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Development of an ecosystem approach to the monitoring and management of Western Australian fisheries

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Contents

2005/063 Development of an ecosystem approach to the monitoring and management of Western Australian fisheries

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Objectives

1. Test the robustness of statistical procedures to identify impacts of multi-sector fishing on community composition using existing fishery data.
2. Assess the level of change in community composition in each bioregion of WA during the previous 30 years.
3. Identify key data to which ecosystem structure and management strategies are most sensitive and which should be collected in the future.
4. Identify critical changes in exploitation and/or environment that would impact marine ecosystems markedly.
5. Identify areas where more detailed research and/or monitoring are needed.

Non-technical summary

Outcomes achieved to date

results obtained from analysis of commercial fishery data have demonstrated that there has been no reduction in mean trophic level or mean maximum length in the finfish catches recorded within the West Coast, South Coast, Gascoyne, Pilbara or Kimberley bioregions, as would be expected if fish faunas were experiencing the phenomenon termed "fishing down the food web". On the contrary, trophic level and mean maximum length have increased, possibly reflecting an expansion of the fisheries onto previously less exploited components of the stocks. The management strategy used for the demersal fishery in the West Coast Bioregion, i.e. maintaining fishing mortality below reference points derived from estimates of natural mortality, was found likely to be very effective in producing a recovery of depleted fish stocks and, provided appropriate abundance, age composition, and other biological data are collected, sustaining key fish stocks in a multi-species fishery and ecosystem. The management strategy was also found to be effective in sustaining those stocks if carrying capacity and/or growth change, e.g. through environmental change affecting biological characteristics or habitat. Collection and analysis of reliable abundance, age composition and biological data for key targeted and non-targeted species should be enhanced (or initiated) to monitor possible changes in biological characteristics.

Diversity and ecosystem-based indicators were calculated for commercial finfish fisheries from 1976 to 2005 for the West Coast, South Coast, Gascoyne, Pilbara and Kimberley bioregions. The ecosystem-based indices, which detect changes in the species composition of the food web within the ecosystem, were mean trophic level (1=herbivores to 5= peak predators), mean size of the fish in the catch (calculated using the maximum lengths for the species), and a Fishery-in-Balance (FIB) indicator. The latter adjusts the magnitude of annual catch to account for changes in observed mean trophic level to determine whether the scaled catch has increased or decreased relative to that within a reference year. The time series of ecosystem-based indices demonstrated that, in each bioregion, both the mean trophic level and the mean maximum length of catches have increased; possibly because the fisheries in some of these bioregions have developed and expanded spatially over the period from 1976 to 2005. In the West and South Coast bioregions, the series appear to have stabilized, but they continue to increase in the other bioregions. There is no evidence from the commercial fishery data that, from 1976 to 2005, there has been any reduction in trophic level or mean maximum length that would be expected from fishing down the food web, and thus, it appears that, at this time, ecosystem services have not been affected by fishing or other factors. It is possible that the indices are being maintained by continued spatial expansion of fishing and/or changes in targeting, and that, if exploitation increases and expansion is no longer possible, the ecosystem-based indices will stabilize and begin to decline.

Statistical analysis of the Western Australian (WA) data using the software package, Primer, demonstrated, however, that the species composition of the catches reported by fishers within each of the bioregions had changed over time. The species that most characterized the changes were identified. The analysis was unable, however, to distinguish whether change in species composition and abundance resulted from fishery practice, recording and reporting processes, management changes, changes in exploitation or targeting, environmental change or a combination of these factors. Thus, while change in species composition had occurred in each bioregion, it was possible that this was due simply to expansion of fisheries to exploit different species groups in different locations within each bioregion. It is also possible that improved reporting by fishers, i.e. reporting of catches at species rather than family level, may also have contributed to the apparent change in species composition.

This and other studies have demonstrated that data collected for key fished and non-fished stocks within an ecosystem should include time series of total catches and reliable relative abundance indices, samples of age, length and sex composition representative both of the catches of each fishing sector and of the wild stocks, and data from studies of population biology, i.e. growth, maturity, sex change, reproduction. Where appropriate and cost-effective, fishery-dependent data should be augmented by fishery-independent relative abundance, age composition and biological data. Limited recreational catch data are currently available, and current estimates of abundance, i.e. cpue data from commercial fishers, are likely to be influenced greatly by changes in fishing power and targeting by fishers.

Management strategies for the West Coast Bioregion were explored in this study. Results of this exploration demonstrated that the indicators, reference points and decision rules that have been adopted by the Department of Fisheries Western Australia for the demersal scalefish fisheries of the West Coast Bioregion are likely to be highly effective. Thus, for Western Australia's finfish fisheries, fishing mortality estimates appear currently to be more reliable indicators of fishery status than abundance estimates, where the reference points for those indicators are those determined from the estimates of natural mortality for the different species. Reference points for spawning biomass such as maximum sustainable yield and virgin spawning biomass rely to a much greater extent on trends in relative abundance, estimates of which are unreliable due to a paucity of accurate abundance data.

Environmental change may affect the growth, reproduction, and carrying capacity of the various stocks. Changes in growth and reproduction can be monitored by appropriate data collection and analysis using traditional methods of fish population biology. Changes in carrying capacity will be more difficult to assess as determination of a stock-recruitment relationship is typically difficult to determine, even when this is assumed to be constant. Although it was not possible to distinguish between fishery and environmental effects, the study demonstrated that the management strategies, which had been accepted for use in the demersal scalefish fishery of the West Coast Bioregion, would be likely to continue to be effective if the species were affected by changes in biological characteristics or carrying capacity.

KEYWORDS: Ecosystem, trophic level, mean maximum length, species composition.

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Background

Internationally, there has been an increasing focus on ecosystem-based management (Pikitch *et al*., 2004). Within Australia, the moves to ensure that Ecologically Sustainable Development (ESD) processes are implemented reflected the increasing community demand for more rigorous assessments of the broader impacts of fishing at an ecosystem level. A clear example of this was the introduction of the Environmental Protection and Biodiversity Conservation (EPBC) Act 1999 by the federal government and the concomitant changes to the Wildlife Protection (regulation of exports and imports) Act, 1982. The outcome of this act was that exports are not permitted from WA's (and all Australian) commercial fisheries unless those fisheries are capable of meeting the requirements of the "Guidelines for the Ecologically Sustainable Management of Fisheries". These comprehensive assessments, which are administered by the federal Department of the Environment, Water, Heritage and the Arts (DEWHA), were to be completed before the end of 2004. In addition to assessing the status of the target species, the assessments cover issues related to the broader ecosystem and most also result in a series of recommendations that will need to be fulfilled before the next assessment of the fishery in five years time.

The National ESD framework, which was developed as part of the FRDC's ESD Reporting and Assessment subprogram, was designed to facilitate the completion of these ESD/EPBC reports for each fishery (Fletcher *et al*., 2002). The current ESD framework and the EPBC assessments examine the effects of an individual fishery on the ecosystem. However, it is becoming clear that, in many circumstances, separating the potential impact of each fishery on the community structure of an area may not be either possible or even appropriate given the overlapping nature of many fishing activities (*i.e.* the same species may be caught by many fisheries). Within WA, the shift towards an Integrated Fisheries Management (IFM) approach has also occurred due to the recognition of the significant interactions between fisheries and that the cumulative impacts of multi-sector fishing within a region need to be managed.

Given the widespread nature of these issues, the Natural Resource Management Standing Committee (NRMSC) supported an initiative to develop an extension to the current ESD framework so that it can deal with cross-fishery issues (such as cumulative impacts and allocation amongst groups) up to multi-sector analyses within the bioregion, leading to regional marine planning which is synonymous with Integrated Ocean Management. A major outcome from this extended framework was the requirement to assess the ecosystem structure within a bioregional context, rather than at an individual fishery level.

The completion of any ecosystem assessment will require appropriate data. Indeed, the quality of such data has been identified in FRDC 2000/311 as being critical for the development of models describing the impact of fishing on exploited marine ecosystems. However, if ecosystem assessments require the establishment of new, dedicated monitoring programs, this would, in most circumstances, be a very expensive and time-consuming operation. Moreover, as these programs would only begin collecting data now, most would suffer from a lack of any historical information for comparisons. Consequently, before any new program is established, the potential for using data already being collected needs to be assessed and, as recognised at its September 2004 meeting by the Research Steering Committee for FRDC 2000/311, if new data are to be collected, the key data to be collected in the future need to be identified.

Collectively, fishing activities in most regions catch a large number of species. These species usually include a broad range of sizes, habits, trophic levels and other characteristics that cover many elements of the ecosystem. It is possible, therefore, that the information within the long term datasets generated from all fishing activities in a region may reflect the structure of the local ecosystem and hence any changes that may have occurred through time.

Changes in the species composition of catches have already been used to demonstrate that some fisheries have had an impact on community structure - such as the reduction in the importance of lutjanid snappers in the North West shelf fishery of WA following extensive Taiwanese pair trawling (Sainsbury, 1988). Similarly, changes in the species composition of commercial catches have been used as evidence that some regions have experienced a "fishing down the food web" impact (*e.g.* Pauly et al., 1998). More recently, the species compositions of commercial catches along with a number of other biological and environmental metrics have been used to assess long term changes in marine ecosystems in the northeast USA (Link *et al*., 2002).

WA has commercial catch data extending over a 30 year period. Consequently, it was considered possible that detailed statistical analyses of these data would be useful for determining if any major changes in community composition have already occurred in any of the bioregions of WA.

Need

There are currently no standard techniques that can be used to assess whether fishing has had significant impacts on ecosystem structure. The current round of EPBC assessments has demonstrated that a more robust assessment of ecosystem impacts will be required when the next application for export exemption is submitted in five years time. It is vital for WA's export fisheries that the types of changes in exploitation and/or environment that could cause marked changes in ecosystem structure are identified, the types of data necessary to assess whether such changes are occurring are determined, and cost-effective methods are developed to provide information on the level of ecosystem changes that have occurred.

Having appropriate techniques to assess whether fishing within a region is significantly modifying the ecosystem is seen as a national priority. At a workshop held by the Research Committee of the Australian Fisheries Managers Forum (AFMF) it was concluded that different types and levels of analysis are likely to be needed for different purposes and it would be "inappropriate to abandon any particular approach prematurely". Given the potential costs associated with the collection of new or additional data at an ecosystem level, it is imperative that attempts are made to see if existing datasets, such as those currently maintained by fisheries agencies on catch and effort, are suitable for this purpose and to apply modelling approaches, such as those developed in FRDC 2000/311, to these refined datasets.

Completing routine ecosystem-level assessments will only be feasible when tools are available to simplify the complex process of analysing the multi-sector, multi-species databases that are present in WA (and most jurisdictions). There is a need, therefore, to identify and test a variety of statistical methods using these datasets to determine if any are suitable for detecting shifts in ecosystems or community structure, and more particularly, whether management strategies are robust to the types of changes that might be expected.

Objectives

The objectives listed in the project application, which were all achieved, were:

1. Test the robustness of statistical procedures to identify impacts of multi-sector fishing on community composition using existing fishery data.
2. Assess the level of change in community composition in each bioregion of WA during the previous 30 years.
3. Identify key data to which ecosystem structure and management strategies are most sensitive and which should be collected in the future.
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5. Identify areas where more detailed research and/or monitoring are needed.

Methods

Databases of fisheries data

The data bases of commercial, recreational and charter boat fishery statistics have been refined (and continue to be further refined) to allow greater flexibility and more ready access. James Scandol, who, at the time, was employed by NSW Fisheries, visited the Department of Fisheries, Western Australia, to advise on database structures and approaches that might be used to automate reporting.

The potential of the available data for use in the current study was reviewed. The need for a long term time series of data was recognised, as, potentially, such data would be more likely to contain the contrast necessary for detecting whether or not the species composition of the catch had changed over the history of fishing. Accordingly, the decision was made to concentrate our analyses on the long term time series of commercial data for those fishing methods that consistently produce information on catch rates for a range of fish species. Thus, although creel surveys have now produced recreational data for various regions on several occasions, and the periods covered by the time series of charter boat catch data in the different regions now extends back to 2002, these data sets still lack the contrast that is present in the catch and effort data derived from the mandatory monthly forms supplied by commercial fishers. However, the charter boat data are recorded at a far finer temporal and spatial resolution than the CAES data and will become increasingly useful as the time series becomes extended and more detailed studies are undertaken.

It should be noted that change in community composition within the reported catch data might relate to modifications in recording or fishing practices rather than reflecting a real change in trophic structure. While data on past recording and fishing practices of commercial fishers in WA have not been collected, there would be value in monitoring such practices in the future. Such data would allow the effect of changes in species composition resulting from changed recording and fishing practices to be taken into account when assessing whether fishing has had an impact on the trophic structure of the ecosystem.

Preliminary analyses

Other studies that have assessed whether there has been a change in species composition and/or biodiversity have typically employed multidimensional scaling ordination and a range of diversity indices (*e.g.* Ley *et al.* 2002; Labropoulou and Papaconstantinou, 2004), although it has been recognized that these latter indices have limitations (*e.g.* Rochet and Trenkel, 2003) (Table 1). Of the numerous diversity indices that have been described in the literature, we have selected a subset of the more typically used indices for use in this study, noting that these are representative of the types of indices that are available. More recently, Pauly *et al.* (1998, 2000) have developed an index of the mean trophic level of the catch and a fishery-in-balance index that reflects whether, in ecological terms, the fishery is balanced, *i.e.* the index declines when catches do not increase as expected when, through consumer demand, fishers choose to exploit species with lower trophic levels. Thus, if the trophic index declines by 1 unit, it is expected that, as the biomass of fish at this level is approximately 10 times greater than the biomass at the higher trophic level, then catches 10 times as great would be expected. If greater catches are taken than would be expected given the mean trophic level, the FIB index would increase, and vice versa. Indices that reflect the changing size spectrum of the ecosystem have also been developed, such as that of the mean maximum length (Piet and Jennings., 2005). We explored each of these indices in our study, calculating the traditional diversity indices using data for the individual fishing gears considered in the study (see below) and the ecosystem-based indices using data both for the individual fishing gears and, as is more typical, for the pooled catches over those fishing gears. Recognizing that different fishing gears target different species and select fish of different sizes, it was considered that the calculation of the ecosystem-based indices using data for the individual fishing gears would allow an assessment of whether the characteristics of the catches of the species caught by each gear type, which are reflected in the indices, displayed trends that were consistent among gear types.

Table 1. Details of the diversity and ecosystem indices used in the study.

The number of species	***S***
The Shannon-Wiener index	$$H' = -\sum_{i=1}^{S} p_i \log_e p_i$$ which has been modified to $$H' = -\sum_{i=1}^{S} p_i \log_e (p_i + 0.0001)$$ in our study, to allow for zero values of p_i (*e.g.* Huang *et al.* 2006). In these formulae, p_i represents the proportion (by number) of individual fish of species *i* in the sample. However, in our study, the index has been calculated using proportion by mass rather than number.
Species richness (Margalef, 1968)	$$d = \frac{(S-1)}{\log_e N}$$

Evenness (Pielou, 1966)	$$J = \frac{H'}{H_{\max}}$$ where $H_{\max} = \log_e S$
Dominance, Simpson's Index (Krebs, 1989)	$$D = \sum p_i^2$$
Mean trophic level (Pauly *et al*., 1998)	$$\frac{\sum B_i TL_i}{\sum B_i}$$ where B_i is the biomass of the i^{th} species and TL_i is its trophic level
Mean of maximum length (Piet and Jennings, 2005)	$$\frac{\sum N_i L_{\max_i}}{\sum N_i}$$ where N_i is the number of fish of species *i* that are caught and $L_{\max_i}$ is the maximum length of individuals of that species. In the calculations in this study, as numbers of fish were not available, B_i was used in place of N_i.
Fishery in Balance Index (Pauly *et al*., 2000)	$$FIB = \log\left[Y_i\left(\frac{1}{TE}\right)^{TL_i}\right] - \log\left[Y_0\left(\frac{1}{TE}\right)^{TL_0}\right]$$ where *Y* is the catch and *TL* is the mean trophic level in year *i*, *TE* is the trophic efficiency and 0 refers to the year selected to be used as the baseline for the series. As in Pauly *et al*. (2000), the value of *TE* has been set to 0.1.

While the above diversity indices (Table 1) are formulated using the numbers of individuals, they have also been calculated using standardized estimates of catch (biomass) per unit of fishing effort (*e.g.* Meuter and Norcross, 2002). Thus, for this study, the indices were calculated using values derived from the biomasses of catches reported by fishers rather than numbers of fish caught, as the latter data were not recorded by commercial fishers.

Commercial fisheries data for the West Coast Bioregion were extracted from the data base and filtered to select those records pertaining to gill netting, hand lining, long lining and drop lining for use in a preliminary evaluation to identify those methods of analysis that appeared most appropriate. These fishing methods are those that produce catches from a range of species and are most likely to provide information relating to the species composition of the fauna. It should be noted, however, that the species landed and reported by commercial fishers are those with a commercial value and that the fishing gear that is used is designed to target individuals of those species that are of a size that is acceptable to the market. Thus, bycatch species that are discarded or species that are poorly selected by the fishing gear are not represented in the data that were provided by the commercial fishers and analysed in this study.

Four tables were produced in the extraction, *i.e.* a catch table, an effort table, a species list table and an area list table. The catch table contained the following fields {Year, Month, Block, Method code, Species code, Live weight (kg), Vessels}, where Block is the value of the code identifying the geographical cell from which the catch was taken (usually a grid cell of 1° latitude by 1° longitude). The associated effort table contained {Year, Month, Block, Method code, Boat days}. The SpeciesList table describing the species recorded in the extracted catch data contained {Species code, Common name}, and the AreaList table comprised details of

{Region name , Block}, allowing a more precise geographic breakdown of data than that provided at the bioregion level, yet not as detailed as that provided by the Block codes.

The data from these four Microsoft Access tables were exported to comma-delimited text files, with field headers and using double-quotes as delimiters for text fields, which, in the case of common names, contain commas. The text files were then processed to create a single record for each year, containing the fields {Year, Method, Boat days, cpue for species 1, cpue for species 2, ... }, where the species were arranged in ascending order of species code (as specified in the SpeciesList table) and catches per unit of effort (cpue) were calculated by dividing the live weight by the total number of boat days of fishing employing that fishing method (*i.e.* non-targeted effort). The data for each method were then selected and species for which no catch was recorded in any year were deleted. The earlier years for which no drop line data were recorded (1976 to 1982 for the West Coast Bioregion, 1976 to 1983 for other bioregions) were also deleted from the analyses for the data derived from this method of capture.

The resulting data were analysed to produce time series of the number of species, Shannon-Wiener's diversity index, Margalef's species richness index, Pielou's evenness index, and Simpson's dominance index (Table 1), which were then plotted. Trends in the different indices were compared and their implications assessed.

Trophic level and risk assessment

An analysis was undertaken by the Department of Fisheries Western Australia (DoFWA) (in a separate study that is not reported in this document; details available from the Department) to determine the trophic level of each of the species and assess its potential vulnerability to fishing. For this analysis, details of the trophic level were extracted, where available, for each of the species groups reported in the database of WA commercial fisheries data, by Amanda Nardi, Department of Fisheries, from the information contained in FishBase (Froese and Pauly, 2007). Other data, including details of maximum age and size, habitat, distribution, resilience, spawning period and diet, were also extracted and recorded. Values of maximum length that were used in calculating the mean of the maximum length indices were fixed at the values reported in published and unpublished research studies of the various commercial fish species. As this database of biological data for WA's fish species is populated using published data, there are numerous gaps representing areas in which no studies have yet been reported. Thus, as in the case of Pink Snapper, where there have been no dietary studies yet undertaken, information on diets and, as a consequence, on trophic level, is missing from FishBase. It should be noted that the groups reported in the WA commercial fisheries catch statistics are those used by fishers when sorting and marketing their catch. Thus, there is occasional overlap of species names. Furthermore, the groups sometimes comprise a mixture of several species. While it is not possible to improve the quality of past data, the Department is taking steps to improve this aspect of future data collection and reporting by fishers, and is encouraging biological studies to fill gaps in crucial biological data, such as trophic level.

A priority assessment of the 146 most important species/stock/bioregion units was then undertaken by the Department of Fisheries, where the assessment for each species/stock represents a combination of the current risk to the stock and the current/likely future value of this species/stock to the community (measured as a combination of the GVP for the commercial sector, participation/use for the recreational sector and also their ecological/social value). It should be noted that not all of the species recorded in the WA commercial fisheries database are of equal importance and value to the community; some appear irregularly in the commercial catches. In order to match the results of the risk assessment to management needs, the Department considered each species on a management unit basis, *i.e.* relating the assessment to the management unit or "stock" within each bioregion.

The assessed risk was based firstly on the biological characteristics of the stock, and considered the habitat occupied by the species, its growth rate, age at attainment of sexual maturity, maximum age, instantaneous rate of natural mortality, spawning dynamics (including seasonality (short – long), relative fecundity, larval behaviour/dispersal, and whether or not the species forms spawning aggregations), sex change (if any), sexual dimorphism, territoriality, size/age related migrations, and potential for mixing amongst regions. Other issues considered when evaluating the risk included the main methods of capture, whether there is likely to be hyperstability in catch rates, whether the biology and behaviour of the species make them more or less vulnerable to fishing, whether stocks of the species have been managed successfully elsewhere, and whether there have been fishery crashes and the subsequent period for recovery. In addition, patterns of annual recruitment, *i.e.* whether recruitment appears relatively consistent among years, is moderately variable about a mean, or exhibits relatively long periods with little recruitment interspersed with good years every decade or so, were also considered.

Consideration was also given to the current status of each stock, reflecting the assessed status of that stock, its current management arrangements and the information available for managing it. Risk assessment followed the framework for national ESD assessment, and considered the likelihood and consequences of risk to future recruitment for each stock.

Changes in diversity and ecosystem indices

The data for each of fishing methods , *i.e.* drop lining, gill netting, hand lining and long lining, within each of the bioregions of Western Australia, were used to calculate the diversity and ecosystem indices listed in Table 1. Plots of the resulting time series were assessed. The catch data for the four fishing methods were combined and values of mean trophic level, mean maximum length and FIB were calculated and plotted.

Changes in species composition

In order to determine more precisely whether a real change in species composition has occurred and, if so, which species have contributed most to the dissimilarity among the data from different years, multivariate analyses such as those provided by the Primer 6 package are essential. Accordingly, MDS ordination analyses were undertaken of the West Coast Bioregion data using Primer 6. For these, the data from 1976 to 1979 were classified arbitrarily as period 75, from 1980 to 1984 as period 80, 1985 to 1989 as 85, 1990 to 1994 as 90, 1995 to 1999 as 95, and 2000 to 2005 as 00. The annual data for the five year period were assumed to be replicate samples of catches that were representative of the trophic structure present within the ecosystem during that period, and thus allowing comparison of species composition among five-year periods. Note, however, that there is no *a priori* basis, other than chronology, for such grouping. The values of cpue were log(x+1) transformed before calculating the resemblance matrices. The ANOSIM R-statistic was calculated to assess whether the species composition differed significantly among (some) periods in the catches that had been recorded by commercial fishers. SIMPER analyses were run to elucidate the species that were responsible for any dissimilarity present in the data.

The results of this preliminary analysis were assessed and likely techniques for more refined analyses were identified. MDS ordination had been shown to have the ability to test whether differences between species compositions of the catches in different periods were statistically significant and to identify the changes that characterised the differences that were detected. However, in these earlier analyses, the potential that the results were dominated by the values of species that had high abundance or unduly influenced by the highly variable and sporadic catch rates of species with low abundance was recognised. To refine the analyses, the MDS analyses of the data for the Western Australian bioregions that were undertaken subsequently applied a filter to the data. Thus, species were excluded from the analyses if, during the period for which data were available, the percentage contribution to the total annual catch taken by the fishing method (dropline, gillnet, handline, or longline) failed to exceed 3% in any of the years. The use of such a filter was recommended by Clarke and Warwick (2001), who noted that a value of 3% was typically used for soft-sediment benthic data. Although this filter appeared to improve the quality of the resulting data, some species with highly sporadic catches and considerable numbers of zero catches remained present in the data set. It was decided that, where there appeared to be no consistent trend in the data, such sporadic data should be excluded from further analysis. Clarke and Warwick (2001) identified the value of data transformation as a tool to change the emphasis placed on the different species, where, without transformation, the more common species tend to dominate the results of the analyses. An improved balance among the more and less common species can be achieved by using transformations such as a square root, double square root or logarithmic transformation. Each view explores different characteristics of the data. After considering the implications of the different transformations available within PRIMER, the decision was made to increase the influence of the less common species by subjecting the data to double square root transformation. Thus, for each of the analyses subsequently undertaken using PRIMER, the data were filtered and subjected to double square root transformation.

Application to a system where changes have occurred.

Data for the analysis were downloaded from the Northeast Fisheries Science Centre (NEFSC) website (see URL http://www.nefsc.noaa.gov/sos/agtt/). Sosebee *et al.* (2006) advise that these data represent summarised values of abundance indices (mean catch per tow) derived from standardised bottom trawl surveys conducted by the NEFSC in autumn (principal groundfish, flounders, other groundfish, other pelagic, squid) and spring (principal pelagic and small elasmobranchs).

The abundance indices downloaded from the NEFSC website were filtered to remove those species that contributed less than 3% of the total survey biomass in any of the years (Clarke and Warwick, 2001, p. 2-7). A factor, period, was assigned to the data, where 1=1968-1970, 2=1971-75, 3=1976-80, 4=1981-85, 5=1986-90, 6=1991-95, 7=1996-2000, and 8=2001-05. To increase the weight given to less abundant species in the analysis, a fourth-root transform was then applied to the abundance data and Bray-Curtis similarities were calculated (Clarke and Warwick, 2001). The data were subjected to multidimensional scaling (MDS) ordination and cluster analysis using Primer V6 (Clarke and Warwick, 2001). An Analysis of Similarities (ANOSIM) test from this software package was then applied to determine whether the multivariate data differed among periods, and to explore the nature of any difference that was detected at the global level. The relative contributions of different species to dissimilarities among the different periods were explored using similarity percentages (SIMPER) calculated by the Primer Version 6 software package (Clarke and Warwick, 2001).

To ensure that changes in overall abundance did not mask changes in relative species composition, the filtered abundance data were used to calculate the proportions of the total (filtered) survey abundance in each year represented by each species. The resulting proportions were analysed using the same approach as was used with the abundance indices, *i.e.* as described above.

Changes in species composition of catches in Western Australia

MDS ordination analysis, together with SIMPER and ANOSIM analyses, were conducted for filtered, transformed catch per unit of effort data from each of the bioregions. The dataset selected initially for study in each bioregion was that for the fishing method with the greatest average annual number of fishing days. Subsequently, it was recognised that the analysis needed to explore the results from different fishing methods, in order to take the different species compositions of the catches by those methods into account. A factor, Period, was defined. This was allocated a value of 1 for data between 1976-80, 2 for 1981-85, ..., 6 for 2001-2005. Shift in species composition evident in the MDS ordination for each region were explored using ANOSIM, while SIMPER was used to determine the changes between the different periods that characterised the differences. It was considered appropriate that, in assessing changes in species composition, the species characterising the change in species composition should first be identified through the objective analysis provided by SIMPER. An assessment could then be made as to whether those species characterising the change were those that had been identified in the risk assessment.

Plots of the time series of catches for the different species and of the cpues of the more frequently-caught species for each of the five bioregions were produced for use by the Department of Fisheries to facilitate consideration of (i) approaches by which artefacts of apparent species change arising from the ways in which data are recorded might be removed and (ii) known changes in management or fishing activity that might have resulted in the observed changes in species composition can be identified.

Operating model and Management Strategy Evaluation (MSE) framework

A Management Strategy Evaluation (MSE) framework was developed to explore the effectiveness of managing multi-species fisheries using alternative data sets and decision rules, in an environment in which the ecosystem was likely to be affected by climate change and increasing exploitation. In summary, the function of this portion of the computer program was to read the data that describe the characteristics of the individual stocks, and the matrix containing details of the fraction of the diet of each predator that comprises each of the other modelled species. The operating model initializes the system state using the assumption that each stock is at equilibrium under a specified initial harvest rate (assumed to be zero in this study), then generates a set of historical fishery data for each of the stocks. Subsequently, for each year of the projection period, the program loops through a cycle of (1) fishery stock assessment and (2) determination of appropriate management controls according to the specified decision rules followed by (3) the application of those rules to the “fishery” and (4) the generation by the operating model of the resultant fishery data for that year and the state of the system at the end of the year. Assessments may be carried out at regular but infrequent intervals, e.g. every five years, and the controls that arise from the application of the decision rules applied by the operating model until the next assessment and review of management controls.

Determination of an appropriate operating model was guided by the results of the recent assessment and review of management arrangements for the demersal finfish off the lower west coast of Western Australia, reported by Wise *et al.* (2007). We deviated from the original plan to base the operating model on an EcoPath/EcoSim model of the fishery as it became apparent from this review that (i) due to the lack of recreational catch and effort data, the time series of fishery data were inadequate for the production of biomass estimates for the various exploited fish stocks; and (ii) limited biological and dietary data were available for non-targeted and bycatch species, thus hampering the development of such an ecosystem model. The most reliable data for the more important commercially-exploited species are the descriptions of their population biology, *i.e.* data on growth and reproduction and, to a lesser extent, estimates of total, natural and fishing mortality and of dietary composition. Accordingly, in defining the operating model, it was decided that the models of the biology of each species should be based on the available data for the population biology of the various species, with the species being linked through their diets. There was, however, also a need to describe how the population would respond when egg production varied and the extent to which the stock was currently depleted from its pristine level. An estimate of current biomass can be obtained if there is an estimate of total catch and of the levels of fishing and natural mortality. Estimates of the extent of depletion or of the likely response of recruitment to changes in stock size require, however, an estimate of the steepness of the stock-recruitment relationship. For this, we used appropriate estimates of steepness selected from those derived for various species, genera and families by Myers *et al.* (1999) from meta-analysis of stock-recruitment data. To provide the basic fishery data that are appropriate for fishery models, the two estimates of recreational catch that are available for the West Coast Bioregion (see Wise *et al,* 2007) were employed, in combination with data for the commercial catches, to fit an exponential curve that was then used as a description of the historical relationship between recreational and commercial catches for the species used in the model. Values of commercial catch per unit of fishing effort were adjusted to account for the changes in efficiency described by Wise *et al.* (2007). The adjusted estimates of commercial cpue were divided into the values of total commercial and recreational catch to obtain estimates of the amount of fishing effort (in terms of commercial days of fishing with the specified fishing gear) expended in each year.

A modular approach to construction of the operating model was taken, such that, as data for additional fish species are collated and subjected to assessment, further species can be added to the suite of species included in the model. The species that are currently included in the MRM are those for which data were sufficient to calculate the parameter estimates required by the operating model. While the species currently included in the MRM represent only a small fraction of the total ecosystem, and interactions among these species are typically through competition, the model allows for size-dependent predation such that the young of some of the species are vulnerable to predation by potential predators.

A formal description of the operating model, a Minimum Realistic Model (MRM) of interacting species in the multi-species fishery, is presented below, noting that the assessment model using the MRM (or, if predation and consumption are ignored, an age-based model for a single stock) is of the same form. The MRM provides a representation of that portion of the ecosystem that is “visible” through the data available from the commercial and recreational fishers, i.e. the catch of target and byproduct species.

Description of model

Subscripts

a = age

t = year

Note – where subscripts/superscripts are not shown, the variable represents the sum over the missing subscripts/superscripts

Superscripts

s = sex (f or m)

j = species

Notation

$a^{j}_{\text{B\&H}}$, $b^{j}_{\text{B\&H}}$ = parameters of Beverton and Holt stock-recruitment relationship for species j

$a^{s,j}_{\text{Wt-Len}}$, $b^{s,j}_{\text{Wt-Len}}$ = parameters of power curve relating the mass (kg) of a fish of species j and sex s to its length (input to the model)

A^{j} = maximum age of species j (input to the model)

$B^{s,j}_{a,t}$ = biomass of fish of species j, sex s, and age a in year t

B^{j}_{t} = biomass of fish of species j in year t

$C^{s,j,s',j'}_{a,a',t}$ = biomass of fish of species j, sex s and age a consumed in year t by fish of species j', sex s', and age a'.

$C^{s,j}_{a,t}$ = biomass of fish of species j, sex s and age a consumed in year t by modelled species (note – other consumption is represented as natural mortality)

$D^{j',j}$= fraction of diet of predator species j', that is comprised of prey species j (input to the model)

E^{j}_{t} = Fishing effort applied to species j in year t (input to the model)

f^{j}_{a} = fecundity (or a proxy such as the biomass of a female) at age a for species j (input to the model)

F^{j}_{t} = fishing mortality of fully-vulnerable fish of species j in year t

$F^{s,j}_{a,t}$ = fishing mortality applied to fish of species j, sex s, and age a in year t

h^{j} = steepness of the Beverton and Holt stock-recruitment relationship for species j

$H^{s,j}_{a,t}$ = proportion of fish of species j, sex s, and age a that are caught in year t

$L_\infty^{s,j}$, $k^{s,j}$, $t_0^{s,j}$ = parameters of von Bertalanffy growth curve for sex s of species j (input to the model)

$L_a^{s,j}$ = length of a fish of species j, sex s and age a

$\mathcal{L}_a^{s,j}$ = maximum length of prey consumed by fish of species j, sex s and age a

M_a^j = natural mortality of fish of species j and age a. Note that this represents the portion of the estimate of natural mortality that is unexplained by the predation of the other species included within the model when the system is at an unexploited equilibrium, where the total natural mortality is typically estimated using methods such as those proposed by Pauly (1980) and Hoenig (1983).

$N_{a,t}^{s,j}$ = number of fish (thousands) of species j, sex s and age a that are alive at the start of year t

n = number of species included in the model

n^j = number of observations of the cpues of species j used when fitting the model, *i.e.* may include projected values when the operating model is being used to assess the effectiveness of alternative harvest strategies.

$p_{a,t}^{s,j}$ = proportion of fish of species j, sex s and age a that are mature at the start of year t (input to model)

$P_{a,t}^{s,j}$ = biomass consumed by fish of species j, sex s and age a in year t

P_t^j = total biomass consumed by fish of species j in year t

q^j = catchability of species j

r^j = ratio of maximum prey length to predator length for fish of species j (input to the model)

R_0^j = annual recruitment of age 0 fish of species j when the stock is unexploited

R_t^j = total recruitment (thousands) of age 0 fish of species j at the start of year t

s^j = standard deviation of observation errors, *i.e.* deviations of the (natural) log-transformed cpues from their predicted values.

S_t^j = total egg production (or a proxy such as spawning biomass) of species j at the start of year t

$T_a^{\zeta,s,j}$ = proportion of fish of species j, sex ζ, and age a that, at the end of the year, will become of sex s, noting that for gonochoristic species $T_a^{\zeta,s,j} = 1$ if $\zeta = s$, and 0 otherwise.

$\theta^{s,j}$ = proportion of recruits of species *j* that are of sex *s*

U_t^j = catch per unit of effort for species *j* in year *t*

$V_a^{s,j}$ = relative selectivity of fish of species *j*, sex *s* and age *a* with respect to the fishing gear used by fishers

$W_a^{s,j}$ = mass of a fish of species *j* and sex *s* at age *a*

$Y_{a,t}^{s,j}$ = yield (tonnes) of fish of species *j*, sex *s*, and age *a* in year *t*

Y_t^j = yield (tonnes) of fish of species *j* in year *t*

Model equations

Growth

$$L_a^{s,j} = L_\infty^{s,j}\{1 - exp[-k^{s,j}(a - t_0^{s,j})]\}$$

Length-weight relationship

$$W_a^{s,j} = a_{\text{Wt-Len}}^{s,j}\left(L_a^{s,j}\right)^{b_{\text{Wt-Len}}^{s,j}}$$

Note that values of length and weight at age *a* and age *a*+0.5 may be calculated and input to the model.

Egg production or spawning biomass

$$S_t^j = \sum_{a=1}^{A^j} p_{a,t}^{f,j} N_{a,t}^{f,j} f_a^j$$

Parameters of stock-recruitment relationship

$$a_{\text{B\&H}}^j = \frac{S_0^j}{R_0^j}\left\{1 - \frac{(h^j - 0.2)}{0.8h^j}\right\}$$

$$b_{\text{B\&H}}^j = \frac{h^j - 0.2}{0.8h^j R_0^j}$$

Recruitment

$$R_t^j = \frac{S_t^j}{a_{\text{B\&H}}^j + b_{B\&H}^j S_t^j}$$

Biomass at age

$$B_{a,t}^{s,j} = N_{a,t}^{s,j} W_{a+0.5}^{s,j}$$

Total biomass

$$B_t^j = \sum_{s=f}^{m} \sum_{a=0}^{A^j} B_{a,t}^{s,j}$$

Consumption

From Liao *et al.* (2005):

$$P_t^j = 3.5011\left(B_t^j\right)^{0.9962}$$

Consumption at age

$$P_{a,t}^{s,j} = \frac{B_{a,t}^{s,j}}{\sum_{s=f}^{m} \sum_{a=0}^{A^j} B_{a,t}^{s,j}} P_t^j$$

Ratio of maximum prey length to predator length

It was assumed that, for each species *j*,

$$r^j = 0.3$$

where this ratio, which varies considerably among species, is determined primarily by the gape size of the predator's mouth. The value of 0.3 was estimated subjectively from the figures presented by Scharf *et al.* (2000) (see also Claessen et al., 2003).

$$\mathcal{L}_a^{s,j} = r^j L_a^{s,j}$$

Prey biomass consumed

$$C_{a,a',t}^{s,j,s',j'} = \begin{cases} 0 & \text{if } L_{a+0.5}^{s,j} > \mathcal{L}_{a'+0.5}^{s',j'} \\ \dfrac{B_{a,t}^{s,j}}{\sum_{s=f}^{m} \sum_{\substack{a=0, \\ L_{a+0.5}^{s,j} \le \mathcal{L}_{a'+0.5}^{s',j'}}}^{a=A^j} B_{a,t}^{s,j}} D^{j',j} P_{a',t}^{s',j'} & \text{if } L_{a+0.5}^{s,j} \le \mathcal{L}_{a'+0.5}^{s',j'} \end{cases}$$

$$C_{a,t}^{s,j} = \sum_{j'=1}^{n} \sum_{s'=f}^{m} \sum_{a'=0}^{A^{j'}} C_{a,a',t}^{s,j,s',j'}$$

Fishing mortality and harvest rate

If effort is the forcing function, *i.e.* the control rule of the MSE requires that future annual effort should be adjusted to E_t^j,

$$F_t^j = q^j E_t^j$$

$$F_{a,t}^{s,j} = V_a^{s,j} F_t^j$$

$$Z_{a,t}^{s,j} = M_a^j + F_{a,t}^{s,j}$$

$$H_{a,t}^{s,j} = \frac{F_{a,t}^{s,j}}{Z_{a,t}^{s,j}}\{1 - exp[-Z_{a,t}^{s,j}]\}$$

$$Y_{a,t}^{s,j} = H_{a,t}^{s,j} N_{a,t}^{s,j} W_{a+0.5}^{s,j}$$

$$\hat{Y}_t^j = \sum_{s=f}^{m} \sum_{a=0}^{A^j} Y_{a,t}^{s,j}$$

$$\hat{U}_t^j = \frac{\hat{Y}_t^j}{E_t^j}$$

Note that the model assumes a single fishery and a single area. There is thus no need to allocate future fishing effort (or future catch) to the commercial and recreational fishing sectors or among different regions of the fishery.

If catch is the forcing function, i.e., the control rule of the MSE requires that future annual yield be adjusted to $\hat{Y}_t^j$, the value of F_t^j that results in an estimated value of yield that is equal to the observed catch was determined using the above equations. An estimate of the amount of fishing effort required to produce this fishing mortality was then calculated as

$$\hat{E}_t^j = \frac{F_t^j}{q^j}$$

and used when estimating the predicted value of catch per unit of effort, $\hat{U}_t^j$.

Number at age

$$N_{a,t}^{s,j} = \begin{cases} \theta^{s,j} R_t^j & \text{if } a = 0 \\ \sum_{\zeta=f}^{m} T_a^{\zeta,s,j} \left(N_{a-1,t-1}^{\zeta,j} - C_{a-1,t-1}^{\zeta,j}\right) exp\left[-Z_{a-1,t-1}^{\zeta,j}\right] & \text{if } 1 \leq a \leq A \\ \sum_{\zeta=f}^{m} T_a^{\zeta,s,j} \left\{\left(N_{a-1,t-1}^{s,j} - C_{a-1,t-1}^{s,j}\right) exp\left[-Z_{a-1,t-1}^{s,j}\right] + \left(N_{a,t-1}^{s,j} - C_{a,t-1}^{s,j}\right) exp\left[-Z_{a,t-1}^{s,j}\right]\right\} & \text{if } a = A \end{cases}$$

The model structure was designed to allow it to be used not only with typical gonochoristic species but also with the types of demographic data typically produced for sharks, in which natural mortality is often assumed to be age-dependent, and with hermaphroditic species.

It was assumed that the ecosystem was at an unexploited equilibrium at the start of the historical fishing record. Although this assumption is unlikely to be justified, the historical data extend back to 1976, and thus transient effects are likely to be relatively minor during the projection period of the MSE.

When the model was used as an operating model to generate synthetic data, each new value of fishing effort (if effort is the forcing function) or catch (if catch is the forcing function) was used by the operating model to update the system state (as specified above) and to generate an estimate of cpue. This latter value was taken as the expected value of cpue and an observed value of cpue for the time step was calculated as

$$U_t^j = \widehat{U}_t^j exp\left(s^j \varepsilon_t^j\right)$$

where $\varepsilon_t^j \sim N(0,1)$. If effort was employed as the forcing function, then the observed value of catch generated for the time step was calculated as

$$Y_t^j = U_t^j E_t^j,$$

whereas, if catch was assumed to be the forcing function, the observed value for effort generated for the time step was calculated using

$$E_t^j = \frac{Y_t^j}{U_t^j}.$$

When used as an operating model, the values of R_0^j, h^j, q^j and s^j that were input to the model were used to generate synthetic data in response to the values of the forcing functions that were supplied. For this, the system state was first set to the estimated equilibrium state for the system. Historical values of effort or catch (depending on which is used as the forcing function) were then used to generate synthetic data through the historical period. The assessment models were then run and decision rules applied (if this was scheduled to occur), and the operating model then used to generate synthetic data for the next year. This process described in the last sentence was repeated for each year within the specified period over which the model was to be projected.

In addition to generating synthetic fishery data for each year, the operating model, a Minimum Realistic Model (MRM) of the ecosystem, also generated random samples of age and size composition of the annual catch of each species. For this, the model accumulated the catch (numbers) of fish of each age and sex class to produce an estimate of the expected distribution of fish of each age and sex within the catch. A random sample of the specified sample size (*e.g.* 500, as was used in this study) was then drawn from the resulting multinomial distribution of catches at age. To produce length samples from the catch-at-age data, the catches at each age were multiplied by the expected proportion of fish of that age in each length class, where lengths at age were assumed to be normally distributed around the lengths predicted by the growth equation and where the standard deviation of this normal distribution (assumed to be constant over age) was specified in the data input to the model. The resulting numbers of fish of each age and sex falling within each length class were accumulated to produce the expected length frequency distribution, from which a multinomial sample of the specified size (*e.g.* 500, as was used in this study) was then randomly drawn.

Assessment models

The MSE framework allows for the use of a number of alternative assessment models, including either an MRM with the same structure as the operating model or one or more of a number of single-species models, including methods to estimate fishing mortality using (1) an age-structured model fitted to catch, effort and cpue data, (2) catch curve analysis, that estimates F from the age composition, (3) an estimate based on mean age, and (4) an estimate based on mean length, which employs the Beverton and Holt (1956) relationship between total mortality and mean length.

The assessment models were applied to (a set of) selected species. When fitting the single-species age-structured assessment models, the values of $C_{a,t}^{s,j}$ were set equal to zero, *i.e.* consumption by other species in the MRM was ignored, although, in hindsight, it would have been more appropriate to adjust the level of natural mortality by including a constant mortality associated with the predation imposed by these other species. If the assessment model to be used was the multispecies model of the ecosystem, the calculations were as described in the mathematical specification above. When fitting this latter form, however, the model proved to be rather stiff suggesting that an alternative fitting algorithm, such as simulated annealing or genetic algorithm, may need to be used instead of the amoeba routine that was initially employed. The parameters estimated when fitting the model were typically R_0^j, h^j, and q^j, however the routine allowed selected parameters to be held fixed to their input values if desired, *e.g.* to constrain the value of steepness to the value selected from Myers *et al.* (1999). The objective function employed when fitting the assessment models was the combined log-likelihood of the deviations of the log-transformed cpues from their expected values, where

$$\lambda^j = -0.5 n^j \left\{ \log_e 2\pi + \log_e \left[\left(s^j \right)^2 \right] + 1 \right\}$$

$$\text{and } s^j = \sqrt{\frac{\Sigma\left(\log_e U_t^j - \log_e \hat{U}_t^j\right)^2}{n^j}}.$$

Species included in model

The species included in the model were Western Rock Lobster *Panulirus cygnus*, Dusky Whaler Shark *Charcharhinus obscurus*, Pink Snapper *Pagrus auratus*, Dhufish *Glaucosoma hebraicum*, and Baldchin Groper *Choerodon rubescens*. An Excel interface was developed for user input of the biological parameters and fishery data for these species. For each species, an assessment was undertaken using the time series of data input to the model to obtain the parameter estimates needed for the operating model. The paucity of quantitative data on the diets of the marine species, *e.g.* Pink Snapper, presents a problem for modelling, however.

Initial exploration of the effectiveness of DoFWA's decision rule

Use of a five-yearly assessment and management cycle and adjusting effort for each species in the MRM to achieve target fishing mortalities, employing a decision rule similar to that used by DoFWA for the demersal fish resources off the Perth metropolitan coastline, was explored using the MSE. Stock assessment for this exploration was carried out by fitting a single-species age-structured model to the simulated data for each species. It was assumed in this simulation that the Department of Fisheries would be able to adjust the effort applied to each of the different fish species to achieve the effort levels output by the decision rule component of the MSE framework.

For an initial assessment of the effectiveness of DoFWA's decision rule for the demersal finfish fishery in waters off the Perth metropolitan area, the model used the calculated value of F_t^j as the indicator variable. This was compared against the target level which was set equal to two thirds of the estimated value of natural mortality (Wise *et al.*, 2007), in order to assess the likely outcome if the decision by the Department of Fisheries to set this as the target level of fishing mortality was to be accompanied by a control rule to achieve this target. If the current level of fishing mortality fell below the target, no action was taken, otherwise the value of the forcing function was reduced such that it was set at the level expected to result in the target level of fishing mortality. To reduce computational time, assessment and adjustment was only carried out every five years in the thirty year projection period that was used in model runs. The results of the runs were examined to assess the effectiveness of the decision rule.

Indicator variables, reference points and decision rules

Following the initial exploration of the effectiveness of the MSE framework as a tool for exploring alternative harvest strategies, an evaluation of alternative indicator variables, reference points and decision rules was undertaken and, after discussion with scientists from DoFWA, the results were used to determine which of these would be included in the MSE. The evaluation was influenced strongly by the decisions that DoFWA had made when considering management options for the demersal finfish stocks off the Perth metropolitan area. Recognising that proportions, and thus species composition, can be influenced by a change in abundance of a single species, it was considered appropriate that the variables selected as indicators should relate to the abundance and fishing mortality associated with each of the individual species not the species composition, *i.e.* the absolute abundances of the different species rather than their proportional contribution to the total abundance. It was also considered appropriate to apply single species stock assessments in the decision rules, provided that these are evaluated in a management strategy evaluation framework that considers the interactions among the different species when assessing the effectiveness of the overall management of the fishery. It appeared likely that, should the indicator variables fall beyond the threshold or limit reference points for a species, managers would attempt to reduce directly the level of exploitation on that species rather than attempt to adjust the exploitation of other species and thereby reduce their indirect influence on the stock that was the intended target of the management actions.

Above all, it was recognised that the indicator variables, reference points and decision rules used in the MSE needed to be consistent with those that had been adopted recently by the Department of Fisheries when assessing the status of stocks of demersal fish in waters off the southwest of Western Australia (Wise et al., 2007). It was also recognised that the information contained within current indices of commercial catch per unit of effort (cpue) was inadequate for production of reliable estimates of stock status, but that, with refinement of data collection and processing procedures, it was possible that these data might become of greater value in the future. Currently, however, the indicator variables that were likely to be of greatest applicability to the Department of Fisheries in assessing stock status were the estimates of fishing mortality derived from age and length composition data collected from fish sampled from catches. These estimates of fishing mortality were considered far more reliable than the estimates of biomass, which, in years in which no data on the often substantial recreational catches were available, were based on incomplete catch data coupled with effort data which were considered likely to be inadequately adjusted for the effects of changes in fishing power. Accordingly, the indicators selected for examination in the Management Strategy Framework were fishing mortality estimates derived from catch curve, mean age, mean length and cpue. The last of these employed a single-species, age-structured model, fitted to the time series of cpue data.

Target, threshold and limit reference points for fishing mortalities were calculated from the estimate of the instantaneous rate of natural mortality, M, and set equal to $2M/3$, M, and $2M$ (if maximum age < 10 years) or $3M/2$ (if maximum age is 10 years or greater), respectively (Wise et al., 2007). This decision ensured consistency with the reference points that had been accepted for use in the recent demersal fish stock assessment.

The decision rules considered in the study were again influenced strongly by the approach adopted by the Department of Fisheries in the recent demersal fish study (Wise et al., 2007). Thus, if the estimate of fishing mortality was found to fall between the threshold and limit reference points, fishing mortality would be reduced by a specified percentage (10, 30 or 50%) and maintained at this level till the next assessment. Similarly, if the estimate was found to fall beyond the limit reference point, fishing mortality would be reduced by a specified (more constraining) percentage (50, 75 or 100%) and maintained at this level till the next assessment. In practice, in the assessment for the demersal fishery, the selection of the percentage reduction to be applied was made subjectively on the basis of a Weight of Evidence approach, in which the risk associated with the characteristics of each species' life history was considered. Such an approach was not able to be implemented in the MSE framework that had been developed for this project, as it was necessary in the MSE model to specify the percentages by which subsequent fishing effort would be reduced. Accordingly, values of 10 and 50% were selected for evaluation. The effectiveness of alternative levels of reduction can readily be explored, however. As an alternative, a decision rule that required fishing effort in excess of the target level to be reduced to the target and maintained at this level till the next assessment was also explored.

The operating model, an MRM of the ecosystem, was extended to implement the production of random samples of age and size composition, and estimation of fishing mortality from catch curve, mean age, and mean size data. Following a suggestion by James Scandol, the catch curve analysis was implemented in R, but was executed under the control of a Visual Basic 2008 program.

An assessment of the effectiveness of the alternative decision rules, indicator variables and reference points was undertaken. The time period between assessments was set to five years, and catch curve, mean age and mean size assessments were based on random samples of 500 fish collected from the (simulated) catches of the previous year. Other time periods and sample sizes could readily be explored. To simplify the interpretation of results, recruitment variability was set to zero. Thus, for the assessments that were undertaken, variability was confined to that generated in the (synthetic) samples of observed data for cpue, age compositions and length compositions. Although the procedure allows the calculation of fishing mortality estimates using a mixture of cpue, catch curve, mean age and mean size analysis, and use of the resulting estimates in the decision rule, with the most conservative of the proposed management actions then being implemented, the analyses undertaken for this study applied each assessment approach separately in order to assess the performance of each.

Value of additional data

The MSE was employed to determine the value of collecting additional types of data. For this, the effectiveness of each of a range of different assessment methods and of a range of different decision rules was assessed using the MSE framework. The MSE evaluated the effectiveness of reducing fishing mortality from an initial heavily-overfished equilibrium state to a level below the threshold reference point for this variable by the end of a specified projection period. The assessment methods included estimation of fishing mortality by fitting an age-structured model to catch-per-unit-effort (cpue) data, by catch curve analysis, and by calculation of total mortality from mean age and mean length then subtraction of the estimate of the value of natural mortality from the resulting value. Each of these analyses used synthetic data that were generated under the assumption that they were collected from recently-initiated data collection programmes, designed to capture representative fishery and biological data for the various species included in the MRM.

The range of decision rules that were assessed included (i) reduction of fishing mortality when the estimate of this value fell beyond the threshold and limit reference points (see the discussion in the previous section); (ii) reduction of fishing mortality to the value of the target fishing mortality if the estimated fishing mortality exceeded the target. Target, threshold and limit reference points for fishing mortalities used in this study were calculated from the estimate of the instantaneous rate of natural mortality, M, and set equal to $2M/3$, M, and $3M/2$ (if maximum age is 10 years or greater), respectively.

Effectiveness of management strategies when stocks are impacted by increasing exploitation or climate change

The MSE was employed to explore the effectiveness of different assessment methods and decision rules and on the species within the MRM when changes in exploitation and climate change were affecting the stocks. For this, the operating model was extended to model, for each of the various species, the effect of (i) linear increases in fishing mortality; (ii) linear decrease in the parameter of the stock-recruitment relationship that determines the carrying capacity of the stock; and (iii) linear decrease in the asymptotic length of the growth curve for the species. The MRM was extended to use a random number generator that facilitated a more reliable comparison of the effectiveness of the management strategy when applied to data generated using the same random number seed with and without the effects of climate change and increasing exploitation. Lastly, the model was further modified to include an option to generate synthetic data using an annual percentage increase in (a) fishing effort, (b) "virgin" recruitment, and (c) asymptotic length. The change in virgin recruitment was used to calculate the new carrying capacity and the new parameters of the Beverton and Holt stock-recruitment relationship. The change in asymptotic length required that, rather than estimating constant lengths at ages at the beginning of the trial, the new lengths at ages were calculated for each year from the length at the previous age employing the new asymptotic length to estimate the annual length increment.

The model was run 100 times (using different random number seeds on each of the 100 trials) with a 30-year projection period, to test the effectiveness of the management strategy used by the Department of Fisheries when applied to data simulated assuming (i) no increase in exploitation or impact of climate change, and (ii) a 1% annual increase in fishing effort and a 1% annual decrease in both virgin recruitment and asymptotic biomass. In these runs the decision rule employed saw effort reduced by 10% if the estimated fishing mortality exceeded the threshold level (i.e. the value of the estimate of natural mortality M), and by 50% if the estimate exceeded the limit reference point ($3M/2$). Stock assessment in the MSE framework was carried out separately for each species at five-yearly intervals, using a single-species approach. Three sets of runs of the operating and assessment models, i.e. the full MSE, were undertaken to estimate the instantaneous rate of fishing mortality F using (A) an age-structured model fitted to catch, effort and cpue data, (B) a catch curve analysis that estimates F from the age composition, and (C) an estimate of F based on mean age.

Results/discussion

Databases of fisheries data

The task of rewriting the commercial Catch and Effort System (CAES), and producing a centralized database for this and other fisheries research data is well underway. The new system is termed the Fisheries Management Information System (FIMS). It is recognised by the Department of Fisheries that the system will continue to be refined and evolve to meet its data and reporting needs. At a later stage in project development, it is intended to automate reporting such that the resulting documents will facilitate production of the Department's annual State of the Fisheries Reports.

Preliminary analyses

The extracted table of commercial catch data for the West Coast bioregion, which was used in the preliminary analysis, comprised 170,054 records covering the period 1976 to 2005, while the associated effort, species and area tables comprised 17,667, 165 and 56 records, respectively.

For the West Coast Bioregion, the number of species recorded as being caught by gillnet fishing increased by approximately 25% over the period from 1976 to 2005 (figures not presented as the results of the preliminary analysis are superseded by those obtained later in the study). Total catch per unit of effort showed a marked, approximately 300% increase over the period (Fig. 1). For the catches per unit of effort for the gillnet fishery of the West Coast Bioregion, the Shannon-Wiener' evenness index decreased slightly from 1976 to the mid-1980s and then increased consistently towards a maximum in 2005. Margalef's index of species richness for the gillnet catches per unit of effort for the West Coast Bioregion suggested a more stable level of richness than the number of species. Pielou's evenness index revealed a similar trend to that produced using the Shannon-Wiener index. Simpson's index of dominance, which is essentially the inverse of the evenness measures, reflected increasing dominance by some species (reduced evenness) in the mid-1980s with a more recent trend towards reduced dominance (greater evenness) in more recent years.

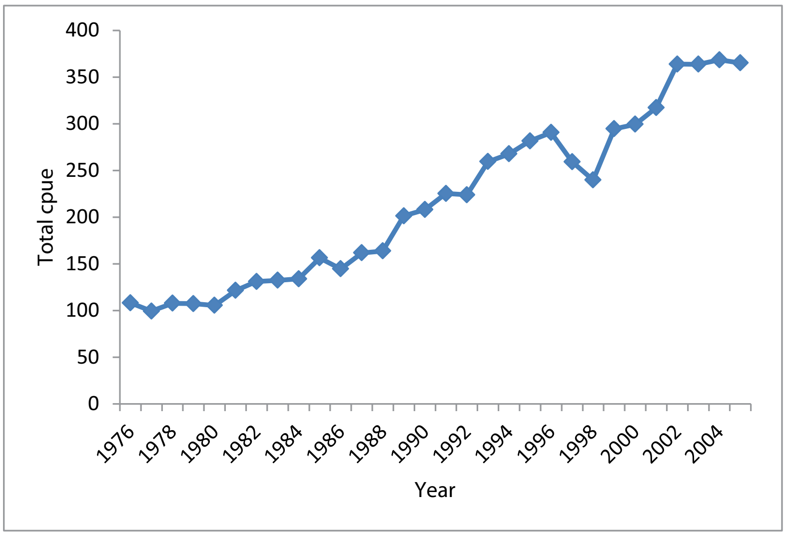


Figure 1. The total commercial catch per unit of effort recorded for gillnet fishing in the West Coast Bioregion between 1976 and 2005.

The results of this analysis demonstrated that, while broad indicators of diversity, such as species richness, evenness and dominance, provide a convenient summary of the changes that have occurred in reported fisheries data, they provide no indication of the nature of the change. These kind of limitations have been previously noted by others (*e.g.* Rochet and Trenkel, 2003). Indeed, for comparability, the indicators should be calculated from replicated research samples. Changes in sample size resulting from changes in fishing effort, and other factors affecting the distribution of fishing, targeting of species, and efficiency of fishing gear will influence the values of the indices. Changes that are evident in the time series of these diversity indices may thus represent changes in fishing practice rather than changes in trophic structure of the ecosystem. For this, it is probably more appropriate to explore the potential effectiveness of ecosystem-based indicators developed for use with fishery data, such as mean trophic level, mean maximum length and fishery-in-balance indices. Such an analysis is presented below.

Changes in diversity and ecosystem indices

The trends in the diversity and ecosystem indices calculated for the cpue data from the different fishing methods within each of the bioregions differ (Fig. 2 to 41). Although the traditional diversity indices, *i.e.* number of species, Shannon-Wiener' diversity index, Margalef's species richness index, Pielou's evenness index, and Simpson's dominance index, have been calculated and plotted, it is recognised that these are sensitive to sample size and, thus, that the trends in these indices will have been affected by the general increase in cpue and fishing effort that occurred within each region. Accordingly, in discussing the results of the analyses, we have concentrated on the ecosystem-based indices, *i.e.* mean trophic level, mean maximum size, and the fishery-in-balance indicators.

While the mean trophic level for droplining in the West Coast bioregion showed a slight gradual increase from the early 1980s to 2003, it exhibited a slight decline in the last two years of the series (Fig. 3; note the scale of the y-axis). The increase before 2003 was accompanied by a corresponding gradual increase in mean maximum size from the early 1980s to 1998, followed by a marked increase in 1999 and a gradual decrease over the subsequent six years. It should be noted that the confidence limits for these data are likely to be wide, and caution should be exercised when assessing the implications of the figures. The marked increase in the value of the FIB from 1983 to 1985 reflects an increase in catches from 1983 that is greater than would have been expected given the change in trophic level, suggesting that fishers were accessing multiple neighbouring ecosystems that were at least partially distinct. The decision to use the first point in the time series as the reference level when calculating the FIB is possibly inappropriate as it is likely that the fishery was developing in this earlier period, and catches would have been expected to increase without marked changes in trophic level. The decline in the FIB between 1985 and 1993 suggests a slight reduction in catches greater than would have been anticipated given the slight increase in trophic level. The FIB then recovers to high levels in the more recent period, indicating that catches have exceeded those that would have been expected through changes in trophic level. Gillnetting in this region appeared to increasingly target larger species in the years prior to 1990, as shown by the increase in the mean maximum length over this period and the associated increase in mean trophic level (Fig. 5). The FIB index exhibited strong growth in the first decade, then remained relatively stable, indicating that catches initially exceeded those that would have been expected from change in trophic level alone, but that, from the mid-1980s, balance has been maintained. The mean trophic level of the fish caught by handlining declined from the mid 1990s and was accompanied by a slight decrease in mean maximum length and, toward the end of the series, an increase in FIB (Fig. 7).

The FIB was highly variable throughout the series, however, and it would be inappropriate to suggest that catches in this latter period have exceeded those that would have been anticipated from the reduction in trophic level. The indices for longlining suggest that there has been a shift towards fishing for species with larger maximum lengths in the last four years and increasing volatility of the various indices, but that, until 2000, catches remained in balance with those expected. In 2002, catches were less than expected, while in subsequent years, they exceeded those that would have been predicted from the changes in trophic level (Fig. 9).

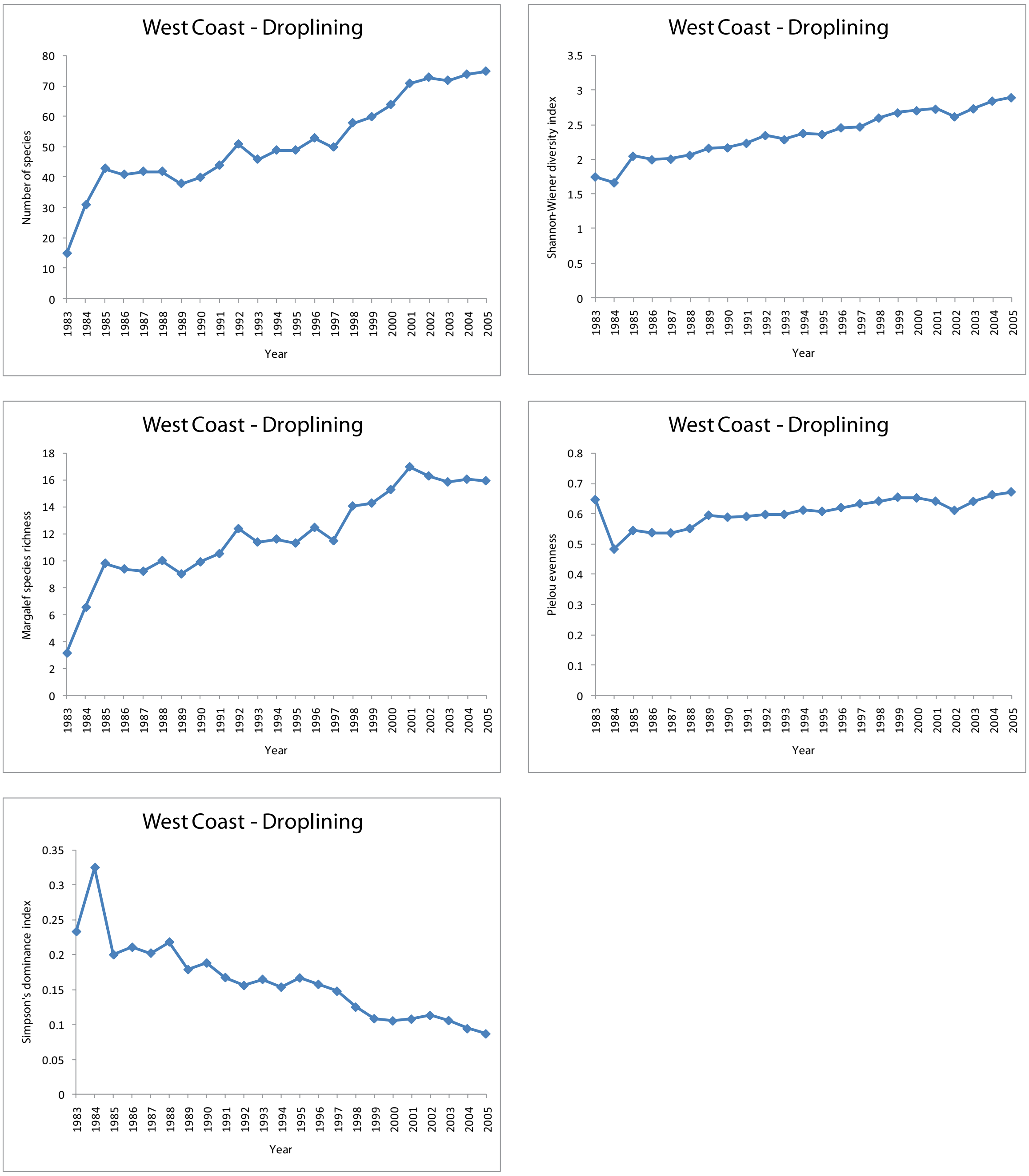


Figure 2. Time series of diversity indices for cpue data derived from commercial fishers using droplines within the West Coast bioregion of Western Australia.

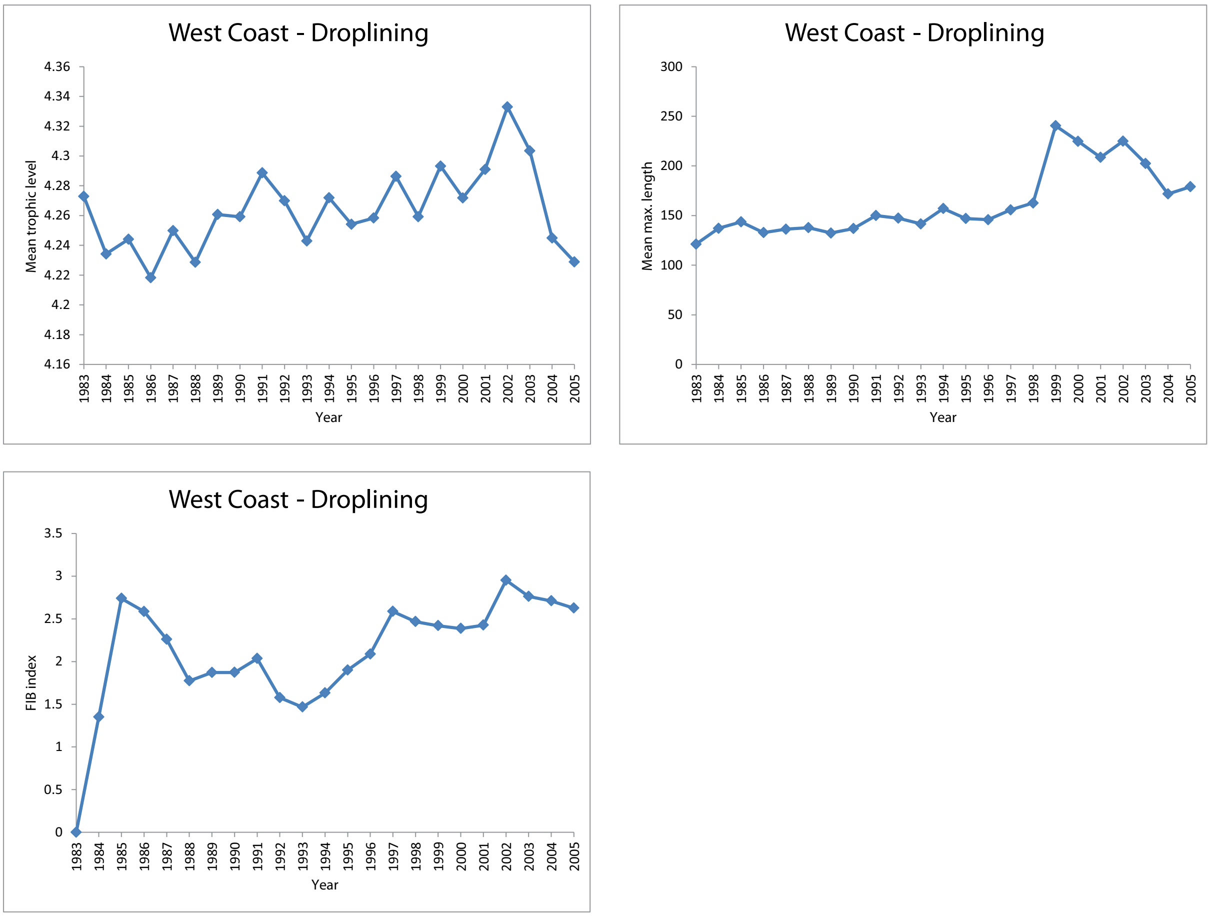


Figure 3. Time series of ecosystem-based indices for cpue data derived from commercial fishers using droplines within the West Coast bioregion of Western Australia.

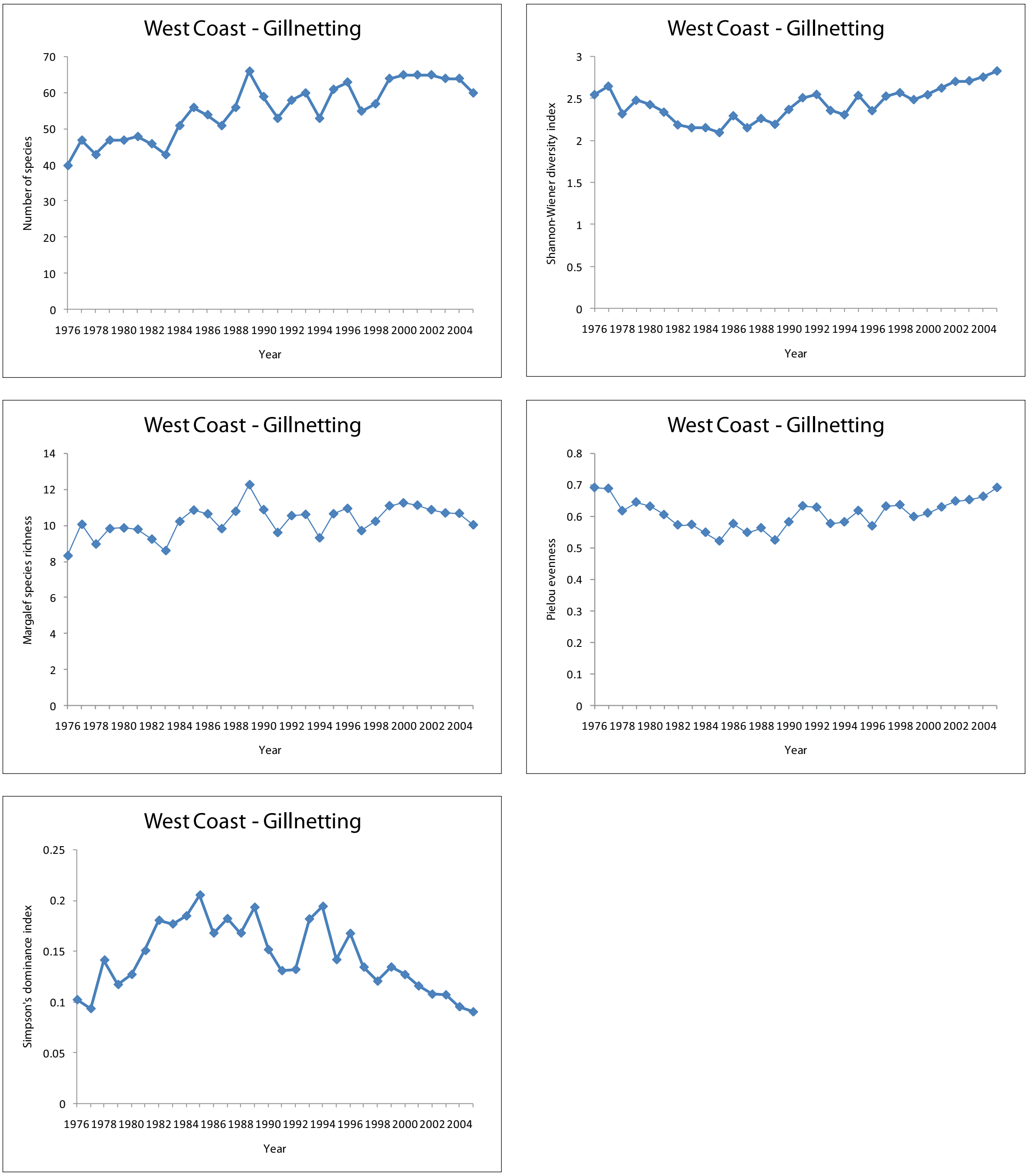


Figure 4. Time series of diversity indices for cpue data derived from commercial fishers using gillnets within the West Coast bioregion of Western Australia.

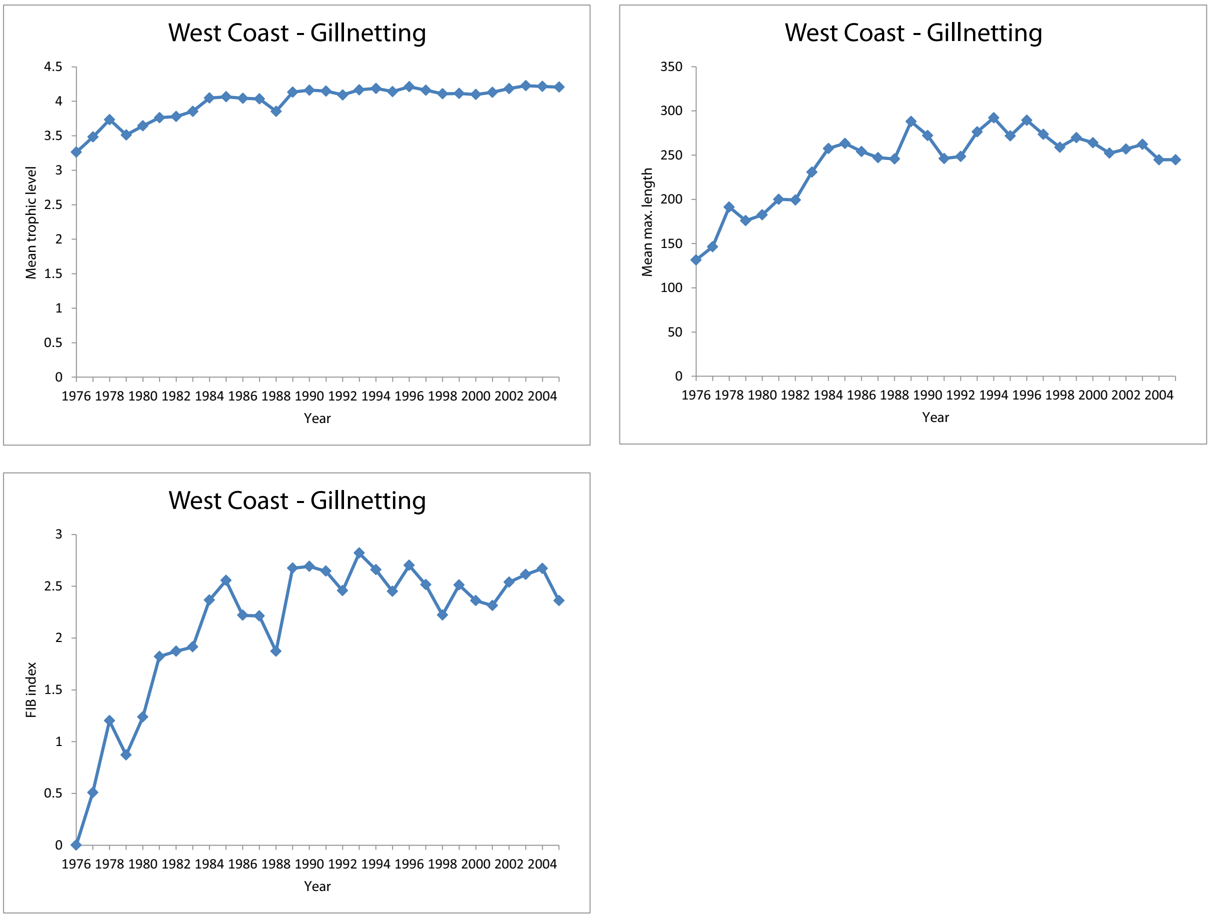


Figure 5. Time series of ecosystem-based indices for cpue data derived from commercial fishers using gillnets within the West Coast bioregion of Western Australia.

Figure 6. Time series of diversity indices for cpue data derived from commercial fishers using handlines within the West Coast bioregion of Western Australia.

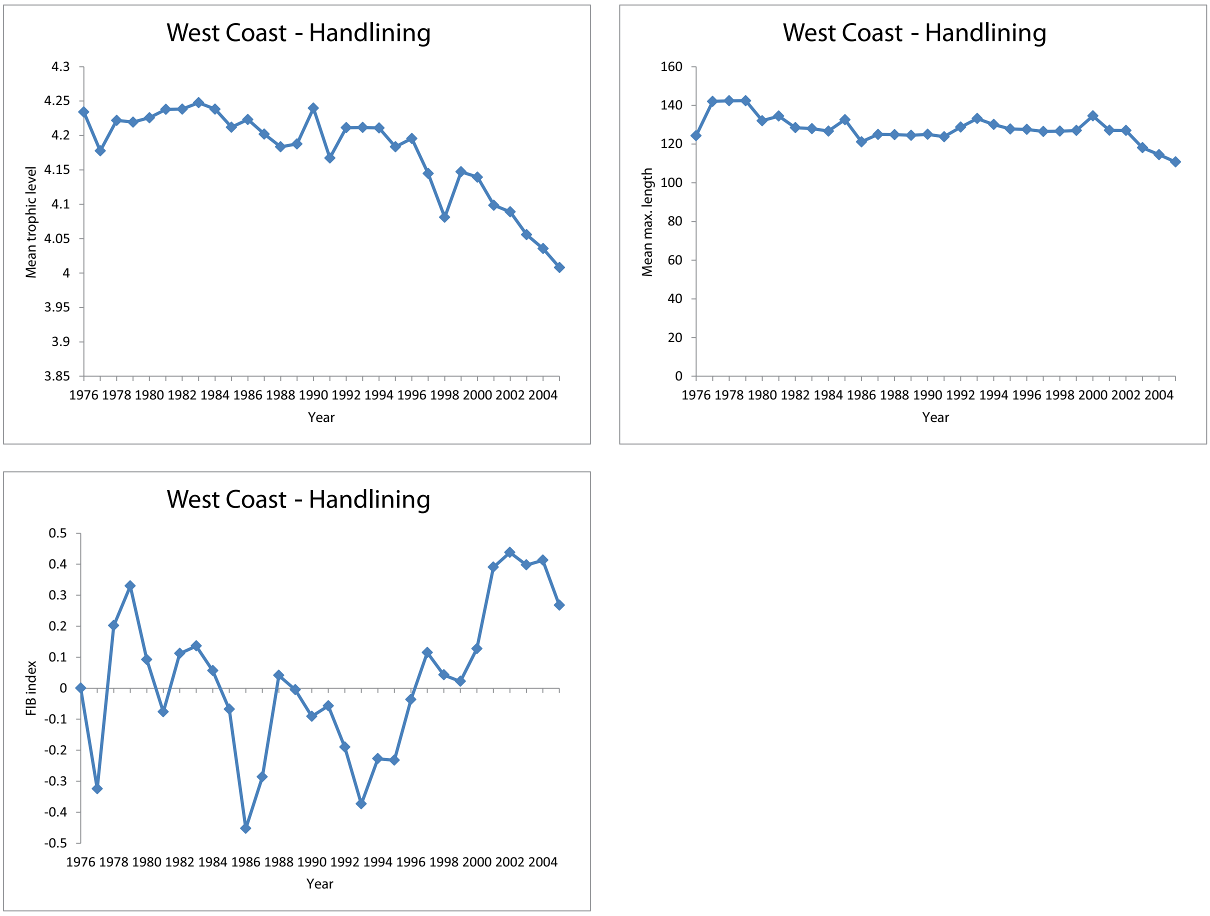


Figure 7. Time series of ecosystem-based indices for cpue data derived from commercial fishers using handlines within the West Coast bioregion of Western Australia.

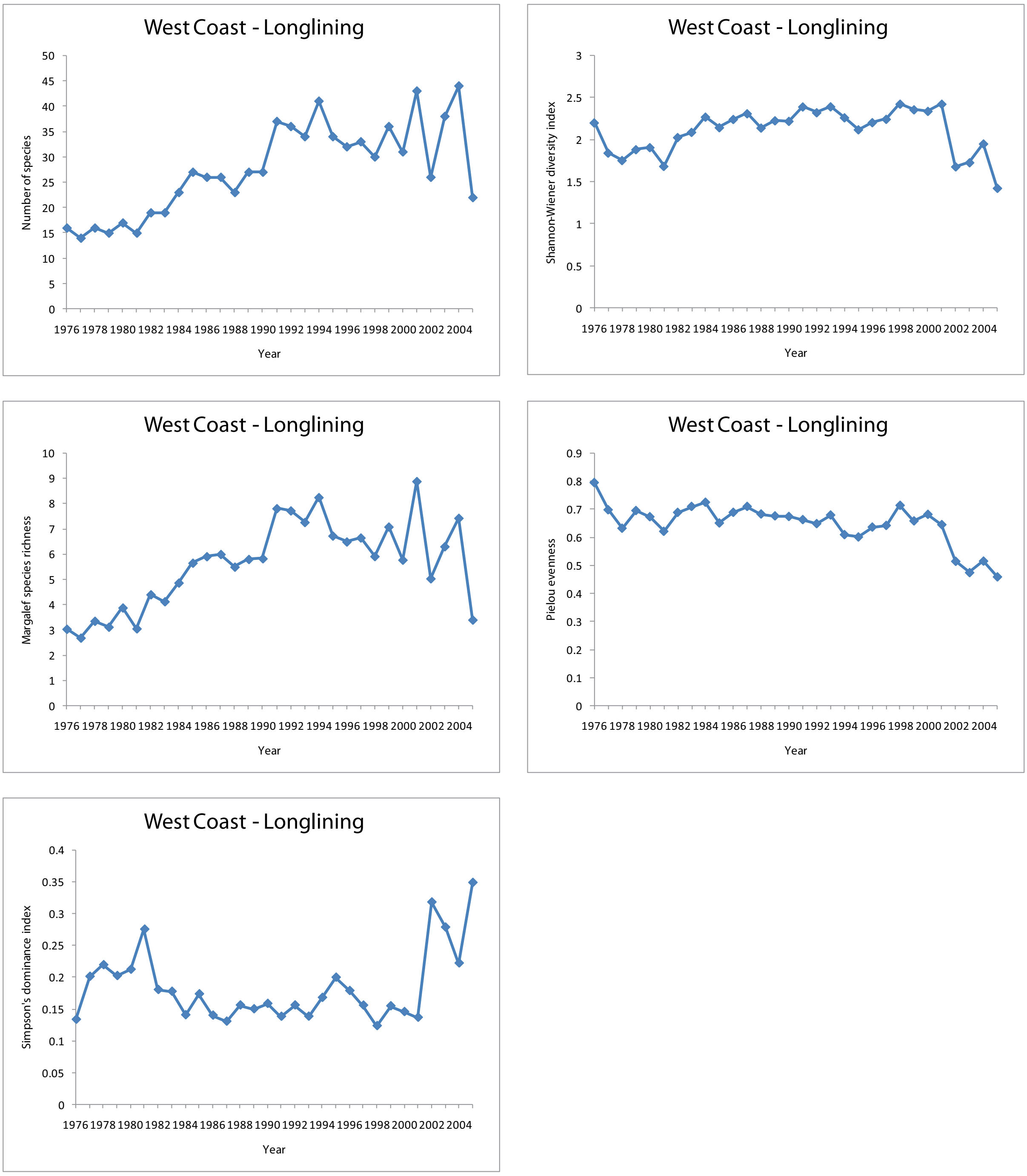


Figure 8. Time series of diversity indices for cpue data derived from commercial fishers using longlines within the West Coast bioregion of Western Australia.

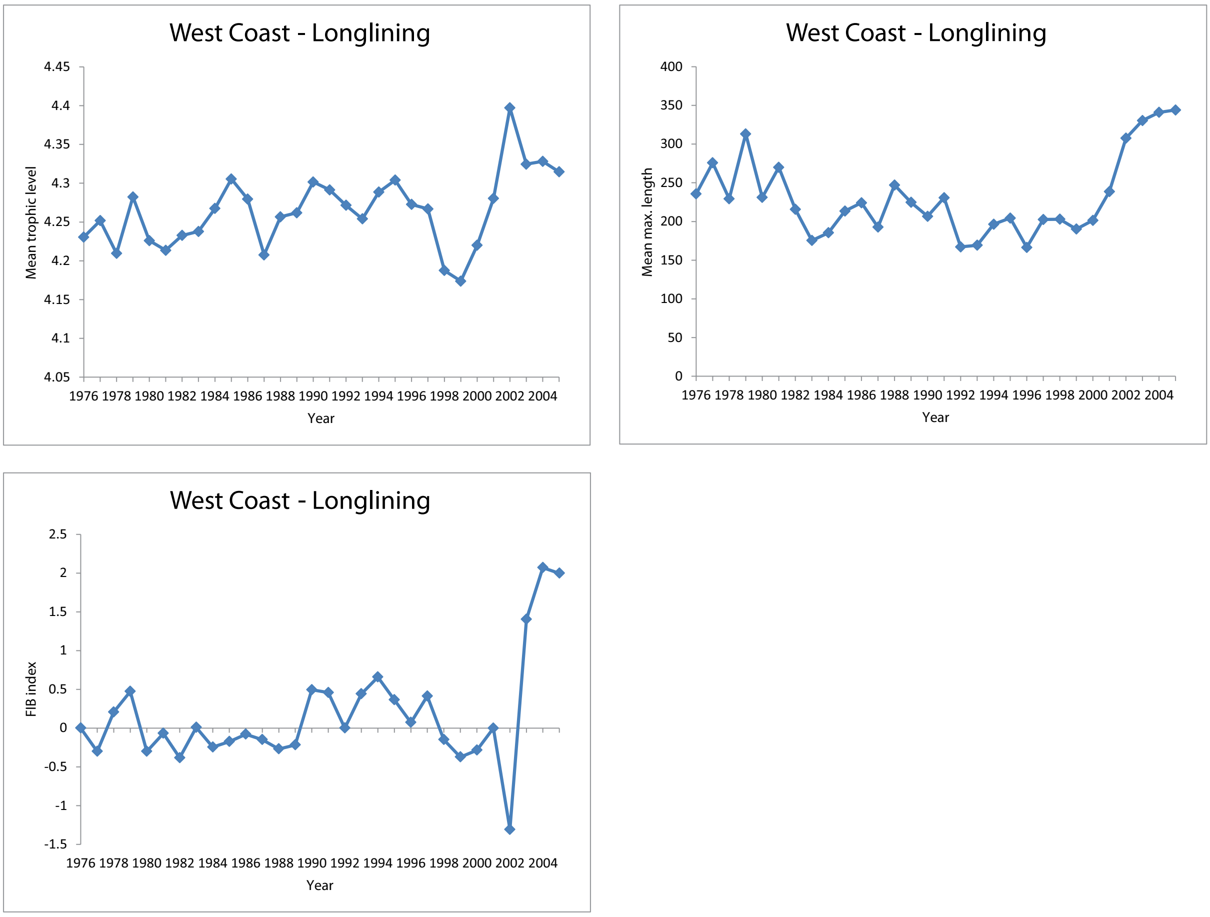


Figure 9. Time series of ecosystem-based indices for cpue data derived from commercial fishers using longlines within the West Coast bioregion of Western Australia.

The FIB for droplining in the South Coast bioregion increased in the period leading up to the early 1990s, then maintained a gradual increase in subsequent years (Fig. 11). The mean trophic level increased gradually over the period, while mean maximum size decreased in the 1980s but then remained relatively stable. There was no indication that balance was not being maintained. For gillnetting, the mean trophic level revealed an increase till the early 1990s but then appeared to stabilise (Fig. 13). Mean maximum length appeared to increase slightly till the late 1980s-early 1990s but then exhibited a slight decline over the last decade. The FIB increased markedly between 1976 and 1990, implying that catches in excess of those predicted from the changes in trophic level were taken and suggesting that that fishers were extending into new systems/subsystems. It subsequently remained relatively stable, indicating that, since the 1990s, balance was being maintained. The mean trophic level of handlining catches in the South Coast bioregion was variable but relatively stable, whereas the mean maximum length decreased between 1976 and 1990, but then appeared to stabilise (Fig. 15). The FIB for handlining revealed that catches less than would have been expected given the observed trophic levels were being taken between 1976 and the late 1980s. The FIB then appeared to become more stable, although rather variable, implying that, in this latter period, balance was being maintained. This suggests that fishers were exploiting a single "effectively closed" system. The increase in mean trophic level exhibited by the longlining data was paralleled by a similar increase in the mean maximum length, although the latter exhibit high variability and a decline in the last three years of the series (Fig. 17). The FIB declined gradually between 1976 and 1993, but has remained variable but relatively stable over the last decade. As with the handlining data, it appears that, in the earlier period, the longlining catches exceeded those that would have been expected given the recorded trophic levels, but that, in the last decade, balance has been maintained.

Figure 10. Time series of diversity indices for cpue data derived from commercial fishers using droplines within the South Coast bioregion of Western Australia.

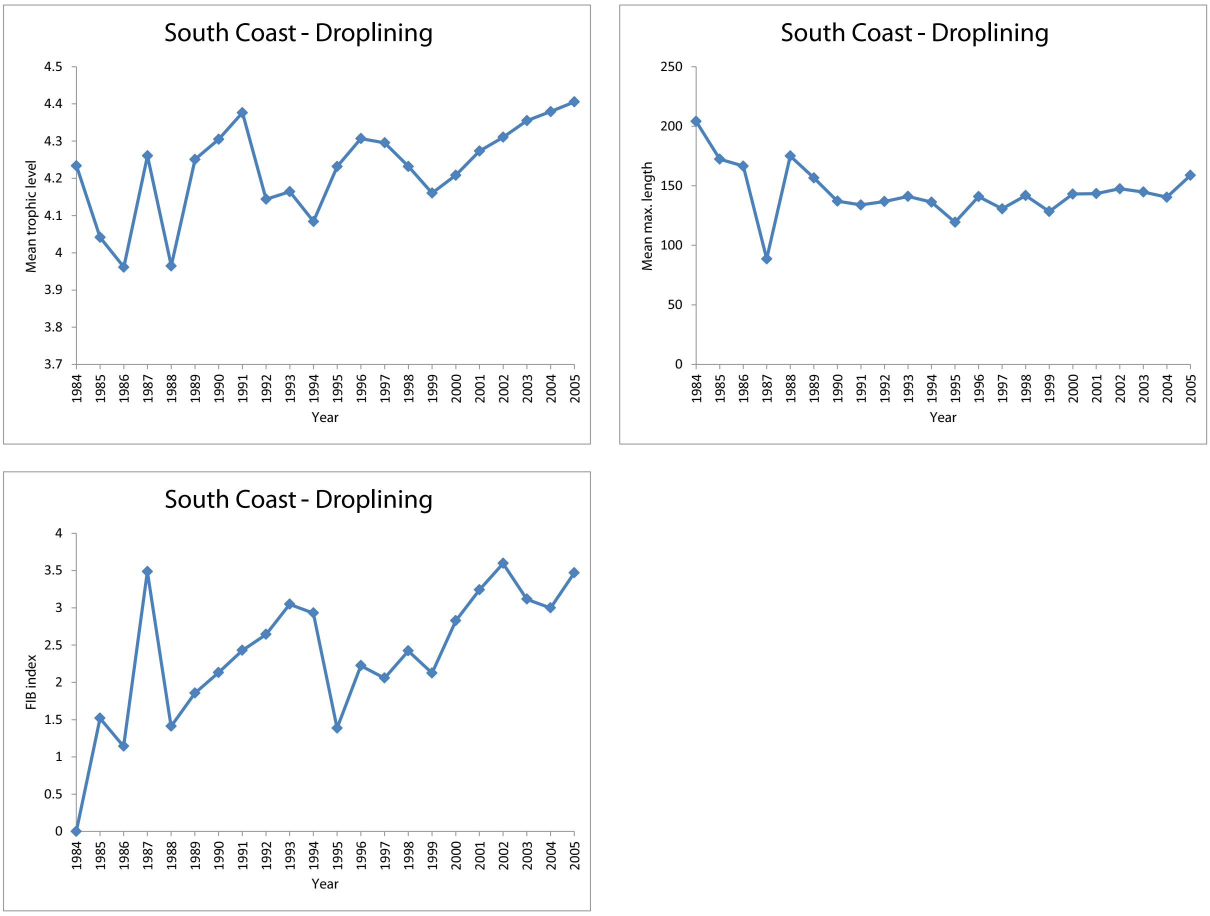


Figure 11. Time series of ecosystem-based indices for cpue data derived from commercial fishers using droplines within the South Coast bioregion of Western Australia.

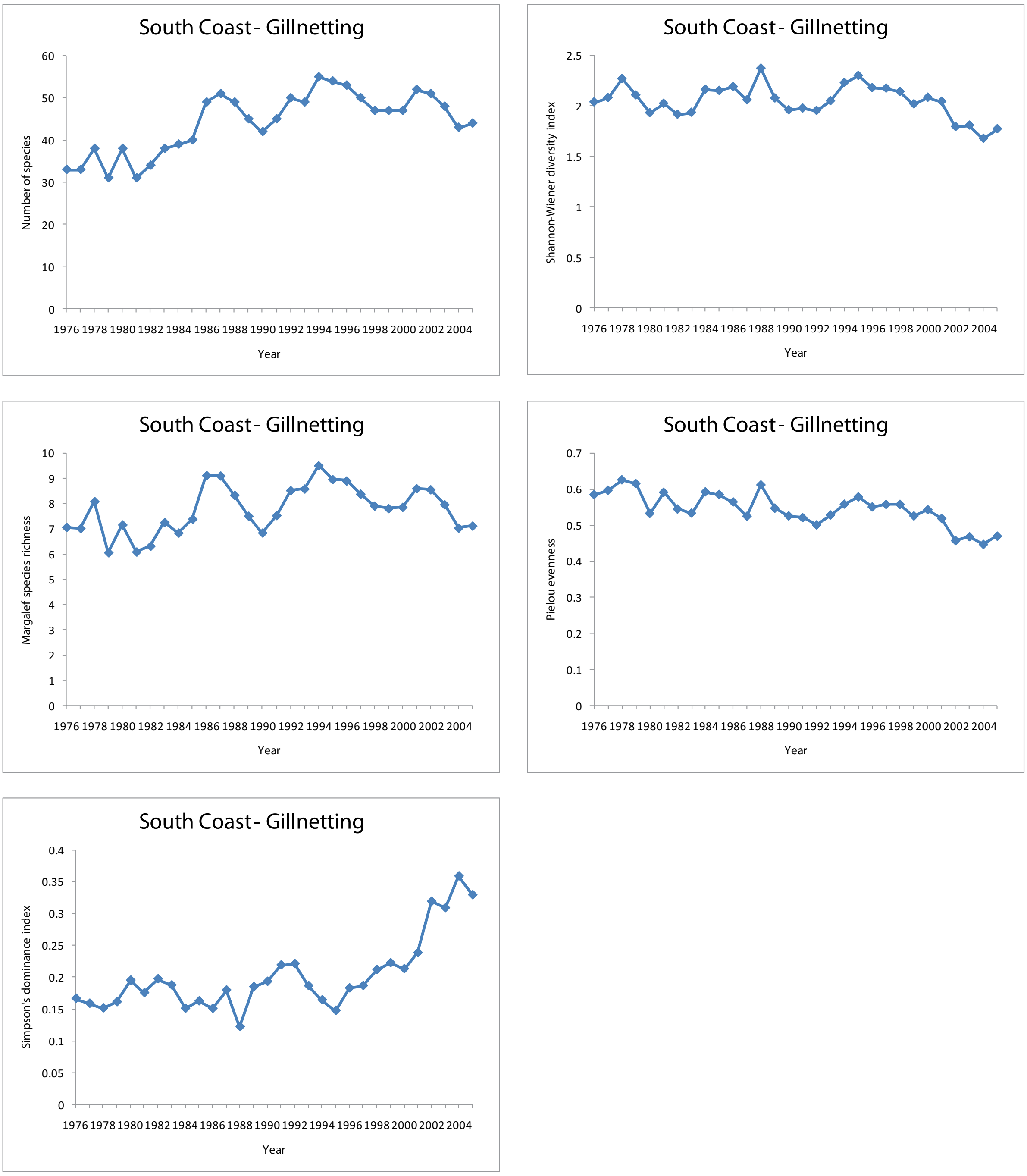


Figure 12. Time series of diversity indices for cpue data derived from commercial fishers using gillnets within the South Coast bioregion of Western Australia.

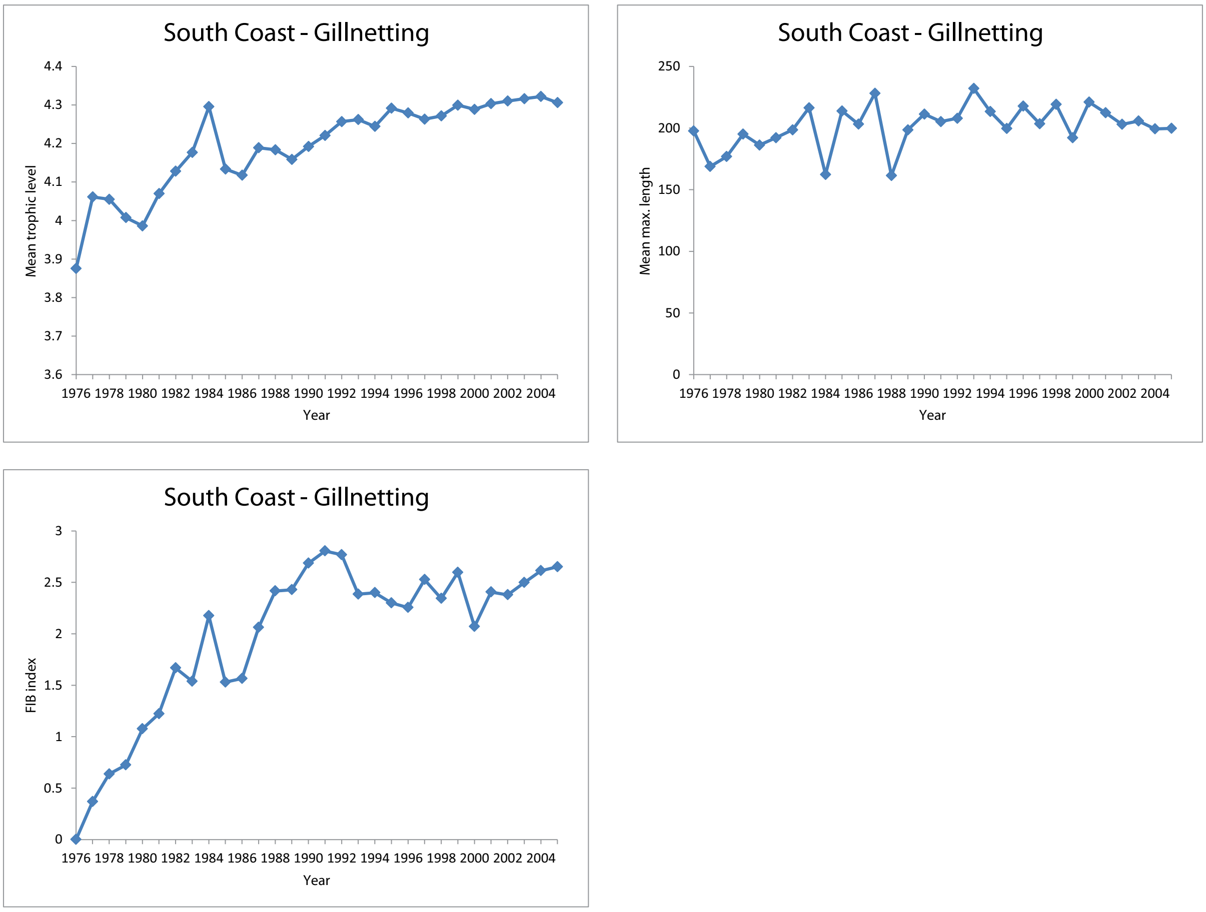


Figure 13. Time series of ecosystem-based indices for cpue data derived from commercial fishers using gillnets within the South Coast bioregion of Western Australia.

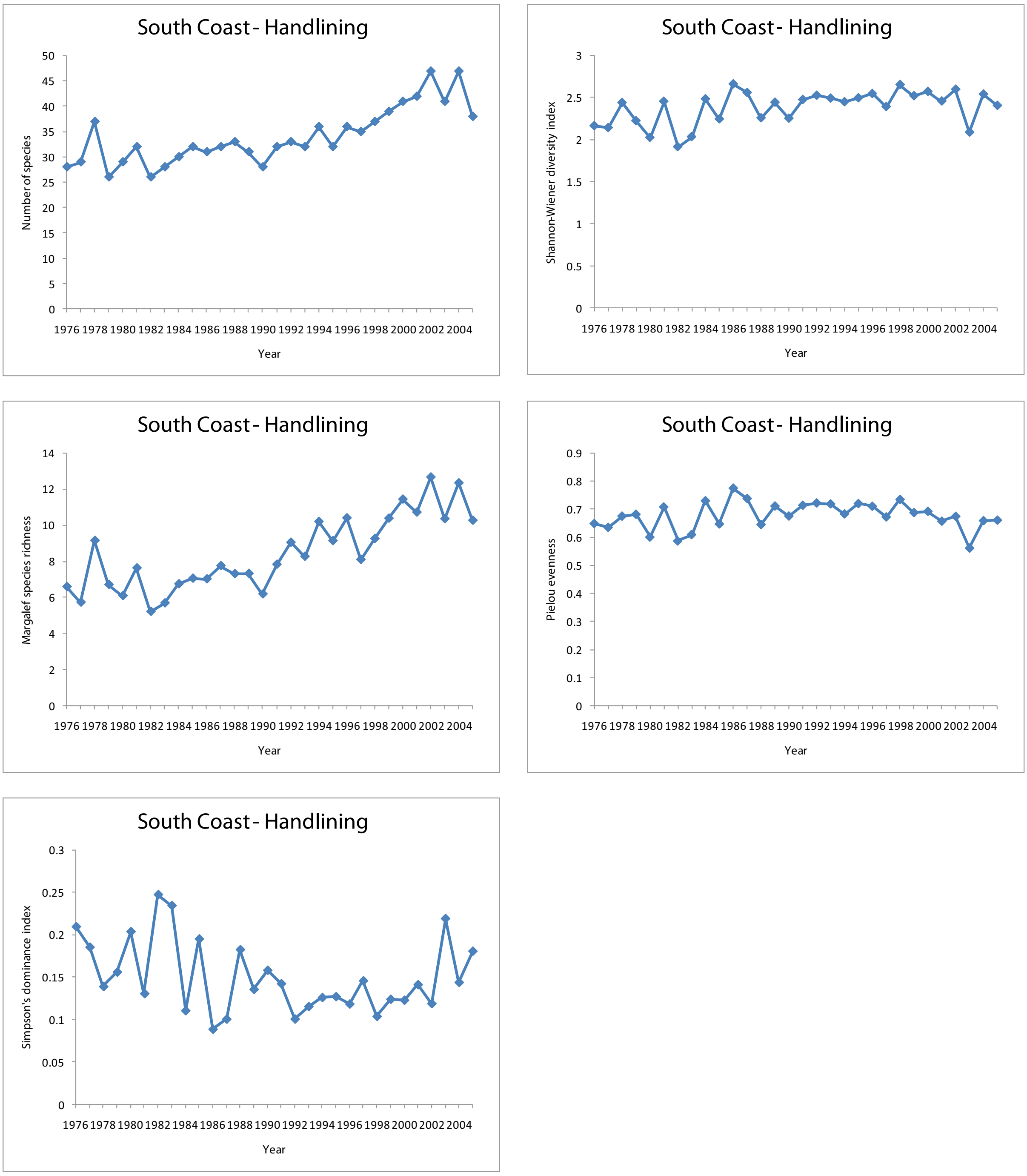
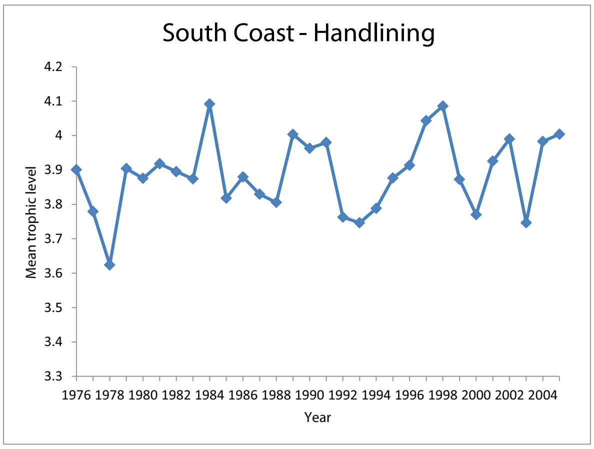
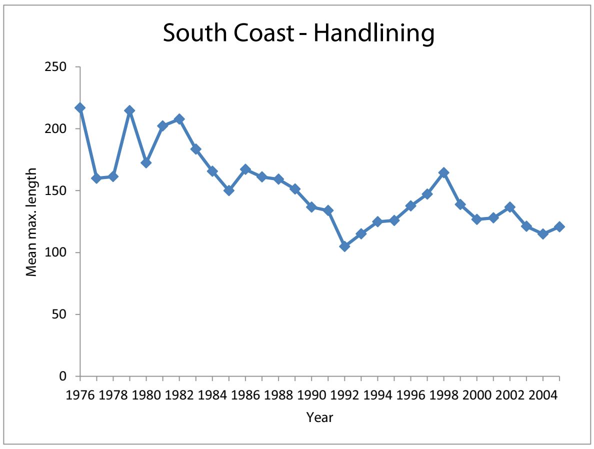


Figure 14. Time series of diversity indices for cpue data derived from commercial fishers using handlines within the South Coast bioregion of Western Australia.





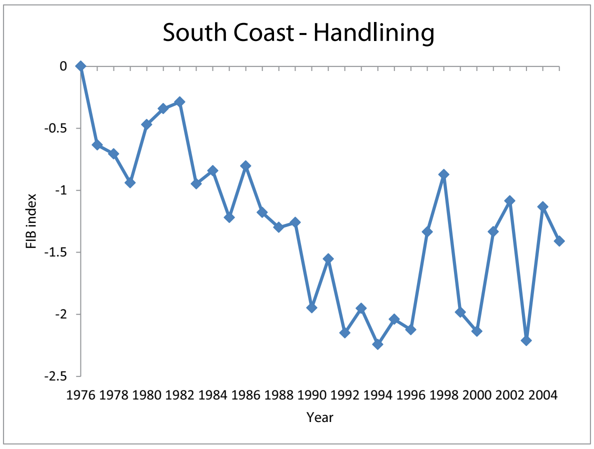
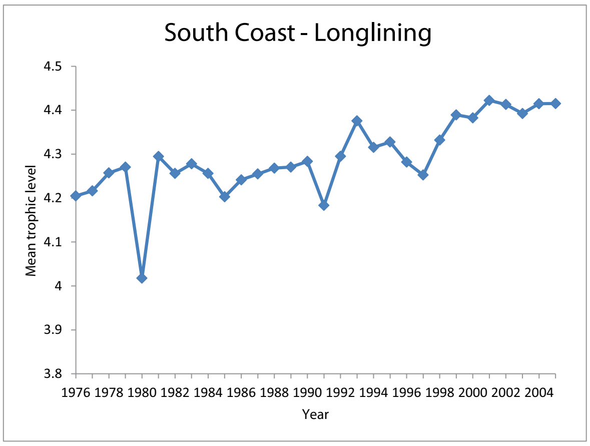
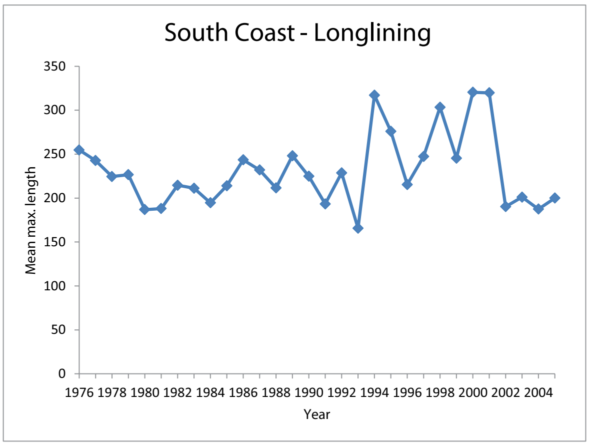


Figure 15. Time series of ecosystem-based indices for cpue data derived from commercial fishers using handlines within the South Coast bioregion of Western Australia.

Figure 16. Time series of diversity indices for cpue data derived from commercial fishers using longlines within the South Coast bioregion of Western Australia.





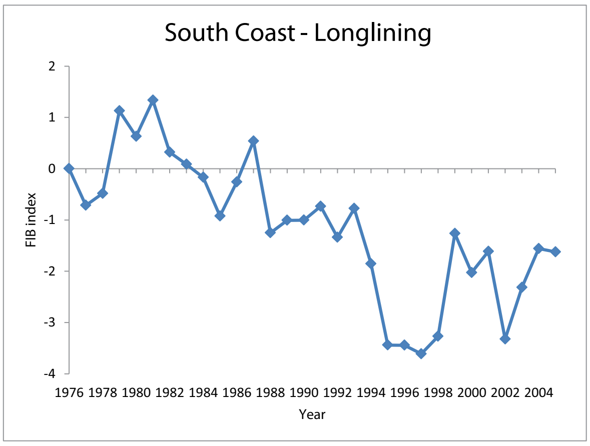


Figure 17. Time series of ecosystem-based indices for cpue data derived from commercial fishers using longlines within the South Coast bioregion of Western Australia.

In the Gascoyne bioregion, while both the mean trophic level and mean maximum length for the droplining fishery remained relatively stable, the FIB exhibited a slight increase over the period, indicating that the fishery was taking catches that slightly exceeded those that would have been predicted given the time series of values of trophic level (Fig. 19). While similar stability of mean trophic length, mean maximum length, and also the FIB, were apparent in the early period for the gillnetting data in this region, these series became highly variable in the 1990s as a probable consequence of the marked increase in catches of sharks in the years corresponding to the peaks in the first two series (Fig. 21). Note that the gaps in the time series of indices that appear in Fig. 20, and subsequent figures, represent years in which data were insufficient for calculation of the indices. Note also that, in some years, sufficient data were available to calculate the mean maximum length but, because trophic levels were not available for all species, there were insufficient data to calculate the mean trophic level or FIB. While the catches of the gillnetting fishery appeared to be less than expected given the trophic levels in 1990 and 1991, they appear subsequently to have recovered to reflect the original balance. It should be noted that this "recovery" may be due to socioeconomic drivers rather than an ecosystem change. The mean trophic level of the handlining catches decreased between 1976 and 2004 but, apart from a short-lived increase in 1980 and 1981, the mean maximum length has remained relatively stable (Fig. 23). The pattern exhibited by the FIB suggests a slight reduction in balance of catches relative to those expected given the trophic levels between 1984 and 1988 from those estimated for 1977 to 1983, and then a subsequent increase in the balance of catches relative to expected catches in the last decade. The data for longlining are more patchy, but the mean trophic level appears to have remained relatively constant, as does the mean maximum size until 1997 (Fig. 25). The latter has increased markedly in 2003 and 2004.

Figure 18. Time series of diversity indices for cpue data derived from commercial fishers using droplines within the Gascoyne bioregion of Western Australia.

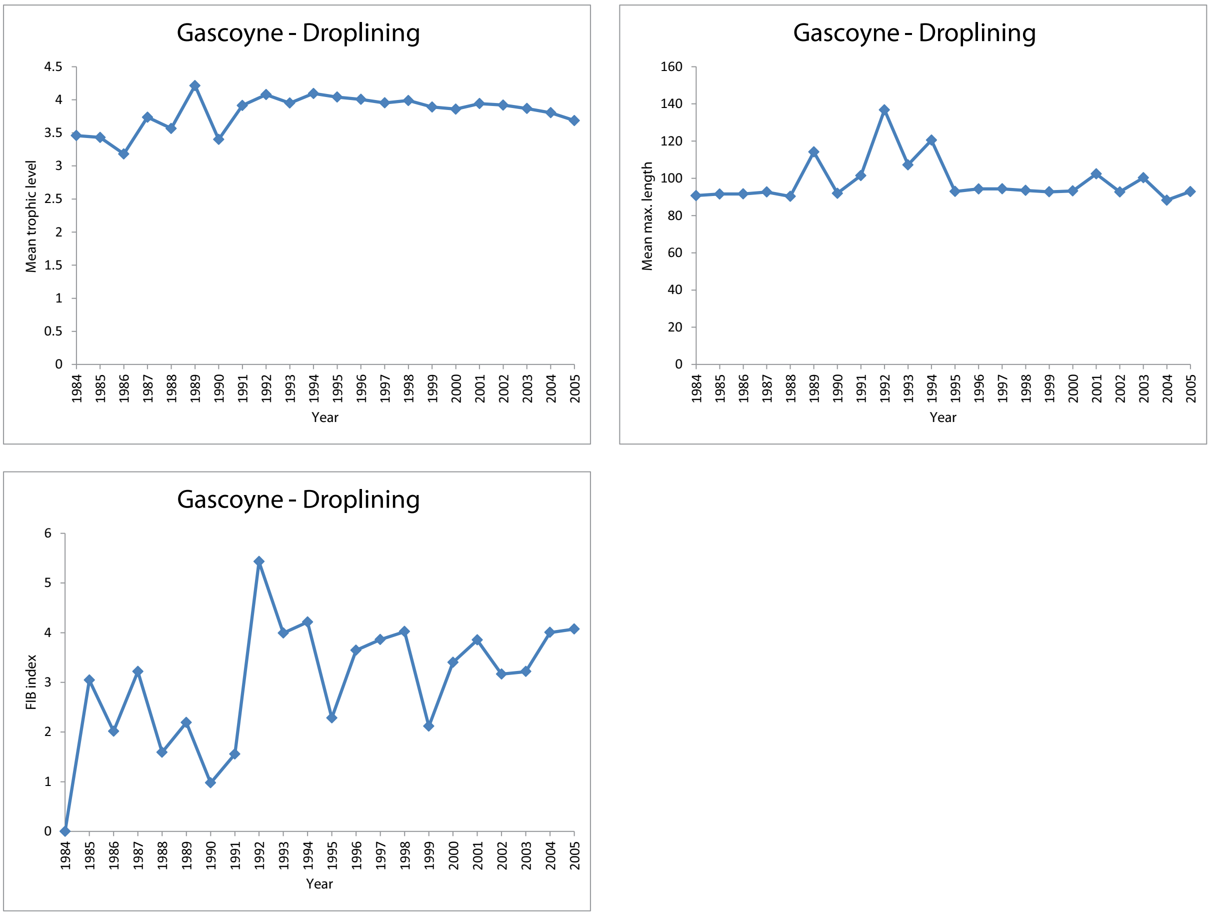


Figure 19. Time series of ecosystem-based indices for cpue data derived from commercial fishers using droplines within the Gascoyne bioregion of Western Australia.

Figure 20. Time series of diversity indices for cpue data derived from commercial fishers using gillnets within the Gascoyne bioregion of Western Australia.

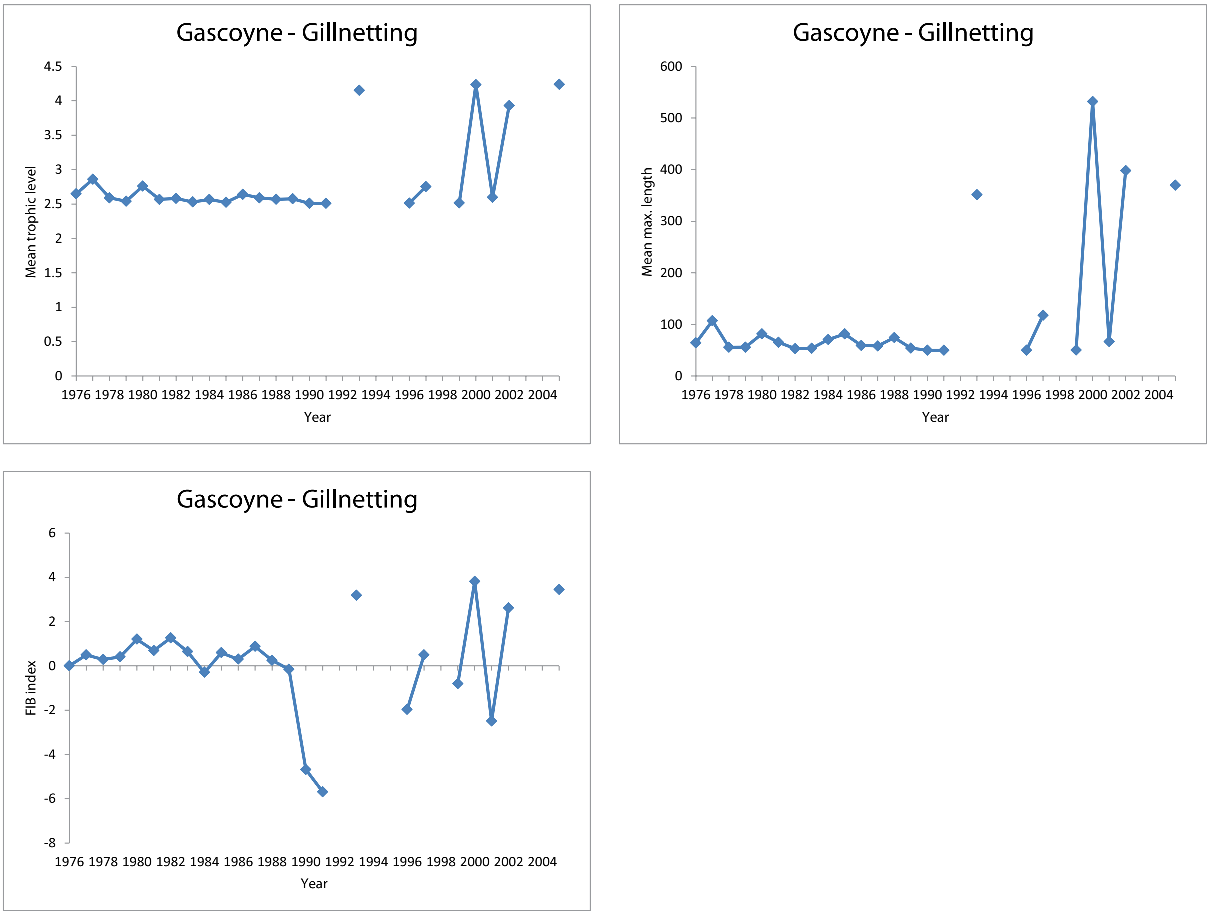
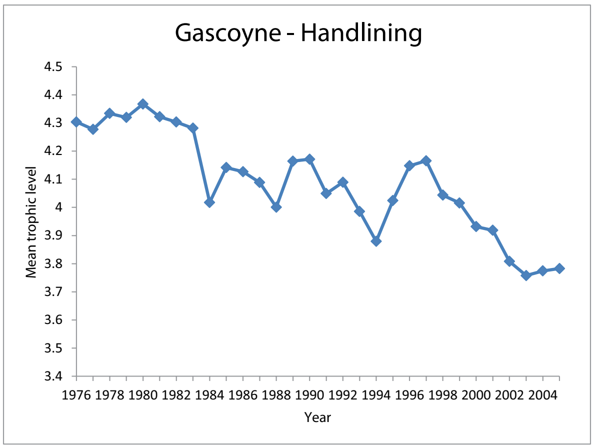
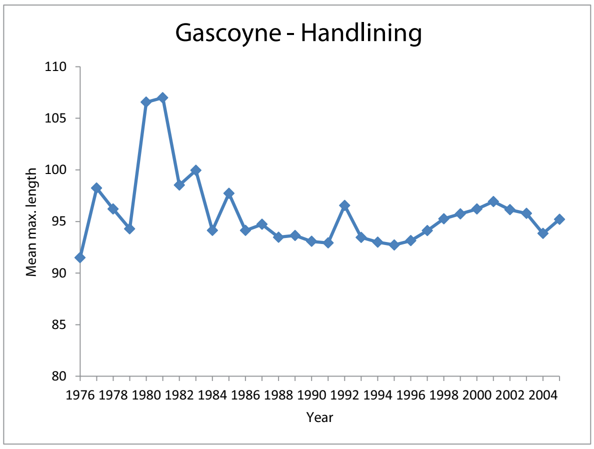


Figure 21. Time series of ecosystem-based indices for cpue data derived from commercial fishers using gillnets within the Gascoyne bioregion of Western Australia.

Figure 22. Time series of diversity indices for cpue data derived from commercial fishers using handlines within the Gascoyne bioregion of Western Australia.





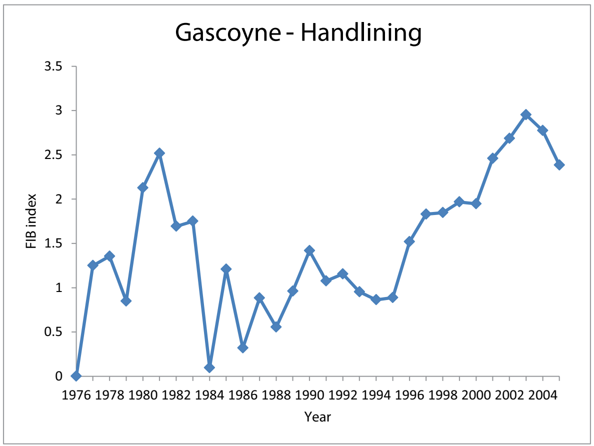
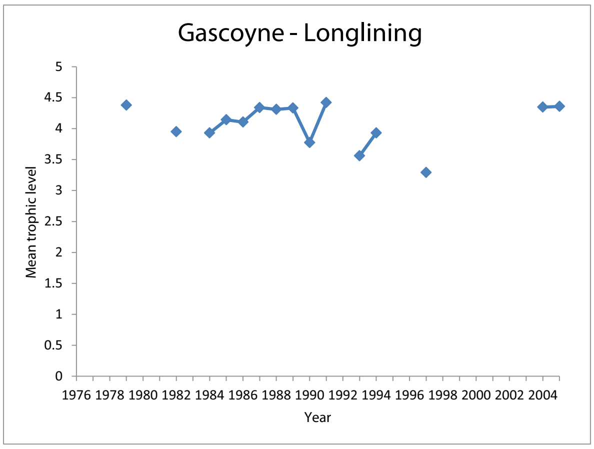
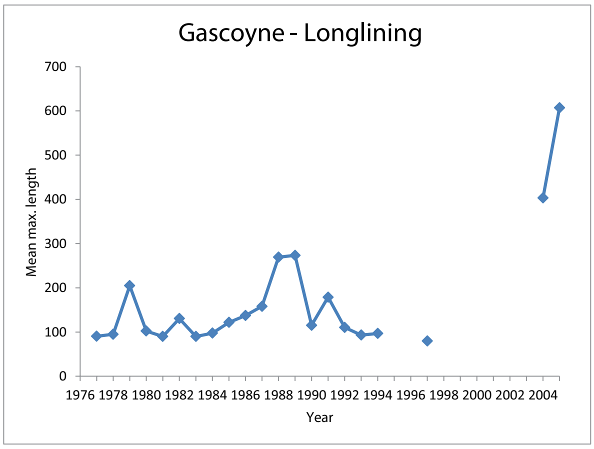


Figure 23. Time series of ecosystem-based indices for cpue data derived from commercial fishers using handlines within the Gascoyne bioregion of Western Australia.

Figure 24. Time series of diversity indices for cpue data derived from commercial fishers using longlines within the Gascoyne bioregion of Western Australia.





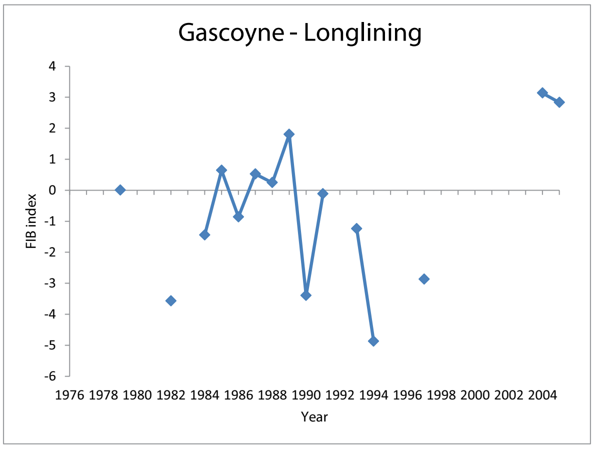
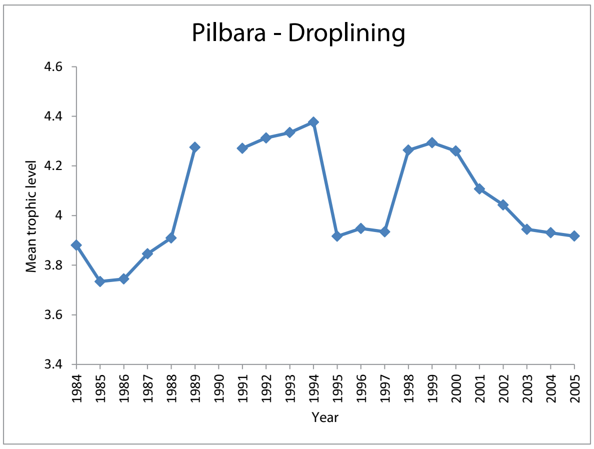
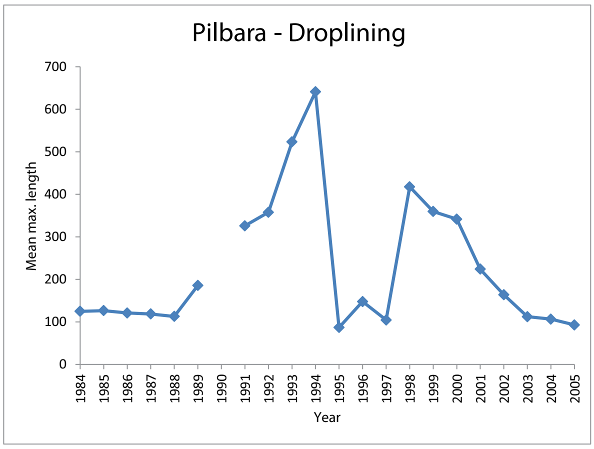


Figure 25. Time series of ecosystem-based indices for cpue data derived from commercial fishers using longlines within the Gascoyne bioregion of Western Australia.

Two peaks are present in the plot of the mean trophic level for droplining catches in the Pilbara bioregion, which are matched by similar peaks in the plot of the mean maximum size, These peaks appear to be associated with increased catches of large sharks (Fig. 27). The FIB declined in the period from 1984 to 1989, but subsequently recovered and then remained relatively stable. For gillnetting, the mean trophic level appeared to increase between 1976 and the late 1990s, but subsequently declined slightly (Fig. 29). The mean maximum size remained relatively stable until the mid 1990s, then rose to a peak between 1996 and 1998 before declining to its previous level by 2004. The FIB increased gradually from 1976 to the mid 1990s but has shown a marked decrease since that time, suggesting that the increase in catches that might have been expected given the decrease in trophic level has not been realised. There appears to have been a slight but gradual decline in the mean trophic level and mean maximum size of handline catches, but the FIB, which is highly variable and displays several peaks, suggests an overall increase rather than a loss of balance (Fig. 31). Longlining data are patchy, but plots of the mean trophic level and the mean maximum size suggest that the values of these variables were initially more patchy, then increased from the lower levels recorded in the 1980s to those determined for the 1990s, which appear to have been relatively stable (Fig. 33). The trend exhibited by the FIB suggests that, since the early 1980s, catches have increased to levels that would not have been predicted given the values of mean trophic level.

Figure 26. Time series of diversity indices for cpue data derived from commercial fishers using droplines within the Pilbara bioregion of Western Australia.





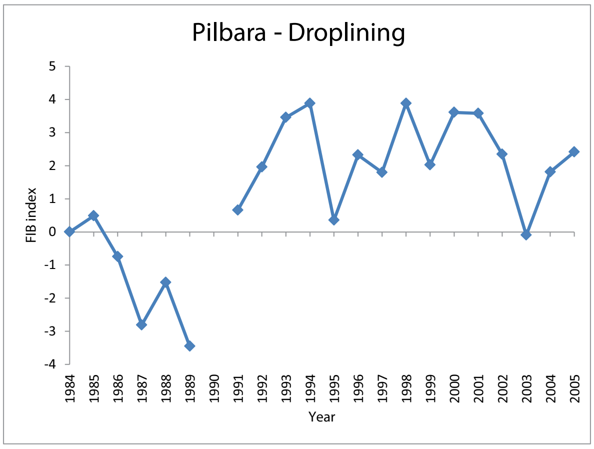


Figure 27. Time series of ecosystem-based indices for cpue data derived from commercial fishers using droplines within the Pilbara bioregion of Western Australia.

Figure 28. Time series of diversity indices for cpue data derived from commercial fishers using gillnets within the Pilbara bioregion of Western Australia.

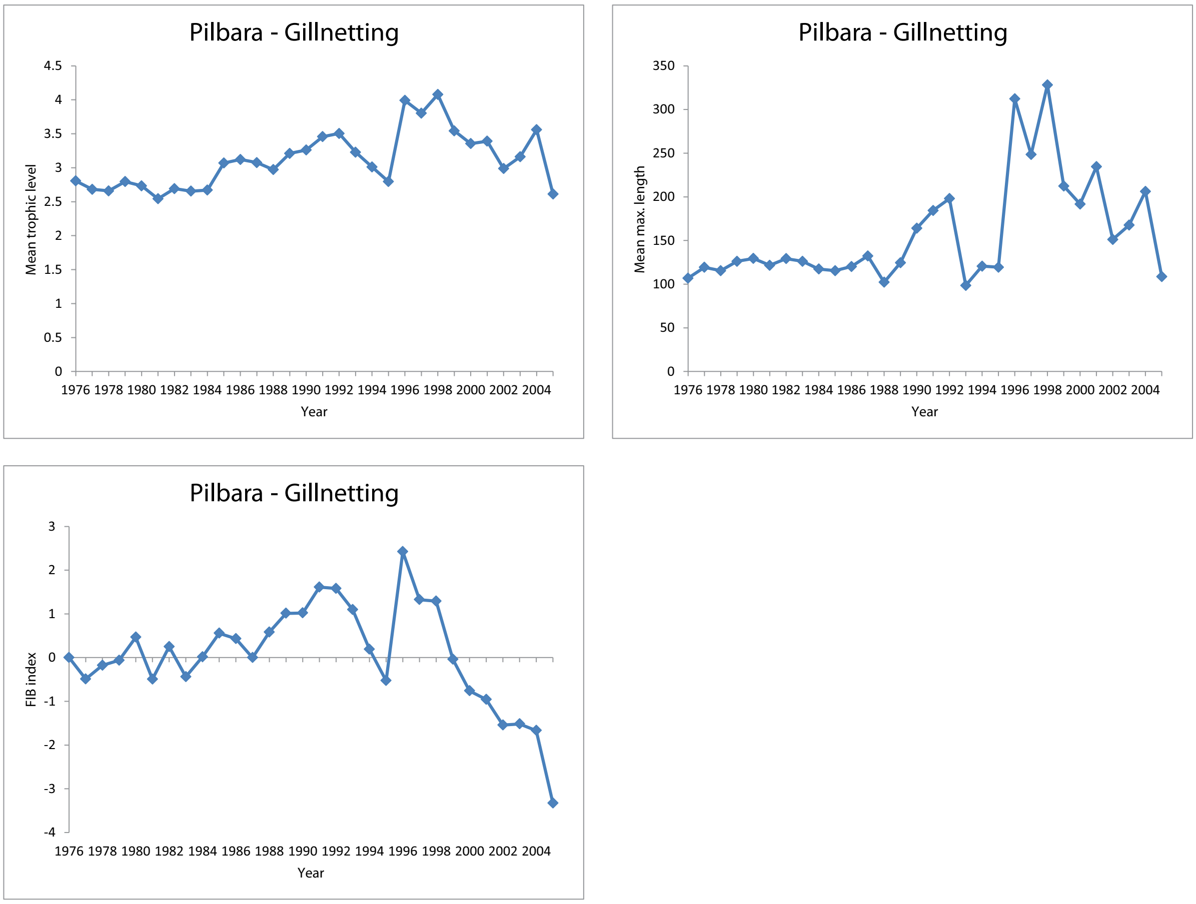


Figure 29. Time series of ecosystem-based indices for cpue data derived from commercial fishers using gillnets within the Pilbara bioregion of Western Australia.

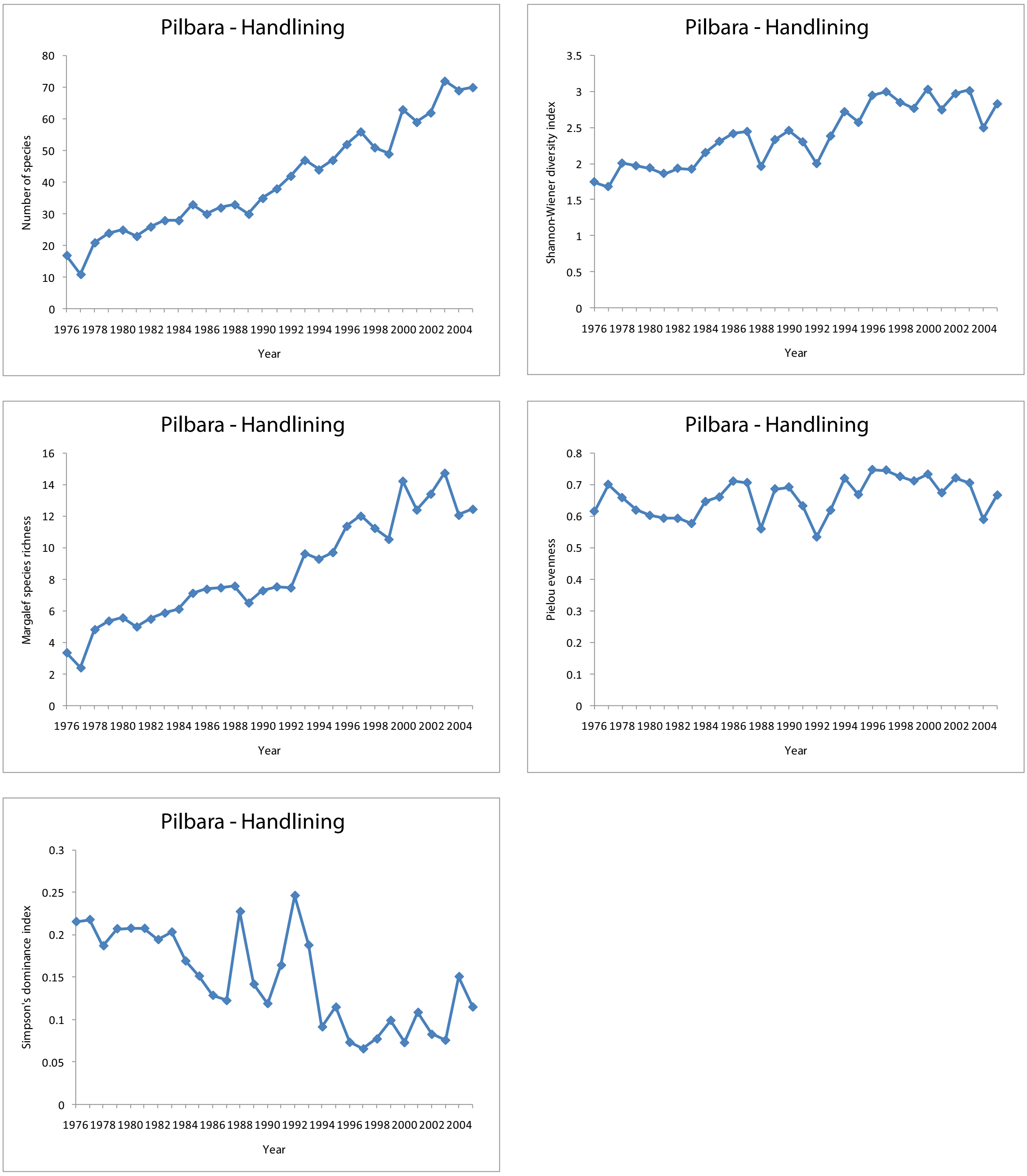


Figure 30. Time series of diversity indices for cpue data derived from commercial fishers using handlines within the Pilbara bioregion of Western Australia.

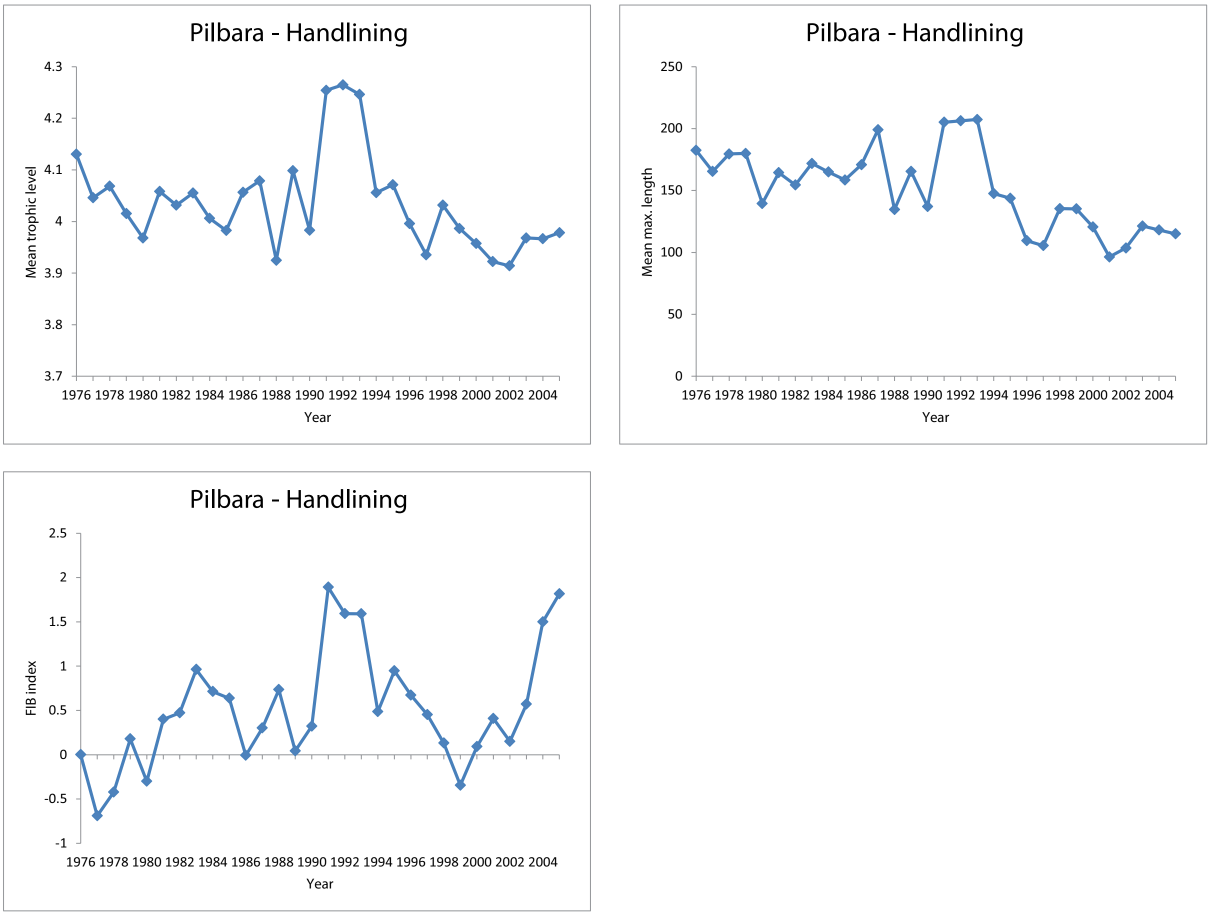


Figure 31. Time series of ecosystem-based indices for cpue data derived from commercial fishers using handlines within the Pilbara bioregion of Western Australia.

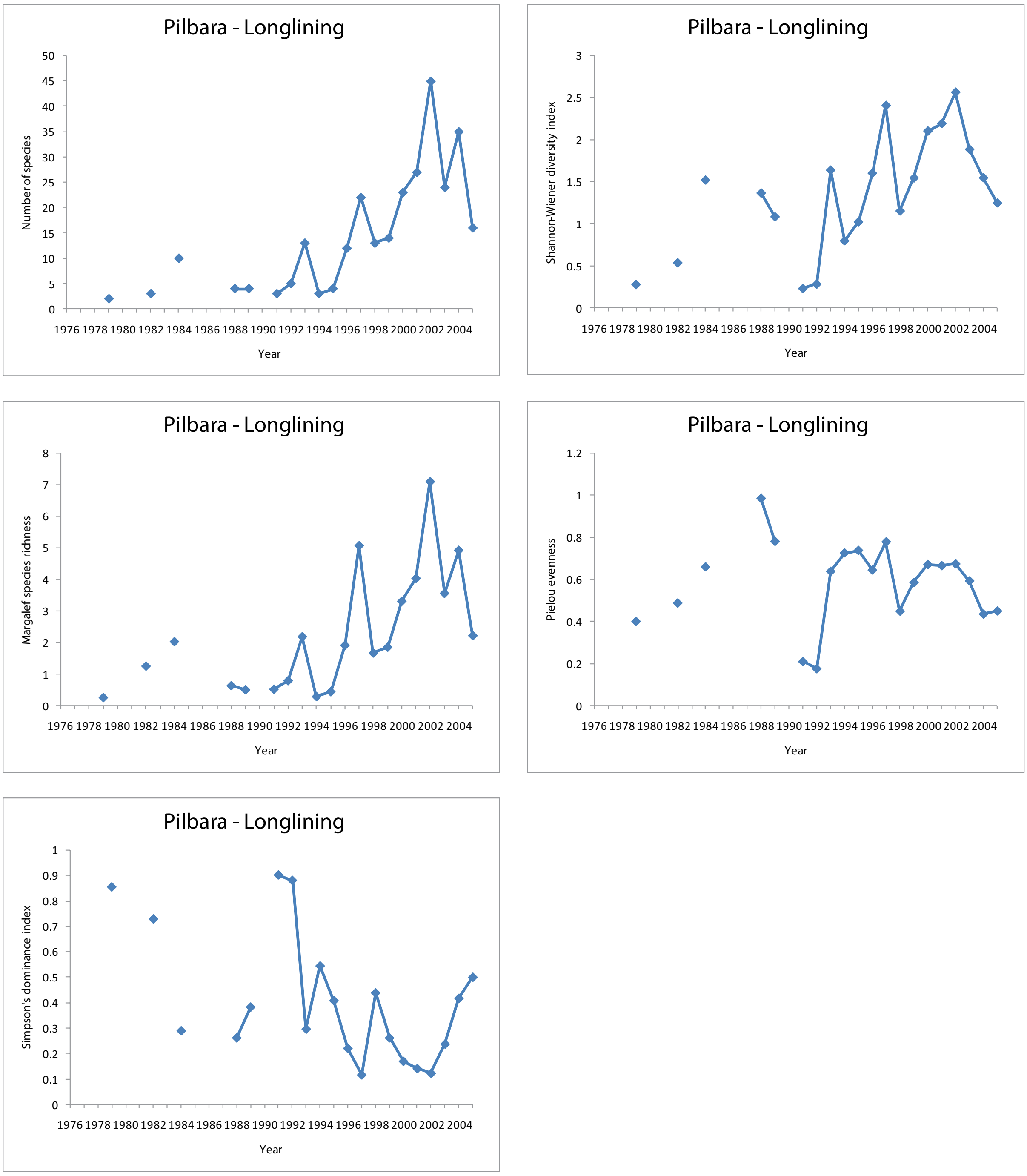
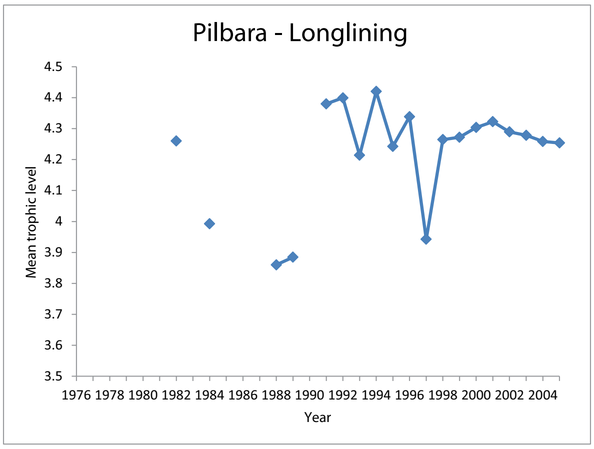
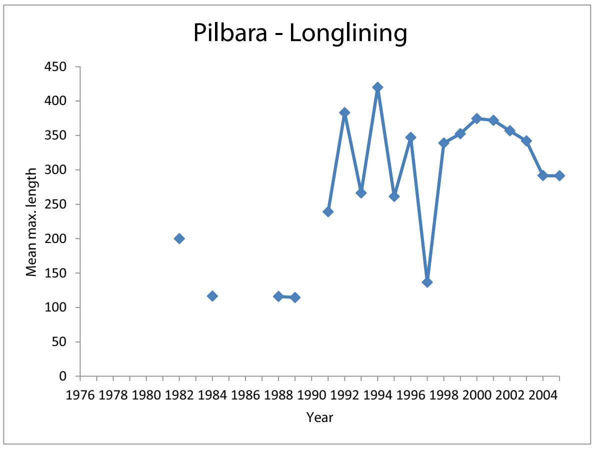


Figure 32. Time series of diversity indices for cpue data derived from commercial fishers using longlines within the Pilbara bioregion of Western Australia.





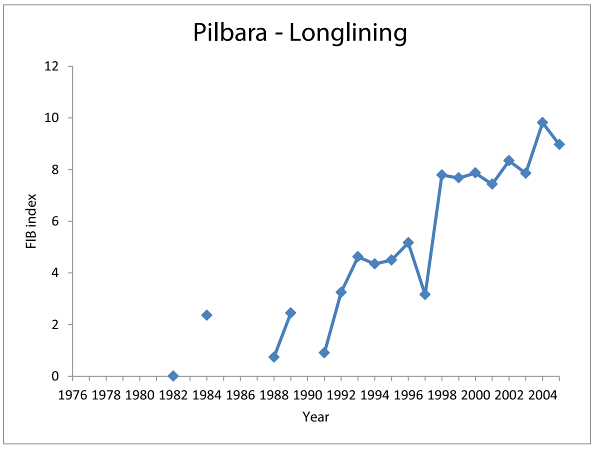


Figure 33. Time series of ecosystem-based indices for cpue data derived from commercial fishers using longlines within the Pilbara bioregion of Western Australia.

Apart from a large increase in the mean maximum size in 1994, and a small increase in mean trophic level in 1993 and 1994, that appear related to increased shark catches, the droplining data for the Kimberley bioregion appear relatively stable (Fig. 35), although the FIB for dropline catches appears to have increased. The mean trophic level for gillnet catches increased over the period however, and has been relatively stable over the last decade (Fig. 37). Mean maximum length has remained approximately constant while the FIB has paralleled the trend shown by the mean trophic level. The mean trophic level for handlining catches rose between 1976 and 1990, but subsequently declined to approximately the 1976 level where it has remained over the last decade (Fig. 39). The plot of mean maximum length mirrored the trend shown by the mean trophic level. While the FIB increased over the period from 1976 to 1996, it subsequently declined and, from 1999 to 2005, has remained at levels similar to those in the mid-1970s. The latter trend suggests that catches in this latter period have not increased to the levels that would have been expected given the reduction in mean trophic level. The longlining data are again patchy, and thus provide inadequate information from which to determine whether there has been any adverse trend in mean trophic level, mean maximum length or FIB (Fig. 41).

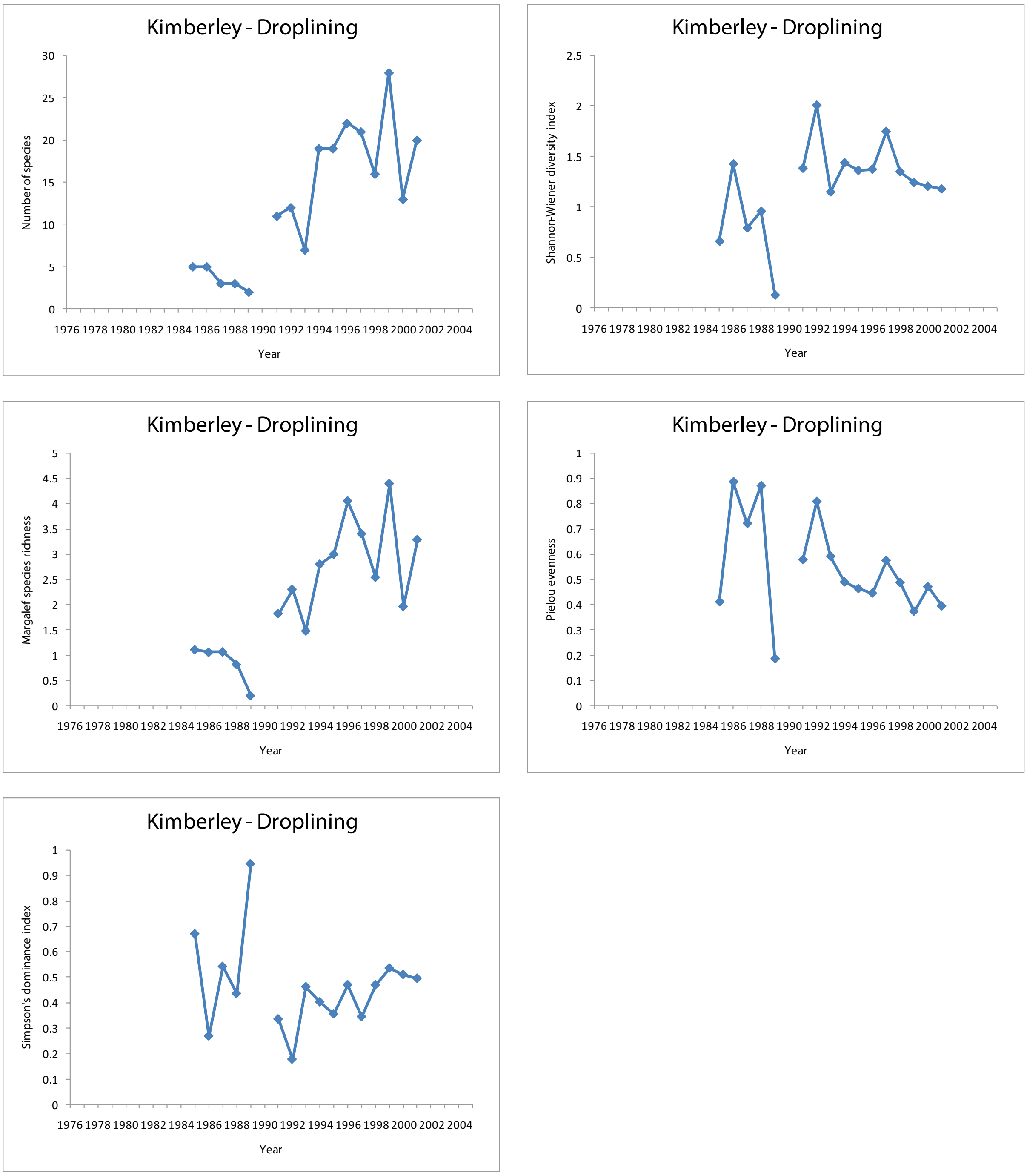
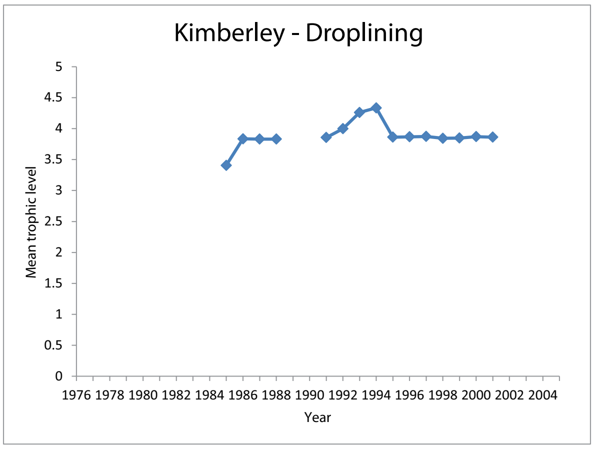
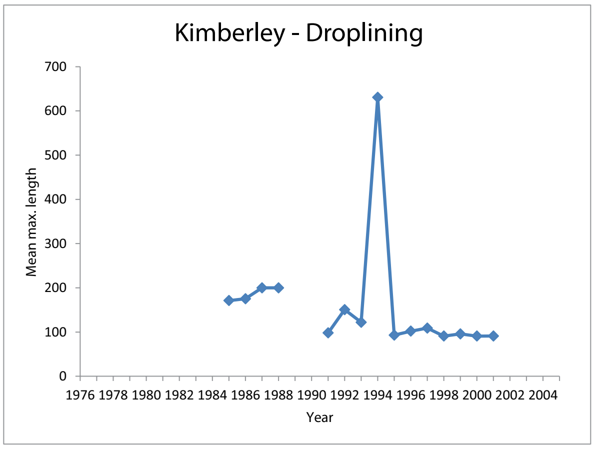


Figure 34. Time series of diversity indices for cpue data derived from commercial fishers using droplines within the Kimberley bioregion of Western Australia.





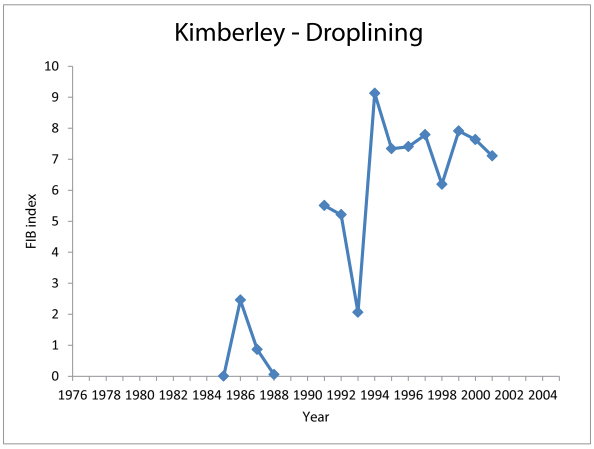


Figure 35. Time series of ecosystem-based indices for cpue data derived from commercial fishers using droplines within the Kimberley bioregion of Western Australia.

Figure 36. Time series of diversity indices for cpue data derived from commercial fishers using gillnets within the Kimberley bioregion of Western Australia.

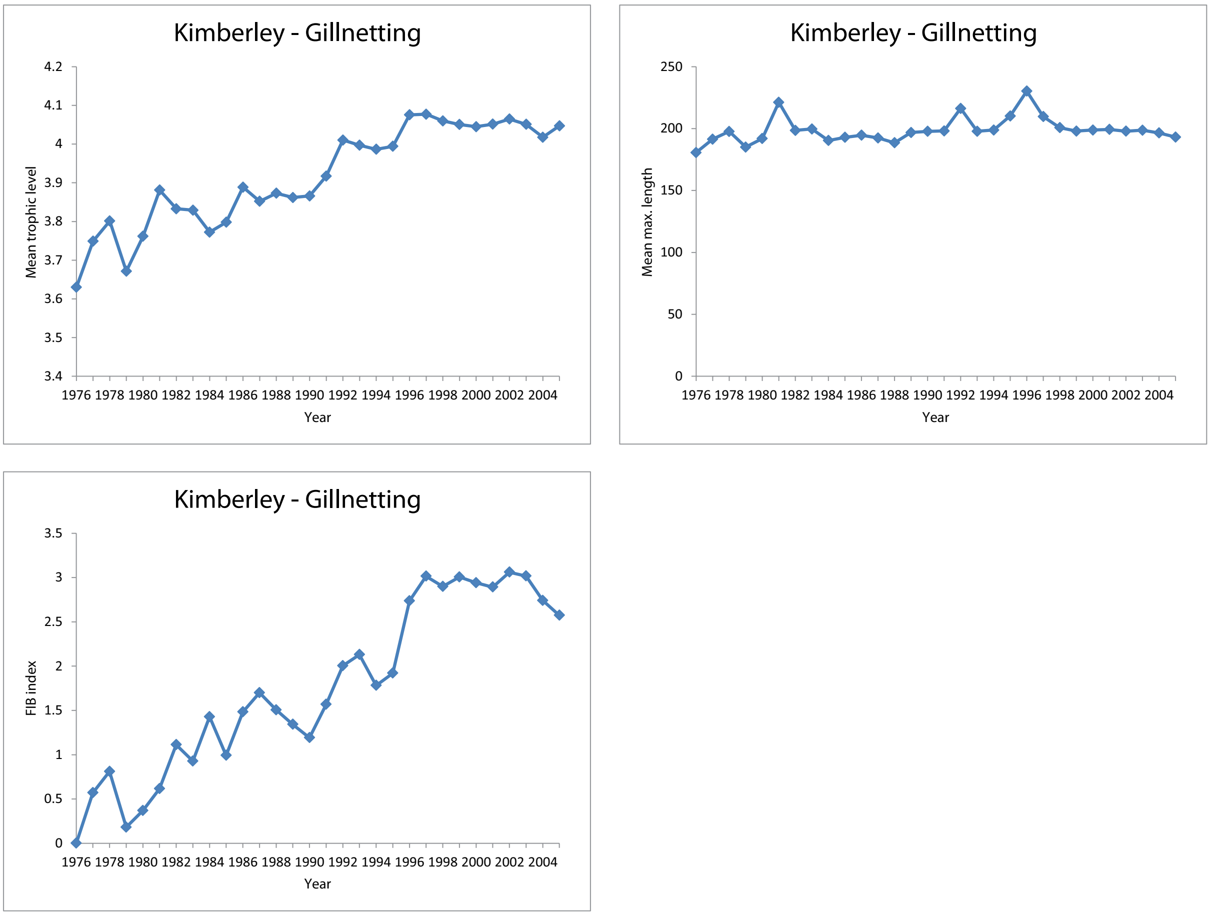


Figure 37. Time series of ecosystem-based indices for cpue data derived from commercial fishers using gillnets within the Kimberley bioregion of Western Australia.

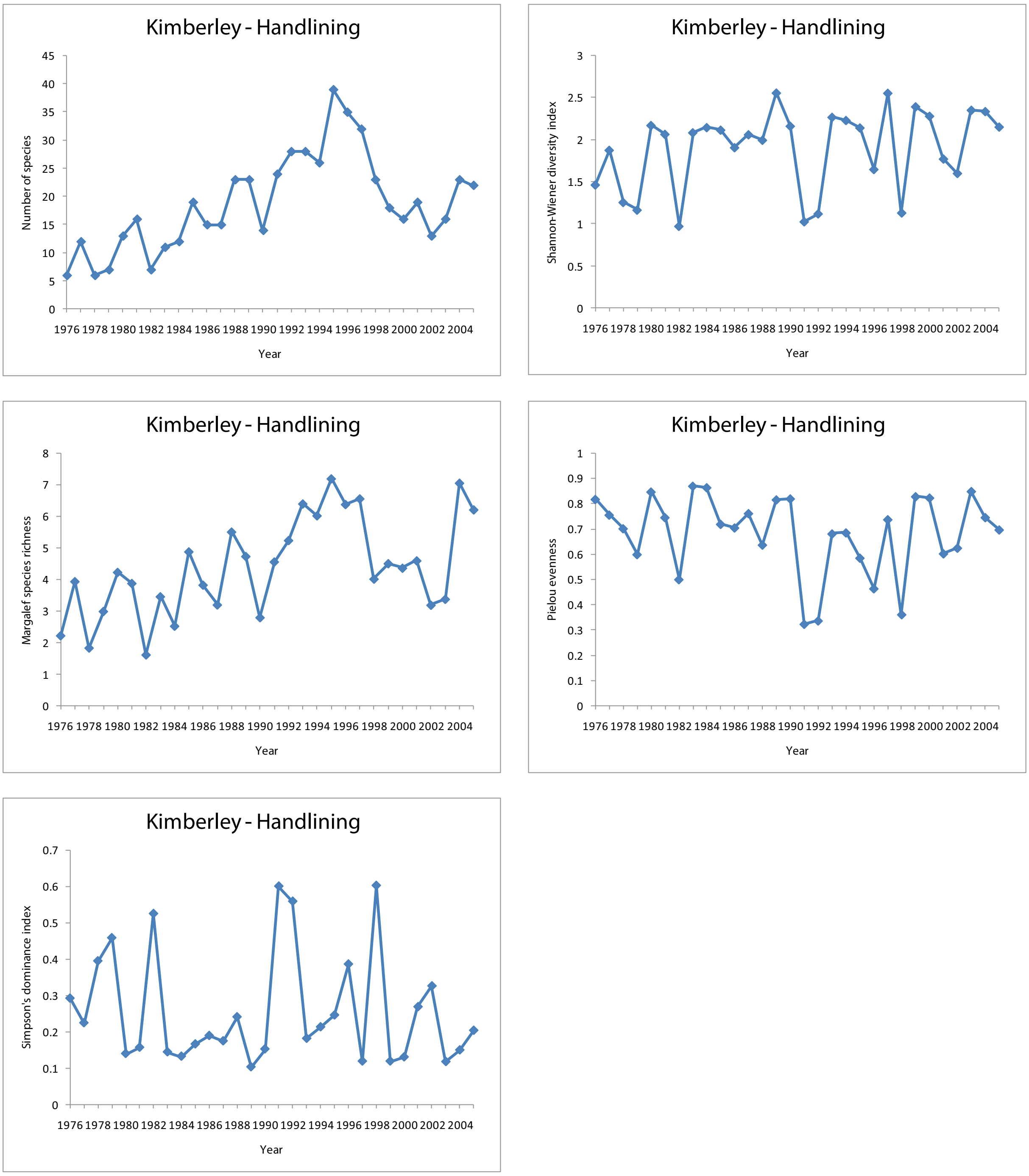


Figure 38. Time series of diversity indices for cpue data derived from commercial fishers using handlines within the Kimberley bioregion of Western Australia.

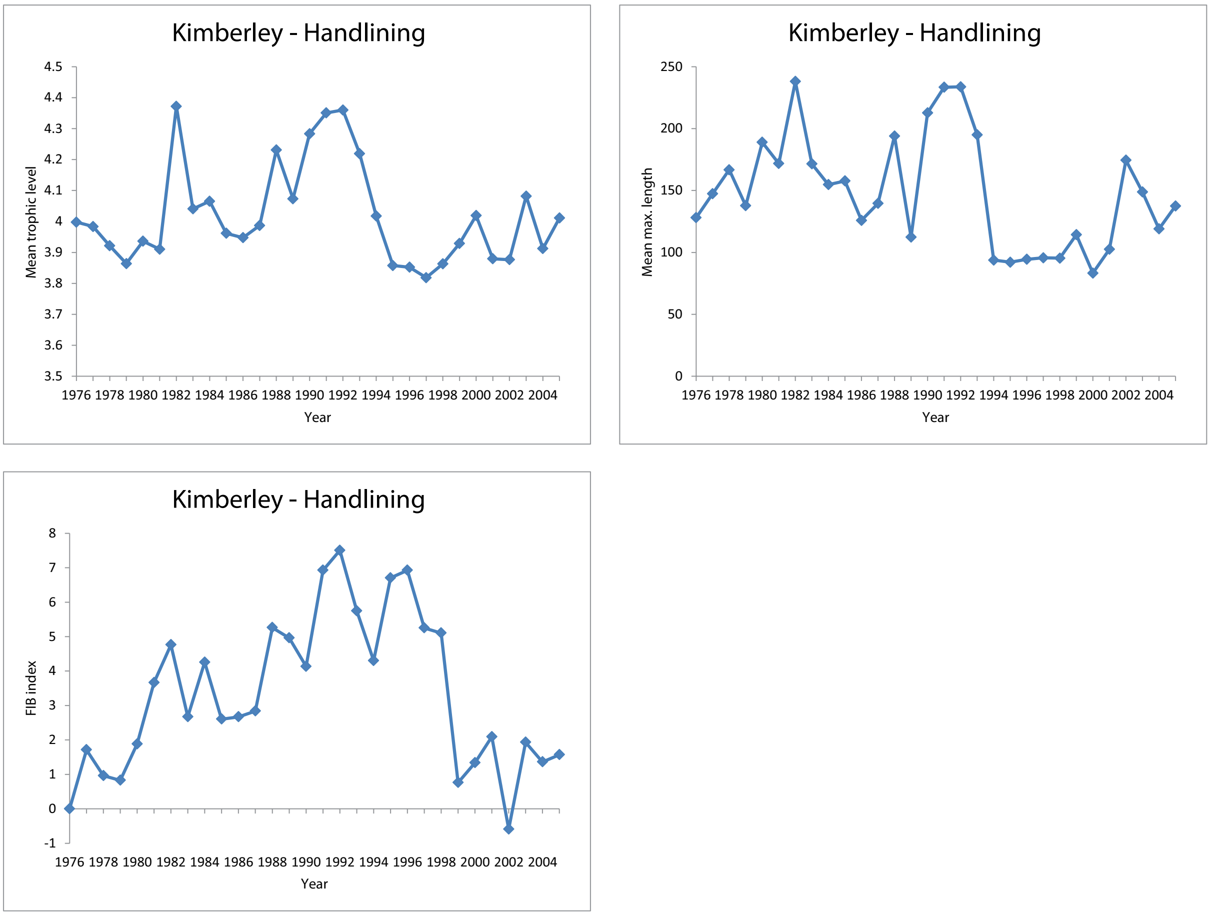


Figure 39. Time series of ecosystem-based indices for cpue data derived from commercial fishers using handlines within the Kimberley bioregion of Western Australia.

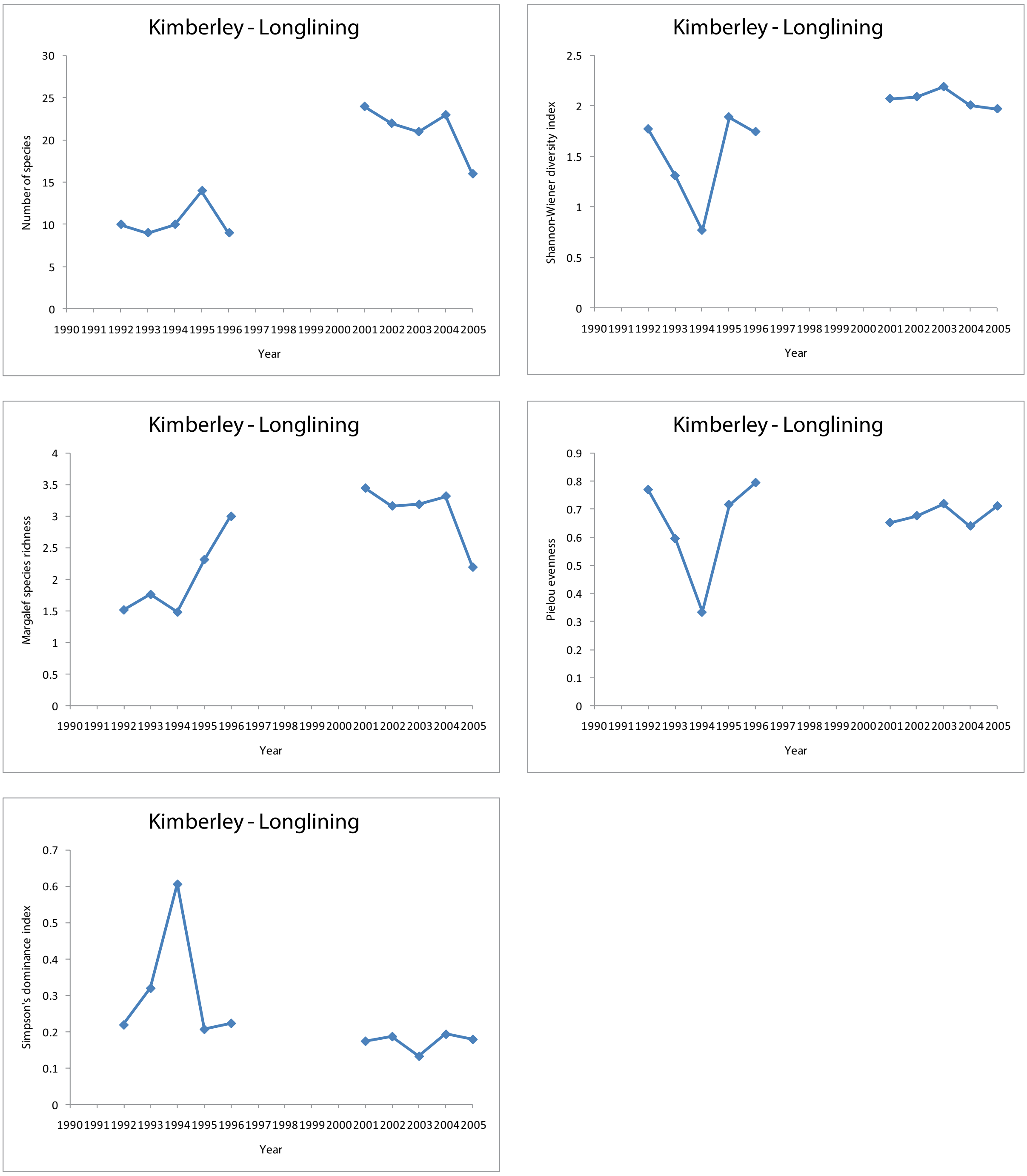


Figure 40. Time series of diversity indices for cpue data derived from commercial fishers using longlines within the Kimberley bioregion of Western Australia.

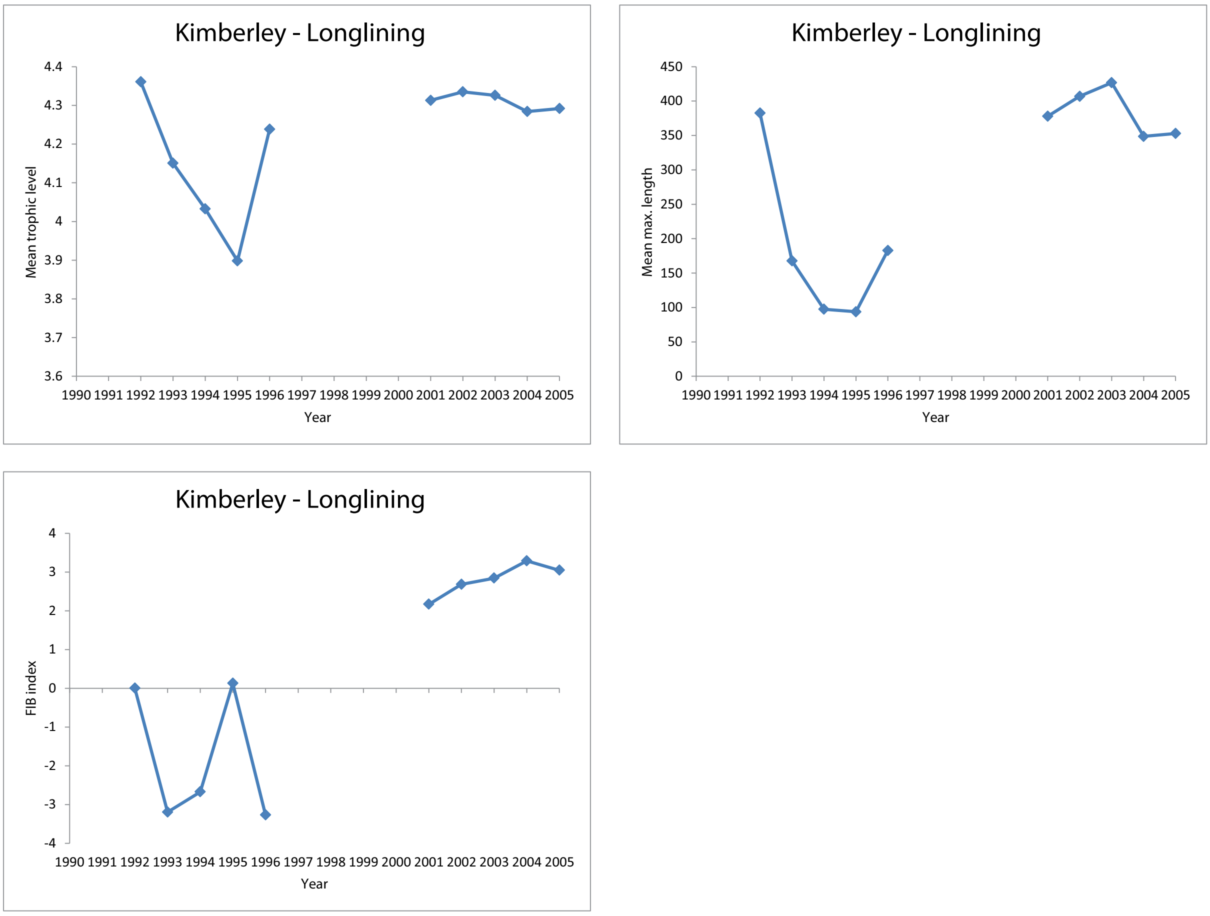


Figure 41. Time series of ecosystem-based indices for cpue data derived from commercial fishers using longlines within the Kimberley bioregion of Western Australia.

For the combined catch data for the West Coast bioregion, mean trophic level increased with catch between 1976 and 1990, but then remained relatively stable (Fig. 42) The mean maximum length also increased but, since the 1990s has become relatively stable. The FIB rose rapidly between 1976 and 1990, but since that time has displayed only a gradual increase, suggesting that, over this latter period, the fishery has maintained its balance. A similar pattern is evident in these indices for the South Coast bioregion, although the mean maximum length in this region appears to have been more stable (Fig. 43). While the mean maximum length has remained very stable in the Gascoyne region, the mean trophic level reveals two distinct levels with the level lying between approximately 3 and 3.5 till 1990 then increasing to lie between 3.5 and 4 for the remaining period (Fig. 44). The FIB also increased in 1990 and remained relatively stable till about 2000, following which it increased further. The increase in the FIB can be attributed to catches in excess of those that would have been predicted given the observed trophic levels. The mean trophic level in the Pilbara bioregion increased almost linearly in the period from 1976 to 2000, but subsequently appears to have remained stable (Fig. 45). An increase in mean maximum length became evident in 1991, with greater variability in mean maximum length since that time. The FIB has also increased approximately linearly in the period from 1976 to 2005, suggesting that there has been a progressive and continuous increase in catches relative to the levels that might have been expected given the observed mean trophic levels. A similar linear increase in mean trophic level is evident in the catch data for the Kimberley bioregion (Fig. 46). Apart from a very marked increase in 1994, the mean maximum length of fish in the catches from the Kimberley remained relatively stable until 2000, but subsequently have increased substantially. An approximately linear increase is evident in the values of the FIB for this region, again implying the catches relative to those predicted from the recorded values of trophic level have progressively increased over the period.

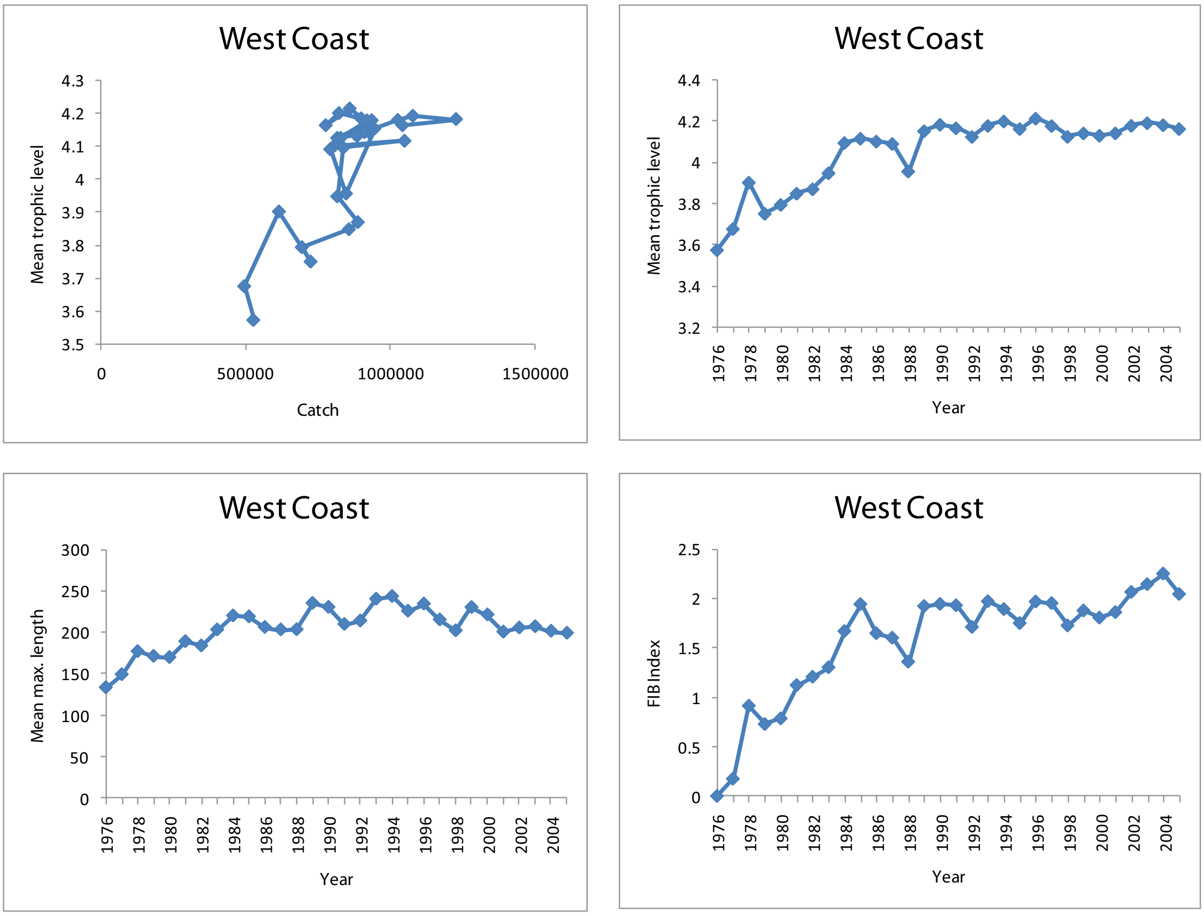


Figure 42. Time series of ecosystem-based indices for catch data derived from commercial fishers using droplines, gillnets, handlines and longlines within the West Coast bioregion of Western Australia.

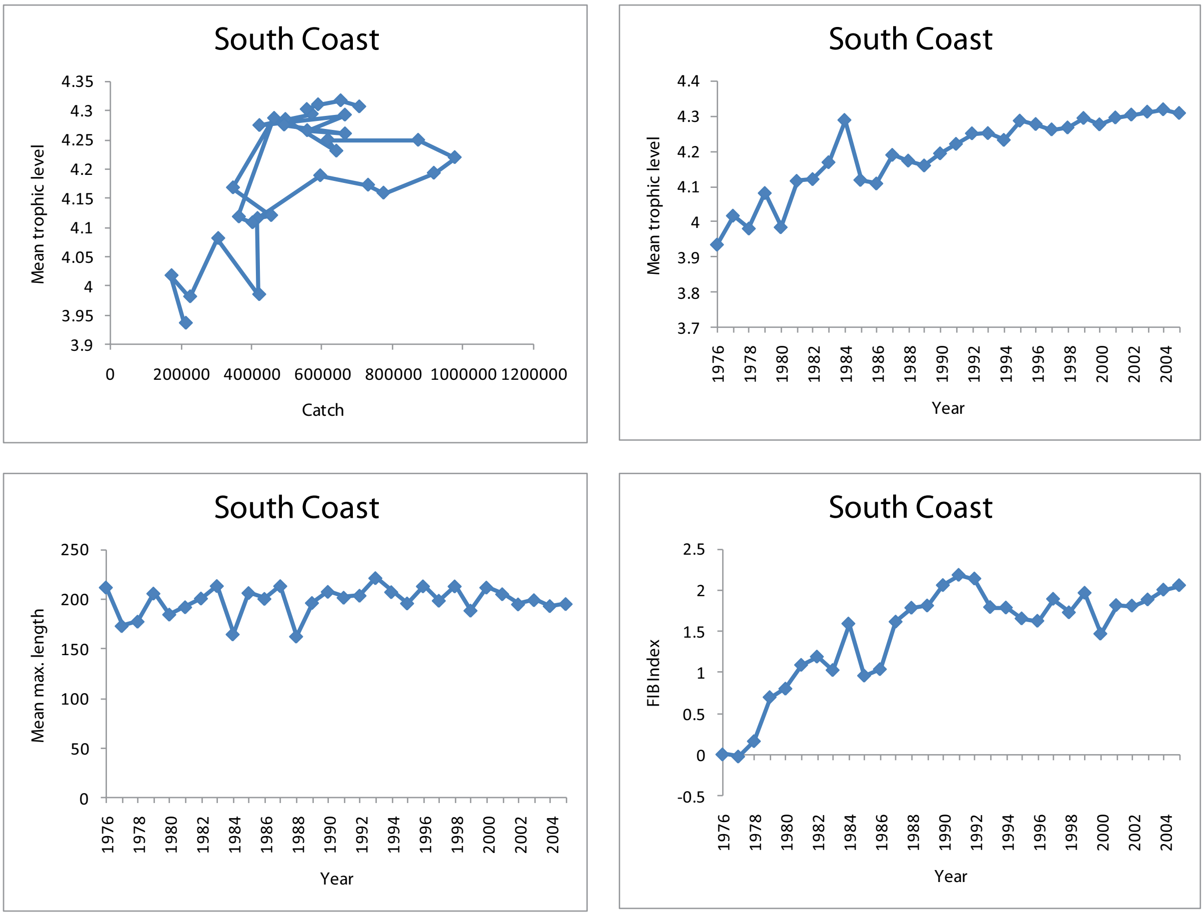


Figure 43. Time series of ecosystem-based indices for catch data derived from commercial fishers using droplines, gillnets, handlines and longlines within the South Coast bioregion of Western Australia.

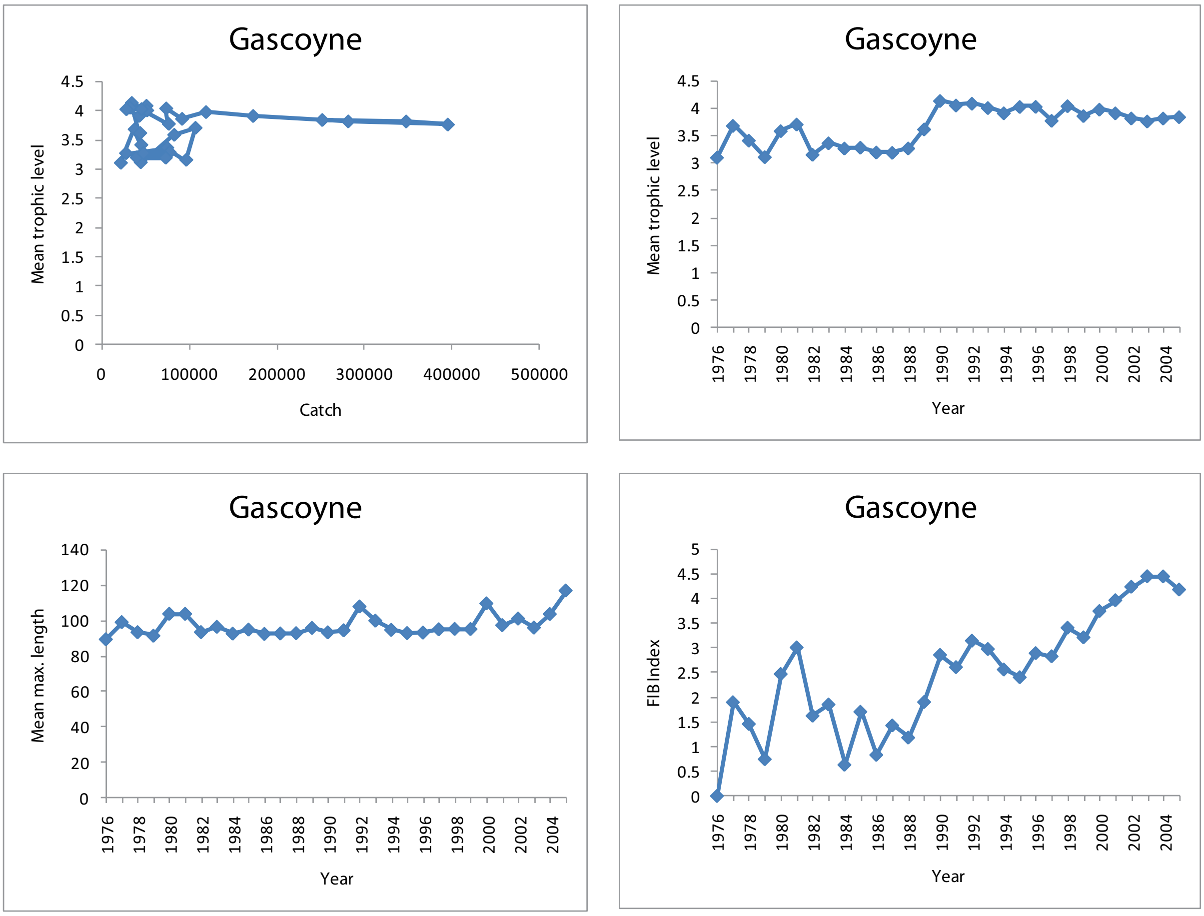


Figure 44. Time series of ecosystem-based indices for catch data derived from commercial fishers using droplines, gillnets, handlines and longlines within the Gascoyne bioregion of Western Australia.

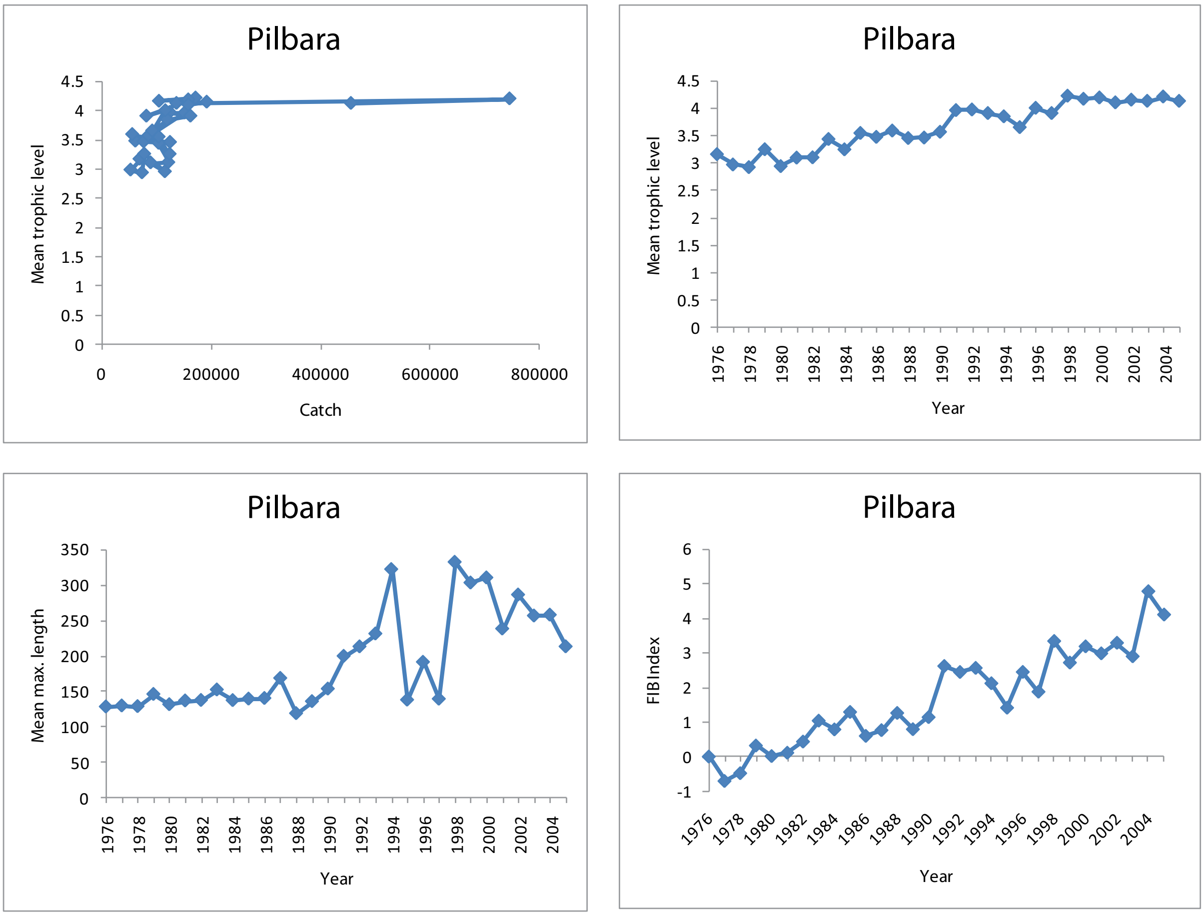


Figure 45. Time series of ecosystem-based indices for catch data derived from commercial fishers using droplines, gillnets, handlines and longlines within the Pilbara bioregion of Western Australia.

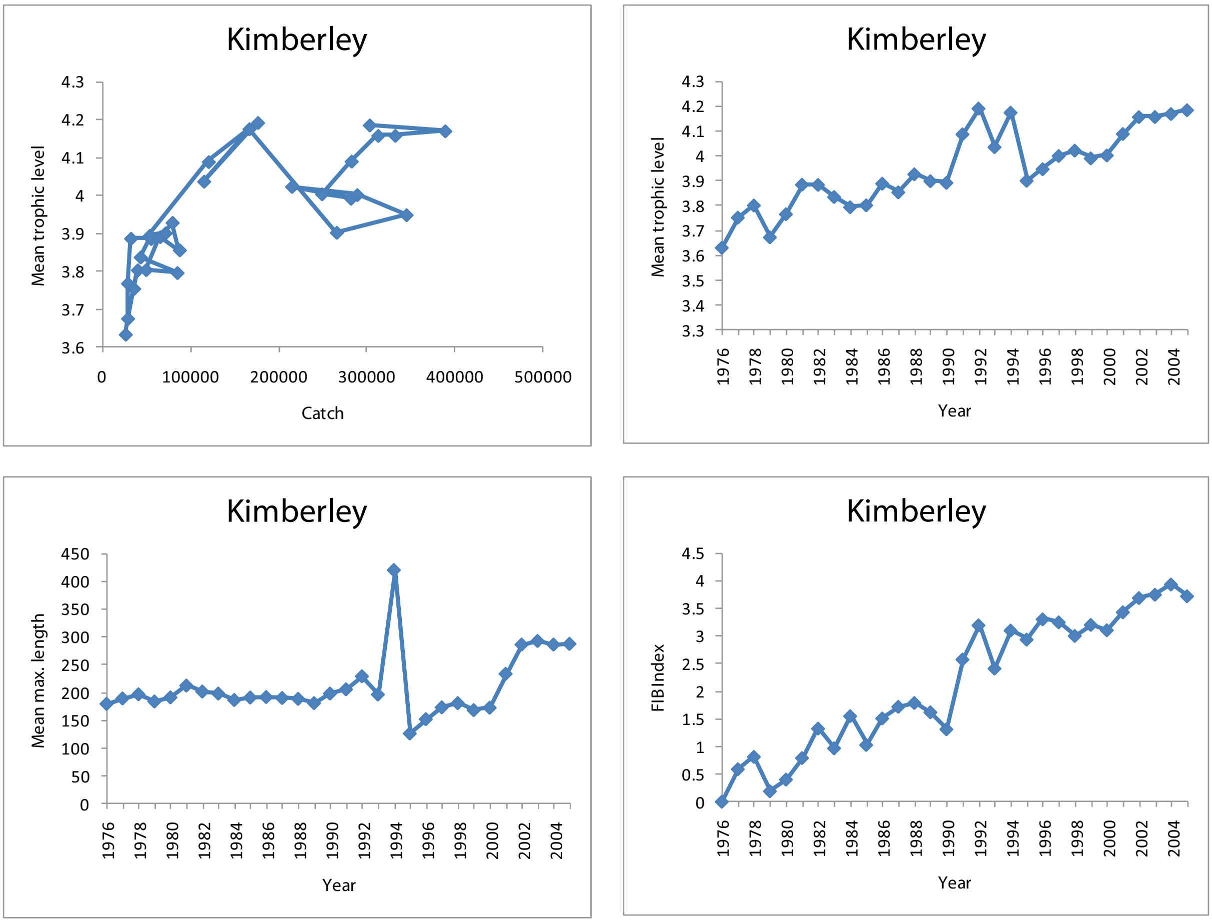


Figure 46. Time series of diversity indices for catch data derived from commercial fishers using droplines, gillnets, handlines and longlines within the Kimberley bioregion of Western Australia.

The results of these analyses of the broad bioregion-wide data suggest that the fisheries in these regions have expanded to catch species of higher trophic levels, and that, through time, catches have exceeded those that would have maintained the balance that was reflected in the 1976 catch data. These results suggest that more than one system is being accessed by commercial fishers. The data provide no evidence that mean trophic level or mean maximum length in any of these five bioregions have been adversely affected by removal of these catches.

While the traditional diversity indices are known to have limitations, they are useful in providing a rapid, broad assessment of the fishery, and potentially allow identification of unusual trends that require further investigation. These indices reflect the fisheries data that are used in the analysis, but the latter are influenced not only by fishing intensity, but also by the changes in species composition that result from changing fishing practices and selective fishing methods. These fisheries-dependent data will almost invariably provide a biased representation of the species composition of the fish stocks within the ecosystem. Multivariate analyses of the fisheries data will allow assessment of hypotheses that the species composition of the catches has changed, but inferences regarding changes in the species composition within the ecosystem rely on the availability of comparable samples, such as might be provided by fisheries-independent sampling. Without careful statistical analysis to distinguish the changes in the underlying species composition of the fish fauna within the ecosystem from the influence of fishing practices, gear, distribution, etc., there will need to be caution when drawing inferences from the results of analyses of diversity and multivariate analyses. It should be noted also that similar caution must be exercised when using the time series of fisheries data to tune ecosystem models, under the assumption that these represent the dynamics of the stocks without recognition of the potential bias in these data resulting from changes in fishing practices.

The ecosystem-based indices of mean trophic level and mean maximum length are derived from catch data and thus, if the underlying fishery data are accurate, provide reliable indicators of those data. Once again, however, changes in these indicators are influenced both by changes in the underlying species composition of the fish fauna and changes in fishing practice. Thus, for example, declines in mean trophic level of catches may relate to fishing down the ecosystem (Pauly *et al.,* 1998), but equally may relate to decisions by fishers to exploit species at lower trophic levels (*e.g.* Essington *et al.*, 2006). While the FIB provides an indication of the extent to which changes in catches are consistent with predictions based on changes in mean trophic level, the index itself provides no indication as to whether the catches for any year are in balance with the fish fauna of the ecosystem itself. Thus, by itself, the FIB cannot be used to determine whether the levels and species mix of catches are appropriate to maintain ecosystem structure. They can, however, be used as indicators of change that can be used to alert scientists and managers of changes within the fishery that may require examination and possible management response.

Changes in species composition

Results of a preliminary multivariate analysis of the commercial fisheries data from the West Coast Bioregion suggested that use of the routines in Primer 6 would allow assessment of the statistical significance of change in species composition of catches and identification of the nature of the changes leading to the observed dissimilarity in those catch compositions. It was considered appropriate, however, to first apply the approach to data for another system, in which changes in community structure were known to have occurred. The results of this exploration are presented below. Note, however, that the analysis below relies on published data and has not had the benefit of critical review by the fisheries scientists or management agencies involved in the study of this U.S. fishery. The results for this fishery should therefore be only considered to be a preliminary exploration to assess the effectiveness of the approach used for analysis and not an authoritative study of the changes in species composition of the fishery.

Application to a system where changes have occurred.

A plot of the smoothed abundance indices derived from the standardised bottom trawl surveys conducted by the NEFSC in autumn (principal groundfish, flounders, other groundfish, other pelagic, squid) and spring (principal pelagic and small elasmobranchs) is presented in Fig. 47.

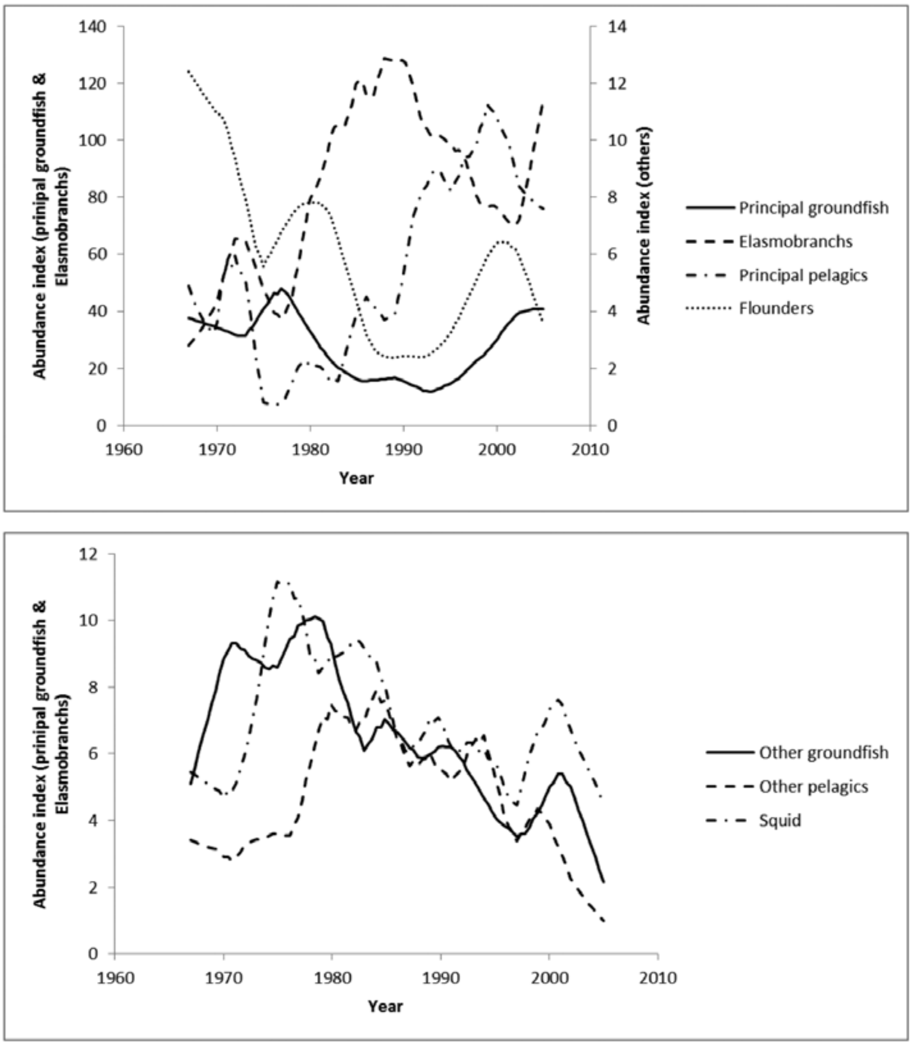


Figure 47. Smoothed abundance indices derived from bottom trawl surveys from north-eastern U.S.A.

The results of MDS ordination for both the forth-root-transformed abundance and proportions data revealed clear separation among the data for the different periods with an obvious change in species composition from the late 1960s to the 1980s, followed by an apparent recovery towards the species composition of the earlier period following 1995 (Figs 48 and 49). The stress levels of 0.12 and 0.11, respectively, which were calculated for these ordinations, suggest that reliable interpretations may be made of the patterns displayed.

Results of the cluster analysis of the fourth-root-transformed data revealed high similarity among years. The groups into which the data for the different years were clustered (Figs 50 and 51) were consistent with the patterns displayed on the 2-dimensional MDS plots, *i.e.* Figs 48 and 49.

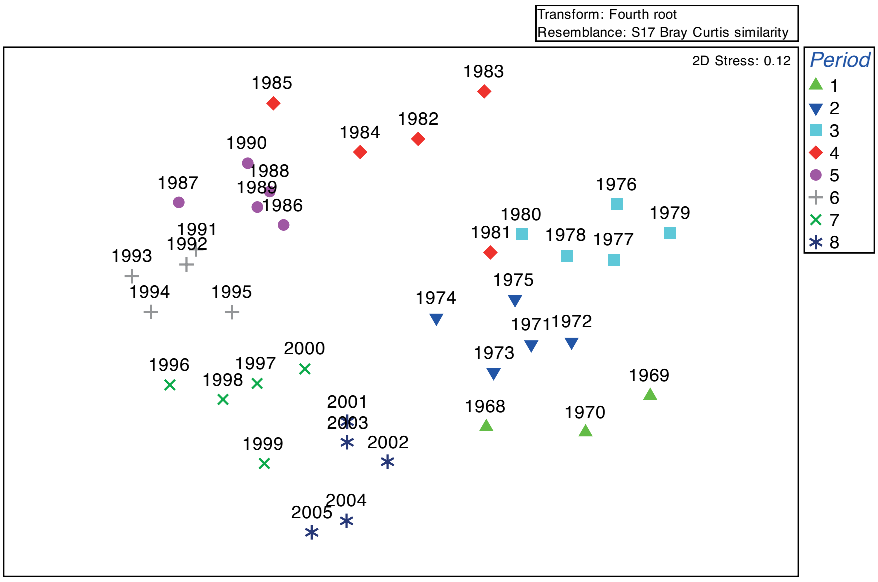


Figure 48. Results of MDS ordination of abundance data derived from bottom trawl surveys from north-eastern U.S.A.

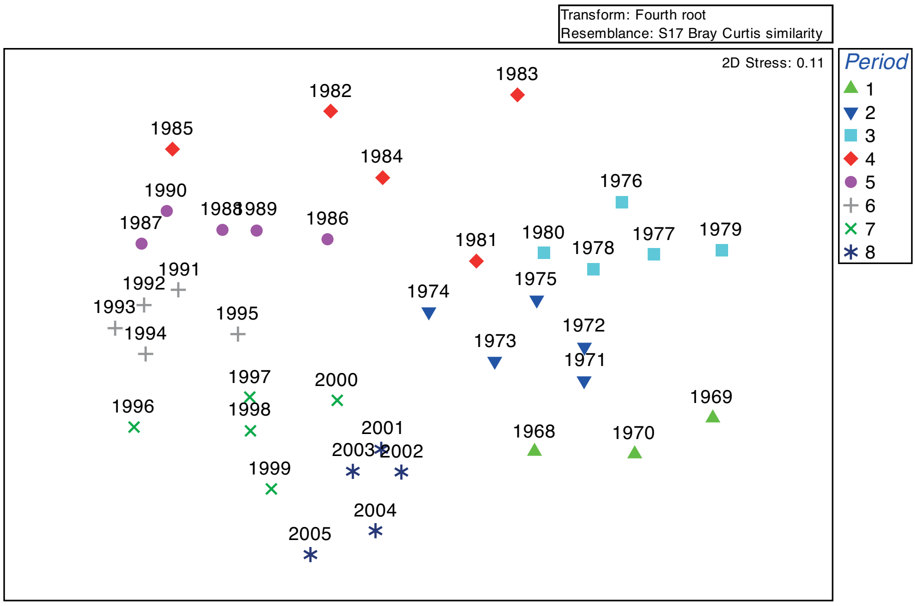


Figure 49. Results of MDS ordination of proportions data derived from bottom trawl surveys from north-eastern U.S.A.

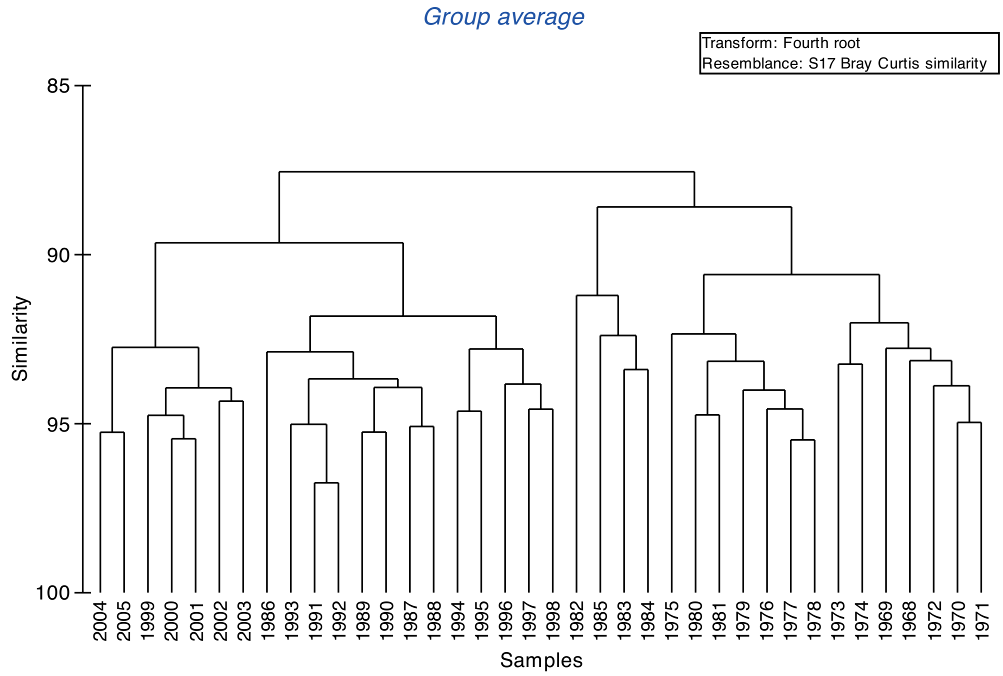


Figure 50. Results of cluster analysis for abundance data derived from bottom trawl surveys from north-eastern U.S.A.

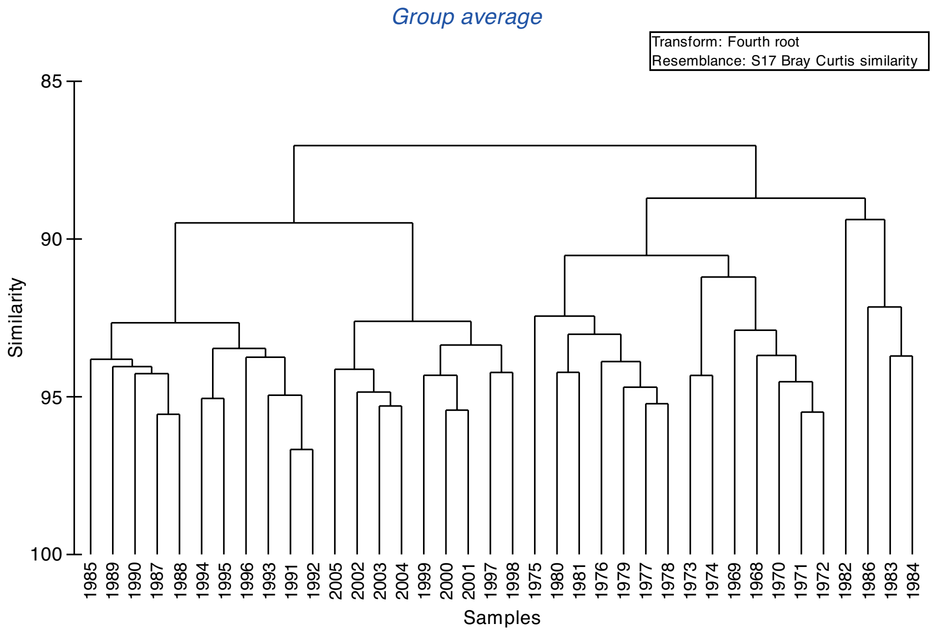


Figure 51. Results of cluster analysis for proportions data derived from bottom trawl surveys from north-eastern U.S.A.

The global R statistic calculated by ANOSIM for the abundance (0.859) and proportions (0.821) data demonstrated that the data differed significantly among periods (P < 0.001 for both). ANOSIM estimates of the R statistic for all pairwise tests of the abundance data for the different periods were also found to be significant (R = 0.364 and P = 3.6 for periods 1 and 2 (*i.e.* 1968-1970 and 1971-75), otherwise P < 0.02 or smaller). While the R statistics calculated almost all of the pairwise comparisons among periods for the proportions data were found to be significant (P < 0.02), that for periods 1 and 2 was not found to differ significantly (R = 0.292 and P > 0.05).

The results produced by SIMPER when analysing the similarity percentages for the two sets of data showed that the level of dissimilarity between the data from the late 1960s increased to a maximum in the early 1990s and then began to decline in subsequent periods. The species that most contributed to this dissimilarity for the fourth-root-transformed abundance data were spiny dogfish (49% increase from the late 1960s to the early 1990s), yellowtail flounder (55% decrease), pollock (46% decrease), haddock (35% decrease), cod (11% decrease), redfish (30% decrease), and thorny skate (34% decrease) (Fig. 52). For the fourth-root-transformed proportions data, the dissimilarity was characterised by yellowtail flounder (59% decrease), pollock (53% decrease), spiny dogfish (34% increase), redfish (36% decrease), cod (39% decrease), haddock (40% decrease), and thorny skate (42% decrease). Pollock, cod, haddock and redfish fall within the suite of species identified as "Principal Groundfish" by the NEFSC, yellowtail flounder in the suite of "Flounders" and both spiny dogfish and thorny skate in the suite of "Small Elasmobranchs".

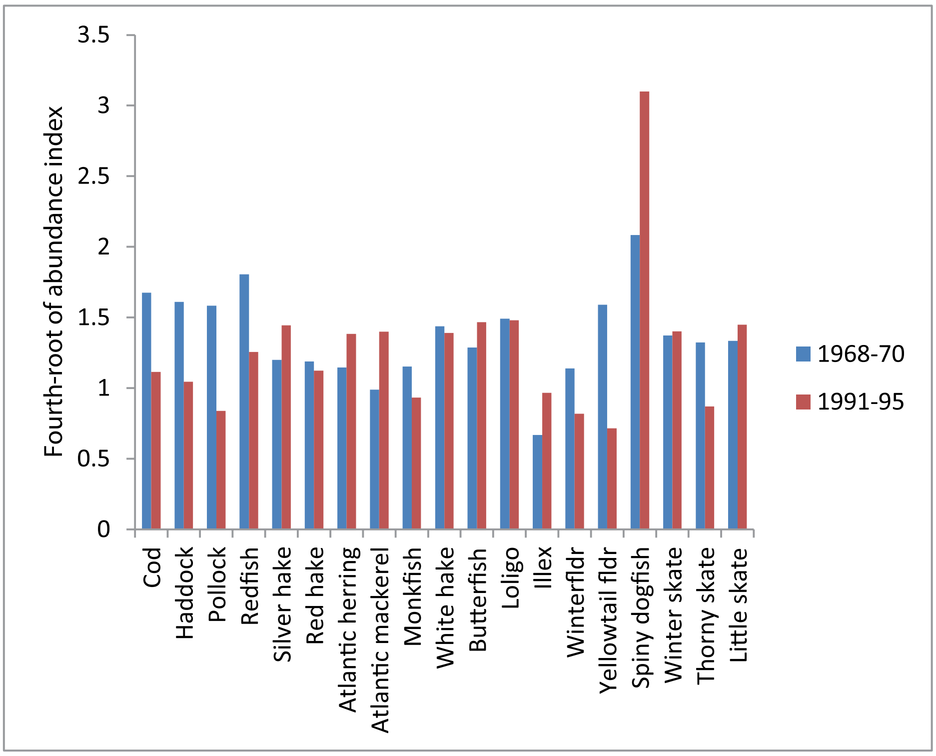


Figure 52. Averages of fourth-root of annual abundance indices in 1968-70 and 1991-95 for data derived from bottom trawl surveys from north-eastern U.S.A.

Sosebee *et al.* (2006) report that the abundances of the principal groundfish increased in the mid to late 1970s in response to management controls that had been introduced in the early 1970s, but subsequently declined to very low levels in the late 1980s and early 1990s before increasing towards levels similar to those experienced in the early 1970s (Fig. 47). The abundance of flounders decreased during the early 1970s before recovering slightly in the late 1970s-early 1980s, then declining to very low levels in the late 1980s-early 1990s, recovering slightly by 2000 but subsequently declining to a very low level in 2005. The abundance of small elasmobranchs rose in the early 1970s, declined slightly in the late 1970s, but then increased to a peak in the late 1980s-early 1990s before declining by the early 2000s to levels similar to those of the early 1970s. Catches reported over the period revealed similar marked changes in species composition with a shift towards species that had previously been considered to be less desirable (Sosebee *et al.*, 2001).

Our analysis has demonstrated that the statistical approaches that we have used to analyse the multivariate data have the ability to detect differences among the species compositions in different periods and to identify the changes in species abundance that characterise those differences. It would be appropriate, however, to use the results of our analysis of the data from the northeastern US with considerable caution as we are not fully aware of the limitations of these data or of the fisheries and biology of the stocks involved. Note also that the analyses that we undertook are intended to explore the similarities and differences in observed species compositions, but are not intended to determine or discriminate among the extents of the direct or indirect effects resulting from possibly both fishing and environmental change that led to the observed changes in species composition.

Trophic level and risk assessment

Results of a risk assessment undertaken by DoFWA indicated that, in the West Coast Bioregion, research/management priority needed to be given to Black Bream in the Swan River Estuary, Tailor, Cobbler, Dhufish, Pink Snapper, Baldchin Groper and Sandbar and Dusky Whaler Shark. In the Southern Bioregion, Marron, Black Bream and Dusky Whaler Shark were identified as high priority, while in the Northern Bioregion, Silver Cobbler, Sawfish, Sandbar Shark and Dusky Whaler Shark were highlighted. In the Gascoyne Bioregion, the high priority species were Pink Snapper, Baldchin Groper, Goldband Snapper and Rosy Jobfish.

Changes in species composition of catches

The ways in which the species compositions of the recorded catches in the West Coast Bioregion changed between 1976 and 2005 were investigated using MDS. The MDS ordination plots for the data from each of the fishing methods, *i.e.* droplining, gillnetting, handlining and longlining, revealed similar marked trends through time.

Plots of the MDS ordination for the cpue data for each region demonstrated a shift in species composition, which, through use of ANOSIM, was demonstrated to be of high statistical significance (Figs 53 to 57). Plots of the percentage change in the five-year average of the fourth root of the cpue data between 1976-80 and 2001-05 are presented in Figs 58 to 63 and reflect the differences in these values for the species identified by SIMPER as characterising the changes in species composition. Comparison of Figs 58 and 63 demonstrates the need to examine the results from analysis of the data for the different fishing methods for a complete understanding of the changes in species composition of the catches per unit of effort that have been recorded from the different regions.

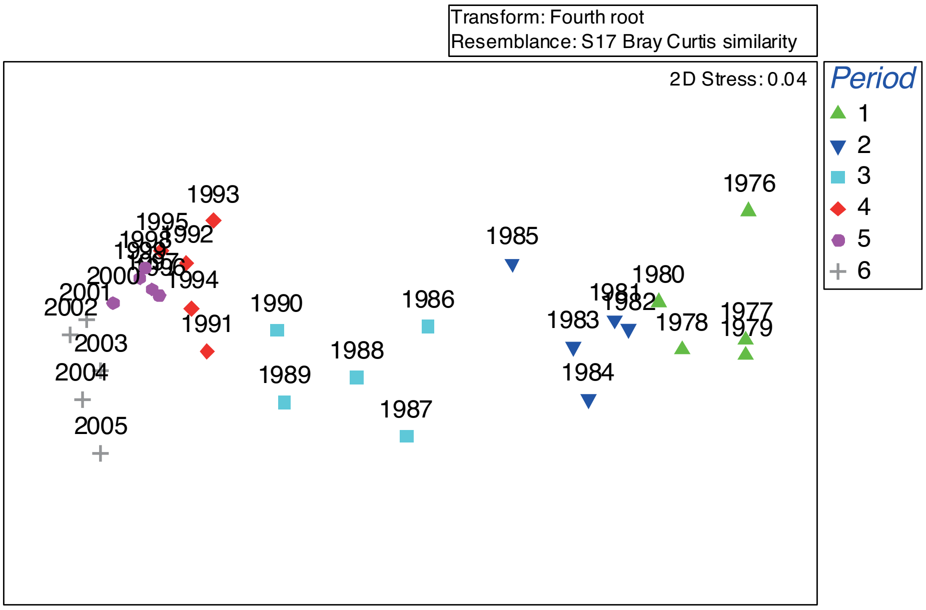


Figure 53. Results of MDS ordination of the fourth root of the cpues for handlining within the West Coast Bioregion.

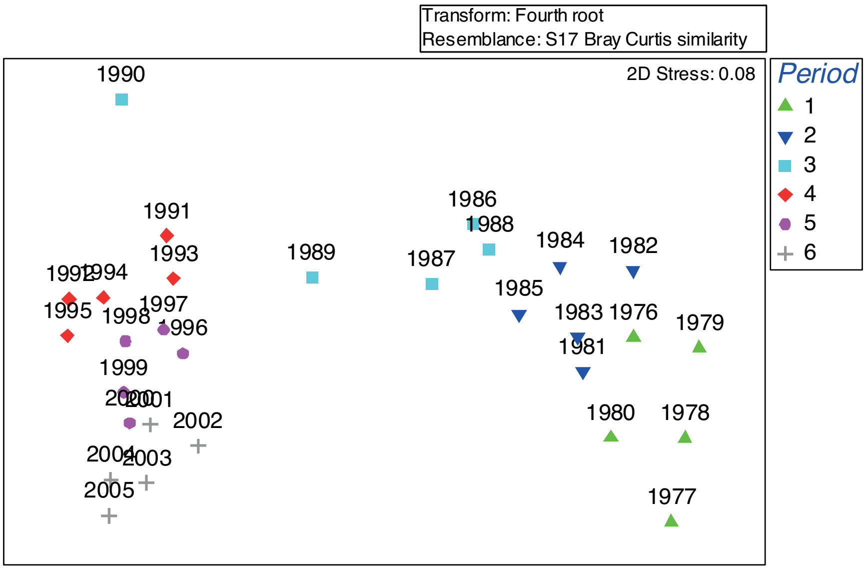


Figure 54. Results of MDS ordination of the fourth root of the cpues for handlining within the South Coast Bioregion.

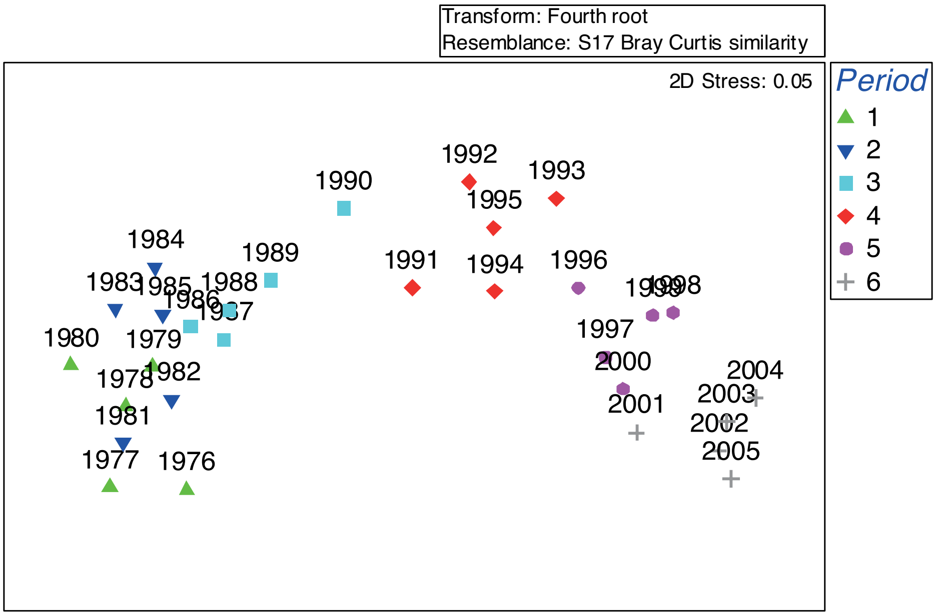


Figure 55. Results of MDS ordination of the fourth root of the cpues for handlining within the Gascoyne Region.

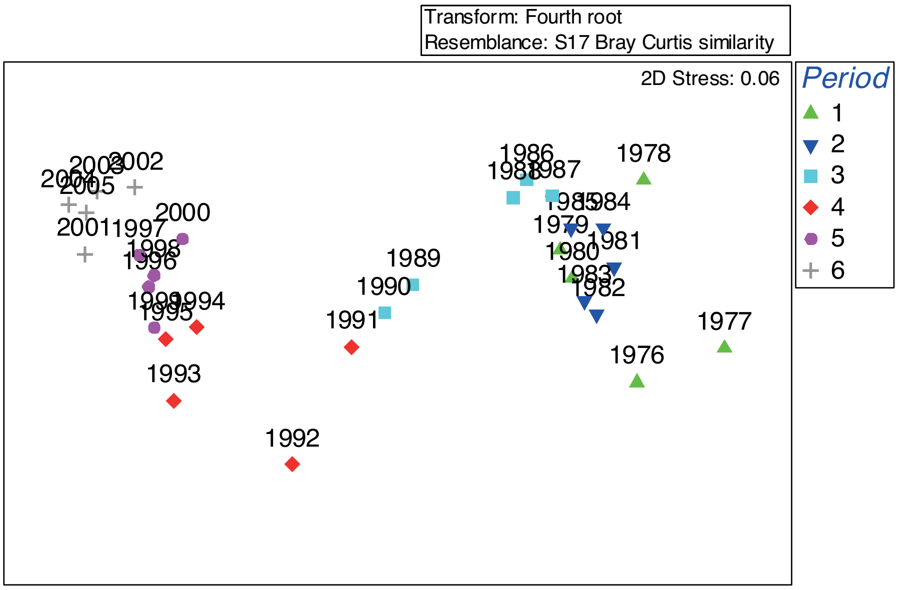


Figure 56. Results of MDS ordination of the fourth root of the cpues for handlining within the Pilbara Region.

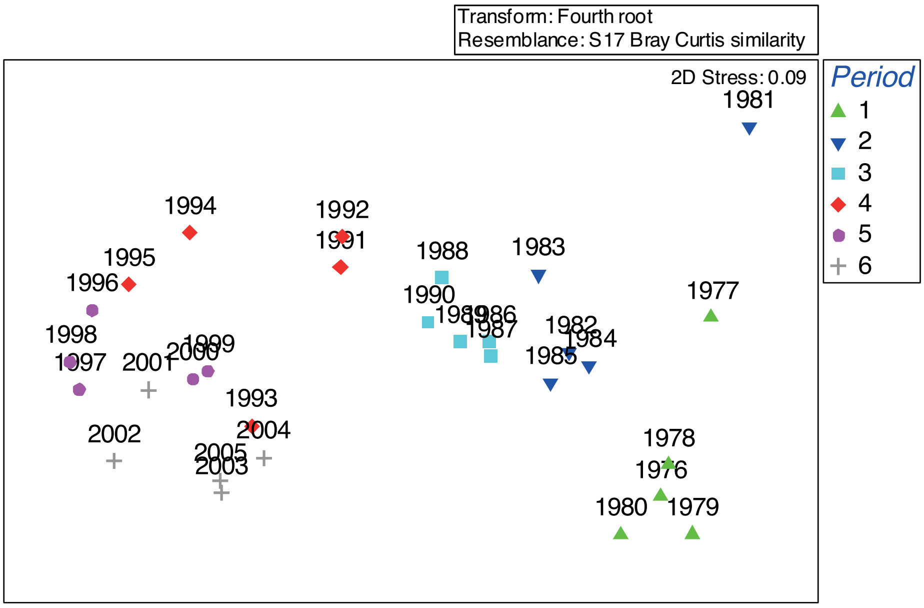


Figure 57. Results of MDS ordination of the fourth root of the cpues for gillnetting within the Kimberley Region.

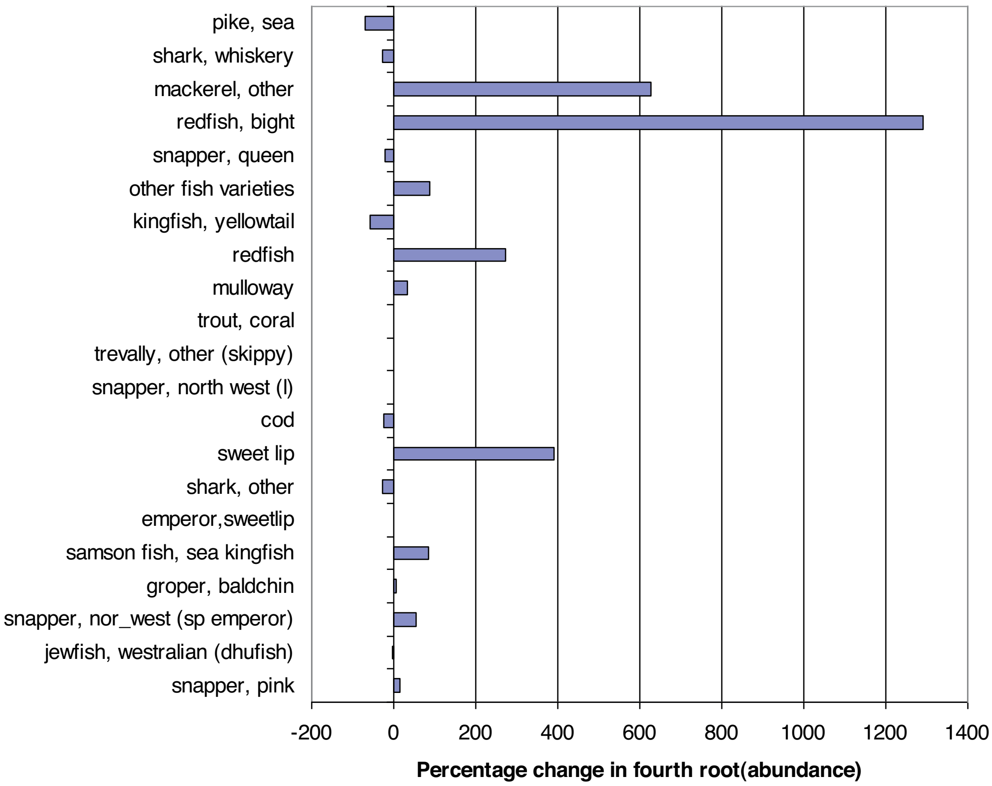


Figure 58. Percentage change in the fourth-root of handline cpues between 1976-1980 and 2001-2005 for the West Coast Bioregion.

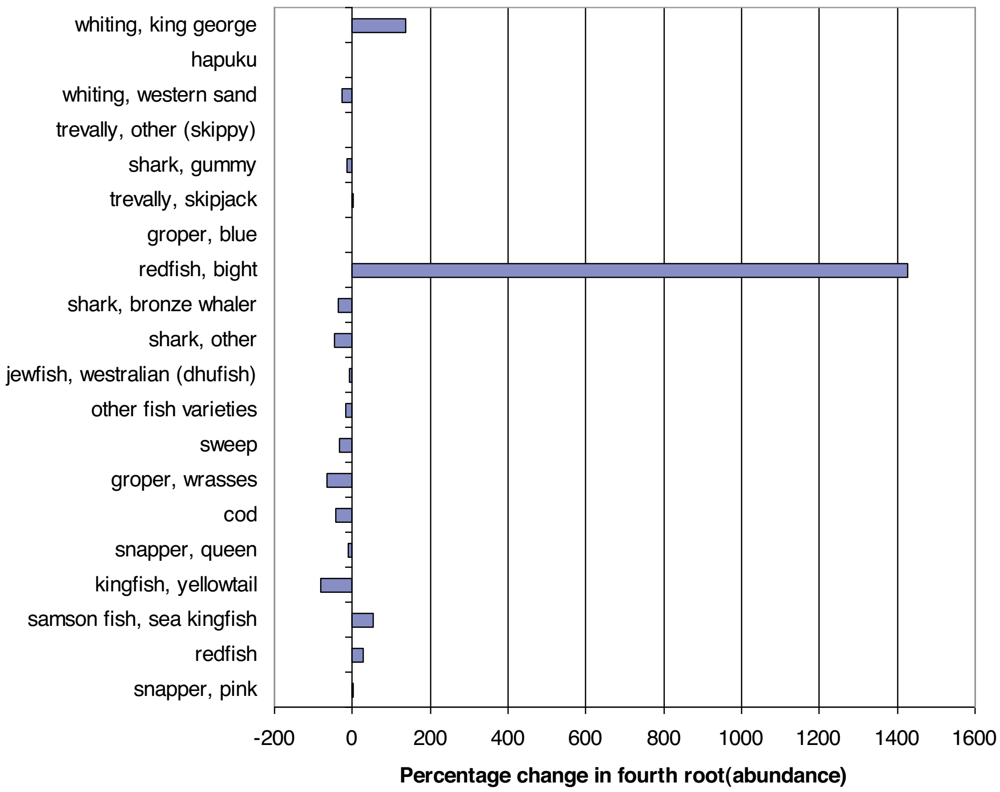


Figure 59. Percentage change in the fourth-root of handline cpues between 1976-1980 and 2001-2005 for the South Coast Bioregion.

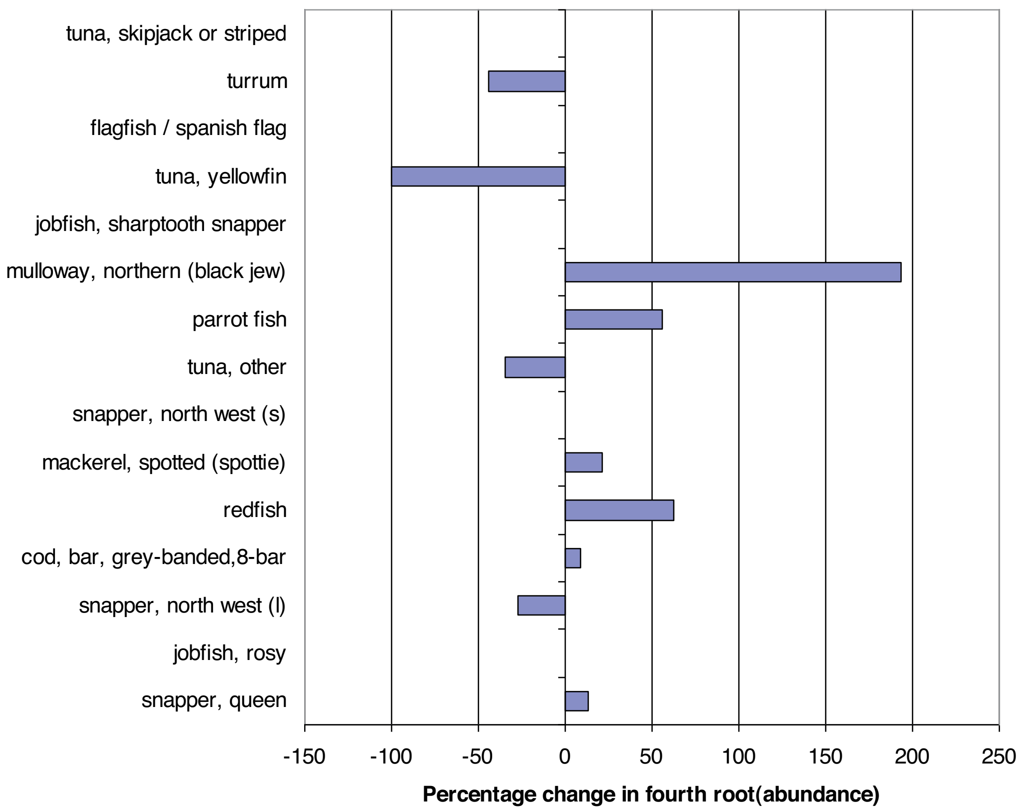


Figure 60. Percentage change in the fourth-root of handline cpues between 1976-1980 and 2001-2005 for the Gascoyne Region.

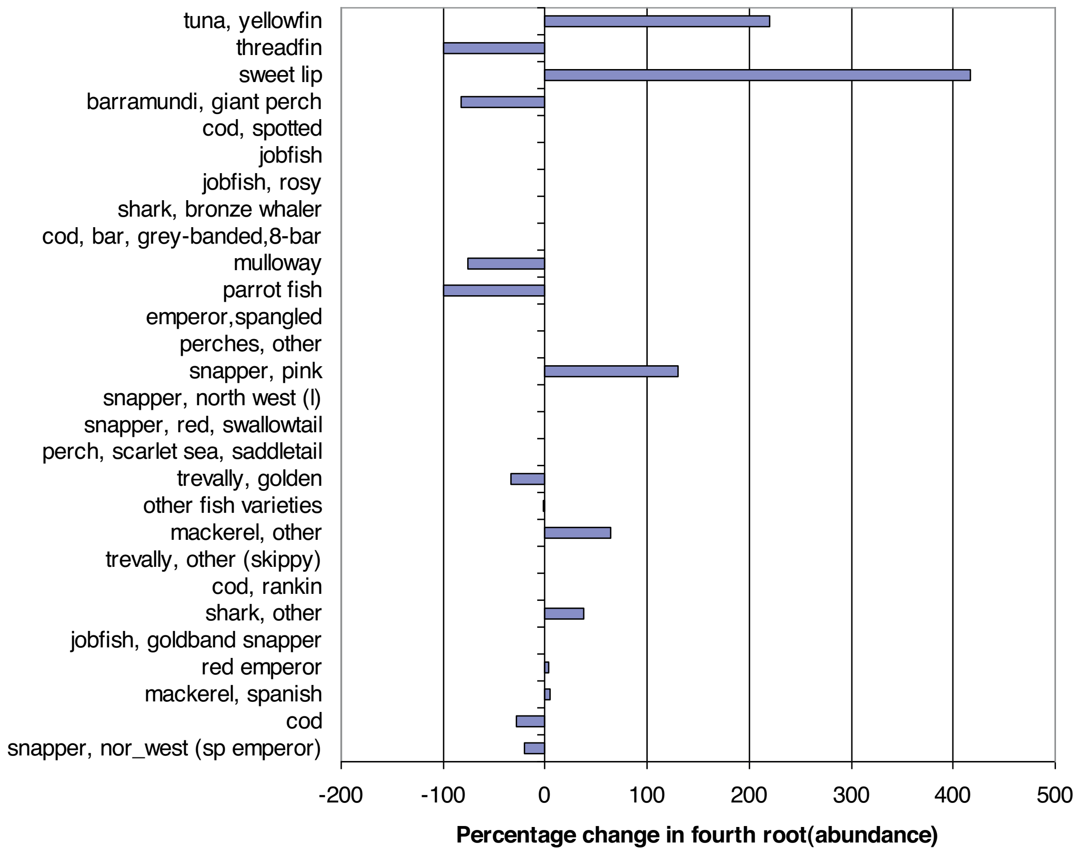


Figure 61. Percentage change in the fourth-root of handline cpues between 1976-1980 and 2001-2005 for the Pilbara Region.

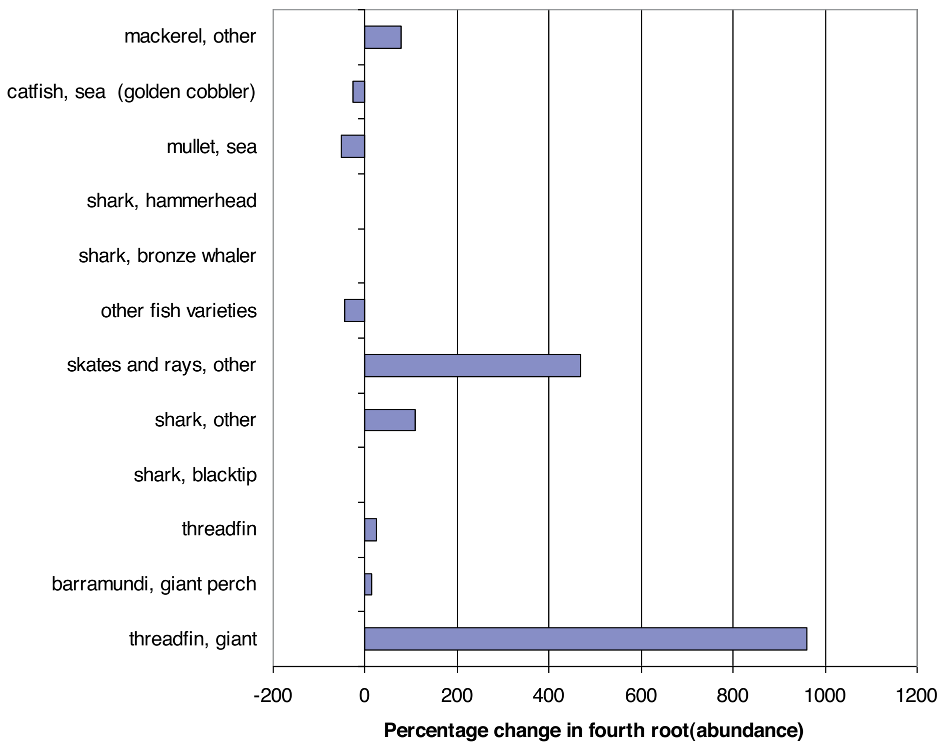


Figure 62. Percentage change in the fourth-root of gillnetting cpues between 1976-1980 and 2001-2005 for the Kimberley Region.

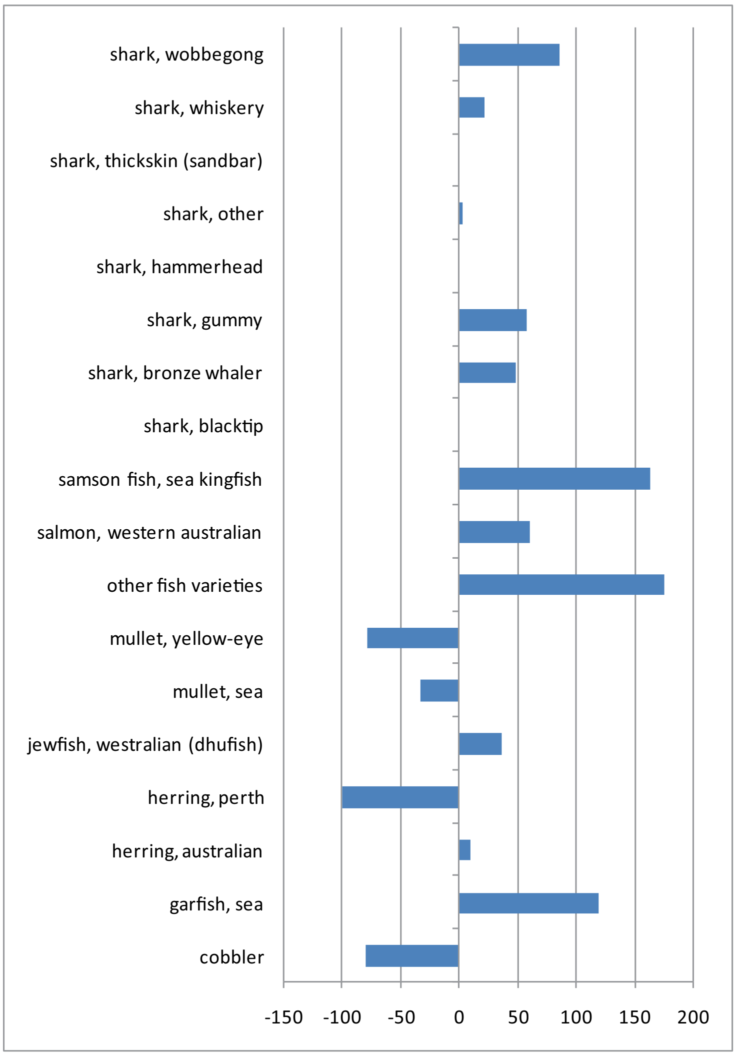


Figure 63. Percentage change in the fourth-root of gillnetting cpues between 1976-1980 and 2001-2005 for the West Coast Bioregion. Note that the cpue of Hammerhead Sharks, Blacktip Sharks and Thickskin Sharks increased from zero, or almost zero, to the values reported for 2001-2005, and are therefore not displayed.

The results of these analyses have demonstrated that the species composition of reported catches by the different fishing methods in the different regions have undergone changes that are statistically significant. However, the cause of the changes cannot be determined using this method of analysis. While changes in the structure of the ecosystems, brought about by the impacts of fishing and/or climate or other environmental change could be implicated, the changes in species composition detected in this study could also be due to changes in management, *e.g.* reported catches of Southern Bluefin Tuna declined markedly after the shift to catch quota and the "flow" of that quota towards South Australia. In other cases, the changes may be due to a change in recording practices with, for example, catches of "Redfish" being recorded as "Bight Redfish" for a period of time. The changes in the Kimberley are considered by Steve Newman to be due, at least in part, to a move towards fishing in deeper water, being reflected in increased catches of deep-water species. Thus, changes in fishing practices in response to changes in market value or the realisation of fishers of opportunities to improve their catches may be responsible for some of the observed changes in the recorded species composition of the commercial catches. Plots of the time series of catches for each of the species and for the cpues of the more frequently-caught species for each of the five bioregions have been produced, creating a 126-page document that was provided to the Department of Fisheries for consideration of approaches by which artefacts of apparent species change arising from the ways in which data are recorded might be removed and known changes in management or fishing activity that might have resulted in the observed changes in species composition can be identified.

Initial exploration of the effectiveness of DoFWA's decision rule

After a brief exploration of the value of employing a delay difference model structure for some species in the operating model, it was concluded that the age structured model that had already been developed offered greater flexibility and more readily allowed use of age and length composition data. Accordingly, development of this model option was abandoned.

When fitting the MRM to the full set of species included in the model, if the fractions of prey species to the diet of the predators were estimated, the model tended to reduce the values of these estimates to very low levels, i.e. there is insufficient information content in the catch and effort data alone to produce reliable estimates of diet fraction for these species.

The MSE runs used to explore a five-yearly assessment and management cycle, using DoFWA harvest control rules to adjust effort for the species included within the Minimum Realistic Model (MRM) to achieve target fishing mortalities show that, for the demersal fish resources off the Perth metropolitan coastline, the rules should allow recovery of the demersal fish stocks. A result of this evaluation for 100 trials of the strategy is presented for Dhufish, one of the more important species in this bioregion, in Fig 64. For this species, which was in a heavily overexploited state at the beginning of the 30 year projection period, successive assessment and application of DoFWA's decision rule to the species in the MRM at five-yearly intervals allowed recovery of the stock to an abundance in excess of that which would correspond to MSY, a result that was implicitly the management goal. Application of the decision rule tended to stabilize the fishery such that the final levels of fishing mortality were approximately equal to the target levels.

Note that these results fail to reveal the full uncertainty of the management strategy, however, as the model used to generate this result allowed only for the variability in catch per unit of effort. Variability in annual recruitment and the uncertainty associated with the fitting of the individual stock assessment models to the different species were not factored into the analysis.

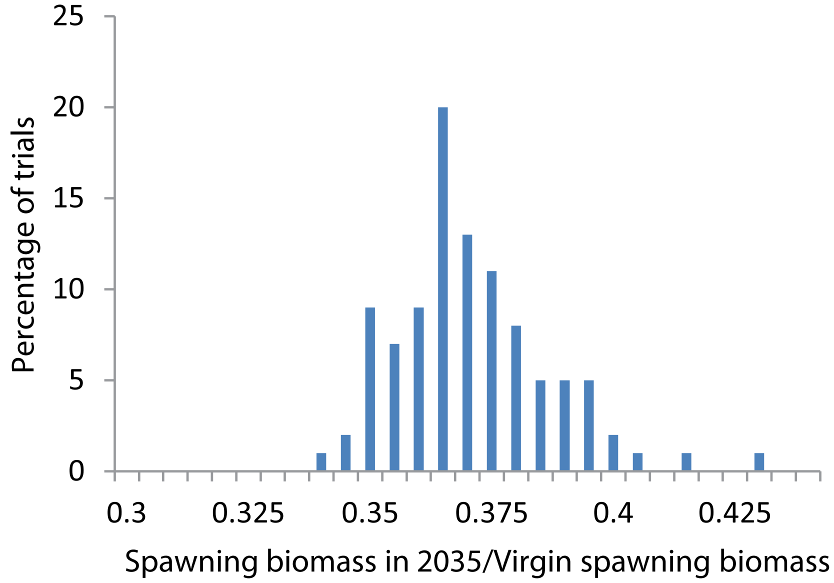


Figure 64. Relative frequency of the ratios of spawning biomass of Dhufish in the West Coast Bioregion in 2035 to the virgin spawning biomass in 100 trials of this selected management strategy (see text).

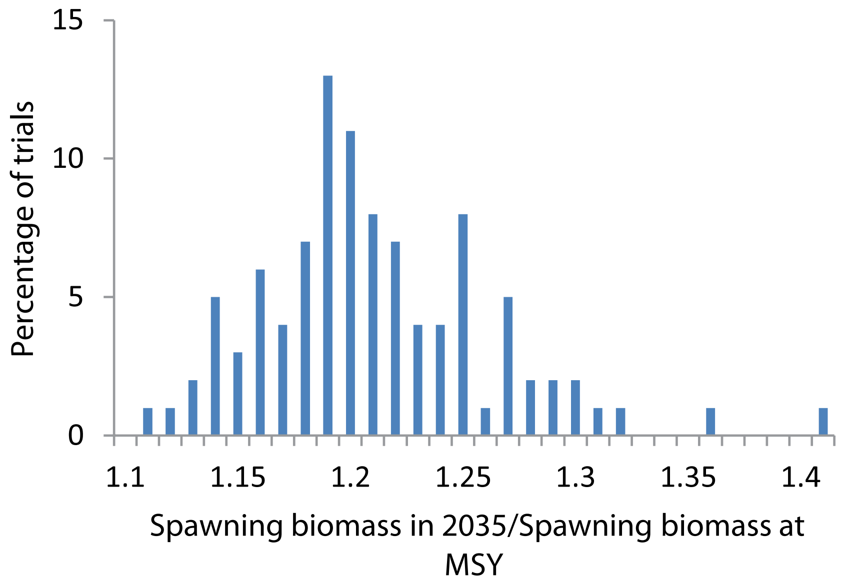


Figure 65. Relative frequency of the ratios of spawning biomass of Dhufish in the West Coast Bioregion in 2035 to the estimate of spawning biomass at Maximum Sustainable Yield in 100 trials of this selected management strategy (see text).

Selection of the indicator variables, reference points and decision rules

Note: the results presented below are drawn from the report that was produced at the conclusion of this segment of the study

Overview

The primary objective of ecosystem-based fisheries management is to maintain all stocks within the ecosystem at levels that will achieve the greatest (long term) value to the state, recognising that this requires the maintenance of ecosystem function through ensuring the sustainability of individual stocks. The associated indicators required to monitor management performance will be those relating to the abundance of, and the fishing mortality experienced by, each of the more important or vulnerable fish stocks. The information that is available from cpue data for many of Western Australia's fish stocks is inadequate to produce reliable estimates of abundance, and improved systems for collecting cpue and fishing power data need to be established. Over the next decade, until adequate reliable cpue data become available, stock assessments will need to be based on estimates of fishing mortality derived from age-or length-composition data. Costs will inevitably preclude sampling all of Western Australia's fish stocks. Nevertheless, it will be important to establish programmes for the collection of appropriate biological data for key important or fishery-sensitive species at intervals of time sufficient to ensure that assessment of the status of (at least some) stocks within the ecosystem is soundly based. Indicators that are currently available to assess stock status are restricted to those derived from age and size compositions, but it is anticipated that, at some future time, reliable estimates of cpue will become available for some species. Accordingly, the management strategies that are proposed to be assessed in this project use model-based estimates of biomass and fishing mortality (with an emphasis on the latter), and sample-based estimates of mean age and mean length as indicators, with estimates of target, threshold and limit reference points derived from the instantaneous rate of natural mortality, M (40, 30 and 20% of unfished biomass and associated fishing mortalities, and 2/3M, M, and 2M (if max. age$<$10 years) or 3/2M (if max age $\geq$ 10 years)). Decision rules that adjust fishing mortality to the target level if the target level is exceeded, or require a reduction of a range of values between 10 and 50% if the indicator lies in the region between the threshold and limit reference points, and a range of values between 50 and 100% if the indicator falls beyond the limit reference point, are proposed to be evaluated to the extent possible with the available computing resources. The effectiveness of alternative biological sampling frameworks and use of alternative decision rules for a species based on assessments for an indicator species will be explored.

Management objectives

Fishery managers endeavour to achieve the objectives set by the State, where those objectives reflect the values of the resource to stakeholders, the community and future generations. Indicator variables and the reference points for those indicator variables are used by those managers as performance measures of the extent to which the objectives have been achieved and are selected on the basis of those objectives. Invariably, there are a number of objectives to be considered, reflecting the views of different stakeholders and the different time horizons of those stakeholders. Weights that are appropriate to the current socio-economic context must be applied when balancing objectives that are inconsistent. Typically, however, the various objectives reflect different views of the appropriate level of abundance of the fish stock and the degree to which it should be subjected to direct exploitation. For ecosystem-based fisheries management, the objectives are expanded from a single species viewpoint to those that reflect the need to sustain ecosystem function, biodiversity and the environment that supports the ecosystem. Note that ecosystem models deal with simplified representations of the ecosystem and that, because of the complexity of trophic interactions, the impact of fishing on some stocks may be inadequately predicted. The need to sustain both exploited and unexploited stocks must be considered, but again the main objective is to maintain the abundance of the stocks within the ecosystem at levels that will achieve the greatest (long term) value to the State. In this case, however, as some stocks are invariably more vulnerable than others to direct and/or indirect exploitation, there is a need to recognise the intrinsic value of sustaining the abundance of all stocks at levels that will ensure the maintenance of ecosystem function, as this is crucial for ensuring the greatest long-term value of the resource.

Indicators

The primary indicators used for single-species stock assessment are typically measures or indices of the abundance of the stock, or proxies for such measures, and measures that reflect the level of direct exploitation, *i.e.* fishing mortality. Other indicators may be considered, *e.g.* the value to ecotourism that may be attributed to the presence of the stock, such as whale watching, Monkey Mia dolphins, etc.

Measures of abundance that may be used as indicators include variables determined from both fishery-dependent and fishery-independent data. The values of catch per unit of effort (cpue) calculated from the catch and effort statistics provided by both commercial and recreational fishers have often been used as measures of abundance. However, in many fisheries, the potential for bias in such estimates has led fisheries managers and scientists to implement research surveys to replace or augment the cpue data provided by fishers. The cpue or research data are typically subjected to statistical analysis to remove the influence of factors such as space, time within year, moon phase, etc., and to derive standardised indices of abundance. For a number of more valuable fisheries, it is becoming common practice to consider all of the available data in an integrated fishery model, deriving estimates of both fishing mortality and of stock size as outputs of the model. It should be noted that comparability of data collected over time is essential when evaluating the trends in the data and determining the necessary reference points against which the resulting indicators may be assessed. Reference points for the abundance of the stock may be determined from analyses using models or based on historical data. Model-based reference points often reflect the theoretical equilibrium abundance at some specified level of exploitation, *e.g.* the unexploited stock, or the stock at which maximum sustainable yield might theoretically be taken.

Indices or measures of fishing mortality are usually derived from model-based analyses of age-composition or fishery data. A typical reference point is, for example, the theoretical fishing mortality for which, when the stock is at equilibrium, the maximum sustainable yield would be taken, or the fishing mortality that would result, at equilibrium, in producing the maximum yield per recruit. In many fisheries, some multiplier of the estimate of the instantaneous rate of natural mortality is used as a reference point for fishing mortality.

As many of the model-based estimates of the absolute value of both the indicator and reference point may vary markedly when analyses are run to update the assessment for a further year of data, the values of the indicators are frequently expressed as ratios of the selected reference points, with the ratio being treated as the indicator and the value 1, or other selected value, as the reference point.

In selecting indicators for use as performance measures, the cost of collecting the necessary data at the required level of precision, and ensuring its accuracy, and the subsequent use of the indicator in decision making must be assessed and balanced against the benefit to the fishery and the State. In many low-valued fisheries, it may be more appropriate to make conservative decisions based on low-cost and more readily-available data than to apply a more refined decision rule that requires the collection of costly data for use in associated assessments.

As costs preclude the collection of adequate fishery-independent data relating to the abundance of each stock in most of Western Australia's fisheries, it is likely that indicators of stock abundance will need to continue to be based on fishery-dependent data. However, changes in fishing practice, fishing power, targeting of different species and distribution of fishing have made past data unreliable. Future data collection will need to address such issues if cpue data are to provide reliable indicators of abundance. Over the next decade and until sufficient reliable cpue data become available, it is likely that stock assessment will need to be based on estimates of fishing mortality derived from age or size compositions of the catches for selected species.

The absence of reliable indices of abundance currently precludes the use of such indices as indicators of management performance. Furthermore, accurate catch data are required to relate changes in relative abundance to absolute abundance. Reliable catch data are only available for those (few) years in which estimates of recreational catch have been produced. Wise *et al.* (2007) propose that biomass levels to be used as target, threshold and limit reference points for Western Australia's demersal finfish stocks of south-western Australia should be set at 40, 30 and 20% of the unexploited biomass. In their table relating to these reference points, they record that fishing mortality proxies for these reference points would be the values of fishing mortality that, at equilibrium, would be associated with these levels of biomass.

Biological data

Fishery assessment requires data on the biological characteristics of the species, as these data determine how fishing is likely to influence the stock. Information on growth, reproduction, and longevity is available for the more important fished species in Western Australia. However, because of cost and the low value of many species, such data are still lacking for some of the less important bycatch and discarded species. There has been no assessment of whether the biological characteristics of the various species has changed over time, although such changes have been detected in other fisheries and might be expected as a consequence of both climate change and fishing pressure. There is inadequate information to allow reliable assessment of the biomass of most species, and information on the diets of the various species is based on the results of a small number of one-off research studies (representing a source of considerable uncertainty for model predictions of indirect effects of fishing on the trophic structure of the ecosystem). While some of the more important species have received research attention, there has been limited collection in the past of age and length composition data for most species.

There is little doubt that the cost of collection of biological data will preclude the collection of age and length-composition data for many species. However, given the quality of the currently-available abundance data, fisheries managers will rely greatly on stock assessment advice derived from the information on fishing mortality that may be extracted from age- and length-composition data for at least the more important exploited species. To neglect the possibility that unexploited and bycatch species might be adversely impacted by the indirect effects of fishing would be unwise, however. A well-designed programme to collect representative samples of fish from the catches of important fish species, particularly those that are likely to be sensitive to the direct and/or indirect impact of fishing, should be funded by the

State to provide the core data necessary to determine the status of these species, using these as proxies of the state of other stocks within the ecosystem. The programme may require the collection of biological data over several years, where such sampling periods occur at regular intervals, *e.g.* three years of sampling every ten years, where appropriate periods of sampling and inter-sample intervals are determined through simulation studies.

Inter-annual variation, observation error and errors associated with parameter estimates

Inter-annual variation may be an issue for many fisheries, which if not addressed would yield erratic management advice from year to year when applying a control rule using the estimates of the indicator and reference points for each new year of data. Approaches to resolve this issue include the calculation of a moving average of the measure which is then taken as the new indicator. Other approaches to overcome variation due to observation error and the imprecision of parameter estimates include explicit calculation of the probability distribution of the values of the ratio of the indicator to its reference point, with the assessment and decision rule being based on the probability of the ratio falling on one side or other of each specified reference value.

Structural uncertainty

Uncertainty resulting from the use of alternative model assumptions or structures, alternative values of parameters (such as natural mortality) that are input to the fishery models, or alternative weights applied to different data sets, are typically considered using a decision table framework.

Integrated assessments and Weight of Evidence

Integrated assessment models employ all available data, *i.e.* fishery-dependent and research data, in an attempt to use the information contained within these data to obtain a more precise estimate of the current abundance of each fish stock. In many cases, however, the various data sets may prove to be inconsistent, either as a consequence of inappropriateness of model assumptions, *i.e.* model structure, or the non-representative nature of the samples, *i.e.* sampling bias. The uncertainty associated with use of alternative data sets may be considered through use of a decision table framework. While integrated assessment models allow a quantitative assessment of all available data, a Weight of Evidence (WoE) approach (Weed, 2005) that also considers other, non-quantifiable risks to the fishery is typically used in Australia when deciding the appropriate course of management action (*e.g.* Wise *et al*., 2007). In such assessment, the potential vulnerability of a stock to exploitation due to its biological characteristics and life history is considered subjectively, together with all other information relating to the fishery or the stock, thereby augmenting the quantitative information derived from the formal stock assessment. The WoE approach may also consider the risk introduced from a paucity of data available for use in quantitative assessment models, from lack of data for many of the stocks within the ecosystem, introduced as a consequence of delays in assessment and implementation of management measures, and through limitations on the effectiveness of management measures.

Indicator variables, reference points and decision rules for this project

The data that appear likely to become available for (at least some of) the fish stocks of the West Coast Bioregion include age and length composition data, and, in the longer term, reliable cpue data. From the age and length compositions, it would be possible to derive estimates of fishing mortality that could be compared against reference points derived from various multipliers of an estimate of natural mortality, where the latter could be estimated from life history data using either Pauly's or Hoenig's equations. Estimates of mean age and mean length could also be derived from the age and size compositions, with reference points for these variables being derived from equations using the assumption that the stock is in equilibrium under a level of fishing mortality equal to a specified multiplier of natural mortality. Later, when reliable cpue data become available, it may be possible to derive estimates of abundance and MSY-based reference points to supplement the data relating to fishing mortality.

It should be noted, however, that, with the effects of climate change becoming increasingly apparent, natural mortality of many fish stocks is likely to be affected. Basing biological reference points on mortality estimates derived from historical relationships between natural mortality and life history characteristics, as suggested in the paragraph above, may prove a risky strategy if potential changes in natural mortality are not taken into account.

The proportional change in fishing mortality that, given the value of the indicator variable, would be required to bring fishing mortality to a specified target mortality was estimated in this project. The indicator variable was compared against reference values calculated for the target, threshold and limit points using fishing mortalities equal to 2M/3, M, and 2M (if maximum age < 10 years) or 3M/2 (if maximum age is 10 years or greater) (Wise *et al*., 2007). The following alternative decision rules were applied and, in the case of multiple indicators, the most conservative decision was accepted for use.

1. If the indicator fell beyond the target reference point, fishing effort was reduced to the target level and maintained at this level till the next assessment.

2. If the indicator fell between the threshold and limit reference points, fishing mortality was reduced by a specified percentage (10, 30 or 50%) and maintained at this level till the next assessment. If the indicator fell beyond the limit reference point, fishing mortality was reduced by a specified percentage (50, 75 or 100%) and maintained at this level till the next assessment.

The availability of computing resources constrains the number of trials used when exploring the different strategies as management strategy evaluation is typically very computer-intensive. In practice, the Department of Fisheries would apply a Weight of Evidence approach and would consider socio-economic input relating to the fisheries and fish stocks before determining an appropriate management response. Such subjective assessments lie beyond the scope of the current FRDC project.

Effectiveness of alternative decision rules, indicator variables and reference points

Exploration of the effectiveness of alternative decision rules, indicator variables and reference points for the MRM produced the following results. The ratios of spawning biomass at the end of a thirty year projection to the virgin biomass for "Dhufish", which provide an indication of the performance of the different assessment approaches and decision rules, demonstrated that, as expected, the more conservative approaches yielded the largest spawning stock sizes at the end of the period. Thus, the decision rule that emulated the Department of Fisheries' rule, but which was constrained to reduce effort by only 10 or 50% produced average ratios for 100 assessments based on analyses using cpue, catch curve, mean age and mean size data produced ratios of 0.28, 0.23, 0.25 and 0.22, respectively. The decision rule based on target fishing mortality produced ratios of 0.36, 0.37, 0.41 and 0.22, respectively. It should be noted, however, that the cpue analyses assumed the availability of cpue data for the entire time series, rather than for just the projection period. It is also possible that, for another species for which increase in length is more linear, a more reliable estimate of total mortality might be estimated from the mean length, thereby producing a more conservative result. The target-reference-point-based decision rule was more conservative than the alternative rule that was tested in this study. Catch curve and mean age-based assessments of fishing mortality yielded similar results, and, at face value, appeared more likely to be reliable (provided representative samples of age composition can be collected).

The value for fisheries management of collection of additional data

Note: the results presented below are drawn from the report that was produced at the conclusion of this segment of the study

Introduction

A question posed for this project was what value there would be for ecosystem-based fisheries management in collecting alternative data to those which are currently collected for Western Australia's fisheries, what improvements might result from use of alternative indicator variables and reference points, and whether alternative decision rules might yield more effective management outcomes.

In addressing these questions, it was recognised that the options investigated would need to be compatible with the approaches that the Department of Fisheries, Western Australia, considered appropriate for management of the fisheries for which it was responsible. Without such compatibility, there would be little possibility that the data resulting from the assessment would be considered useful to the agency. Accordingly, advice was sought from Dr Brent Wise, Fisheries Department, Western Australia, as to what types of data could be collected for Western Australia's finfish fisheries, recognising the constraints on resources and the changing structures of, and management arrangements for, those fisheries. Because of its cost, fishery-independent data collection was considered not to be an option for most finfish fisheries in Western Australia,. Thus, data collection options are constrained to the collection of (improved) fishery data and (more representative) age and size composition data.

Although the Department considered the current time series of catch per unit effort (cpue) data for most finfish species to be unreliable because of the lack of sound information on changes in fishing power, there was potential for collection of improved cpue data (coupled with data on fishing efficiency), such that a new, more reliable time series could be developed in the future. Fisheries data collection was already being enhanced for some fisheries by collecting data on the number of fish in the catch, to supplement information on catch weight. Enhancement of existing biological monitoring programmes was also being considered or undertaken to provide data on both age and size composition of catches.

Advice was also sought from Dr Wise as to the decision rules that would be acceptable to the Department of Fisheries. The decision framework was unlikely to change, with data derived from stock assessment being considered in a broader weight-of-evidence context. Thus, fishing mortality (F)-based indicators are likely to continue to be compared against reference points calculated from estimates of natural mortality (M). If those estimates of fishing mortality exceed the values of the associated reference points, recommendations regarding the management response will continue to be formulated using a weight-of-evidence approach and taking into account the risk of overfishing for the particular species involved, as a consequence of the characteristics of its life history (*i.e.*, in accordance with the approach described by Wise *et al*., 2007).

Guided by this advice, the questions addressed in the project became focused on the effectiveness of management decisions based upon indicators of fishing mortality derived from trends in cpue, and age or size compositions of catches. These indicators were to be determined using data from newly-initiated data collection programs, *i.e.* with data only becoming available for use in stock assessment from the most recent years. The set of natural-mortality-based threshold and limit reference points for fishing mortality established and used by the Department of Fisheries for the demersal finfish stocks in the West Coast Bioregion (Wise *et al*., 2007), i.e. $F_{\text{target}} = 2/3\ M$, $F_{\text{threshold}} = M$, and $F_{\text{limit}} = 3/2\ M$ for species with maximum ages ≥ 10 years or = 2 M for fish with lesser life spans, would continue to be applied. Assessment was required, however, of the effectiveness of the different decision rules applied within the weight-of-evidence approach, *i.e.* the effectiveness of effort reductions of between 10 and 50%, depending on the assessed risk of overfishing for the species, if $F_{\text{threshold}}<F<F_{\text{limit}}$, and between 50 and 100% if $F>F_{\text{limit}}$.

Accordingly, the specific objectives of this element of the project were to assess, for an ecosystem in which the fish stocks were considerably over-exploited and where fishery assessments are conducted every five years (a period likely to correspond to the interval between successive future assessments for most of Western Australia's finfish fisheries), the effectiveness of (1) indicators based on cpue, and age and size composition, using only data collected from a newly-initiated data collection programme. (2) fishing mortality reductions of 10, 30 and 50% when $F_{\text{threshold}}<F<F_{\text{limit}}$, in combination with reductions of 50, 70 and 90% when $F>F_{\text{limit}}$, respectively. (3) fishing mortality reduction to the target fishing mortality when fishing mortality exceeds the target.

For this study, the time period between assessments was set to five years, and catch curve, mean age and mean size assessments were based on random samples of 500 fish collected from the (simulated) catches of the previous year. Other time periods and sample sizes could readily be explored.

Currently, the model uses the calculated value of F_t^j as the indicator variable. This is compared against the target level and both the threshold and limit reference points. When exploring the effectiveness of indicators based on cpue, and age and size composition, using only data collected from a newly-initiated data collection programme, *i.e.* specific objective 1 of this study, fishing mortality was reduced by 10% if $F_{\text{threshold}}<F<F_{\text{limit}}$ or by 50% if $F>F_{\text{limit}}$, but otherwise was left unchanged. When considering the second objective of the study, to assess the effectiveness of three alternative combinations of threshold and limit reference points, fishing mortality was reduced by 10, 30 and 50% when $F_{\text{threshold}}<F<F_{\text{limit}}$, and by 50, 70 and 90% when $F>F_{\text{limit}}$, respectively. For the third objective, fishing mortality was reduced to the target fishing mortality when it was found to be in excess of this target.

Further details of this aspect of the study

The analyses undertaken for this study applied each assessment approach separately in order to assess its performance. However, to explore a further assessment option, a further set of trials was conducted using both age composition and mean age data, *i.e.* both catch curve and estimation of F from the mean age, as assessment methods, with the decision rule being applied to each result and the most conservative of the resultant management recommendations being accepted for use in managing the "fishery". This was possible as the assessment component of the MSE allows the calculation of fishing mortality estimates using a mixture of cpue, catch curve, mean age and mean size analysis, and use of the resulting estimates in the decision rule, with the most conservative of the proposed management actions then being implemented.

The effectiveness of each assessment approach or decision rule was assessed by setting the initial equilibrium fishing mortality for each stock to 1.5 times the natural mortality thereby simulating a highly overfished state for each stock, and by running 100 simulation trials for each of the approaches (*i.e.* type of data used in estimating *F*, or decision rule), projecting the data for a thirty year period during which, at five-yearly intervals, a new assessment was undertaken and, if required, a new level of effort was set. The proportion of these 100 random trials in which the "true" fishing mortality fell below the threshold reference point at the end of the projection period was used as a measure of the effectiveness of the various assessment/decision rule approaches.

To simplify the interpretation of results, recruitment variability for each species was initially set to zero when considering the effectiveness of using estimates of fishing mortality derived from cpue, age composition, mean age and mean length (*i.e.* objective 1), *i.e.* when using the age-structured assessment model, catch curve analysis, and estimation of total mortality, and hence *F*, from mean age and mean length, respectively.. Thus, for this objective, variability was confined to that generated in the (synthetic) samples of observed data for cpue, age compositions and length compositions. Subsequently, for the second and third objectives of the study, recruitment variability (*i.e.* the standard deviation of the natural logarithms of the recruitment deviations) was set to 0.3, the low end of the range of recruitment variability reported for fish by Mertz and Myers (1996). Although the recruitment of WA's fish stocks varies, the extent of inter-annual variation has yet to be quantified. The results from the first set of trials were used to select the most effective of the assessment approaches and this was applied when exploring the second and third objectives.

To increase discrimination between the measures of effectiveness for the different decision approaches, the analyses for objectives 2 and 3 were also run using a 10-year projection period.

Results

Use of the current assessment approach based only on mean length and employing the Beverton and Holt method of estimating total mortality proved highly ineffective (Table 1). Further investigation demonstrated that the estimate of total mortality becomes increasingly underestimated as the standard deviation of lengths at age is increased, This bias resulted in failure of the estimate of fishing mortality to trigger reduction of fishing mortality and hence low effectiveness of this assessment approach. A variant of the Beverton and Holt method was developed, in which estimates of mortality were calculated using the lower bound of each length class as the length at first capture, and selecting the maximum of the mortality estimates as the estimate of total mortality from which the value of fishing mortality employed in the decision rules was calculated. This approach produced similar under-estimates of total mortality when applied to length-composition data generated with a distribution of lengths at age, with bias increasing when the standard deviation of the distribution of lengths at age was increased. Alternative, more refined methods of assessment of length composition data should be developed, including approaches that employ both length and occasional age composition data, as these would prove valuable for fisheries for which the length composition of the fish can be monitored but resource limitations constrain other data collection

The results of the simulations employing estimates of *F* derived from analysis of cpue data using an age-structured model demonstrated that, in general, assessments employing this approach were less effective than those that employed catch curve analyses of age composition data or estimates of F derived from the mean age of fish in the catch (Table 1). Use of a combination of these last two approaches, employing the larger estimate of F in the decision rule, proved the most effective strategy, resulting in the "true" fishing mortality for each species at the end of the projection period lying below the threshold limit in over 98% of the simulation trials.

Table 1. Percentage of trials in which "true" F was below the threshold level at the end of the thirty year projection period when estimation of F was based on cpue data, age composition data, mean age and mean length. Note: 100 is good, 0 is bad.

	Lobster	Dusky Whaler	Snapper	Dhufish	Blue Groper
Cpues	90	87	74	73	82
Age composition	93	100	100	100	100
Mean age	82	100	0	100	100
Mean length	0	6	80	0	0
Age composition + mean age	98	100	100	100	100

Exploration of the effectiveness of the alternative management responses, i.e. different levels of reduction of fishing mortality, in cases for which the estimate of fishing mortality exceeded the threshold and/or limit reference points (objective 2), or cases in which this mortality exceeded the target fishing mortality (objective 3) was hampered by the lack of discrimination among the different approaches when assessed over a thirty-year projection period, *i.e.* all strategies appeared almost equally effective (Table 2). Greater resolution, particularly for the smaller reductions in F, was obtained by running the simulations over a ten-year projection period (Table 3).

Although exhibiting some noise in the results obtained for lobsters when employing reductions when $F_{\text{threshold}}<F<F_{\text{limit}}$, and $F>F_{\text{limit}}$ of 10 and 50%, respectively, and of 30 and 70%, respectively, the results of the simulations demonstrated that greater reduction in fishing mortality was more effective in ensuring that the "true" fishing mortality at the end of the projection period did not exceed the threshold limit (Table 3). However, all three of the effort reduction strategies were highly effective in producing this outcome. The strategy of reducing fishing mortality to the target fishing mortality when the estimate of F exceeds the target fishing mortality was equally as effective as the strategy that reduced fishing mortality by 50 and 90% when the threshold and limit reference points were exceeded.

Table 2. Percentage of trials in which "true" F was below the threshold level at the end of the thirty year projection period when estimation of F was based on a combination of age composition and mean age data using decision rules where fishing mortality was reduced when $F_{threshold}<F<F_{limit}$, and $F>F_{limit}$ by 10 and 50%, 30 and 70%, and 50 and 90%, respectively, or to the estimated target fishing mortality. Note: 100 is good, 0 is bad.

	Lobster	Dusky Whaler	Snapper	Dhufish	Blue Groper
10 and 50%	97	100	100	100	100
30 and 70%	99	100	100	100	100
50 and 90%	100	100	100	100	100
Target F	100	100	100	100	100

Table 3. Percentage of trials in which "true" F was below the threshold level at the end of the ten year projection period when estimation of F was based on a combination of age composition and mean age data using decision rules where fishing mortality was reduced when $F_{threshold}<F<F_{limit}$, and $F>F_{limit}$ by 10 and 50%, 30 and 70%, and 50 and 90%, respectively, or to the estimated target fishing mortality. Note: 100 is good, 0 is bad.

	Lobster	Dusky Whaler	Snapper	Dhufish	Blue Groper
10 and 50%	88	100	55	98	58
30 and 70%	77	100	100	100	100
50 and 90%	100	100	100	100	100
Target F	100	100	100	100	100

Discussion

The results of the simulations using the MRM within the Management Strategy Evaluation framework have demonstrated that the three management responses proposed by the Department of Fisheries for use in cases when the estimate of fishing mortality exceeds the threshold and limit reference points for that variable (*i.e.* reduction of effort by amounts ranging from 10 to 50% and 50 to 100%, respectively) are likely to be highly effective in reducing fishing mortality to a level below the threshold level over a thirty-year projection period. The greater the reduction in effort, the more effective the approach will be and the greater the probability that fishing mortality will be reduced to a level below the threshold fishing mortality in a shorter projection period of ten years. For the fishery, however, the increase in effectiveness achieved by greater reduction in effort would be accompanied by increased cost through loss of opportunity to fish and greater reduction in catch. The MSE reveals however that, although effort may be constrained, fish stocks will take time to recover and, in the case of species such as Dusky Whaler, because of the late age at which females mature, response to effort reduction may be considerably delayed.

The results of the analysis of the effectiveness of obtaining an estimate of fishing mortality by analysing cpue using an age-structured model were somewhat surprising. It had been anticipated that the results of analysis of the very short time series of new fisheries data collected only from recently-initiated data collection programmes and without contrast in fishing mortality would reveal far-less effective outcomes than those that were recorded. Initial values of the parameter estimates used in the assessments were randomly-selected from a broad range of feasible values, and thus it is unlikely that these results are an artifact of the assessment method. Further investigation is being undertaken to ascertain why the approach was as successful as shown in the results, albeit less effective than catch curve or mean age approaches, While there may be value in employing mean length data in stock assessment, the bias of the Beverton and Holt (1956) method when applied to length data that reflect variation in length at age, rather than those which arise in theory from deterministic length at age, suggests that such an approach will be ineffective when coupled with current threshold and limit reference points. A change in the decision rule would be required to ensure that an appropriate management response is made when a biased estimate of F is encountered that corresponds to a "true" F that exceeds the reference points. That is, an appropriate "correction factor" would be required to adjust the biased estimate of F to a more accurate estimate. Alternatively, methods that take the variation in length at age into account when estimating fishing mortality by analysing the length-composition data need to be employed.

Use of a strategy that adjusts fishing mortality to the target level when the estimate of fishing mortality exceeds the target reference point would be equally effective as a strategy that reduces fishing mortality by 50 and 90% when $F_{\text{threshold}}<F<F_{\text{limit}}$, and $F>F_{\text{limit}}$. The most effective of the assessment approaches currently incorporated in the Management Strategy Evaluation program is one that employs a combination of assessment approaches, using the largest of the resulting estimates of fishing mortality in the decision rule to determine the appropriate management response.

Effectiveness given increasing exploitation and climate change

Note: the results presented below are drawn from the report that was produced at the conclusion of this segment of the study

Overview

There is little doubt that climate change, i.e. long-term environmental change, will have an impact on fish stocks, and that stock assessment strategies and management plans need to accommodate the resulting changes in the population dynamics of those stocks (Hobday *et al.*, 2008). An assessment of whether it would be possible to detect such impacts on Western Australia's fish stocks and of the effectiveness of management strategies to sustain those stocks if impacted by climate change was a task included in this project. This document reports findings of that assessment. In summary, it will be possible to determine climate change impacts on growth, maturity-length relationships, and diets using traditional studies of fish population biology, and to consider changes in predation due to changing abundances of predators in multi-species assessment models. However, it is unlikely that changes in natural mortality due to factors other than predation could be detected or assessed. If data are sufficient to facilitate development of age-structured models in which recruitment deviations from a (constant) stock-recruitment relationship can be estimated, the effect of climate change on the stock-recruitment relationships might best be assessed by calculating trends in a moving average of those deviations. Management strategy approaches that have been developed for Western Australia's finfish stocks in the "pre-climate change era" are likely to be robust and equally successful in maintaining those stocks in the face of climate change.

Assessing the impacts of climate change

The effects of climate change on Western Australia's fish stocks are likely to vary among stocks and regions. Changes in the environment are likely to affect both productivity and species distribution. For an individual fish, fecundity at length is likely to be unaffected but changes in growth, size and age at maturity, and natural mortality are likely to occur (Pauly, 1980; Mangel, 2003; Griffiths and Harrod, 2007; Thresher *et al.*, 2007). As with juveniles and adults of the species, survival and growth of larvae are likely to be affected, with implications for the stock-recruitment relationships for the various species (Rijnsdorp *et al*., 2009). While dietary compositions are expected to change in response to the changes in species composition, distributions and availability of the fish within the ecosystem, the potential exists for a fundamental change in the types of species potentially predated as a result of increasing CO_2 levels and reduction of olefactory cues (Dixson *et al*., 2010).

Stock assessment models are based on available biological data coupled with a set of assumptions. One assumption that is typically adopted in those models is that growth and maturity relationships remain constant. The reality is that biological processes are subject to change with their environment and there is growing acceptance among fishery scientists that, for some fish stocks, evolutionary change in biological parameters may also have been induced by fishing. For example, it is common in U.S. fishery assessments to consider variation in annual mean weight at age. In addition, there are now numerous reports that length and age at maturity have changed over time. Although not currently monitored, changes in the age and size compositions, growth, weight-length relationship, and the relationship between proportion mature and fish length may be assessed relatively easily using traditional population biology studies. Dietary studies could prove valuable to assess the implications of climate change on interactions among species, and provide insight into the changes in population biomass of prey species that are not caught by fishers.

Changes in natural mortality may arise from change in either predation or the component of natural mortality of individuals that is unrelated to predation. Changes associated with predation may be taken into account through the changing abundance of predators, but change in the level of natural mortality unassociated with predation cannot easily be assessed. The mortality that is reflected in the age composition of the fish within the stock is the sum of mortalities due to fishing, predation and other mortality. In traditional single-species stock assessment, estimates of natural mortality are typically estimated using Hoenig's (1983) equation relating maximum age to mortality for lightly fished stocks. It is doubtful that the maximum age for Western Australia's fully-exploited stocks would provide a reliable estimate of natural mortality. Nevertheless, tagging studies and/or analysis of future age composition data from large marine reserves might provide some information on a new level of natural mortality. However, given the types of data currently or likely to be collected from Western Australia's fisheries, estimation of the effect of climate change on the component of natural mortality unassociated with predation will be extremely difficult.

While methods are available for exploring the effect of environmental variability on the stock-recruitment relationship (e.g. Iles and Beverton, 1998), there is considerable difficulty in determining the parameters of that relationship, even in a single-species stock assessment model without considering the potential for change. The data relating to the stock-recruitment relationship that are collected from a fishery are typically very noisy. Thus, the ability to evaluate any changes in the stock-recruitment relationships of the individual stocks will be very limited. If data are sufficient and it is possible to develop an age-structured assessment model for a fish stock, it may be possible to calculate the deviations in recruitment from the values predicted using a (constant) stock-recruitment relationship and assuming a normal distribution of those deviations. Trends in a moving average of those recruitment deviations may allow detection of a climate change effect. Lack of contrast in the available fishery data and the paucity of reliable catch-per-unit-of-effort data currently preclude the use of such an approach for many of Western Australia's finfish fisheries. However, there appears a high probability that environmental change will lead to change in the stock-recruitment relationships for many of WA's fish stocks (e.g. Dippner, 1997; Brander, 2005).

After concluding the above assessment, and considering the quality of the cpue data available for Western Australia's finfish stocks, there appeared little value in attempting to detect a climate change effect in simulated data. The more immediate concern was to determine whether the current management strategies adopted by the Department of Fisheries are likely to be sufficiently robust to sustain the stocks when the effects of climate change are felt by WA's fish stocks.

Effectiveness of current management strategies when climate change impacts WA's fish stocks

The analyses considered in this project task included not only an evaluation of the effectiveness of the current management strategies when the fish stocks were influenced by climate change, but also when those stocks experienced increasing levels of exploitation.

The results from these trials demonstrated that, for all three assessment methods, the management strategy was equally (and highly) effective whether or not exploitation rate was increasing or climate change was affecting the growth curve or the carrying capacity of the stock. Thus, for Dhufish, at the end of the 30 year projection period, the ratio of the resulting fishing mortality to the threshold level when assessed using CPUE data, catch curve, and mean age was 0.35, 0.67, and 0.67, respectively, when no exploitation or climate change was considered, and 0.25, 0.90, and 0.90, respectively, when exploitation rate was increasing or climate change was affecting growth or carrying capacity. Note that, when effort was adjusted, the level of exploitation was reset to the value required by the management strategy and exploitation then began to increase again for those runs where exploitation rate increased and climate change affected the stock.

Conclusions from explorations using the MSE framework

The ability to determine the status of key fished and non-fished stocks within an ecosystem requires that time series of total catches and reliable relative abundance indices, reflecting contrasting levels of exploitation or other forcing functions, representative samples of age, length and sex composition from the catches of each fishing sector and from the stock, and data from studies of population biology, i.e. growth, maturity, sex change, reproduction for each of those species, are available for analysis.

Western Australia's finfish fisheries are typically multi-species, multi-gear, multi-sector fisheries. Fishery-dependent data for such fisheries are influenced markedly by factors such as market price, the relative abundances of the different stocks, changes in spatial and temporal distribution of fish and fishers, competition among different sectors, and developments in technology and fishing efficiency. Although the analyses reported by Wise *et al.* (2007) determined the effect on cpues of changes in fishing power of the commercial fleet and the cpue data from the commercial fishery may be adjusted for this factor, there is no simple method of assessing the impacts of other factors on observed cpue and the resulting series of data is still considered to be too unreliable to be used for stock assessment. While samples from the catches of a species are essential to understand the age and length composition of removals from the stock, they are only representative of the population if catches made by fishers are random samples from that population. While considered to provide a better indicator of population status than cpue data, they are still influenced markedly by the spatial and temporal distribution of the fishery relative to that of the fish stock and by the selectivity of the fishing gear.

The MRM that was developed for the the West Coast Bioregion employed sound data on the population biology of the species, the available estimates of exploitation, limited dietary data and very limited data for catches from the recreational fishery. However, the only time series of abundance data available for creating the operating model of the MRM was the fishery-dependent cpue data, which, as noted above, are considered to be of limited reliability. The MRM should therefore be considered to represent a multi-species fishery with characteristics similar to that of the modeled fishery, but should not be considered a sound representation of that fishery. Nevertheless, conclusions regarding the effectiveness of alternative harvest strategies derived from analyses based on the MSE are likely to be sound.

Stock assessment of a multi-species fishery with predator-prey interactions demands greater accuracy of observed data than single species models. Yet single species models ignore the uncertainty that results from the interactions among species that are present within the ecosystem. The issues that affect the quality of fishery dependent data as a source of accurate information regarding the state of the various fish stocks can, in reality, only be addressed by introducing fishery-independent, research studies that collect the key abundance, age composition and biological data required for stock assessment and fishery management through a consistent and well-designed sampling programme. While such fishery-independent studies are expensive, their long term value for providing reliable information on the state of the fisheries cannot be overstated. Fishery-dependent studies simply cannot provide data of equivalent value and stock assessments that fail to complement fishery-dependent data with appropriate fishery-independent data will inevitably be of limited reliability.

The potential benefits of collecting additional types of data were considered. Such data need to be compatible with the stock assessment methods and management strategies of DoFWA to be most effective. The indicators that were found to be most useful in informing the stock assessment and enhancing the effectiveness of the management strategy were the estimates of fishing mortality derived from abundance and age composition data. Estimates of spawning biomass and reference points based on maximum sustainable yield or virgin biomass rely on the accuracy of the time series of relative abundance. Clearly, such abundance data would need to be derived from fishery-independent rather than fishery-dependent data. The collection of appropriate age composition and associated biological data for key fish species, which would also improve the stock assessment and effectiveness of management strategies, would also be enhanced if collected using a fishery-independent data collection program.

Reference points for Western Australia’s finfish fisheries are appropriately based on fishing mortality estimates derived from age composition and cpue data. Results from exploration using the MSE demonstrated that the decision rules that have been adopted for the demersal fisheries of the West Coast Bioregion were likely to be highly effective. The response of each stock was determined by the biological characteristics of the species and thus the time to recover from an overexploited state depended on the species.

The effects of increasing exploitation and climate change on the ecosystem were considered. While increasing exploitation would directly reduce the biomass of the spawning stock of the targeted species, traditional stock assessment is able to detect the changes in age composition and cpue data provided adequate and appropriate data are collected. The flow-on indirect effects of increased exploitation of targeted species would be expected to affect the abundance and mortality of other stocks. Again, if appropriate abundance and age composition data are collected for those stocks, and dietary data are available for both targeted and non-targeted species, multi-species MRM models may be used to assess the direct and indirect impacts of increasing exploitation. Climate change was expected to affect the biological characteristics, such as growth and reproduction, and carrying capacity of the various stocks. Fisheries agencies will be able to monitor the effects of climate change on growth, etc., by appropriate data collection and analysis using the traditional methods of fish population biology. Changes in carrying capacity will be more difficult to assess as typical single species stock assessment models find it difficult to estimate parameters of the stock recruitment relationship even when this is assumed to be constant. Thus, although not able to detect and quantify the effects of change, the question addressed in the study was whether a management strategy with decision rules based on fishing mortality would be able to sustain a multispecies fishery when the carrying capacities of the stocks in that fishery were subjected to the impacts of climate change. Although the MRM was not developed to discriminate between the effects of increasing exploitation and climate change, the MSE demonstrated that the management strategies that had been accepted for use in the demersal fishery of the West Coast Bioregion would be likely to be effective if the species were affected by those changes.

Lags inevitably exist between the implementation of regulations and the response of the biological system. Thus, the measures that were introduced to reduce exploitation of the demersal finfish stocks of the West Coast Bioregion will take some years to accomplish the recovery that is expected. For example, a recovery signal contained in age composition data will require several years to become evident as additional year classes must first recruit before the age composition changes. Similarly, while cpue will not decline as rapidly with reduced exploitation, the recruitment of new year classes is required before the stock is seen to rebuild. The harvest strategy that is adopted by a fisheries agency, i.e. the combination of data collection, stock assessment, and decision rule, needs to reflect the lag in response. Without this, it is possible that, as a result of the harvest strategy, regulations may be introduced to reduce exploitation, but the harvest strategy is then employed a second time before the stock has had sufficient time to respond to those regulations and the decision rules recommend further reduction in exploitation. That is, there is a need to develop decision rules that respond not only to the state of indicator variables with respect to their associated reference points, but also reflect whether the trend in those indicators is appropriate given the current regulations or whether further action is required. Similarly, simple catch-curve based analyses of age composition data need to be extended such that they can accommodate the change in total mortality that follows the introduction of regulations to reduce exploitation, and thus improve the reliability of assessments that currently rely very greatly on such data.

Benefits and adoption

This study has found no evidence to suggest that there is a decline in the mean trophic levels or mean maximum lengths of catches taken in the West Coast, South Coast, Gascoyne, Pilbara or Kimberley bioregions. Although catches have increased beyond those that would have been expected given the observed mean trophic levels, suggesting potential creep into new subsystems and sequential spatial exploitation, there is currently no indication at the entire bioregion level that the fish faunas of these five regional ecosystems have been impacted by the development of their fisheries to the extent that ecosystem services have been affected. The study has also demonstrated that the diversity and ecosystem-based indicators used in this study provide valuable tools for summarising, and thereby monitoring, the effects of fishing on the ecosystem, and that multivariate analyses provides an approach for determining which species have experienced changes that underlie any significant changes that are detected in the species composition of the catches. Commercial and recreational fishers will benefit from the study through the improved ability of the Department of Fisheries to use these approaches in monitoring and detecting adverse changes in the species composition of the catches, thus ensuring that fisheries are managed in accordance with needs for Ecosystem Based Fisheries Management.

The results from the MSE demonstrated that the management strategy that had been implemented by DoFWA for the West Coast Bioregion was likely to be highly effective in allowing the depleted stocks of the demersal fishery to recover. This management strategy was also shown to be likely to succeed in sustaining the key stocks within an ecosystem impacted by increasing exploitation or climate change. Collection of reliable data on the relative abundance, age composition and population biology for the key fished and non-fished species, using fishery-independent methods, will be a critical element of such management strategies, however.

The study has demonstrated that, provided sound data on the abundance, age composition and population biology of key species are available, it is possible to apply management strategies of the type employed by DoFWA to sustain those stocks successfully. Thus a crucial element of future ecosystem-based fishery management will be the need to review the extent to which key species have been identified, and for which such data are being collected. The importance of fishery-independent data collection programmes has been identified. DoFWA has indicated that it intends to analyse commercial catch data and report values of ecosystem-based indicators in future State of the Fisheries annual reports.

Further development

The Department of Fisheries shall be continuing to refine the data base of commercial fisheries data, and to explore ways in which the influence of fishing practices and the selectivity of different fishing gears can be taken into account before subjecting the data to the types of analyses used in this study, or when using the data to tune ecosystem models. For example, the forms used by commercial fishers to record finfish catches now include fields to capture both the number and mass of fish of the different species that are caught. Decision rules, which recognize that the response of a stock lags the implementation of management action that follows initial application of the decision rule, need to be developed. Methods of analysing age composition data that reflect changes in exploitation that have yet to be experienced by all age classes need to be developed to assist in determining whether the response of the species to the changed regulations matches the predicted response.

Planned outcomes

This project has explored simple and rapid statistical approaches that can be used to assess the ways in which the species compositions of catches vary through time.

The project outputs enable more robust and defined assessment of potential changes in ecosystem and community structure resulting from fishing activities, although consideration must also be given to the ways in which the fishery itself is changing its fishing practices and its target species. The methods explored in this study would assist many fisheries, particularly multi-species fisheries, in meeting their ESD/EPBC obligations and therefore in maintaining the export status of their catches. This will, therefore, provide more certainty in the outcomes of the assessment for industry and will increase the level of confidence of the general community in the decisions made.

The study was intended to ensure that resources for further ecosystem research work are focused on areas where there have been identifiable changes that warrant attention. The results of the analyses, however, have demonstrated that there is currently no evidence to suggest any change in trophic structure likely to affect ecosystem services within any of Western Australia's five bioregions. The study has also identified methods by which catches may be monitored to facilitate detection of any change in trophic composition of catches that might arise from increasing exploitation, to which there will need to be a management response.

The Department of Fisheries' component of the study has achieved its objectives of (1) testing the robustness of statistical procedures to identify impacts of multi-sector fishing on community composition using existing fishery data, and (2) assessing the level of change in community composition in each bioregion of WA during the previous 30 years.

The key data to which ecosystem structure and management strategies are most sensitive were assessed in the context of the models that were constructed to describe the dynamics of the species within the ecosystem, and the response of the species to the management changes that were proposed. The biological system is far more complex than the abstract system described by the model and is likely to be sensitive to numerous variables that are not considered within the model. However, the complexity of the models used to describe the ecosystem is constrained by the data that are available, and the structure imposed by the assumptions that are required to reduce the system's complexity to a level that is manageable. The results from the MSE demonstrated that, within the context of the MRM that was used to describe the ecosystem and the use of fishing mortality as the key indicator of the status of the fishery, the key data that facilitate estimation of fishing mortality were time series of abundance and age composition data for the species included in the model, combined with sound information on the population biology, i.e. growth, longevity and reproduction. Such data are essential if the status of each of the stocks is to be determined and appropriate management actions are to be taken.

Changes in exploitation will affect the relative abundance of different species in the ecosystem, either directly or indirectly, thus affecting the trophic structure and potentially affecting ecosystem function. Management strategies that maintain the abundance of the stocks at appropriate levels need to account for the indirect effects of fishing on the key species within the ecosystem. Reference points that are appropriate for single species, which are the target of fishing, may not be appropriate in an ecosystem context in which vulnerable species may be exposed to the cumulative indirect impacts of exploitation of a number of target species. Changes in the environment due to climate change are likely to affect the productivity of the different species, through impacts on growth, reproduction, and mortality and through the impact on the stock-recruitment relationship, i.e. the viability of eggs, and survival of the larvae and pre-recruits. While changes in biological parameters can be assessed through the collection of biological data and studies of population biology, fishery scientists will have difficulty in assessing the impacts on natural mortality and on the stock-recruitment relationship. Unlike increases in exploitation, managers are unable to control the impacts of climate change. It will be necessary, however, to recognize that levels of exploitation that may have been appropriate under earlier climatic conditions may no longer be appropriate when climate change affects the productivity and carrying capacity of the stock. Thus, adaptive management strategies that adjust to the effects of climate change will be required. The decision rules that have been adopted by DoFWA for use with the demersal fisheries of the West Coast Bioregion have been shown to be robust to the effects of climate change, provided appropriate and reliable data are available for stock assessment.

It was assumed in this study that the abundance, age composition and biological data required for assessment of the state of each of the fish stocks in the MRM were both available and accurate. With the increasing demands placed on the data used in models by the need to consider the direct and indirect effects on the ecosystem of both exploitation and climate change, it is essential that fishery agencies review their data collection programs. Adequate biological data will need to be collected to monitor changes in population biology, *e.g.* growth and reproduction, and both accurate abundance and age composition data will need to be available for the key species. The implementation of fishery-independent data collection programs would provide valuable information that would enhance the reliability of assessment and improve the effectiveness of management. Decision rules need to be developed that reflect not only the values of the indicator variables with respect to reference points, but also whether the trends in the indicator variables are within the range that is acceptable given the regulations imposed at a previous assessment, i.e. the decision rules need to accommodate the lag in response expected given the biological characteristics of each species. Methods of analyzing catch curve data, which allow for the fact that not all age classes will have experienced the change in fishing mortality that accompanies a recent change in fishery regulations, need to be developed to determine whether fisheries such as the demersal finfish fishery of Western Australia are recovering in accordance with expectation or whether additional regulations will be required to sustain the stocks.

Conclusion

The objectives of the study were achieved, and the study has resulted in the following:

1. Demonstration that the mean trophic level, mean maximum length and FIB provide valuable indicators of possible changes in the species composition of the catches that are likely to relate to changes in the trophic structure of the ecosystem. Such indicators facilitate a preliminary rapid assessment of whether changes in catches are consistent with expectations derived from observed mean trophic levels of those catches.
2. Demonstration that, at the current time, there is no indication that catches within any of the five bioregions have declined in trophic level or mean size, or that ecosystem services have been affected by the fisheries in those bioregions.
3. Demonstration that the diversity indicators used in this study provide valuable tools for summarising the effects of fishing on the catches. It is noted, however, that the data used in calculating these indices are not comparable, and that the indices will be affected not only by changes in fishing practice and gear selectivity, but also by changes in the amount of fishing effort applied within the different years.
4. Demonstration that multivariate analyses, such as employed within the Primer software package, allow determination of the statistical significance of changes in the species composition of catches, and facilitate identification of the species that have contributed most to characterising the dissimilarity in species composition.
5. Demonstration that the methods used in this study provide valuable tools for future monitoring of the catches with the various bioregions for detection of changes that might indicate changes in trophic structure within the fish fauna, and which might be cause for concern that ecosystem services are likely to be affected.
6. Demonstration that accurate abundance, age composition, and biological data are crucial for informing the multi-species models that are used for stock assessment and management, and thus these are the key data required for determining whether ecosystem structure is compromised by the impacts of increased exploitation or climate change, and for determining whether management strategies and responses are appropriate. Fishery-independent data collection programmes are recommended if data are to be better representative of key stocks within the ecosystem. Use of such programmes may be essential for key non-fished species within the ecosystem.
7. Demonstration that the management strategy and decision rules adopted by DoFWA for use with the demersal finfish fishery of the West Coast Bioregion would be likely to be effective if the key species within the ecosystem are affected by those changes.
8. Increasing exploitation would directly and indirectly reduce the biomasses of both fished and non-fished species. However, if appropriate abundance and age composition data are collected for those stocks, and dietary and biological data are available for those species, multi-species MRM models may be used to assess the direct and indirect impacts of increasing exploitation.
9. Climate change was expected to affect the biological characteristics, such as growth and reproduction, and carrying capacity of the various stocks. Fisheries agencies will be able to monitor the effects of climate change on growth, etc., by appropriate data collection and analysis using the traditional methods of fish population biology. Changes in carrying capacity will be more difficult to assess as typical single species stock assessment models

find it difficult to estimate parameters of the stock recruitment relationship even when this is assumed to be constant. Thus, although not able to detect and quantify the effects of change, the question addressed in the study was whether a management strategy with decision rules based on fishing mortality would be able to sustain a multispecies fishery when the carrying capacities of the stocks in that fishery were subjected to the impacts of climate change.

10. Adequate biological data will need to be collected by fisheries agencies to monitor changes in population biology, *e.g.* growth and reproduction, which might result from the impacts of climate change.
11. Consideration needs to be given to the implementation of fishery-independent data collection programs to collect accurate abundance, age composition and biological data for both fished and unfished stocks.
12. Decision rules that accommodate the lag in response expected given the biological characteristics of each species need to be developed.
13. Methods of analyzing catch curve data, which allow for the fact that not all age classes will have experienced the change in fishing mortality that accompanies a recent change in fishery regulations, need to be developed.

Appendices

Appendix 1: Intellectual property

The information produced in the study is not suited to commercialization.

Appendix 2: Staff list

Staff employed on the project included:

Dr Norman Hall

Dr Brent Wise

Appendix 3: References

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