-type approach with existing representation of exchanges. Existing Ecospace-

type exchanges based on a single layer, constant wind, and no tides is unrealistic and unlikely to produce the required range of behaviors.

2. Ecospace-type approach with exchanges based on existing circulation model outputs. This requires some software development to utilize exchange estimates.

3. Ecospace-type approach with exchanges based on new circulation models of idealized systems. This again requires some software development to link exchange estimates into the software. Significant resources would be needed, however, to set up and run new circulation models.

4. Box model with exchanges based on existing circulation model outputs. The methodology to do this is already well developed.

5. Box model with exchanges based on new circulation models of idealized systems. As for the Ecospace-type approach, significant resources would be needed to set up and run new circulation models.

6. Exchanges based on satellite-derived currents for pelagic systems. Circulation fields are available for the past decade and scenarios might be crudely represented by increasing or decreasing all exchanges by some factor. There are additional choices about the representation of biogeochemical process in this case; which fields (e.g. primary productivity) are prescribed and which are calculated? Prescribing fields will require less process understanding and be faster, but the fields may be inconsistent with other components of the simulation.

It should be noted that existing circulation models are quite sophisticated and available for a range of Australian systems (north west shelf, Gulf of Carpentaria, south east shelf and slope, Port Phillip Bay etc), but have been run under a limited range of conditions / scenarios.

B6.5 Management scenarios

This study does not involve a full MSE evaluation of the use of indicators in adaptive management. However there is still is a need to identify the kinds of management scenarios that are likely, so that prospective indicators can be tested for their ‘signal strength’ and robustness to management circumstances, as well as to how the ecosystem may work and the fishery may respond. The types of management scenarios that could be considered include:

1. Changes over time in the fraction of by-catch that is reported (either because of increased use of observer programs or because markets are identified for species that are presently by-catch species) – this could be included in the operating model through its observation component.
2. Increasing use of by-catch reduction devices that either reduces by-catch or causes ‘invisible’ mortality.
3. Spatial zoning of capture methods based on whether habitat modification occurs.
4. No-take reverses (small, large, variable enforcement).

B6.6 Scenarios for the operating models

The workshop attempted to identify the range of scenarios for inclusion in the operating model. Table B3 summarizes the key factors and an initial selection of levels for those factors. It was acknowledged that it would not be possible to “cross” every factor with every other factor due to the computational demands of the calculations. It was noted that both BM2 and Ecosim / Ecospace could, in principle, act as a generic operating model. Both models have their advantages (e.g. the speed of Ecosim / Ecospace and its lack of an explicit stock-recruitment relationship, the explicit modeling of nutrients in BM2).

In discussion, the workshop identified a number of issues that need to be taken into account when designing the operating models.

- If indicators based on “reproductive success” are to be considered, recruitment of large predators may have to be added to BM2.
- The scenarios for fishing effort should involve transients (increases, stability, decline) with various levels of intensity to evaluate the sensitivity of the performances of the indicators in fisheries of different levels of “maturity” / fishing intensity.
- (Increasing) trends in marine mammals should be considered.
- The transport model in BM2 currently relies on vectors of flow defined by a complex hydrodynamic model that is run separately. New flow vectors would be needed to use the model for a system other than Port Phillip Bay (see Section 6.4).
- The assumptions in BM2 that nitrogen is the limiting nutrient and that Redfield ratio is always maintained is the primary reason the model fails to work in oligotrophic conditions. This problem could be rectified by including more nutrient pools and other formulations for the uptake of nutrients by autotrophs.
- In BM2, some of the benthos (infauna) occasionally show almost exponential growth, but space limitation terms should solve this.
- Fleet dynamics, recruitment and fish movement are all very simplistic or prescribed in BM2.
- The operating models may need to include species (rather than species groups) at various levels (not just vertebrates). If species are included explicitly (particularly for the vertebrate groups) then care should be taken to ensure contrasting life histories are captured (e.g. turtles vs seals).
- Inclusion of the variable speed features of Ecosim into BM2 could substantially improve its performance.
- Scenarios could be scoped using Ecosim and then included in BM2.
- There needs to be a link between habitat and the dynamics of several of the functional groups.
- The operating models are not designed to deal with processes that act at the sub-grid level. Judicious choice of observation models may help to overcome this limitation.
- There is considerable value in examining sensitivity to different choices for the species modeled explicitly and the choice of functional groups, as this may impact the performance of indicators as well as the behavior of the system itself.
- The choices of spatial structure and vertical stratification in the operating model (and even the necessity of modeling nutrients) will depend on the type of system being modeled (bay, coastal, etc.) Fulton (2001) developed some “rules of thumb” regarding the number of spatial boxes needed.
- There may need to be links between damage to habitat and juvenile mortality.
- There is a need to explicitly include species with a range of productivities in the operating model.
- The impact of flux into bay systems can be modeled by including some “bay / estuary” cells in a coastal model; the workshop agreed that operating models that include multiple systems (see Table B3) were not necessary.
- The behavior of the values of the attributes for the different types of forcing (fishing, other anthropogenic, and oceanographic) should be examined.
- The impact of trawling could be modeled in BM2 through bottom stress. There is a

need to ensure that the spatial sub-structure is accounted for when modeling the impact of habitat damage – the impact on habitat types that tend to be close in space should be correlated.

- The operating model should be assumed to be “closed” with respect to as many components as possible.
- Modeling the impact of trawling on the habitat may require developing observation models that implicitly account for this at the sub-grid level scale.
- Consideration should be given immediately to selecting “system types” based on existing oceanographic models.

Table B3: Factors to include in the operating models

Factors	Scenarios
System type	3+ (bay / coastal (fast and slow flushing); shelf; slope / pelagic (upwelling and production poor))
Natural forcing	2 scenarios regarding oceanographic forcing
Ecological model	2 levels of species / functional group aggregation
	5 ecological scenarios
Anthropogenic impacts	3 scenarios regarding nutrient influx (bay / coastal only)
Fishing	5 time-sequences of fishing effort
	3 gear-types
Other management options	2 spatial management options
	2 changes in regulations

B7. Final discussion and summary

Much of the above can be summarized by three questions.

- What are the impacts of fishing on ecological systems?
- Which impacts matter?
- How well can those impacts be detected?

The first question relates to consideration of the choice of operating models, together with ecological, fishing and management scenarios. As with any Management Strategy Evaluation application, the operating models need to be credible, but not exquisitely detailed. Performance in terms of run time is also a major consideration.

The second question relates to the choice of attributes. Not all attributes will be included in the project, and, to some extent, the choice of attributes will be dictated by pragmatism, based on the availability of existing operating models. However it is envisaged that a number of important ecological attributes will be able to be dealt with, including aspects of biodiversity, habitat impacts, and trophic cascades.

The third question relates to the choice of indicators and metrics for their performance. The list of indicators in Table B2 needs further elaboration and specification. As noted during discussion, the project may also lead to the identification of “new” indicators.

In closing, the Chair thanked all participants for their contributions, and the rapporteurs for their efforts.

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Annex A : Agenda

1. Preliminaries
2. Review of current modeling approaches
    - Trophic models
    - Habitat models
    - Multi-species / size models
    - Single species models
3. Selection of attributes
4. Review of indicators
5. Metrics to evaluate indicators
6. Specifications for ecological operating models
    - 6.1 Overview of the BM2 model
    - 6.2 Fishery models
    - 6.3 Observation models
    - 6.4 Forcing models
    - 6.5 Management scenarios
    - 6.6 Scenarios for the operating model
7. Final discussion and summary
8. References

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APPENDIX C – ATLANTIS OPERATING MODEL EQUATIONS

Table C.1: List of main terms used in the equations in this appendix. All terms, variables, constants and expressions are defined in the relevant sections, but this table may be a useful quick reference for the main terms and conventions.

Term	Meaning
E	Excretion (ammonia produced by a consumer)
F	Fishing (catch)
G	Growth
M	Mortality
P	Uptake
R	Remineralisation
S	Sediment chemistry (nitrification or denitrification, the subscript will denote which on a case-by-case basis)
W	Waste (detritus produced by a consumer)
XX	All doubles (and triples) refer to components of the model (see Tables 4.2 and 4.4 for definitions). They do not represent multiplications at any time and any multiplications will be explicitly denoted by a "·".

Note that the due to the large number of symbols needed to describe the many equations that make up the operating model, some symbols have had to be reused with sub or superscripts to differentiate between their multiple meanings. Also please consider the symbols here in isolation to those used elsewhere in the report (in the equations or discussion of the sampling model and indicators) as the reuse of symbols does not mean they represent the same thing in the different parts of the report – where this is the case it is specifically stated. I have tried to keep this reuse and potential confusion to a minimum and where it has been necessary to do it as logical as possible, I'm sorry for any inconvenience or confusion this may cause.

C.1 Rate of change equations

C.1.1 Autotrophs

Rate of change for standard water column primary producer (PX):

$$\frac{d(PX_{\mathrm{w}})}{dt} = G_{\mathrm{PX_w}} - M_{\mathrm{lys,PX_w}} - \sum_{\substack{\mathrm{i=predator}\\ \mathrm{groups}}} P_{\mathrm{PX_w,i}} \qquad \text{(C.1)}$$

$$\frac{d(PX_{\mathrm{sed}})}{dt} = -M_{\mathrm{nat,PX_{sed}}} \qquad \text{(C.2)}$$

Where G_{PX} stands for the growth of PX, $M_{\mathrm{lys,PX}}$ is the loss of PX due to lysis, $M_{\mathrm{nat,PX}}$ is the natural mortality losses of PX when in the sediments and $P_{\mathrm{PX,I}}$ are the losses of PX due to predation. The equations for the benthic primary producers are slightly different. The rate of change of microphytobenthos is given by:

$$\frac{d(MB_{\mathrm{w}})}{dt} = G_{\mathrm{MB_w}} - M_{\mathrm{lys,MB_w}} - \sum_{\substack{\mathrm{i=water\ predator}\\ \mathrm{groups}}} P_{\mathrm{MB_w,i}} \tag{C.3}$$

$$\frac{d(MB_{\mathrm{sed}})}{dt} = G_{\mathrm{MB_{sed}}} - M_{\mathrm{nat,MB_{sed}}} - \sum_{\substack{\mathrm{i=sed\ predator}\\ \mathrm{groups}}} P_{\mathrm{MB_{sed},i}} \tag{C.4}$$

The macrophytes (MX) are restricted to the epibenthic layer and have no water column or sediment pools. The general form of their rate of change is as follows:

$$\frac{d(MX)}{dt} = G_{\mathrm{MX}} - M_{\mathrm{MX}} - \sum_{\substack{\mathrm{i=predator}\\ \mathrm{groups}}} P_{\mathrm{MX,i}} \tag{C.5}$$

C.1.2 Bacteria

The general formulation for the dynamics of aerobic attached bacteria (where *XB* stands for Pelagic Attached Bacteria (PAB) or sediment bound Aerobic Bacteria (AEB)) is:

$$\frac{d(XB)}{dt} = G_{\mathrm{XB}} - M_{\mathrm{XB}} - \sum_{\substack{\mathrm{i=consumer}\\ \mathrm{groups}}} P_{\mathrm{XB,i}} \tag{C.6}$$

C.1.3 Invertebrate Consumers

Rate of change for a standard invertebrate consumer (CX):

$$\frac{d(CX)}{dt} = G_{\mathrm{CX}} - M_{\mathrm{CX}} - \sum_{\substack{\mathrm{i=predator}\\ \mathrm{groups}}} P_{\mathrm{CX,i}} - F_{\mathrm{CX}} \tag{C.7}$$

where F_{CX} stands for losses due to fishing on this group. Invertebrate consumers are restricted to having only a water column or epibenthic or sediment pool and can not have pools in multiple layers.

C.1.4 Vertebrate Consumers

The following are the rates of change for a vertebrate group (FX).

$$\frac{d(FX_{\mathrm{i,s}})}{dt} = G_{\mathrm{FX_{i,s}}} \tag{C.8}$$

$$\frac{d\left(FX_{\mathrm{i,r}}\right)}{dt} = G_{\mathrm{FX}_{\mathrm{i,r}}} \tag{C.9}$$

$$\frac{d\left(FX_{\mathrm{i,d}}\right)}{dt} = T_{\mathrm{IMM,FX_i}} - T_{\mathrm{EM,FX_i}} - M_{\mathrm{FX_i}} - \sum_{\substack{\mathrm{j=predator}\\ \mathrm{groups}}} P_{\mathrm{FX,j}} - F_{\mathrm{FX_i}} \tag{C.10}$$

Where the subscript s stands for structural weight (skeletal and other material that can not be reabsorbed), r for reserve weight (fats and other tissues that can be broken down when food is scarce), d for density and i represents age class i (or age class i if life phases not age classes are used – an age class may be a single year class or it may represent a proportion of the total life span of the animal, the terms age class can be used interchangeably with age class for the remainder of the document), there is one equation for each age class included. The T terms represent the movement of vertebrates in to ($T_{\mathrm{IMM,Fxi}}$) and out of ($T_{EM,\mathrm{Fxi}}$) the cell. In addition there are short-term spawning and recruitment events which effect the various FX pools. At the same point each year (the exact day dependent on the vertebrate and with a window of +/- 14 days) the vertebrates reproduce and the materials required to do this is removed from the reserve weight of FX. At this point the proportion of the age class aging into the next age class (if in a scenario with external adult stocks then the oldest age class in the system leaves the system at this time). Sometime later (the exact period dependent on the group) the recruits settle out and their weights and density are assigned to the youngest age class.

The amount of reserve weight (mg N per individual) that is used up during spawning is given by:

$$s_{\mathrm{FX}_i} = \begin{cases} \mathrm{U}_{\mathrm{FX}_i} \cdot \boldsymbol{max}\left(0, \left(\mathrm{Z}_{\mathrm{FX}} \cdot \left(1 + \mathrm{X}_{\mathrm{RS}}\right) \cdot FX_{\mathrm{i,s}} - \mathrm{Y}_{\mathrm{FX}}\right)\right) & , \ FX_{\mathrm{i,s}} + FX_{\mathrm{i,r}} > \left(1 + X_{\mathrm{RS}}\right) \cdot FX_{\mathrm{i,s}} \\ \mathrm{U}_{\mathrm{FX}_i} \cdot \boldsymbol{max}\left(0, \left(\begin{matrix} \mathrm{Z}_{\mathrm{FX}} \cdot \left(1 + \mathrm{X}_{\mathrm{RS}}\right) \cdot FX_{\mathrm{i,s}} + \left(FX_{\mathrm{i,s}} + FX_{\mathrm{i,r}}\right) \\ - \mathrm{Y}_{\mathrm{FX}} - \left(1 + \mathrm{X}_{\mathrm{RS}}\right) \cdot FX_{\mathrm{i,s}} \end{matrix}\right)\right) & \boldsymbol{,} \ FX_{\mathrm{i,s}} + FX_{\mathrm{i,r}} < \left(1 + X_{\mathrm{RS}}\right) \cdot FX_{\mathrm{i,s}} \end{cases} \tag{C.11}$$

where $\mathrm{U}_{\mathrm{Fxi}}$ is the proportion of age class i that is reproductively mature, Z_{FX} is the fraction of the weight of FX used in spawning, Y_{FX} is the spawning function constant and X_{RS} is the ratio of structural to reserve weight in well fed vertebrates.

The formulations for recruitment are given by either:

$$b_{tj} = J_t \tag{C.12}$$

for constant recruitment where J_t is element t of the recruitment vector (constant spatially and temporally). More commonly a Beverton and Holt recruitment function was implemented using the following formulation:

$$b_{tj} = \frac{\left(\dfrac{\alpha \cdot L_{tj}}{\beta + L_{tj}}\right)}{t_x} \tag{C.13}$$

is used, where α is the Beverton-Holt α for the vertebrate group; β is the Beverton-Holt β for the vertebrate group, t_x is total length of recruit period; and the biomass of the offspring of vertebrate group FX in cell j at time t (L_{tj}) is determined as follows:

$$L_{\mathrm{tj}} = \sum_{\mathrm{i=age\ class}} s_{\mathrm{FX_i}} \cdot FX_{\mathrm{i,d}} \cdot \left(1+\omega_{recruit} \cdot \delta[t]\right) \tag{C.14}$$

where s_{FXi} is the spawn from age class I of FX, $\omega_{recruit}t$ is the scalar for episodic recruiment and $\delta[t]$ is an impulse function, which is only non-zero when time t is a multiple of the period of the recruitment pulses.

C.1.5 Inanimate Pools

Rates of change for ammonia (NH) in the water column is:

$$\frac{d\left(NH_{\mathrm{w}}\right)}{dt} = -\sum_{\mathrm{i=PX_w}} P_{\mathrm{NH_w,i}} - P_{\mathrm{NH_w,MB_w}} - P_{\mathrm{NH_w,MA}} - P_{\mathrm{NH_w,PFB}} + \sum_{\mathrm{i=CX_w\ ,BF}} E_{\mathrm{i}} + \sum_{\mathrm{i=FX}} E_{\mathrm{i}} + \sum_{\substack{\mathrm{i=pelagic}\\ \mathrm{bacteria}}} E_{\mathrm{i}} - S_{\mathrm{NIT,PAB}} + R_{\mathrm{NET,w}} \tag{C.15}$$

and in the sediment:

$$\frac{d\left(NH_{\mathrm{sed}}\right)}{dt} = R_{\mathrm{NET,sed}} - S_{\mathrm{NIT,sed}} - P_{\mathrm{NH_{sed},MB_{sed}}} - P_{\mathrm{NH_{sed},SG}} + \sum_{\mathrm{i\neq BF,CX_w}} E_{\mathrm{i}} \tag{C.16}$$

where $P_{\mathrm{NH,XX}}$ is the uptake of NH by the autroph XX, E_{CX} is the production of NH by the consumer CX, $S_{\mathrm{NIT,XB}}$ is the amount of NH lost due to nitrification by the bacteria XB, R_{NET} is the amount of NH produced by denitrification.

The rates of change for nitrate (NO) in the water column is given by:

$$\frac{d\left(NO_{\mathrm{w}}\right)}{dt} = -\sum_{\mathrm{i=PX_w}} P_{\mathrm{NO_w,i}} - P_{\mathrm{NO_w,MB_w}} - P_{\mathrm{NO_w,MA}} + S_{\mathrm{NIT,PAB}} \tag{C.17}$$

and in the sediment:

$$\frac{d\left(NO_{\mathrm{sed}}\right)}{dt} = S_{\mathrm{NIT,sed}} - S_{\mathrm{DENIT,sed}} - P_{\mathrm{NO_{sed},MB_{sed}}} - P_{\mathrm{NO_{sed},SG}} \tag{C.18}$$

The rates of change of dissolved silicate (Si) in the water column is:

$$\frac{d\left(Si_{\mathrm{w}}\right)}{dt} = R_{\mathrm{DSisol,w}} - \sum_{\mathrm{i=PL_w,MB_w}} P_{\mathrm{Si_w,i}} \tag{C.19}$$

and the rate of change of detrital silica (DSi) in the water column is given by:

$$\frac{d(DSi_{\mathrm{w}})}{dt} = \mathrm{X}_{\mathrm{SiN}}\left(\sum_{\mathrm{i=PL_w,MB_w}}\left(M_{\mathrm{lys,i}} + \sum_{\mathrm{j=CX_w}} P_{\mathrm{i,j}}\right)\right) - R_{\mathrm{DSisol,w}} \tag{C.20}$$

where $\mathrm{X}_{\mathrm{SiN}}$ is the Redfield ratio of silicon and nitrogen (set at 3.0 (Murray and Parslow 1997)) and R_{DSisol} is the amount of detrital silica remineralised. Note that the equations for $\mathrm{Si}_{\mathrm{sed}}$ and $\mathrm{DSi}_{\mathrm{sed}}$ are as for (C.19) and (C.20) except that $\mathrm{CX}_{\mathrm{sed}}$ is used in the place of CX_{w} and MB is the only PX present in the sediment that uses Si.

The rates of change for dissolved oxygen (O2) in the water column is given by:

$$\frac{d(O2_{\mathrm{w}})}{dt} = \mathrm{X}_{\mathrm{ON}}\left(\sum_{\mathrm{i=PX_w}} G_{\mathrm{i}} + G_{\mathrm{MB_w}} + G_{\mathrm{MA}} + \frac{G_{\mathrm{SG}}}{2} - \sum_{\substack{\mathrm{i\neq infauna,}\\ \mathrm{MZ,BG}}} E_{\mathrm{i}} - \sum_{\mathrm{i=FX}} E_{\mathrm{i}} - \sum_{\substack{\mathrm{i=pelagic}\\ \mathrm{bacteria}}} E_{\mathrm{i}} - R_{\mathrm{DON,w}}\right) \tag{C.21}$$

and in the sediment:

$$\frac{d(O2_{\mathrm{sed}})}{dt} = \mathrm{X}_{\mathrm{ON}}\left(G_{\mathrm{MB_{sed}}} + \frac{G_{\mathrm{SG}}}{2} - \sum_{\substack{\mathrm{i=infauna,}\\ \mathrm{MZ,BG}}} E_{\mathrm{i}} - R_{\mathrm{DON,sed}}\right) \tag{C.22}$$

where X_{ON} is the Redfield ratio of oxygen and nitrogen (set at 16.0 (Murray and Parslow 1997)) and R_{DON} is the DON lost due to remineralisation.

The rates of change of dissolved organic nitrogen (DON) in the water column is:

$$\frac{d(DON_{\mathrm{w}})}{dt} = W_{\mathrm{DON,w}} - R_{\mathrm{DON,w}} - P_{\mathrm{DON,PFB}} \tag{C.23}$$

and in the sediment:

$$\frac{d(DON_{\mathrm{sed}})}{dt} = W_{\mathrm{DON,sed}} - R_{\mathrm{DON,sed}} \tag{C.24}$$

where W_{DON} is the DON produced by bacteria, R_{DON} is the DON lost due to remineralisation and $P_{\mathrm{DON,PFB}}$ is the DON taken up by pelagic free bacteria (PFB).

The rates of change of labile detritus (DL) in the water column is:

$$\frac{d(DL_{\mathrm{w}})}{dt} = \sum_{\mathrm{i=CX_w}} W_{\mathrm{DL_w,i}} + \sum_{\mathrm{i=FX}} W_{\mathrm{DL_w,i}} + \sum_{\substack{\mathrm{i=pelagic}\\ \mathrm{bacteria}}} W_{\mathrm{DL_w,i}} + \sum_{\mathrm{i=PX_w}} M_{\mathrm{lys,i}} + M_{\mathrm{lys,MB_w}} + M_{\mathrm{MA}} - P_{\mathrm{DL_w,PAB}} - P_{\mathrm{DL_w,BF}} \tag{C.25}$$

and in the sediment:

$$\frac{d(DL_{\mathrm{sed}})}{dt} = \sum_{\mathrm{i=PX_{sed}}} M_{\mathrm{nat,i}} + M_{\mathrm{nat,MB_{sed}}} + M_{\mathrm{lys,MB_{sed}}} + M_{\mathrm{SG}} + \sum_{\mathrm{i=infauna}} \left(W_{\mathrm{DL,i}} - P_{\mathrm{DL_{sed},i}}\right) + \sum_{\mathrm{i=epifauna}} \left(W_{\mathrm{DL,i}} - P_{\mathrm{DL_{sed},i}}\right) - \sum_{\mathrm{i=FX}} P_{\mathrm{DL_{sed},i}} \qquad \text{(C.26)}$$

where $W_{\mathrm{DL,CX}}$ is the amount of DL in the waste products from consumer CX and $P_{\mathrm{DL,CX}}$ is the DL consumed by CX.

The rates of change of refractory detritus (DR) in the water column is given by:

$$\frac{d(DR_{\mathrm{w}})}{dt} = \sum_{\mathrm{i=FX}} W_{\mathrm{DR_w,i}} - \sum_{\mathrm{i=CX_w}} P_{\mathrm{DR_w,i}} - P_{\mathrm{DR_w,PAB}} - J_{\mathrm{DR}} \qquad \text{(C.27)}$$

and in the sediment:

$$\frac{d(DR_{\mathrm{sed}})}{dt} = \sum_{\mathrm{i=infauna}} W_{\mathrm{DR_{sed},i}} - \sum_{\mathrm{i=infauna}} P_{\mathrm{DR_{sed},i}} + J_{\mathrm{DR}} \qquad \text{(C.28)}$$

where $W_{\mathrm{DR,CX}}$ is the DR in the wastes of consumer CX, $P_{\mathrm{DR,CX}}$ is the amount of detritus consumed by CX, infauna includes sediment bacteria and J_{DR} is the amount of DR transferred from the water column to sediment pool due to the feeding activities of the benthic filter feeders.

C.2 Process equations

C.2.1 Growth of primary producers

$$G_{\mathrm{PX}} = \mu_{\mathrm{PX}} \cdot \delta_{\mathrm{irr}} \cdot \delta_{\mathrm{N}} \cdot \delta_{\mathrm{space}} \cdot PX \qquad \text{(C.29)}$$

with μ_{PX} is the maximum growth rate, the nutrient limitation factor due to nitrogen is given by:

$$\delta_{\mathrm{N}} = \frac{DIN}{\kappa_{\mathrm{N,PX}} + DIN} \qquad \text{(C.30)}$$

(where DIN=NH+NO) except for those primary producers which are also limited by the availability of Si then nutrient limitation is given by:

$$\delta_{\mathrm{N}} = \boldsymbol{min}\left(\frac{DIN}{\kappa_{\mathrm{N,PX}} + DIN}, \frac{Si}{\kappa_{\mathrm{Si,PX}} + Si}\right) \qquad \text{(C.31)}$$

and light limitation is given by:

$$\delta_{\mathrm{irr}} = \boldsymbol{min}\left(\frac{IRR}{\kappa_{\mathrm{irr,PX}}}, 1 \right) \tag{C.32}$$

with the κ representing the half saturation constants for the respective processes, and space limitation as follows:

$$\delta_{space} = 1 - \frac{PX}{\theta_{\mathrm{PXmax}} \cdot \delta_{substrate} \cdot \delta_{habdegrad}} \tag{C.33}$$

where $\delta_{substrate}$ is the proportion of the available space that is of the correct substrate type to support this type of primary producer and $\delta_{habdegrad}$ is the scalar for local habitat degradation scenarios (see equation 4.1 in the main text for the equation used). These last two only apply to macrophytes.

Using the above formulations for growth and nutrient limitation the nutrient uptake functions for the primary producer PX are given by:

$$P_{\mathrm{NH,PX}} = G_{\mathrm{PX}} \cdot \frac{NH}{\kappa_{\mathrm{NH,PX}} + NH} \cdot \frac{\kappa_{\mathrm{NH,PX}} + DIN}{DIN} \tag{C.34}$$

$$P_{\mathrm{NO,PX}} = G_{\mathrm{PX}} \cdot \frac{NO}{DIN} \cdot \frac{\kappa_{\mathrm{NH,PX}}}{\kappa_{\mathrm{NH,PX}} + NH} \tag{C.35}$$

where $\kappa_{\mathrm{NH,PX}}$ is the half saturation constant for the uptake of NH. In addition, for PL and MB there is the uptake of Si as follows:

$$P_{\mathrm{Si,PX}} = \mathrm{X}_{\mathrm{SiN}} \cdot G_{\mathrm{PX}} \tag{C.36}$$

C.2.2 Growth of mixotrophic primary producers

Total growth of mixotrophic primary producers (G_{DF}) is given by

$$G_{\mathrm{DF}} = G_{\mathrm{phs,DF}} + \varepsilon_{\mathrm{DF}} \cdot G_{\mathrm{phag,DF}} \tag{C.37}$$

where photosynthetic growth ($G_{\mathrm{phs,DF}}$) is given by

$$G_{\mathrm{phs,DF}} = \mu_{\mathrm{DF}} \cdot \delta_{\mathrm{irr}} \cdot \delta_{\mathrm{N}} \cdot DF \tag{C.38}$$

while the phagotrophic contribution ($G_{\mathrm{phag,DF}}$) to total growth is given by

$$G_{\mathrm{phag,DF}} = \boldsymbol{min}\left(\sum_{\text{prey groups}} P_{\mathrm{i,DF}} \;, \; \frac{\mu_{\mathrm{DF}}}{\varepsilon_{\mathrm{DF}}} \cdot \delta_{\mathrm{irr}} \cdot \left(1 - \delta_{\mathrm{N}}\right) \cdot DF \right) \tag{C.39}$$

$\varepsilon_{\mathrm{DF}}$ is the assimilation efficiency of the mixotrophic dinoflagellates (set at 0.6); μ_{DF} is the temperature dependent maximum daily growth rate of the dinoflagellates (set at 0.5 mg N d^{-1}, Murray pers. com.), δ_{irr} is the light limitation factor, δ_{N} the nutrient limitation factor and $P_{\mathrm{i,DF}}$ the amount of prey group i grazed by the predator DF, which is calculated in the same way as for all other grazers in Atlantis. The light and nutrient limitation factors were largely calculated as for the pure autotrophs in Atlantis. Since there is strong evidence that dinoflagellates show an increase in efficiency at low light levels (Jeong *et al.* 1999, Li *et al.* 1999), there were some modifications made to the formulation of light limitation for this group. The modification is based on general observations that, due to increased efficiency at low light levels, mixotrophic growth rates are two- to three-fold higher than those of strict phototrophic growth under identical (low light) conditions (Skovgaard 1996, Legrand *et al.* 1998, Li *et al.* 1999). The final form of the light limitation factor (δ_{irr}) is:

$$\delta_{\mathrm{irr}} = \begin{cases} \boldsymbol{min}\left(IRR \cdot 0.01 + 0.018\,, 1\right), & 0 < IRR \le 0.1 \\ \boldsymbol{min}\left(\dfrac{IRR}{\kappa_{\mathrm{irr,DF}}}, 1\right), & \text{otherwise} \end{cases} \tag{C.40}$$

and the nutrient limitation factor as

$$\delta_{\mathrm{N}} = \frac{DIN}{\kappa_{\mathrm{N,DF}} + DIN} \tag{C.41}$$

where *DIN* represents the total inorganic nitrogen pool (made up of ammonia and nitrate).

C.2.3 Growth of bacteria

Growth of the bacteria (G_{XB}) is given by

$$G_{\mathrm{XB}} = \mu_{\mathrm{XB}} \cdot XB \cdot \boldsymbol{max}\left(0\,, \left(1 - \rho_{\mathrm{XB}}\right)^{\psi}\right) \tag{C.42}$$

and

$$\rho_{\mathrm{XB}} = \frac{XB}{\left(\tau_{\mathrm{DL,XB}} \cdot DL + \tau_{\mathrm{DR,XB}} \cdot DR\right) \cdot \delta_{\mathrm{O2}} \cdot \delta_{\mathrm{stim}}} \tag{C.43}$$

with μ_{XB} representing the maximum temperature-dependent daily growth rate for the group XB. *XB* is the current pool of bacteria and *DL* and *DR* are the labile and refractory detrital pools (all in mg N m^{-3}); $\tau_{\mathrm{DL,\,XB}}$ and $\tau_{\mathrm{DR,\,XB}}$ represent the maximum possible biomass of XB per biomass of that grade of detritus; ψ is the exponent dictating the reduction in growth as the bacterial pool approaches its maximum attainable levels (set to 3) and δ_{O2} is the oxygen limitation factor, which is given by:

$$\delta_{\mathrm{O2}} = \begin{cases} \dfrac{\gamma_{\mathrm{O2}}}{\gamma_{\mathrm{O2}} + \gamma_{\mathrm{XB}}}, & \text{XB benthic} \\ 1, & \text{otherwise} \end{cases} \tag{C.44}$$

where γ_{XB} is the half oxygen mortality depth for XB, and the oxygen horizon (γ_{O2}) is given by:

$$\gamma_{\mathrm{O2}} = \frac{2 \cdot O2_{\mathrm{sed}} \cdot \gamma_{\mathrm{sed}}}{O2_{\mathrm{bw}}} \tag{C.45}$$

with $O2_{\mathrm{sed}}$ the concentration of oxygen in the sediments, $O2_{\mathrm{bw}}$ the concentration in the bottom water and γ_{sed} the depth of the sediment layer considered in the model. Finally δ_{stim} indicates the degree of stimulation of the bacteria by bioturbation and it is calculated as follows:

$$\delta_{\mathrm{stim}} = \begin{cases} \dfrac{\delta_{\mathrm{te}} \cdot 250 \cdot (POR\text{-}0.225)}{193.75}, & \text{XB benthic} \\ 1, & \text{otherwise} \end{cases} \tag{C.46}$$

C.2.4 Growth of consumers

The growth of an invertebrate consumer (CX) is given by:

$$G_{\mathrm{CX}} = \left(\varepsilon_{\mathrm{CX}} \cdot \sum_{\substack{\mathrm{i=living} \\ \mathrm{prey}}} P_{\mathrm{i,CX}} + \sum_{\mathrm{j=DL,DR}} \left(P_{\mathrm{j,CX}} \cdot \varepsilon_{\mathrm{CX,j}} \right) \right) \cdot \delta_{\mathrm{space}} \cdot \delta_{\mathrm{o2}} \tag{C.47}$$

with $\varepsilon_{\mathrm{CX}}$ the growth efficiency of CX when feeding on live prey, $\varepsilon_{\mathrm{CX,j}}$ the efficiency when feeding on detritus (DL treated separately to DR), space limitation given by:

$$\delta_{space} = \begin{cases} 1 - \dfrac{\delta_{substrate} \cdot \delta_{habdegrad} \cdot \left(CX - \theta_{\mathrm{CXlow}}\right) \cdot \frac{\left(CX - \theta_{\mathrm{CXlow}}\right)}{CX - \theta_{\mathrm{CXlow}} + \kappa_{\mathrm{CXsat}}}}{\delta_{substrate} \cdot \delta_{habdegrad} \cdot \left(CX - \theta_{\mathrm{CXlow}}\right) \cdot \frac{\left(CX - \theta_{\mathrm{CXlow}}\right)}{CX - \theta_{\mathrm{CXlow}} + \kappa_{\mathrm{CXsat}}} + \kappa_{\mathrm{CXthresh}}}, & CX = \mathrm{BF} \text{ and } CX > \theta_{CXlow} \\ 1, & \text{otherwise} \end{cases} \tag{C.48}$$

where θ_{CXmax} is the maximum biomass per area allowed for CX, θ_{CXlow} is the crowding lower threshold, κ_{CXsat} is the crowding half saturation level, $\kappa_{\mathrm{CXthresh}}$ is the crowding threshold (this formulation is based on that of the European Regional Seas Ecosystem Model II (ERSEM II) (Blackford 1997)), $\delta_{substrate}$ is the proportion of the available space that is of the correct substrate type to support this type of primary producer and $\delta_{habdegrad}$ is the scalar for local habitat degradation scenarios (see equation 4.1 in the main text for the equation used).

The oxygen limitation is given by:

$$\text{or } \delta_{\mathrm{o2}} = \begin{cases} \dfrac{\gamma_{\mathrm{o2}}}{\gamma_{\mathrm{o2}} + \kappa_{\mathrm{CX,Mo2}}} & \text{, if epifauna or infauna} \\ 1 & \text{, if pelagic} \end{cases} \tag{C.49}$$

where γ_{o2} is the depth of the oxygen horizon and $\kappa_{\mathrm{CX,Mo2}}$ is the half oxygen mortality depth.

The growth for each vertebrate group, is calculated by equation of the same form as (C.47), but per age class of each vertebrate, the result is then apportioned to structural and reserve weight increases such that:

$$G_{\mathrm{FX}_{\mathrm{i,s}}} = \Lambda \cdot G_{\mathrm{FX}_{\mathrm{i}}} \tag{C.50}$$

$$G_{\mathrm{FX}_{\mathrm{i,r}}} = (1 - \Lambda) \cdot G_{\mathrm{FX}_{\mathrm{i}}} \tag{C.51}$$

where

$$\Lambda = \begin{cases} \dfrac{\frac{1}{\mathrm{X_{RS}}} + \mathrm{X_{pR,FX}} \cdot \left(\frac{FX_{\mathrm{i,r}}}{\mathrm{X_{RS}} \cdot FX_{\mathrm{i,s}}} \right)}{\frac{1}{\mathrm{X_{RS}}} + \left(\frac{FX_{\mathrm{i,r}}}{\mathrm{X_{RS}} \cdot FX_{\mathrm{i,s}}} \right)} & \text{, if} > 0 \text{ and } G_{\mathrm{FX_i}} > 0 \\ = 0 & \text{, otherwise} \end{cases} \tag{C.52}$$

with $\mathrm{X_{RS}}$ the maximum ratio of reserve to structural weight FX can have and $\mathrm{X_{pR,FX}}$ is the relative degree to which FX concentrates on replenishing reserves rather than undergoing structural growth when underweight.

The grazing term is given by:

$$P_{prey,CX} = \frac{CX \cdot \mathrm{k_{CX}} \cdot \mathrm{p_{prey,CX}} \cdot \delta_{refuge} \cdot prey}{1 + \mathrm{k_{CX}} \cdot \dfrac{\varepsilon_{CX} \cdot \left(\sum\limits_{\text{j=live prey groups}} \mathrm{p_{j,CX}} \cdot \mathrm{j} \right) + \varepsilon_{\mathrm{CX,DL}} \cdot \mathrm{p_{DL,CX}} + \varepsilon_{\mathrm{CX,DR}} \cdot \mathrm{p_{DR,CX}}}{\mu_{CX}}} \tag{C.53}$$

where “prey” is the group being consumed by CX, $\mathrm{k_{cx}}$ is the clearance rate of CX and $\mathrm{p_{prey,CX}}$ is preference (or availability) of that prey for the predator CX. This last parameter is similar to the “vulnerability” parameters in ECOSIM (Christensen *et al.* 2000) and represents the fact that the entire prey population will not be available to the predators at any one time (some may be hiding for instance). The availability of the food is further modified by δ_{refuge} if the group is dependent on biogenic habitat refuges, or size refuges (if they are physically outside the gape range of their vertebrate predators) or if the spatial range of the predator and prey do not completely overlap (and so explicit spatial refuges exist). If the group is dependent on biogenic habitat then δ_{refuge} is given by:

$$\delta_{refuge} = \exp\left(-\kappa_{\mathrm{cov}er} \cdot d_{\mathrm{cov}er} + \theta_{\mathrm{cov}er}\right) + \frac{1}{\theta_{\mathrm{cov}er}} \tag{C.54}$$

where $-\kappa_{cover}$ is the refuge magnitude coefficient, θ_{cover} is the habitat steepness coefficient and d_{cover} is the relative cover in the cell for the prey, which is calculated by:

$$d_{\mathrm{cov}er} = \left(\delta_{substrate,habdegrad} \cdot \rho_{substrate} + \sum_j \rho_{biogenic,j} \right) \cdot \left(1 + \rho_{canyon}\right) \tag{C.55}$$

where $\delta_{substrate,habdegrad}$ is degradation in the physical habitat due to coastal development (e.g. reefs broken up), $\rho_{substrate}$ is the proportion of the cell covered with suitable substrate types, $\rho_{substrate,j}$ is the proportion of the cell covered by biogenic habitat defining group *j* and ρ_{canyon} is the proportion of the cell covered by canyons (which is treated as an enhancement factor here as they are known concentrate production but their absence does not prevent the establishment and growth of the groups (Alan Williams *pers. com.*)).

If the refuges is size-based (only true for vertebrates) then the available prey in cohort i of vertebrate group FX (FX_i), for the vertebrate predator (FY_j), is given by:

$$prey \cdot \delta_{refuge} = \begin{cases} \sum \mathrm{p}_{\mathrm{FX_i,FY_j}} \cdot \frac{(FX_{\mathrm{i,s}} + FX_{\mathrm{i,r}}) \cdot FX_{\mathrm{i,d}}}{\mathrm{cell\_vol}} & , \ \Theta_{\mathrm{low,FY}} \cdot \mathrm{FY}_{\mathrm{j,s}} \leq FX_{\mathrm{s,i}} \leq \Theta_{\mathrm{up,FY}} \cdot \mathrm{FY}_{\mathrm{j,s}} \\ 0 & , \ \text{otherwise} \end{cases} \tag{C.56}$$

where $\Theta_{\mathrm{low,FY}}$ is the lower prey selection size limit for FY and $\Theta_{\mathrm{up,FY}}$ is the upper prey selection size limit. The δ_{refuge} for benthic prey (vertebrate and invertebrate alike) is calculated as follows:

$$\delta_{refuge} = \begin{cases} 0 & , \ \gamma_{\mathrm{CX}} < \gamma_{\mathrm{top}} \\ \frac{(\gamma_{\mathrm{CX}} - \gamma_{\mathrm{top}})}{(\gamma_{\mathrm{o2}} - \gamma_{\mathrm{top}})} & , \ \gamma_{\mathrm{top}} < \gamma_{\mathrm{CX}} < \gamma_{\mathrm{o2}} \\ 1 & , \ \gamma_{\mathrm{top}} < \gamma_{\mathrm{o2}} < \gamma_{\mathrm{CX}} \end{cases} \tag{C.57}$$

and if anaerobic

$$\delta_{refuge} = \begin{cases} 1 & , \gamma_{\mathrm{CX}} < \gamma_{\mathrm{top}} \\ \left(1 - \frac{(\gamma_{CX} - \gamma_{top})}{(\gamma_{o2} - \gamma_{top})} \right) & , \gamma_{\mathrm{top}} < \gamma_{\mathrm{CX}} < \gamma_{\mathrm{o2}} \\ 0 & , \gamma_{\mathrm{top}} < \gamma_{\mathrm{o2}} < \gamma_{\mathrm{CX}} \end{cases} \tag{C.58}$$

where γ_{CX} is the depth in the sediment that the predator CX can forage down to and γ_{top} is set to zero (as there is only one sediment layer).

C.2.5 Mortality and loss functions

The mortality terms for invertebrate consumers and autotrophs are in terms of lost biomass while those for vertebrates refer to the number of individuals lost. Nevertheless the general form of the equations is the same (but the units of the coefficients obviously differ between the vertebrates and other groups). The natural mortality term for group XX is given by

$$M_{\mathrm{XX}} = \mathrm{m}_{\mathrm{lin,XX}} \cdot XX + \mathrm{m}_{\mathrm{quad,XX}} \cdot XX^2 + (1 - \delta_{\mathrm{o2}}) \cdot \mathrm{m}_{\mathrm{o2,XX}} \cdot XX + \mathrm{m}_{\mathrm{special,XX}} \cdot XX + \mathrm{m}_{\mathrm{top,XX}} \cdot XX \tag{C.59}$$

where $\mathrm{m}_{\mathrm{lin,XX}}$ is the coefficient of linear mortality for XX, $\mathrm{m}_{\mathrm{quad,XX}}$ is the coefficient of quadratic mortality for the group XX, $\mathrm{m}_{\mathrm{O2,XX}}$ is the coefficient of oxygen dependent mortality and $\mathrm{m}_{\mathrm{special,XX}}$ is the special (additional) loss rate for XX. This rate of "special" mortality is usually set to zero, except in the following cases:

$$\mathrm{m}_{\mathrm{special,MA}} = STRESS \cdot \mathrm{m}_{\mathrm{STRESS}} \tag{C.60}$$

$$\mathrm{m}_{\mathrm{special,SG}} = DIN \cdot \mathrm{m}_{\mathrm{DIN}} \tag{C.61}$$

where $\mathrm{m}_{\mathrm{STRESS}}$ and $\mathrm{m}_{\mathrm{DIN}}$ are the coefficient of mortality due to mechanical stress and fouling by epiphytes, respectively. Lastly:

$$\mathrm{m}_{\mathrm{special,FX_i}} = \begin{cases} \dfrac{\mathrm{m}_{\mathrm{starve,FX}} \cdot \theta_{\mathrm{starve}} \cdot (1+\mathrm{X}_{\mathrm{RS}}) \cdot \mathrm{FX}_{\mathrm{i,s}} - (\mathrm{FX}_{\mathrm{i,s}} + \mathrm{FX}_{\mathrm{i,r}})}{(1+\mathrm{X}_{\mathrm{RS}}) \cdot \mathrm{FX}_{\mathrm{i,s}}} & \text{, if} > 0 \\ = 0 & \text{, otherwise} \end{cases} \tag{C.62}$$

with $\mathrm{m}_{\mathrm{starve,FX}}$ is the threshold ratio of reserve to structural weight at which death due to starvation is likely. The final term of equation (C.61) was adopted from ERSEM I (Bryant *et al.* 1995) to represent the impact of seabirds and any other top predators not represented explicitly by the dynamic groups in the model. While all the groups in the model had a linear mortality term, some groups (the vertebrates and higher trophic level zooplankton and benthic groups) suffered mortality described by a quadratic term. Only benthic consumers had oxygen dependent mortality, the macrophyte and vertebrate groups had special mortality as shown above and $\mathrm{m}_{\mathrm{top}}$ is only applied to the vertebrate groups.

The final loss term is one that is applied to the microscopic primary producers only and it represents lysis. The losses of a primary producer (PX) to lysis is formulated as follows:

$$M_{\mathrm{lys,PX}} = \frac{\mathrm{m}_{\mathrm{lys,PX}} \cdot PX}{\delta_{\mathrm{N}} + 0.1} \tag{C.63}$$

with $\mathrm{m}_{\mathrm{lys,PX}}$ the rate of lysis.

C.2.6 Waste processes

The production of waste products by invertebrate and vertebrate consumers are handled in the same way, but in the case of vertebrates the mortality term has to be converted from a density to a biomass before being used in the following equations. The production of labile detritus (DL) by consumer group XX is given by:

$$W_{\mathrm{DL}} = \left(\begin{array}{l} (1-\varepsilon_{\mathrm{XX}}) \cdot \Gamma_{\mathrm{XX}} \cdot \sum\limits_{\substack{\mathrm{i=living\ prey} \\ \mathrm{group}}} P_{\mathrm{i,XX}} + (1-\varepsilon_{\mathrm{XX,DL}}) \cdot \Gamma_{\mathrm{XX,DL}} \cdot P_{\mathrm{DL,XX}} \\ + (1-\varepsilon_{\mathrm{XX,DR}}) \cdot \Gamma_{\mathrm{XX,DR}} \cdot P_{\mathrm{DR,XX}} + \varphi_{\mathrm{XX}} \cdot \mathrm{M}_{\mathrm{XX}} \end{array} \right) \cdot \mathrm{f}_{\mathrm{XX,DL}} \tag{C.64}$$

with φ_{XX} the proportion of mortality losses assigned to detritus, Γ_{XX} the proportion of the growth inefficiency of XX when feeding on live prey that is sent to detritus, $\Gamma_{\mathrm{XX,DL}}$ the proportion of the growth inefficiency of XX when feeding on DL that is sent to detritus, $\Gamma_{\mathrm{XX,DR}}$ the proportion of the growth inefficiency of XX when feeding on refractory detritus (DR) that is sent to detritus and $\mathrm{f}_{\mathrm{XX,DL}}$ is the proportion of the total detritus produced that is of the type DL. The same equation is used for the production of DR (W_{DR}), except that the final multiplication of the brackets by $\mathrm{f}_{\mathrm{XX,DL}}$ is replaced by multiplication by ($1-\mathrm{f}_{\mathrm{XX,DL}}$).

The other main waste product is excreted ammonia. The general formulation used for the production of ammonia by a consumer XX (invertebrate or vertebrate) is as follows:

$$E_{\mathrm{XX}} = (1-\varphi_{\mathrm{XX}}) \cdot M_{\mathrm{XX}} + (1-\varepsilon_{\mathrm{XX}}) \cdot (1-\Gamma_{\mathrm{XX}}) \cdot \sum_{\substack{\text{i=living prey}\\ \text{group}}} P_{\mathrm{i,XX}} + (1-\varepsilon_{\mathrm{XX,DL}}) \cdot (1-\Gamma_{\mathrm{XX,DL}}) \cdot P_{\mathrm{DL,XX}}$$
$$+ (1-\varepsilon_{\mathrm{XX,DR}}) \cdot (1-\Gamma_{\mathrm{XX,DR}}) \cdot P_{\mathrm{DR,XX}} \qquad \text{(C.65)}$$

C.2.7 Physical processes

The only physical processes in Atlanits that differ from those in the Port Phillip Bay Integrated Model (PPBIM, detailed in Murray and Parslow 1997, Walker 1997) are bioturbation, bioirrigation and the calculation of the light attenuation coefficient.

The formulation of the light attenuation coefficient is from Fulton (2001) and is given by:

$$n = n_{\mathrm{w}} + n_{\mathrm{DON}} \cdot DON + n_{\mathrm{D}} \cdot (DL + DR) + n_{\mathrm{P}} \cdot \sum_{\mathrm{i=PX}} PX + n_{\mathrm{susp}} \cdot SUSP \qquad \text{(C.66)}$$

with n_{w} the background extinction coefficient, n_{DON} the contribution due to DON, n_{D} the contribution due to detritus, n_{P} the contribution due to phytoplankton (PX) and n_{susp} the contribution due to suspended sediments (SUSP).

The equations for bioirrigation are as detailed in Walker (1997) for PPBIM, but it is tied to the dynamical sediment fauna via an "enhancement" term similar to that of ERSEM I (Ebenhöh *et al.* 1995).

As Atlantis uses explicit sediment layers it can approximate particulate diffusion, expulsion (whereby material at depth is moved to the surface) and exchange with the surface by transferring sediment between the appropriate layers of the model. Only those particulate components (tracers) that are allowed in the sediments and are not macrobenthos (sediment grains, settled phytoplankton, microphytobenthos, meiobenthos, detritus and sediment bacteria) can be acted upon by bioturbation. The formulation implemented expresses the tracer concentration in the *i*th sediment layer ($BX_i(t)$) at the end of a time step as:

$$BX_i(t+\Delta t) = \frac{BX_{i+1}(t) \cdot f_{i+1} + BX_{i-1}(t) \cdot f_{i-1} + BX_i(t) \cdot z_i - 2 \cdot BX_i(t) \cdot f_i - BX_i(t) \cdot c_i - BX_i(t) \cdot w_i + BX_0(t) \cdot w_0}{f_{i+1} + f_{i-1} + z_i - 2f_i - c_i - w_i + w_0} \qquad \text{(C.67)}$$

$$f_i = \frac{\varpi \cdot \chi \cdot o \cdot \Omega_i}{z_i} \qquad \text{(C.68)}$$

$$b_i = \sigma \cdot \chi \cdot o \cdot \Omega_i \qquad \text{(C.69)}$$

$$w_i = \eta \cdot \chi \cdot o \cdot \Omega_i \qquad \text{(C.70)}$$

Where k_i represents the thickness transferred from i due to particulate diffusion, c_i is the thickness moved to the surface from layer *i* by expulsion and w_i is the thickness moved from layer *i* due to exchange with surface layers and z_i is the thickness of layer *i*. The thicknesses f_i, c_i and w_i only differ in a single parameter. For the parameters they share, χ represents the base density of biological activity; o represents the modification to the baseline to reflect dynamic sediment fauna activity in the ecological sub-model (calculated in much the same way as that of ERSEM (see Ebenhöh *et al.* 1995)); and Ω_i is the depth dependence of the mixing process (this is a simple functional form, as of PPBIM, and though usually constant it is also possible to implement linear, parabolic and half-Gaussian forms (Walker 1997)). The parameter which does differ in the calculation of f_i, c_i and w_i is the base rate of each process - ϖ is the rate of particle diffusion (m^2 per Δt per unit biomass of bioturbative benthos per m^2), σ is the rate of expulsion (m per Δt per unit biomass of bioturbative benthos per m^2) and η is the rate of exchange between the surface and deeper layers (m per Δt per unit biomass of bioturbative benthos per m^2). A small amount of burial of sediments and associated detrital particles is also enabled using a similar formulation.

C.2.8 Equations for Sediment Chemistry

Use of a compound effect of enhanced bioturbation (δ_e), and porosity (POR) is based on observations by Alongi (1998) and the relationship detailed by Blackburn (1987). Using equations (C.42) to (C.45), the utilisation of labile detritus by aerobic bacteria is given by:

$$P_{\mathrm{DL,XB}} = G_{\mathrm{XB}} \cdot \frac{\rho_{\mathrm{XB}} \cdot \tau_{\mathrm{XB,DL}} \cdot DL}{XB \cdot \varepsilon_{\mathrm{XB,DL}}} \tag{C.71}$$

where $\varepsilon_{\mathrm{XB,DL}}$ is the assimilation efficiency of the bacteria on labile detritus. The uptake of refractory detritus is calculated similarly. The natural mortality term (M_{XB}) is as for the other invertebrates, but the term representing predation losses to predator group i ($P_{\mathrm{XB,i}}$) is given by:

$$P_{\mathrm{XB,i}} = P_{\mathrm{DL,i}} \cdot \rho_{\mathrm{X}B} \cdot \tau_{\mathrm{XB,DL}} + P_{\mathrm{DR,i}} \cdot \rho_{\mathrm{XB}} \cdot \tau_{\mathrm{XB,DR}} \tag{C.72}$$

The waste handling equations for bacteria are also different to those for other invertebrates since wastes are channelled into DON not DL. All of the equations for the Anaerobic Bacteria (ANB) are as for XB here, except that any δ_{O2} factors in the equations are replaced by ($1-\delta_{\mathrm{O2}}$). Adopting these equations for the attached bacteria made it easier to identify a method of introducing dynamic flexibility to the empirical nitrification-denitrification model proposed by Murray and Parslow (1999a) for PPBIM.

To integrate a more interactive form of the processes governing nitrification and denitrification into Atlantis, the empirical sediment chemistry model used in PPBIM (Murray and Parslow 1999) is linked directly to the activities of sediment bacteria and infauna. The amount of ammonia produced by the remineralisation of DON (R_{DON}) is handled as in PPBIM, that is:

$$R_{\mathrm{DON}} = \Phi \cdot DON \cdot POR \tag{C.73}$$

where Φ is the temperature-dependent rate of breakdown for DON (set at 0.00176 d^{-1}, Murray pers. com.). In PPBIM, equations similar to (C.73) were used to calculate the production of ammonia due to the breakdown of DL and DR (Murray and Parslow 1997). This is not the case in Atlantis, where the production of the remainder of the ammonia is dependent upon the activity of sediment dwelling fauna and flora. Thus, the total ammonia available for nitrification and denitrification (R_{NET}) is:

$$R_{\mathrm{NET}} = \boldsymbol{max}\left(0\,, R_{\mathrm{DON}} + E_{\mathrm{AEB}} + E_{\mathrm{ANB}} + \xi \cdot \left(E_{\mathrm{OB}} + E_{\mathrm{BD}}\right) - P_{\mathrm{NH,MB}}\right) \tag{C.74}$$

where $P_{\mathrm{NH,MB}}$ is the uptake of NH by MB (see equations for autotrophs), E_{XX} is the ammonia released by XX and ξ is the fraction of the excreted NH by infauna that contributes available nitrogen for nitrification and denitrification (set to 0.95). The form of E_{XX} for OB and BD is of the general form given for heterotrophs in Appendix B, but that for AEB and ANB is slightly different and is given by:

$$E_{\mathrm{XB}} = P_{\mathrm{DL,XB}} \cdot \left(1-\varepsilon_{\mathrm{XB,DL}}\right) + P_{\mathrm{DR,XB}} \cdot \left(1-\varepsilon_{\mathrm{XB,DR}}\right) + M_{\mathrm{XB}} - W_{\mathrm{DON}} - W_{\mathrm{DR}} \qquad \text{(C.75)}$$

where E_{XB} is the release of NH by XB, $\varepsilon_{\mathrm{XB,DX}}$ is the efficiency of XB on the detritus fraction DX, and the production of DON (W_{DON}) and DR (W_{DR}) are calculated as follows:

$$W_{\mathrm{DON}} = \left(P_{\mathrm{DL,XB}} \cdot \left(1-\varepsilon_{\mathrm{XB,DL}}\right) + P_{\mathrm{DR,XB}} \cdot \left(1-\varepsilon_{\mathrm{XB,DR}}\right) + M_{\mathrm{XB}} \cdot \varphi_{\mathrm{XB}}\right) \cdot \mathrm{f}_{\mathrm{XB,DON}} \qquad \text{(C.76)}$$

$$W_{\mathrm{DR}} = \left(P_{\mathrm{DL,XB}} \cdot \left(1-\varepsilon_{\mathrm{XB,DL}}\right) + M_{\mathrm{XB}} \cdot \varphi_{\mathrm{XB}}\right) \cdot \mathrm{f}_{\mathrm{XB,DR}} \qquad \text{(C.77)}$$

where φ_{XB} indicates the fraction of the losses of XB due to natural mortality that are not released as NH and $\mathrm{f}_{\mathrm{XB,DX}}$ is the fraction of the products of growth inefficiency and mortality directed to the detritus fraction DX. Using equation (C.74) the processes of nitrification and denitrification were completed using the form of the empirical model of Murray and Parslow (1999a), giving nitrification (S_{NIT}) as:

$$S_{\mathrm{NIT}} = R_{\mathrm{NET}} \cdot \theta_{\mathrm{DMAX}} \cdot \boldsymbol{max}\left(0, 1 - \frac{R_{\mathrm{NET}} \cdot \gamma_{\mathrm{SED}}}{{}_{\mathrm{r0}}}\right) \qquad \text{(C.78)}$$

and denitrification (S_{DENIT}) as:

$$S_{\mathrm{DENIT}} = S_{\mathrm{NIT}} \cdot \boldsymbol{min}\left(1, \frac{R_{\mathrm{NET}} \cdot \gamma_{\mathrm{SED}}}{\theta_{\mathrm{rD}}}\right) \qquad \text{(C.79)}$$

where θ_{DMAX} is the maximum rate of denitrification (set at 0.25, Murray pers. com.), θ_{r0} is the temperature-dependent minimum rate of respiration that supports nitrification (set at 200, Murray and Parslow 1997) and θ_{rD} (set at 10, Murray and Parslow 1997) is the peak of the nitrification-denitrification curve (as defined by Murray and Parslow 1999). This general form is adopted from PPBIM due to its demonstrated performance and robustness (Murray and Parslow 1999, Fulton 2001).

The equations for oxygen are modified from (Walker 1997), due to the more interactive representation of the sediment processes, and oxygen dynamics are now governed by:

$$\begin{aligned} O2_{\mathrm{bw,t+1}} = {} & \frac{\left(O2_{\mathrm{bw,t}} \cdot VOL_{\mathrm{bw}} + O2_{\mathrm{sed,t}} \cdot VOL_{\mathrm{por}}\right)}{VOL_{\mathrm{bw}} + VOL_{\mathrm{por}}} \\ & + e^{-\phi_{irr} \cdot \left(\frac{1}{VOL_{\mathrm{bw}}} + \frac{1}{VOL_{\mathrm{por}}}\right)} \cdot \left(O2_{\mathrm{bw,t}} - \frac{\left(O2_{\mathrm{bw,t}} \cdot VOL_{\mathrm{bw}} + O2_{\mathrm{sed,t}} \cdot VOL_{\mathrm{por}}\right)}{VOL_{\mathrm{bw}} + VOL_{\mathrm{por}}}\right) \end{aligned} \qquad \text{(C.80)}$$

$$O2_{\mathrm{sed,t+1}} = O2_{\mathrm{sed,t}} - \frac{VOL_{\mathrm{bw}}}{VOL_{\mathrm{por}}} \cdot \left(O2_{\mathrm{bw},t+1} - O2_{\mathrm{bw,t}}\right) \qquad \text{(C.81)}$$

where ϖ_{irr} is the exchange rate due to irrigation, $O2_{\mathrm{SED,t}}$ is the concentration of oxygen in the sediment at time t, $O2_{\mathrm{bw,t}}$ is the concentration of oxygen in the bottom water at time t, VOL_{bw} is the volume of the bottom water layer and the porewater volume above the oxygen horizon is given by:

$$VOL_{\mathrm{por}} = POR \cdot \frac{\gamma_{\mathrm{o2}} \cdot \chi_{\mathrm{cell}}}{VOL_{\mathrm{sed}}} \tag{C.82}$$

with VOL_{sed} being the volume of the entire sediment layer and χ_{cell} is the area of the cell.

C.2.9 Equations for Vertebrate Movement

The default vertebrate movement (in terms of the density d of vertebrate group FX, age class i, in cell j) is given by:

$$FX_{i,d,j} = \begin{cases} FX_{i,tot} \cdot \left(\vartheta \cdot \left(FX^{D}_{j,qrt+1,FX} - FX^{D}_{j,qrt,FX}\right) + FX^{D}_{j,qrt,FX}\right), & \mathrm{qrt} < 4 \\ FX_{i,tot} \cdot \left(\vartheta \cdot \left(FX^{D}_{j,1,FX} - FX^{D}_{j,qrt,FX}\right) + FX^{D}_{j,qrt,FX}\right), & \mathrm{qrt} = 4 \end{cases} \tag{C.83}$$

where $FX_{\mathrm{i,tot}}$ is the total number of FX in age class i in the entire system (that is the sum over all cells), ϑ is the proportion of the current quarter of the year which has already passed, $\mathrm{FX}^{\mathrm{D}}_{\mathrm{j,qrt,FX}}$ is the proportion of the population of FX found in cell j in the qrt quarter of the year.

For the forage and density dependent vertebrate movement scheme, the following formulation is used:

$$G_{\mathrm{FX,i,j,potential}} = \begin{cases} g_{\mathrm{roc\_mult}} \cdot G_{\mathrm{FX,i,j}}, & G_{\mathrm{FX,i,j}} > g_{\mathrm{thesh}} \\ \dfrac{G_{\mathrm{FX,i,j}}}{g_{\mathrm{roc\_mult}}}, & \mathrm{otherwise} \end{cases} \tag{C.84}$$

$$G_{\mathrm{FX,i,tot}} = \sum_{\mathrm{all\ j}} G_{\mathrm{FX,i,j}} \tag{C.85}$$

$$FX^{D}_{i,d,j,t} = \frac{G_{\mathrm{FX,i,j,potential}} \cdot d_{\mathrm{cov}er} \cdot \delta_{depth} \cdot FX^{D}_{i,d,j,t,other}}{G_{\mathrm{FX,i,tot}}} \tag{C.86}$$

$$FX_{\mathrm{i,d,j}} = FX_{\mathrm{i,tot}} \cdot \left(FX_{i,vel} \cdot \left(FX^{D}_{i,d,j,t} - FX^{D}_{i,d,j,t-1}\right) + FX^{D}_{i,d,j,t-1}\right) \tag{C.87}$$

where $G_{\mathrm{FX,i,j,potential}}$ is a measure of the potential attractiveness of the cell j based on the available forage, $G_{\mathrm{FX,i,j}}$ is calculated as of G_{CX} in equation C.34, $g_{\mathrm{roc\_mult}}$ is a constant reflecting how much more attractive a site with forage sufficient to support $\mathrm{FX_i}$ is over a site with poor food resources, g_{thresh} is the potential growth rate (as an index of the quality of the resources) where $\mathrm{FX_i}$ switch from finding the site desirable to undesirable and d_{cover} is the relative cover in the cell for the group $\mathrm{FX_i}$ (set to one for all groups that are not habitat-dependent). To take into account other pressures on fish movement (such as seasonal or spawning migration) the calculation of the proportion $\mathrm{FX}^{\mathrm{D}}_{\mathrm{i,d,j,t}}$ is weighted by the ideal distribution for those other migration factors and then the final distribution is determined by interpolating between the current distribution and the ideal distribution (taking the maximum swim speed of the vertebrate into account so that individuals can not move further than they could actually swim in reality). These $FX_{i,d,j}$ values are then normalised so that their sum is one. If a vertebrate group is site attached then it only moves vertically at most, and if the group employs maternal care then the movement scheme is calculated for the mothers and then applied to them and the juvenile age classes.

D FISHERIES AND MANAGEMENT MODEL EQUATIONS

D.1 Fishing mortality and management

Many different fishing mortality equations were used depending on the scenario of fishery of interest. The general form of the amount caught at time t is given by:

$$F_{\mathrm{FX,t}} = C_{\mathrm{eff}} \cdot \left(FX_{\mathrm{s,i}} + FX_{\mathrm{r,i}}\right) \cdot FX_{\mathrm{d,i}} \cdot \delta^{FC}_{space,FX,i} \cdot \delta^{FC}_{depth,FX,i} \cdot \left(1 - FX^{FC}_{escape}\right) \qquad \text{(D.1)}$$

where, $\delta^{FC}_{depth,FX,i}$ is the proportion of $\mathrm{FX_i}$ at the depth the fishing gear is sweeping, FX^{FC}_{escape} is the proportional escapement (either constant or size-based), the accessibility of $\mathrm{FX_i}$ for the fishery FC ($\delta^{FC}_{space,FX,i}$) is given by either simple percentage overlap of the habitats occupied by $\mathrm{FX_i}$ using :

$$\delta^{FC}_{space,FX,i} = \rho_{FC,FX} \cdot \mathrm{h}_{\mathrm{FX,i}}$$

or the distribution model developed by Ellis and Pantus (2001):

$$\delta^{FC}_{space,FX,i} = \left(1 - \min\left(1, \left(\rho_{FC,FX} \cdot \exp\left(-\frac{\eta_{FC}}{\rho_{FC,FX}} \cdot \frac{\log\left(1+\mathrm{h}_{\mathrm{FX,i}} \cdot \eta_{pattern}\right)}{\eta_{pattern}}\right) + 1 - \rho_{FC,FX}\right)^{\eta_{patch}}\right)\right) \qquad \text{(D.2)}$$

where $\mathrm{h_{FXi}}$ is the catchability of the age class *i* of FX (see equations (D8) to (D12)), $\rho_{FC,FX}$ is the proportional overlap of fishery FC and vertebrate group FX, η_{FC} is the cover of the fishery, $\eta_{pattern}$ is the distribution of the fishing within the area, η_{patch} is the number of patch types in the area; and finally the effort applied is given by

$$C_{\mathrm{eff}} = \delta_{depth,FC} \cdot \mathrm{m}_{\mathrm{FC,FX}} \cdot \mathrm{q}_{\mathrm{FC,FX}} \cdot \mathrm{a}_{\mathrm{j,FC,FX}} \qquad \text{(D.3)}$$

with $\mathrm{m_{FC,FX}}$ the current coefficient of fishing mortality for FX by fishery FC (it can change through time as fishing pressure changes) – it is the percentage of total effort (of the FC fishing fleet) concentrated in cell j if the fleet was allowed to act at will; $\mathrm{q_{FC,FX}}$ is the selectivity coefficient of the gear used by fishery FC on FX, and $\mathrm{a_{j,FC,FX}}$ is the adjustment to the final rate due to management actions currently applied in cell j (this can also change through time). As indicated by (D.1) and (D.2) the fishing implemented for is a variation on the simple catch equation.

D.2 Effort

The effort coefficient $\mathrm{m_{FC,FX}}$ can be calculated in a number of ways, ranging from prescribed effort matrices to dynamic processes (such as basing all effort allocation on past catches or making minor modifications to long term trends of effort distribution based on recent catches in each area) and is given by one of the following equations. If effort is a temporally prescribed constant (i.e. effort per quarter is fixed):

$$\mathrm{m}_{\mathrm{FC,FX}} = \begin{cases} \left(\vartheta \cdot \left(\mathrm{m}^{D}_{qrt+1,FC} - \mathrm{m}^{D}_{qrt,FC}\right) + \mathrm{m}^{D}_{qrt,FC}\right), & \mathrm{qrt} < 4 \\ \left(\vartheta \cdot \left(\mathrm{m}^{D}_{1,FC} - \mathrm{m}^{D}_{qrt,FC}\right) + \mathrm{m}^{D}_{qrt,FC}\right), & \mathrm{qrt} = 4 \end{cases} \tag{D.4}$$

where ϑ is the proportion of the current quarter of the year which has already passed, $\mathrm{m}^{\mathrm{D}}_{\mathrm{j,qrt,FX}}$ is the effort of fishery FC in the qrt quarter of the year. This value can be used as is (i.e. homogeneously distributed across all cells) or spatially weighted based on the CPUE from previous time step. If the spatial distribution of the effort is prescribed rather than the temporal component then the equation used is the same as for (D.4) but with $\mathrm{m}^{\mathrm{D}}_{\mathrm{qrt,FX}}$ replaced with $\mathrm{m}^{\mathrm{D}}_{\mathrm{j,qrt,FX}}$ the effort of fishery FC in cell *j* in quarter qrt.

If the effort distribution is calculated dynamically then $\mathrm{m}_{\mathrm{FC,FX}}$ in cell *j* time *t* is given by:

$$\mathrm{m}_{\mathrm{FC,FX,t,j}} = \mathrm{F}_{vel} \cdot \left(\mathrm{m}_{\mathrm{eff,t,j}} - \mathrm{m}_{\mathrm{FC,FX,t\text{-}1,j}}\right) + \mathrm{m}_{\mathrm{FC,FX,t\text{-}1,j}} \tag{D.5}$$

where the ideal new distribution $\mathrm{m}_{\mathrm{eff,t,j}}$ given by:

$$\mathrm{m}_{\mathrm{eff,t,j}} = \frac{\omega_{FC,\max} \cdot B_{CPUE,j}}{\sum_{j} B_{CPUE,j}} \tag{D.6}$$

with $\omega_{FC,max}$ is maximum allowable effort and $B_{CPUE,j}$ is the CPUE in the cell in the previous time step and the velocity F_{vel} is based on the distance between cells (if CPUE based only) or the distance to the ports if using a fleet dynamics model.

The equations (D4) to (D6) are used for the commercial fleets, but the recreational fishery is represented in a slightly different way:

$$\mathrm{m}_{\mathrm{eff,t,j}} = F_{recvel} \cdot \kappa_{pop} \cdot \sum_{port} N_{pop,k} \cdot \begin{cases} \left(\vartheta \cdot \left(\mathrm{m}^{D}_{j,qrt+1,rec} - \mathrm{m}^{D}_{j,qrt,rec}\right) + \mathrm{m}^{D}_{j,qrt,rec}\right), & \mathrm{qrt} < 4 \\ \left(\vartheta \cdot \left(\mathrm{m}^{D}_{j,1,rec} - \mathrm{m}^{D}_{j,qrt,rec}\right) + \mathrm{m}^{D}_{j,qrt,rec}\right), & \mathrm{qrt} = 4 \end{cases} \tag{D.7}$$

where $N_{pop,k}$ is the population in port *k*, κ_{pop} is the proportion of the population that fishes recreationally, F_{recvel} is the velocity of recreational vessels based on distances to port, and $\mathrm{m}^{D}_{j,qrt,FX}$ the effort of the recreational fishery in cell *j* in quarter qrt (usually used to constrain recreational and charter-boat effort to coastal cells).

If the effort displacement option is being used then if the total effort for a fishery would drop using any of these dynamic formulations (either due to the imposition of MPAs or declining stocks) then the difference in each area is redistributed to the adjacent cells with the greatest biomass of the target groups.

D.3 Selectivity

The selectivity coefficient $\mathrm{q}_{\mathrm{FC,FX}}$ may be given by one of five functions depending on the gear used by the fishery. It may be a constant proportion applied to all age classes or it may be size based and calculated based on a normal, logistic, lognormal or gamma distribution. The selectivity of the gear with regard to cohort *i* of vertebrate group FX ($q_{,i,FX}$) is given by one of the following equations (depending on the selectivity curve of the gear). If the selectivity curve is a constant

$$\mathrm{q}_{\mathrm{FX}_\mathrm{i}} = \mathrm{\kappa}_{\mathrm{i},0} \tag{D.8}$$

if a logistic selectivity curve is used for the gear then

$$\mathrm{q}_{\mathrm{FX}_\mathrm{i}} = \left(1 + \exp\left(-\kappa_{i,1} \cdot \left(l_{i,FX} - \kappa_{i,2}\right)\right)\right)^{-1} \tag{D.9}$$

if a normal selectivity curve is used for the gear then

$$\mathrm{q}_{\mathrm{FX}_\mathrm{i}} = \exp\left(\frac{-\left(l_{i,FX} - \kappa_{i,2}\right)^2}{2 \cdot \left(\kappa_{i,1}\right)^2}\right) \tag{D.10}$$

if a lognormal selectivity curve is used for the gear then

$$\mathrm{q}_{\mathrm{FX}_\mathrm{i}} = \frac{\exp\left(-\left(\log\left(l_{i,FX}\right) - \kappa_{i,2}\right)^2 \cdot \left(2 \cdot \left(\kappa_{i,1}\right)^2\right)^{-1}\right)}{\kappa_{i,1} \cdot \sqrt{2 \cdot \pi}} \tag{D.11}$$

and if a gamma selectivity curve is used for the gear then

$$\mathrm{q}_{\mathrm{FX}_\mathrm{i}} = \left(\frac{l_{i,FX}}{\kappa_{i,2}}\right)^{\kappa_{i,2} \cdot \left(0.5 \cdot \left(\sqrt{\left(\kappa_{i,2}\right)^2 + 4 \cdot \left(\kappa_{i,1}\right)^2} - \kappa_{i,2}\right)\right)^{-1}} \cdot \exp\left(\left(\kappa_{i,2} - l_{i,FX}\right) \cdot \left(0.5 \cdot \left(\sqrt{\left(\kappa_{i,2}\right)^2 + 4 \cdot \left(\kappa_{i,1}\right)^2} - \kappa_{i,2}\right)\right)^{-1}\right) \tag{D.12}$$

where $\kappa_{\iota,0}$ is the selectivity constant for the fishery on the *i*th age class of vertebrate group FX, $\kappa_{\iota,1}$ is the selectivity coefficient (spread of the curve) for the fishery on the *i*th age class of vertebrate group FX, $\kappa_{\iota,2}$ is the selectivity coefficient (length at which 50% of the population is selected) for the fishery on the *i*th age class of vertebrate group FX and the length ($l_{j,FX}$) of a vertebrate from cohort *i* in vertebrate group *FX* is given by:

$$l_{i,FX} = \left(\frac{\left(\kappa_{wet} \cdot \mathrm{X}_{\mathrm{CN}} \cdot \left(FX_{s,i} + FX_{r,i}\right)\right)}{\alpha_{w,FX} \cdot 1000}\right)^{1/\beta_{w,FX}} \tag{D.13}$$

with X_{CN} the redfield ratio of carbon to nitrogen, κ_{wet} is the conversion coefficient from weight to dry weight, $\alpha_{w,FX}$ is a scaling coefficient in the length-weight relationship for FX and $\beta_{w,FX}$ is the exponent in the length-weight relationship. In practice the logistic or normal selectivity curves are used (unless otherwise specified for that scenario). The parameters used to specify the selectivity curve may be adjusted in response to “management decisions” made under some of the alternative management scenarios (i.e. gear restrictions can translate into changes in selectivity).

D.4 Management

The final coefficient ($\mathrm{a}_{\mathrm{j,FC,FX}}$) represents management actions that influence fishing, but not by modifying the gear and its selectivity. This includes effort reduction and spatial and temporal closures or zoning. All of these involve calculating or specifying the proportion of "at will" effort that is allowed under the current management strategy. If a trigger event (such as target or vulnerable stock decline) occurs it initiates a gradual reduction in effort, which can be reversed later if the trigger recovers. The following equation is used to determine the final management coefficient ($\mathrm{a}_{\mathrm{j,FC,FX}}$):

$$\mathrm{a}_{\mathrm{j,FC,FX}} = \begin{cases} \left(1 - \dfrac{\left(t - t_{trigger}\right)}{t_{reduction}} \cdot \left(1 - d\right)\right) \cdot \delta_{TAC} \cdot \delta_{season} \cdot \delta_{\mathrm{zone}} & \text{, if trigger event has occured} \\ \delta_{TAC} \cdot \delta_{season} \cdot \delta_{zone} & \text{, if no trigger event} \end{cases} \tag{D.9}$$

where δ_{TAC} is one normally (set to zero if a TACs are present and has been exceeded), δ_{season} is one if the fishery is not under a temporal closure and δ_{zone} is one if there is no spatial management in the cell (it is zero if the cell is closed and there is no infringement), t is current time, $t_{trigger}$ is the time a trigger event occurred, $t_{reduciton}$ is the time period over which the reduction in effort is to take place and d is the final level of effort allowed relative to the original level when the effort reduction was triggered. Infringement of any of the management conditions is represented by keeping the management scalars at a user defined level above the values they would drop to if there was no infringement. Management actions such as setting TACs or defining fishing seasons year (which are scaled back as stocks decline) occur on the first timestep of each year and are in place for the entire year.

D.5 Discarding

Three forms of discarding were included in the model and the specific one used in any particular case depended on the scenario under consideration. The simplest form of discarding saw a set proportion of the total catch of vertebrate FX by fishery FC discarded – the proportion of each age class actually discarded matched the percentage make-up of the catch. Another form of discarding saw a fixed proportion of all age classes discarded. The final, more realistic formulation used was based on size where the biomass of age class i of FX discarded and is given by

$$FX_i^{discarded} = \mathrm{F}_{\mathrm{FX,t}} \cdot \begin{cases} \left(1 - \rho_{illegal}\right) & l_{i,FX} < l_{legal} \\ \rho_{discard} & l_{i,FX} \geq l_{legal} \end{cases} \tag{D.10}$$

and the length of the vertebrate ($l_{i,FX}$) is given by (D.8).

APPENDIX E – 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 E1: 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
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

Parameter	Value	Parameter	Value
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
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 E.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		

Table E.3: Constant ECOPATH parameters used in the sampling model. As many groups have the same values for these parameters, rather than repeat the parameters for each group individually, the parameters are given per type-of-group and all groups within that type would use those parameter values.

Type-of-group	**Consumer**	**Assimilation Efficiency**	**Imports**	**Proportion that Flows to Detritus**				**Discard Fate**			
				DLpel	**DRpel**	**DLsed**	**DRsed**	**DLpel**	**DRpel**	**DLsed**	**DRsed**
large phytoplankton	0	-	0	1.0	0.0	0.0	0.0	0.8	0.0	0.0	0.2
small phytoplankton	0	-	0	1.0	0.0	0.0	0.0	0.8	0.0	0.0	0.2
small zooplankton	1	0.4	0	1.0	0.0	0.0	0.0	0.8	0.0	0.0	0.2
large zooplankton	1	0.3	0	1.0	0.0	0.0	0.0	0.8	0.0	0.0	0.2
large pelagic invertebrates	1	0.2	0	1.0	0.0	0.0	0.0	0.8	0.0	0.0	0.2
infauna	1	0.2	0	0.0	0.0	0.9	0.1	0.8	0.0	0.0	0.2
epifauna	1	0.2	0	0.0	0.0	0.9	0.1	0.8	0.0	0.0	0.2
macrophytes	0	-	0	0.1	0.9	0	0.0	0.8	0.0	0.0	0.2
vertebrates (juvenile and adult)	1	0.2	0	0.1	0.9	0	0.0	0.8	0.0	0.0	0.2
pelagic labile detritus	-	-	0	0.2	0.3	0.4	0.1	0.8	0.0	0.0	0.2
pelagic refractory detritus	-	-	0	0.0	0.0	0.4	0.6	0.8	0.0	0.0	0.2
benthic labile detritus	-	-	0	0.0	0.45	0.05	0.5	0.8	0.0	0.0	0.2
benthic labile detritus	-	-	0	0.0	0.05	0.05	0.9	0.8	0.0	0.0	0.2

APPENDIX F – SAMPLING MODEL EQUATIONS

Note that the due to the large number of symbols needed to describe the many equations that make up the sampling model some symbols have had to be reused with sub or superscripts to differentiate between their multiple meanings. Also please consider the symbols here in isolation to those used elsewhere in the report (in the equations or discussion of the operating model and indicators) as the reuse of symbols does not mean they represent the same thing in the different parts of the report – where this is the case it is specifically stated. I have tried to keep this reuse and potential confusion to a minimum and where it has been necessary to do it as logically as possible, I'm sorry for any inconvenience or confusion this may cause.

The following equations describe the formulations implemented at each sampling step to generate "noisy" data from the original deterministic output of the operating model.

F.1 Invertebrates

For each invertebrate group (*i*) the "sampled" biomass, consumption or production ф_i is given by one of the following expressions. If error was assumed to be lognormal (the usual case) the formulation used for ф_i:

$$\text{ф}_i = \exp\left(\left(\sqrt{-2\cdot\log(\text{я}_1)}\cdot\cos(2\cdot\pi\cdot\text{я}_2)\right)\cdot\text{ъ} - \frac{\text{ъ}}{2}\right)\cdot B\cdot\varsigma \qquad \text{(F.1)}$$

where B is the true biomass, ς is the (potentially biased) sample mean as a proportion of the true mean and is Ъe variance of the error, Я_1 and Я_2 both ~U(0,1).

F.2 Physical characteristics

Each physical variable sampled had error added in the same way as for the invertebrate pools (F.1). The only major difference to biological pools was that the Ъ for physical variables is usually substantially smaller than for biological variables (see the parameter values in Appendix E).

F.3 Vertebrates

F.3.1 Charismatic vertebrates

The biomass, numbers and condition for all waders and any seabirds are sampled to reflect bird survey data (all sampling spots are included here as they all fall within the usual survey distance from shore restrictions used in reality). The data is lumped into juvenile, fledged and adult groups and error added in the same way as for invertebrates (F.1). The total numbers of seabirds caught (and discarded) are also recorded (with error calculated in the standard way). The average weight and total numbers counted is then used to give a biomass estimate for the waders and seabirds.

Mammal stock numbers are collected per age class (with error added) to represent aerial and boat surveys of the populations. To determine the mammal size structure the population is sub-sampled (with error as given by (F.1)) twice within the sampling period. This information is binned into size classes and the average condition of the individual in each size class plus the distribution of numbers and age across the size-classes is determined for the stock as a whole and for the animals which have been taken during fishing operations. Normally distributed aging error is included in this sub-sampling process. The distributions are normalized so that they can be used to calculate proportional contributions to network values. The index of condition for the *j*th size class $(\text{Ж}_{i,j})$ is calculated using the following formulation

$$\text{ж}_{ij} = \sum_{a=1}^{n_{age}} \sum_{k=1}^{n_j} \frac{\text{ђ}_{ia}^{r}}{\left(\text{ђ}_{ia}^{s} + \text{ђ}_{ia}^{r}\right) \cdot n_j} \qquad \text{(F.2)}$$

where n_{age} is the number of age classes for group *i* (in this case there is 20 for mammals), n_j is the number of individuals in the sub-sample of the correct size for that age class, ђ_{ij}^{s} is the structural weight of an average individual in cohort *a* of group *i*, and ђ_{ij}^{r} is the reserve weight of an average individual in cohort *a* of group *i.*

The total production and consumption by mammals, waders and seabirds is calculated by summing the actual values across the age classes. An estimate of this value is then given by adding error in the same way as for invertebrates (F.1).

F.4 Fisheries data

For each fishery the total catch, effort and discards information is aggregated (summed) at the level of the fishing zone (or fishing region) and then error is added in the same way as given in (F.1). The fine scale information is also sub-sampled (with error as given by (F.1)) and binned in the same way as outlined above for the mammals. This data is used to produce an index of condition for the individuals in the catch and to estimate a distribution of the numbers in the catch across the size-classes.

F.5 Fisheries independent sampling (fished vertebrates)

For fish and sharks the population is sampled by “research fishing” (which has associated catchability, selectivity and availability coefficients – see Appendix D for the form of the equations used) twice within the sampling period . This information is binned into size classes and the average condition of the individual in each size class, plus the distribution of numbers and age across the size-classes, is determined for the stock as a whole. Aging error is included in this sub-sampling process. All of these estimates and distributions are then used to produce estimates for the numbers in the stock, catch and discards. Selectivity is then estimated and used to correct the estimates of stock size and numbers in the catch and discards (by dividing the value by the selectivity estimate for that size class). Thus the final estimate of abundance for vertebrate group *i* in the zone (or region) ( ) $\aleph_i$ given by

$$\aleph_i = \sum_{j} \frac{\text{ф}_{i,z} \cdot \text{ч}_{i,j,z}}{\text{Ш}_{i,j,z}} \qquad \text{(F.3)}$$

where *j* is the size class, $\text{ф}_{i,z}$ is the total estimated biomass of group *i* in zone *z*, and $\text{ч}_{i,j,z}$ is the proportion of the population of *i* in size class *j* in zone *z* (from the selectivity corrected size distribution) and $\text{Ш}_{i,j,z}$ is the average weight of the individual in size class *j* of group *i* in zone *z*.

The estimate of the number of fish caught uses the same method, except that the size information used in the calculation of the average weight of the fish comes from catches not fisheries independent samples; and the catch landed rather than the biomass estimate ($\text{ф}_{i,z}$) is used in the calculation of $\aleph_i$.

F.6 Diet composition

Diet composition is not calculated on time or spatial scales as fine as for the other data types. The temporal resolution is dealt with in section 4.7 of the main report and spatially only an inshore and an offshore diet composition is estimated. This matches the resolution used in the past, though finer scale sampling is beginning to be used in some of the most highly valuable fisheries of the wealthier nations.

The contribution to the diet of group i by group k in area z ($Q_{i,k,z}$) is calculated using

$$Q_{i,k,z} = \sum_{y} \text{Д}_{i,k} \cdot B_{k,z} \cdot \text{Ц}_{i,k} \tag{F.4}$$

where y is the sampled individuals in that area (inshore or offshore), $\text{Д}_{i,k}$ is the availability of group k to group i, B_k is the biomass of group k in that area (for vertebrate group k as the prey, B_k is actually the biomass of group k where the individual vertebrate is within the gape limits of the predator group i) and $\text{Ц}_{i,k}$ is a random multinomial patchiness scalar. To cope with the fact that very rare groups would hardly ever appear in diet matrices if the multinomial alone was used to mimic patchiness the diet composition was calculated in two steps. The first step $\text{Д}_{i,k} \cdot B_k$ (equivalent of sampling 1000 individuals with something identifiable in their stomachs) involved using the $Q_{i,k,z}$ as written with the multinomial determining which actual groups appeared in the stomach of the sampled individuals. The process was then repeated for 10 individuals, but with the $\text{Д}_{i,k} \cdot B_k$ rescaled so that all groups with a $\phi_{i,k} \cdot B_k$ of more than 0.0 but less than 0.01 were set to 0.1, this new set of $\text{Д}_{i,k} \cdot B_k$ were normalised and then the multinomial selection of actual prey included in the sample was repeated with this new set of potential contributions. In this way it was possible to represent the under and over-contribution of certain groups to sampled diet composition. This contribution $Q_{i,k,z}$ was then normalized to produce the proportion contribution to the diet of group i by group k ($Q'_{i,k,z}$) using

$$Q'_{i,k,z} = \frac{Q_{i,k,z}}{\sum_{k} Q_{i,k,z}} \tag{F.5}$$

Once this diet composition's has been determined using (F.4) and (F.5), the trophic level of group i in an area z ($\text{Ŧ}_{i,z}$) is the weighted average of its prey's trophic levels +1, and is given by

$$\text{Ŧ}_{i,z} = 1 + \sum_{k} \text{Ŧ}_{k,z} \cdot Q'_{i,k,z} \tag{F.6}$$

with detritus and primary producers having a trophic level of 1.