

Chapter 7 Lessons learnt from the comparison of three ecosystem models for Port Phillip Bay, Australia

Abstract

Three ecosystem models were tuned to data from Port Phillip Bay, Australia. Several general conclusions can be drawn by comparing predictions of the three models across a range of fishery management strategies and scenarios for environmental change. First, the large, shallow and enclosed physical structure of Port Phillip Bay and the dependency of many fish groups on spawning stocks from outside the immediate area may see the bay react more strongly to eutrophication than to fishing. Second, a selected set of indicator groups (in this case, sharks, seagrass and chlorophyll a) seems to capture the major ecosystem impacts of alternative management scenarios, whether the biomass and productivity of important groups declines, persists or increases. This has obvious implications for system monitoring in an adaptive management approach. Third, multispecies or ecosystem models can identify potential impacts that a series of single species models cannot, such as non-intuitive changes in biomass when species interactions outweigh fishery induced pressures. Finally, policies that focus on protection of species or groups only at higher trophic levels can fail to achieve sensible ecosystem objectives and may push systems into states that are far from pristine.

Keywords

biogeochemical, ecosystem, model, ECOSIM, ECOPATH, Port Phillip Bay

Note: This chapter resulted from work undertaken at an ECOPATH with ECOSIM workshop on the use of ecosystem models to investigate multispecies management strategies for capture fisheries held at the University of British Columbia, Vancouver

Canada, in July 2000. The work involved fitting the models to the current conditions of Port Phillip Bay and then considering the potential effects of management strategies. As a result, some of the terminology used in this chapter differs from that used in the rest of the thesis. Most importantly, whereas the term “standard run” is used to refer to the default run of the full models in all the other chapters the term “base case” is used here. Further, whereas “baseline” is used to refer to the default nutrient loadings and fishing pressures in all the other chapters the terms “current conditions” or “*status quo*” are used in this chapter. I apologise for any confusion this may cause.

7.1 Introduction

Concern over sustainable and responsible management of ecosystems, rather than particular species of interest, has grown over the last century, and particularly within the last decade. This is reflected in the increasing adoption in international treaties and national legislation of policies that take explicit account of such concerns. For example, Australia has adopted a national oceans policy that requires, *inter alia*, development of regional marine plans based upon principles of ecosystem management. Despite this level of interest and activity, the scientific and management tools to underpin such policies are poorly developed. In particular, the tools to predict the impact of alternative uses and management strategies are still being developed (Sainsbury et al. 2000). In the context of ecosystem management, the use of trophic models has been proposed to predict such impacts (Walters et al. 1997, Walters et al. 1999). A range of such models has been developed, but to date there has been little effort to compare and contrast such models, or to compare their strengths and weaknesses (Baretta et al. 1994, Baretta et al. 1996, this thesis).

In this study, three separate ecosystem models are used to explore the utility of such models for the assessment of potential management strategies and their likely

consequences. These three models span a range of process detail, from ECOSIM with its relatively compact formulation (Christensen et al. 2000), to the more nutrient-oriented Bay Model 2 (BM2) (chapter 2), and finally to a highly detailed process model, the Integrated Generic Bay Ecosystem Model (IGBEM) (chapter 1). Between them, these three models reflect much of the range of detail found in simulation models currently being built and used to understand and predict the ecosystem effects of fishing and eutrophication. Unfortunately, their respective histories and the varying purposes for which they were developed mean that the models do differ in many ways and that there is no systematic variation in assumptions. This can make extraction of organising principles or conclusions difficult. However, there is value in determining whether various existing ecosystem models of different forms predict similar outcomes in response to changing conditions and management policies. This kind of information can be instructive with regard to understanding the implications of the formulations used and whether results are robust across modelling methodologies.

The real world system used as the data source and guide for this model comparison is Port Phillip Bay (PPB), adjacent to Melbourne, Australia. During the 160 years since European settlement, PPB has come under increasing pressure from nutrient enrichment and fisheries exploitation. Approximately 15% of Australia's population live within the catchment area of PPB, and the bay has become the focal point for many of their recreational pursuits. It has been estimated that the annual recreational effort is around 670,000 angler hours, which results in the landing of about 470t of fish (Anon 1997). The bay's commercial finfish fisheries land over 60 species, with a total annual take of between 700 and 2000t worth about \$3 million AUD wholesale. A number of invertebrate species are also landed, though many only opportunistically as bycatch. The main targeted invertebrate harvests are cultured mussels and wild abalone, with annual landings of 600t worth \$1.5 million AUD and 50t worth \$1 million AUD

respectively. Until the late 1990s scallops were the most intensively harvested and valuable fishery in PPB, with up to 10,000t (shell weight) being landed per year. However, the fishery was highly variable (fluctuating by two orders of magnitude in 3 years) and the sediment plumes associated with the dredges used in the fishery led to public concern. The fishery was closed in 1997.

PPB is a relatively “easy” system to model as it is a shallow and nearly enclosed temperate bay. It has an area of approximately 1930 km², mostly of waters less than 8m deep, though it reaches 24m at its deepest points. With only 8 drainage basins feeding the bay, and a large sand bar delta and narrow seaward opening restricting exchange with Bass Strait, it has minimal boundary condition problems. The bay has also been the subject of a number of quite extensive environment studies, one or more per decade since the late 1940s, and so there is a large amount of good quality information available for use in model construction and validation (Harris et al. 1996).

Eutrophication and fishing are the two most significant anthropogenic impacts on PPB. The difficulties inherent in interpreting and managing the consequences of human actions on marine systems have meant that fisheries and nutrient related concerns have traditionally been dealt with separately. This is not just the case for Australian bays and near shore waters, but is commonplace worldwide. However, there is increasing evidence that primary productivity and fisheries are more tightly linked than previously thought (Houde and Rutherford 1993, Nielsen and Richardson 1996). With this in mind, the three models are compared across a range of levels of fishing pressure and nutrient inputs.

7.2 Methods

7.2.A Model Descriptions

A brief description of the general form of each model is given below, but space precludes an extended discussion of each of their features, structures and assumptions. To allow for an informed comparison, the essential details of the models and how they vary are listed in Table 7.1. Further, to give some idea of the degree of variation in process detail between the models, the formulation for phytoplankton production is given in Table 7.2. This example is a fair representation of the difference in formulation detail for the lower trophic levels. However at the highest levels, especially those where split pools (juvenile and adults) are included in ECOSIM, the detail can swing back more the other way. At these levels all the models include age structure, explicit recruitment and split food intake into portions allocated to growth and reproduction, though the exact method of doing this does vary between models (Christensen et al. 2000, chapters 1 and 2).

The variation in process detail between the models is one of the motivating forces for the comparison of the models and their predictions. However, other major differences in underlying model assumptions must be kept in mind in comparing the models and their dynamics. These include:

- (a) the consumption formulations (a forage arena approach is used in ECOSIM, but Holling Type functional responses are used in the other two models),
- (b) the lack of bycatch in the standard versions of IGBEM and BM2,
- (c) the omission of invertebrate fisheries from IGBEM,
- (d) the absence of explicit spatial structure in ECOSIM
- (e) the static nature of the birds, mammals and sharks in BM2 and IGBEM and
- (f) the stock structure of the fish groups with its inherent implications for the form of the stock-recruit relationship used.

Table 7.1: Comparison of the underlying structure and assumptions of the three ecosystem models, ECOSIM, IGBEM and BM2. The standard set-ups used for the runs in this study are given, any feature where alternatives may be implemented are marked with a *.

Feature	Model		
	ECOSIM	BM2	IGBEM
General features			
Biomass units	t/km2 (wet weight)	mg N/m3 (dry weight)	mg/m3 of C, N, P, Si (dry weight)
Input forcing	yes (of primary producers), interannual	yes (of nutrients and physics), interannual, seasonal, tidal functional group	yes (of nutrients and physics), interannual, seasonal, tidal functional group
Level of group detail	variable (age group of species up to entire trophic levels)		
Formulation related			
Consumption formulation	forage arena	type II*	mixed (type II, type III)
Formulation detail	simple (expansion of ECOPATH master equation)	general (growth, mortality, excretion explicit)	physiological (assimilation, basal/ activity/stress respiration, defecation, excretion, ingestion, mortality all explicit)
Light limitation	no	optimal irradiance fixed	phytoplankton can adapt to changes in ambient light levels
Mixotrophy	no (no mixed consumers-producers defined in the ECOPATH model)	yes (Dinoflagellates)*	no
Nutrient limitation	no	yes (external)	yes (internal)
Nutrient ratio	-	Redfield	internal nutrient ratio
Oxygen limitation	no	yes	yes
Sediment burial	no	no	yes*
Sediment chemistry	no	yes (dynamic, with sediment bacteria)	yes (empirical, sediment bacteria are a tracer only)
Shading of primary producers	no	yes	yes

Table 7.1: Continued

Feature	Model		
	ECOSIM	BM2	IGBEM
Spatial structure	no explicit spatial structure (ECOSPACE model not considered here)	explicit (8 and 59 box versions)*	explicit (8 and 59 box versions)*
Temperature dependency	no	yes	yes
Model closure			
Status of birds	dynamic	static loss term on fish only	static loss term on fish only
Status of mammals	dynamic	static loss term on fish only	static loss term on fish only
Status of sharks	dynamic	static loss term on fish only	static loss term on fish only
Fish and fisheries related			
Age structured fish	yes (juvenile + adult)	yes (9 age classes)	yes (9 age classes)
Fishery Discards	target and bycatch species	target species only*	target species only
Invertebrate fisheries	yes	yes	no ('fix' implemented by adjusting the mortality terms of the groups concerned)
Stock-recruit relationship	dynamic	constant recruitment	constant recruitment
Stock structure	self-seeding (entire stock in the bay)	external (reproductive stock outside the bay produces the recruits, oldest age classes migrate out of the bay to join this stock)	external (reproductive stock outside the bay produces the recruits, oldest age classes migrate out of the bay to join this stock)

Table 7.2: Process detail involved in the phytoplankton production for each model.

ECOSIM	BM2	IGBEM
$= \frac{r \cdot B}{1 + B \cdot h}$ <p>where B = Biomass of the phytoplankton r = the maximum P/B ratio that can be realised (at low B) and $\frac{r}{h}$ = the maximum net primary production when the biomass is not limiting to production (at high B)</p>	$= \mu \cdot \delta_{irr} \cdot \delta_N \cdot B$ <p>where B = Biomass (mg N m⁻³) of the phytoplankton group (4 types) μ = maximum temperature dependent growth rate δ_N = nutrient limitation factor $= \frac{DIN}{\kappa_N + DIN}$ and DIN = Ammonia + Nitrate unless also limited by Silica (Si) then $= \min \left(\frac{DIN}{\kappa_N + DIN}, \frac{Si}{\kappa_{Si} + Si} \right)$ κ_{XX} = is the half saturation constant for the uptake of the nutrient XX δ_{irr} = light limitation factor $= \min \left(\frac{IRR}{\kappa_{irr}}, 1 \right)$ IRR = Light κ_{irr} = is the half saturation constant for the uptake light</p>	$= \mu \cdot \delta_{irr} \cdot \delta_{Nut} \cdot B_C$ <p>where B_C = Biomass (mg C m⁻³) of the phytoplankton group (4 types)** μ = maximum temperature dependent growth rate δ_{Nut} = nutrient limitation factor $= \min(\delta_N, \delta_P)$ $\delta_N = \min \left(1, \max \left(0, \frac{B_C - \beta_{CNlow}}{\beta_{CNint} - \beta_{CNlow}} \right) \right)$ $\delta_P = \min \left(1, \max \left(0, \frac{B_C - \beta_{CPlow}}{\beta_{CPint} - \beta_{CPlow}} \right) \right)$ β_{CXXlow} = minimum permissible nutrient ratio of C:XX for the cell β_{CXXlow} = standard internal nutrient ratio of C:XX for the cell unless also limited by Silica (si) then $= \min \left(\min(\delta_N, \delta_P), \min \left(1, \frac{Si}{2 \cdot \kappa_{Si}} \right) \right)$ δ_{irr} = light limitation factor $= \min \left(1, \frac{IRR}{\kappa_{irr_adjusted}} \right)$ IRR = Light κ_{irr} = is the half saturation constant for the uptake light adjusted for acclimation to new ambient light conditions</p>

** There are also N and P pools of the biomass explicit in IGBEM and the production in these is based on the internal nutrient ratio and the production for the B_C pool.

These assumptions are identified as having the most potential to cause differences in model outcome. The assumptions relating to bycatch, invertebrate fisheries, static top predators and the form of the stock structure and recruitment relationship are likely to affect the higher trophic levels (the fish in particular). More generally, consumption (predation) and spatial structure have been major research topics in theoretical ecology for a large part of the past century, and have been found to have significant effects in other ecological model studies. For example Hassell and May (1973) and Holmes et al. (1994) have shown that, for predator-prey and competition models, different forms of the consumption formulation, or the addition of spatial structure, can lead to very different sets of population behaviour and species interactions. In some ways it is unfortunate that so many factors vary at once between the three models being considered. However only the ECOPATH with ECOSIM model presented here was built explicitly for PPB. The other two models were built as part of a more general and theoretical study of ecosystem models (this thesis) and then tuned to data from PPB to allow for the comparisons considered here.

IGBEM was created by tying together two existing models, the Port Phillip Bay Integrated Model (PPBIM) (Murray and Parslow 1997, 1999a) and the European Regional Seas Ecosystem Model II (ERSEM II) (Baretta et al. 1995, Baretta-Bekker and Baretta 1997). The biological groups from ERSEM II, as well as a few additional groups formulated in the same way, were tied into the physical and biogeochemical submodels of PPBIM (chapter 1). In its final form, IGBEM is a highly physiologically based process model, which tracks the carbon, nitrogen, phosphorus and silicon pools of 7 primary producer groups (3 benthic, 4 planktonic), 4 zooplankton groups, 3 infaunal groups, 3 epifaunal groups and 4 fish groups (Figure 7.1). This is done within a spatial geometry of 59 polygons (boxes) which parallel the geographical form of PPB, with the size of the polygons reflecting the speed with which physical variables change within

Figure 7.1: Schematic diagram showing the groups in BM2 and IGBEM and their relative trophic positions. The bacterial groups are omitted from the diagram and the model comparisons as no equivalent group exists in the ECOPATH model. The greyed box indicates the position of the dinoflagellates when mixotrophy was allowed in BM2. The dashed boxes indicate the position of the static predator groups implemented for the nutrient model runs used here. Groups marked with a # are fished explicitly in IGBEM and those marked with a * are fished in BM2.

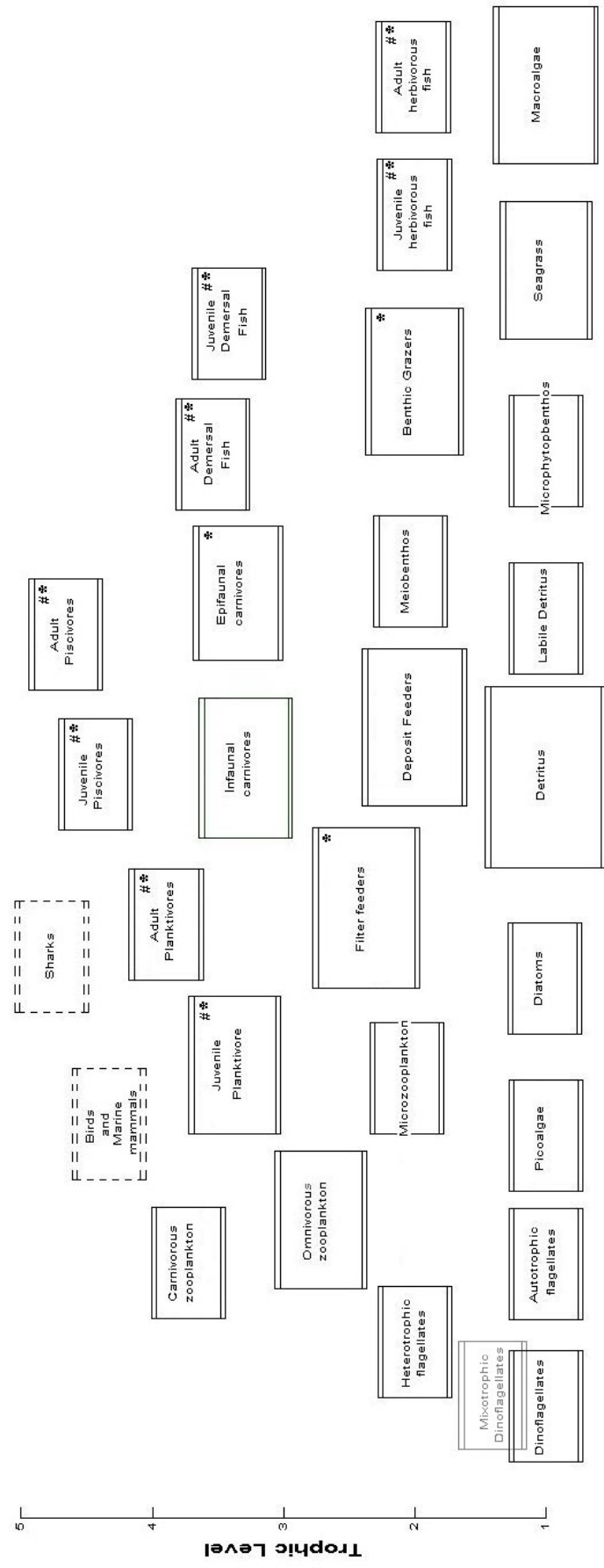
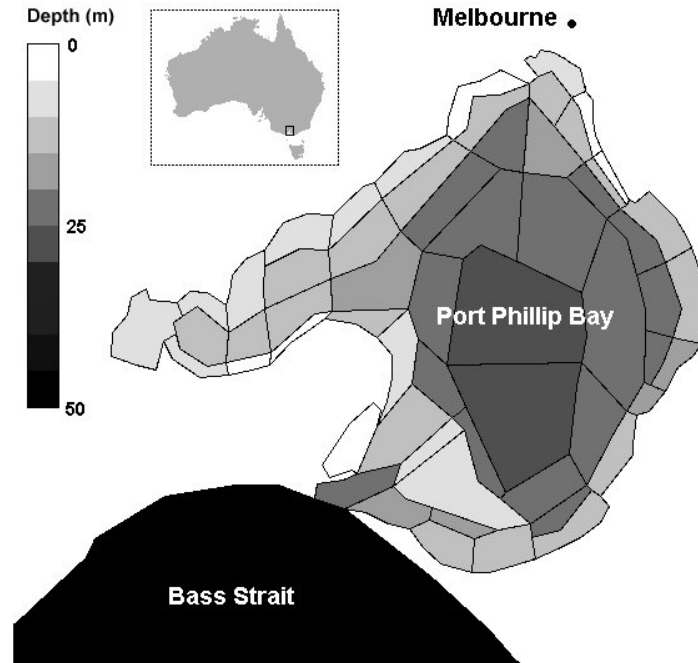


Figure 7.2: Depth map of Port Phillip Bay, Melbourne, Australia. The inset map of Australia indicates the location of Port Phillip Bay. The polygons marked on the large map are those of the standard 60 box geometry used for BM2 and IGBEM.



that part of the bay (Figure 7.2). The level of detail included in this model is considered to be at the upper level, but not beyond, that which is employed in ecosystem models currently in use (e.g SSEM (Sekine et al. 1991), ERSEM II (Baretta-Bekker and Baretta 1997) and ATLSS (DeAngelis et al. 1998)).

BM2 has the same trophic groups, general submodels (water column, sediment, epibenthic) and physical (transport, mixing and spatial geometry) details as IGBEM. BM2 does not have sediment burial, but does have dinoflagellate mixotrophy, while IGBEM has the reverse. The greatest difference between IGBEM and BM2 lies in the level of process detail. BM2 is an order of magnitude simpler, with very similar dynamical representation to PPBIM (Murray and Parslow 1997, 1999a). Processes such as excretion and respiration are not modelled explicitly, but are subsumed into

generalised assimilation and waste production equations (chapter 2). Further, BM2 only models the nitrogen component, and relies on Redfield ratios, rather than internal nutrient ratios, to determine the form of nutrient dependent activities. This level of detail is more representative of the most commonly utilised eutrophication and water column trophic models (Fransz et al. 1991).

Data from the Port Phillip Bay Environmental Study (PPBES), primarily for the years 1994 – 1995, were used to calibrate IGBEM and BM2 to achieve a satisfactory representation of the biological conditions in PPB (in mg N m^{-3}). In a few instances it was necessary to draw values from the general literature or use data from other years to fill in gaps, but this was kept to a minimum. The same data were then used to construct an ECOPATH model, converting from mg N m^{-3} to t km^{-2} wet weight under the assumption that N makes up 1% of an organism's wet weight. The resulting model contains one phytoplankton group, small and large zooplankton groups, 9 benthic invertebrate groups, 3 benthic primary producers, 16 fish groups (some of which were species split into juvenile and adult groups), marine mammals, birds and detritus. While the level of aggregation of species in the ECOPATH model does not match that in BM2 and IGBEM, it is felt that it was better suited to, and more typical of, what is usually found in ECOPATH models. Moreover some of the group splitting was necessary to avoid high levels of cannibalism, though this is not possible for the zooplankton or piscivore groups due to a lack of relevant information. The identity of the various groups and the value and source of the input parameters for the ECOPATH model are given in Table 7.3 and a schematic diagram of the system in Figure 7.3. Eight harvesting enterprises are also included in the model. These are the purse seine, scallop dredge, haul seine, longline, mesh net, dive and pot fisheries as well as the culture of mussels (details summarised in Table 7.4). There is not enough information to specify a separate recreational fishery, so the recreational catches are aggregated with those for

Table 7.3: The basic input parameters for the Port Phillip Bay ECOPATH model. All values shown are those of the final balanced model, but changes made during balancing are noted. In addition the original P/B and Q/B for pilchards are halved and all those for all other fish (juvenile snapper down to rays) were quartered during balancing. Those columns marked by * were calculated by ECOPATH. The TS values stated are the correction values used to correct the trawl data to give total biomass estimates for that group.

Group name	Group description	Habitat area	B (t/km ²)	P / B (/yr)	Q / B (/yr)	Unassim Q	EE*	P/Q*	Catch (t/km ² /yr)	Discards (t/km ² /yr)	Notes on Balancing	References
Phytoplankton	Lumped phytoplankton	1	7.617	250	-	-	0.604	-	0	0	P/B increased 15%	Murray and Parslow, 1997
Small zooplankton	Zooflagellates and small copepods	1	6.477	36.8	59.781	0.3	0.713	0.616	0	0	Q/B increased 32%	Beattie et al. 1996 Holloway and Jenkins 1993
Large zooplankton	Mesozooplankton	1	9.974	23.8	38.609	0.3	0.866	0.616	0	0	Q/B increased 32%	Beattie et al. 1996 (Q/B, B) Holloway and Jenkins 1993 (P/B)
Deposit feeders	Sediment eating invertebrates	0.835	69.948	4.8	66.7	0.3	0.73	0.072	0	0	B decreased 40%	Poore 1992
Scallops and mussels	Wild scallops and cultured mussels	0.835	4.922	3.1	10.9	0.3	0.25	0.284	0.862	0.25	B decreased 10%	Wilson et al. 1993 Kailola et al. 1993 Poore 1992
Filter feeders	Non commercial filter feeders (including oysters)	0.85	73.511	2.8	11.8	0.3	0.756	0.237	0	0.025	B decreased 35%	Wilson et al. 1993 Poore 1992
Infaunal predators	Burrowing worms and other predatory infauna	0.4	13.575	5.4	58.4	0.2	0.999	0.092	0	0	B decreased 20%	Wilson et al. 1993 As for filter feeders
Epifaunal predators	Crustaceans, gastropods and starfish.	0.4	2.363	2.9	21.9	0.2	0.978	0.132	0	0.026	B decreased 10%	As for filter feeders
Southern rock lobster		0.37	0.068	0.73	12.41	0.2	0.874	0.059	0.003	0	B decreased 2%	Anon 1996
Abalone	Green and black lip abalone	0.37	0.699	0.73	12.41	0.3	0.994	0.059	0.048	0	B decreased 2%	Wilson et al. 1993 Anon 1996 Poore 1992 Wilson et al. 1993

Table 7.3: Continued

Group name	Group description	Habitat area	B (t/km ²)	P / B (yr)	Q / B (yr)	Unassim Q	EE*	P/Q*	Catch (t/km ² /yr)	Discards (t/km ² /yr)	Notes on Balancing	References
Other grazers	All other grazers, only urchins fished.	0.4	2.249	0.88	11.68	0.3	0.758	0.075	0	0	B decreased 10%	Anon 1996 Kailola et al. 1993 Poore 1992 Wilson et al. 1993
Scavengers	All epifauna opportunistic scavenging feeders	0.4	9.326	6.86	55.48	0.25	0.834	0.124	0	0	B decreased 10%	Poore 1992 Wilson et al. 1993
Microphytobenthos		1	18.135	44	-	-	0.14	-	0	0	P/B increased 25%	Murray and Parslow 1997
Seagrass		0.1	2.591	24	-	-	0.345	-	0	0.01	P/B increased 25%	As for microphytobenthos
Macroalgae		0.7	25.907	20	-	-	0.301	-	0	0.01	P/B increased 25%	As for microphytobenthos
Clupeoids	Pilchards, anchovy and sprat.	1	2.85	1.15	30.15	0.2	0.993	0.038	0.812	0	P/B and Q/B decreased 50%	Anon 1996 Hall 1992
Juvenile snapper	<3 yrs. VBGF k = 0.1079, Wavg/Wk = 1.16	0.7	0.469	0.548	2.737	0.2	0.973	0.2	0.012	0	P/B and Q/B decreased 75%	Parry et al. 1995 Officer and Parry 1996 Parry et al. 1995
Snapper	3+ yrs (splits based on recruitment to fishery and 50% maturity)	0.7	0.376	0.493	2.737	0.2	0.785	0.18	0.033	0.001	P/B and Q/B decreased 75%	Gunthorpe et al. 1997 As for juvenile snapper
Juvenile flatfish	<3 yrs. VBGF k = 0.19, Wavg/Wk = 1.2	0.75	2.319	0.821	2.737	0.2	0.977	0.3	0.004	0.001	P/B and Q/B decreased 75%	As for juvenile snapper
Flatfish	3+ yrs	0.75	2.285	0.411	2.737	0.2	0.902	0.15	0.143	0.011	P/B and Q/B decreased 75%	As for juvenile snapper
Juvenile KG whiting	<3 yrs. VBGF k = 0.16, Wavg/Wk = 1.1. TS = 5	0.8	0.142	0.821	2.737	0.2	0.936	0.3	0.06	0	P/B and Q/B decreased 75%	As for juvenile snapper
KG whiting	3+ yrs. TS = 5	0.8	0.117	0.548	2.737	0.2	0.293	0.2	0.001	0	P/B and Q/B decreased 75%	As for juvenile snapper
Juvenile piscivores	<3 yrs. VBGF k = 0.42, Wavg/Wk = 1.1. TS = 20	0.72	0.567	0.821	2.737	0.2	0.999	0.3	0	0	P/B and Q/B decreased 75%	As for juvenile snapper
Piscivores	3+ yrs. TS = 20	0.72	0.288	0.411	2.737	0.2	0.92	0.15	0.001	0	P/B and Q/B decreased 75%	As for juvenile snapper

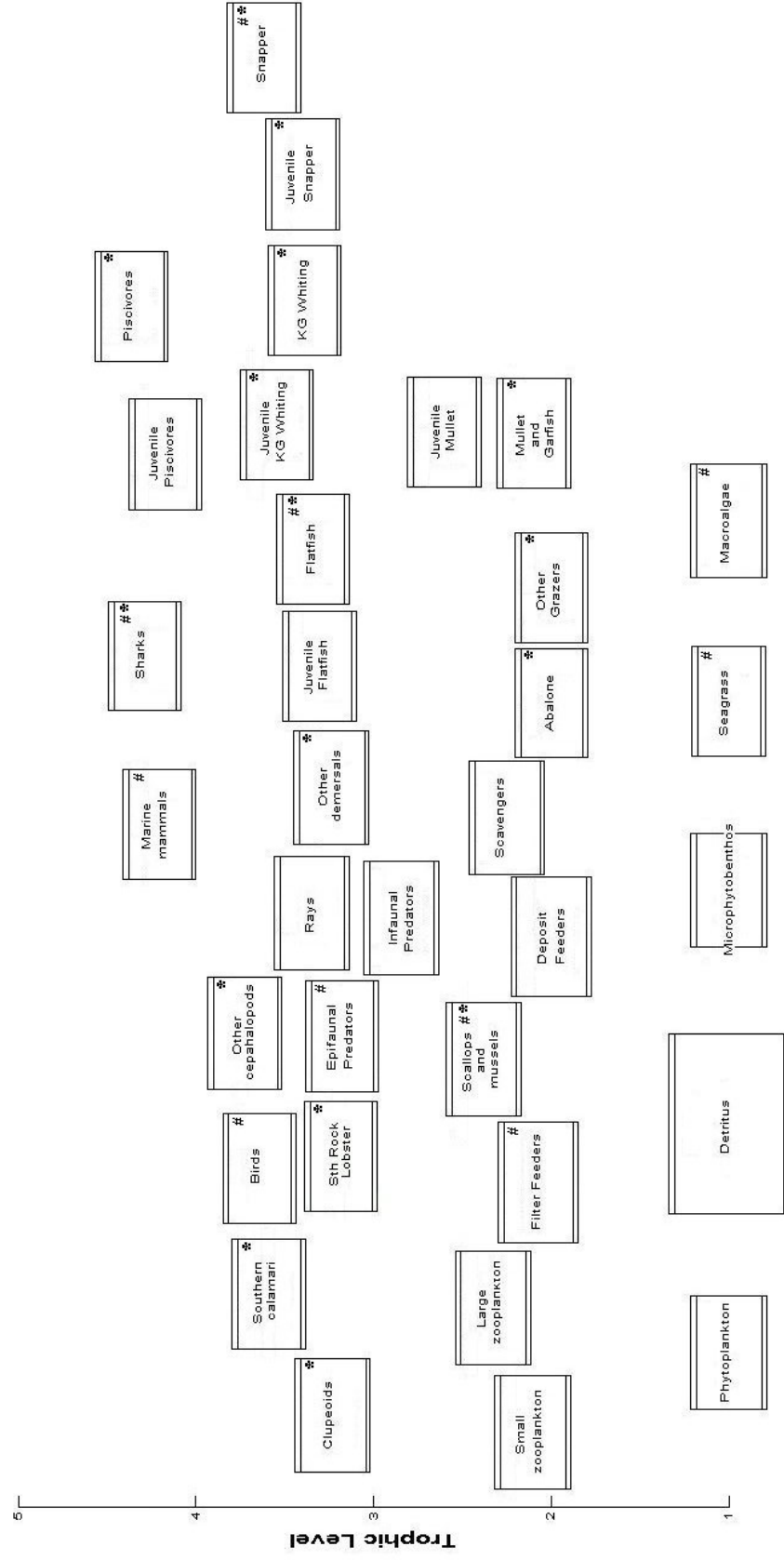
Table 7.3: Continued

Group name	Group description	Habitat area	B (t/km ²)	P / B (yr)	Q / B (yr)	Unassim Q	EE*	P/Q*	Catch (t/km ² /yr)	Discards (t/km ² /yr)	Notes on Balancing	References
Juvenile mullet	<3 yrs. VBGF k = 0.271, Wavg/Wk = 1.15. TS = 4.45	0.825	0.526	0.411	2.737	0.3	0.993	0.15	0	0	P/B and Q/B decreased 75%	As for juvenile snapper
Mullet and garfish	3+ yrs. TS = 4.45	0.825	0.383	0.329	2.737	0.3	0.909	0.12	0.053	0	P/B and Q/B decreased 75%	As for juvenile snapper
Other demersals	All other demersal finfish. TS = 2	0.965	4.899	0.548	2.737	0.2	0.849	0.2	0.041	0	P/B and Q/B decreased 75%	As for juvenile snapper
Southern calamari	TS = 4	0.93	0.319	1.825	18.25	0.2	0.785	0.1	0.05	0		Officer and Parry 1996 Parry et al. 1995 Gunthorpe et al. 1997 Lee 1994
Other cephalopods	Octopus. TS = 4	0.945	0.415	1.369	9.125	0.2	0.952	0.15	0	0		As for southern calamari
Rays		0.9	6.166	0.234	1.56	0.2	0.007	0.15	0	0	P/B and Q/B decreased 75%	Officer and Parry 1996 Parry et al. 1995 Gunthorpe et al. 1997 Schmid et al. 1993 As for rays
Sharks	To get B used demersal trawl data, TS = 13.35.	0.62	0.148	0.234	1.56	0.2	0.639	0.15	0.002	0.001		
Birds	Shorebirds (based on representative densities of shorebirds in similar habitats elsewhere)	1	1.018	0.07	1.69	0.2	0.145	0.041	0	0		Briggs et al. 1997 PICES 1998
Marine mammals	Dolphins and seals	1	0.02	0.09	19.88	0.2	0.006	0.005	0	0		Dolphin Research Institute 2000 Nicholson et al. 1996 Harris et al. 1996
Detritus	Biomass represents top 20cm of sediment. 398.96 t imported per yr.	1	14766.84	-	-	-	0.994	-	0	0		

Table 7.4: Landings, discard and total value information for the harvest methods included in the Port Phillip Bay ECOPATH model.

Fishery	Target Species	Bycatch Species	Landings (t/km²/yr)	Discards (t/km²/yr)	Value (\$AUD)
Purse seine	Clupeoids		0.812	0	1,500,000
Scallop dredge	Scallops	Scallops Filter feeders Epifaunal predators Seagrass Macroalgae Flatfish Snapper	0.551	0.3205	8,000,000
Haul seine	Juvenile and adult King George whiting Mullet and garfish Southern calamari Other cephalopods		0.1639	0.001	930,000
Longline	Juvenile and adult snapper	Juvenile and adult flatfish Sharks	0.041	0.012	500,000
Mesh nets	Juvenile and adult snapper Juvenile and adult flatfish Piscivores Other demersals Sharks	Marine mammals	0.1945	0.00001	780,000
Dive	Abalone Other grazers	Epifaunal predators	0.0483	0.0005	1,175,000
Aquaculture	Mussels		0.311	0	1,500,000
Pot	Southern rock lobster		0.003	0	125,000

Figure 7.3: Schematic diagram of the Port Phillip Bay ECOPATH model, showing the constituent groups and their relative trophic positions. Groups marked with a * are landed by fisheries included in the model, whereas groups marked with a # are species that are taken and discarded.



the commercial fishery using the same gear. As with all ECOPATH models some balancing is required. This process involved adjusting some, ideally the most uncertain, of the diet compositions, biomasses and the Production/Biomass ratios (P/B) and Consumption/Biomass ratios (Q/B) input until all of the ecotrophic efficiencies are less than 1 and the gross food conversion efficiencies are within sensible bounds for each group. However all parameter values remain within the ranges given in the PPBES technical reports (though many are moved from the centre to one end or the other of the quoted ranges). The biomass of the deposit feeders proves to be a critical value in the model, but this is also one of the most uncertain.

In hindsight a potentially confounding issue arises from the fact the nutrient models (IGBEM and BM2) are calibrated, while the ECOPATH model undergoes balancing. The nutrient models are calibrated by setting their parameters (growth, mortality etc) based on the species composition of PPB and then minor adjustments are made to these so that the biomass levels output are reasonable reflections of those reported in the PPBES technical reports. While the biomasses from the technical reports are also used in the ECOPATH model they undergo some significant changes during balancing. Thus the biomass values in the final version of the ECOPATH model no longer match those used to calibrate the nutrient models. (While not cripplingly important for all groups in the model, it became obvious that in an ideal situation more tuning would have been beneficial in spots and these will be identified in the following discussion.)

When the ECOPATH model is extended into an ECOSIM model, a mediation effect is added, to reflect the critical importance of seagrass to juvenile King George whiting. This does not have a deleterious impact on model stability and behaviour. As the vulnerability settings used in an ECOSIM model can be crucial (Christensen et al. 2000), alternative sets of vulnerabilities are tested. To avoid confusion with the term

vulnerability as used in fisheries science, the ECOSIM vulnerabilities are referred to as “refuge parameters” for the remainder of this chapter. In setting the refuge parameters for the fished groups estimates of virgin levels of egg production are used to give initial estimates of the refuge parameters and then these are adjusted slightly to obtain stable equilibria (no inherent rate of increase or decline) under *status quo* fishing mortalities (Fs). Refuge parameters for all groups are set at 0.5 with the following exceptions. Scallops, abalone, clupeoids, and both piscivore age classes are at lower levels (0.48, 0.45, 0.45, 0.4 and 0.4 respectively) while southern rock lobster, juvenile snapper, marine mammals and both King George whiting groups are at higher levels (0.8, 0.6, 0.9, 0.8 and 0.8 respectively).

IGBEM and BM2 are spatially explicit models, but an ECOSPACE form of the ECOSIM model for PPB has not been fully developed. Consequently, the final four years of the 20 year runs from IGBEM and BM2 are spatially averaged to give values for the entire bay and these are then used in the comparisons discussed here. The biomass units used in the internal model calculations of the three models also differ, so all biomasses have been converted to t/km² to facilitate comparisons. Despite this, BM2 and IGBEM are referred to as nutrient models (as they deal with nutrient pools rather than total biomasses).

7.2.B Comparison of the three models

Once tuned to the data for PPB, the comparison of the three models and the potential management strategies was undertaken in a multi-step process. The following is an outline of the overall process, with each step briefly outlined in its own subsection.

This multi-step process allows for the clearest grasp of the individual and combined changes in the systems due to the effects of anthropogenically induced change in nutrient load and fish mortality and management intervention to do with these

factors. Base cases are compared first so that differences in the models when there is no change, or simple changes, can be understood before the more complex interaction of optimal and changing management strategies are applied.

7.2.B.1 Comparison of the “base case” results

Comparison of biomasses and Comparison of P/B and Q/B ratios

The spatially averaged output of the nutrient models is compared to the values used to construct the balanced ECOPATH model. As mentioned above, the Biomass (B), Production/Biomass ($P/B \approx$ Total mortality) and Consumption/Biomass (Q/B) values used in the ECOPATH model came from estimates reported in the PPBES. In contrast, the Biomass, P/B and Q/B values produced by the nutrient models are the outcome of tuning the physiological and process parameters to represent the species of PPB. Thus, there is uncertainty associated with the values produced by all three models. However, a model comparison is easier if there is an identifiable baseline to compare against and the easiest way of doing this is to designate one of the three models as that baseline. Moreover, the work presented here was done in the context of a larger evaluation of ECOSIM as a tool for considering the effect of fisheries policy. As a consequence, it was decided that the ECOPATH values are as good a baseline as any and so the nutrient models are measured against them. Ideally, all three models should be compared to an independent data set for the bay, but this is not possible at this time.

System-level indices

A comparison of the biomass, P/B and Q/B values provides only a partial summary of how well each model captures the overall state of the system. Production, consumption, biomass and catch results for each group are output routinely by the nutrient models and estimates of these values exist for PPB (the same data used to

construct the ECOPATH model). It is possible, therefore, to calculate an extended list of system-level indicators for the “base case” of each model. Unfortunately, this is not so easily done for any of the later simulation runs. Thus, only Mean Trophic Level (for the entire system), Mean Trophic Level for the Catch (including bycatch groups) and Total System Biomass are presented for these later runs.

Temporal dynamics

An important validation of simulation models is how well temporal dynamics are captured. Overall averages may be close, but this can conceal quite large phase lags and mismatches in the patterns produced through time by the models. Therefore, the temporal dynamics of biomass for each of the models is compared.

Alternative forcing scenarios

The models are compared under alternative nutrient and fishing mortality regimes. The regimes examined included:

- increase in nutrients (fivefold increase on current levels)
- decrease in nutrients (decrease to a fifth of current levels)
- increase in fishing mortality (across the board fivefold increase on current levels)
- decrease in fishing mortality (across the board reduction to a fifth of current levels)

The various regimes all represent longterm gradual change and are implemented in ECOSIM by taking the nutrient input files used to force IGBEM and BM2 and using them to force the phytoplankton and macrophyte groups in the ECOSIM model.

Changes in F are sketched into the ECOSIM model directly via the scribble pad provided in the software (Christensen et al. 2000). The level of change implemented in

the scenarios is not an arbitrary choice. A fivefold increase in nutrients is known to cause eutrophication in the Port Phillip Bay Integrated Model (Murray and Parslow 1999a), and fivefold changes in F have not been uncommon during the history of some of the PPB fisheries.

Summary of conclusions for “base case” results

A summary of the major findings are presented from the “base case” runs before any fisheries policy analysis is implemented.

7.2.B.2 Fishing policy analysis

ECOSIM, with no change in nutrient load

The open and closed loop policy analysis option in ECOSIM is used to find optimal fishing strategies under a number of economic, social and ecological criteria (defined in Table 7.5).

“Optimal” policies implemented in BM2 and IGBEM, with no change in nutrient load

The optimal policies derived in ECOSIM are trialed in IGBEM and BM2 to see what effects are produced in these models.

Change in F scenarios vs the results of the optimisations

As the optimisation with an economically weighted objective function produces an increase in most F s and the optimisation with an ecologically weighted objective function produces a general drop in F s, another comparison is possible. The changes in biomass that result from the scenarios where there are across the board changes in F are compared with the results of the optimisations.

Fishing policy analysis by ECOSIM, with a change in nutrients

The optimisation process is repeated for those cases where there is a long-term change in nutrients. As the nutrients (and thus productivity) undergo large changes with time, a single F applied for the entire period is unlikely to be optimal. Thus, the policy analysis is completed twice in this case: once where a single F is found, per fishery, for the entire simulation period and once where there is one policy for the first twenty years and a second for the final twenty years of the simulation. The second policy does not begin until there has been an obvious change in productivity and system conditions.

“Optimal” policies implemented in nutrient models, with a change in nutrients

The two-stage policies (split policies) that are suggested as optimal by ECOSIM are implemented in IGBEM and BM2.

Nutrient scenarios vs the results of the optimisations with a change in nutrients

The change in nutrients has a much larger impact upon the system-level measures of catch and economic value than the results of the optimisations when there is a change in nutrients. Therefore, the outputs of these optimisations are compared to the scenarios where there is a change in nutrients, but no change in F.

Summary of conclusions for fishing policy results

A synthesis is made of the general results to come out of the fishing policy analysis and the many comparisons.

Table 7.5: Criteria used to define the objective functions used in the ECOSIM policy analysis routines.

(a) ecological weightings used in the ECOSIM analysis

Group	Strategies			
	Charismatic		All species represented	
	Mandated Relative Biomass	Relative Weight	Mandated Relative Biomass	Relative Weight
Phytoplankton	1	0	1	0
Small zooplankton	1	0	1.5	0
Large zooplankton	1	0	1.5	0
Deposit feeders	1	0	1.5	0.2
Scallops and mussels	1	0	5	0.4
Filter feeders	1	0	1	0.4
Inf. predators	1	0	1	0.2
Epi. predators	1	0	2	0.4
Sth rock lobster	1	0	5	1.4
Abalone	1	0	3	1.4
Other grazers	1	0	1.5	1.2
Scavengers	1	0	1	0.2
Microphytobenthos	1	0	1	0
Seagrass	1	1	1	0
Macroalgae	1	0	1	0
Clupeoids	1	0	2	0.8
Juvenile snapper	1	0	1	1.8
Snapper	4	0.1	4	2
Juvenile flatfish	1	0	1	1.2
Flatfish	1	0	2	2.4
Juvenile KG whiting	1	0	1	1.2
KG whiting	5	0.1	5	1.8
Juvenile piscivores	1	0	1	1.2
Piscivores	1	0	3	2.4
Juvenile mullet	1	0	1	2.4
Mullet and garfish	1	0	2	3
Other demersals	1	0	2	1.8
Southern calamari	1	0	2	0.6
Other cephalopods	1	0	1.5	0.8
Rays	1	0.1	1	4.2
Sharks	2	1	2	4.2
Birds	3	1	3	14.2
Marine mammals	4	1	4	11.2

(b) social weightings used in the ECOSIM analysis

Gear Type	Jobs/Catch
Purse seine	1
Scallop dredge	1
Haul seine	1
Longline	0.5
Mesh nets	1
Dive	0.2
Aquaculture	0.1
Pots	0.5

7.3 Results

7.3.A Comparison of the “base case” results

The final parameterisations of IGBEM and BM2 result in simulated systems that are fairly close to the summarised form of the bay as captured by the ECOPATH model.

Comparison of Biomasses

The majority of groups, both in IGBEM and BM2, have biomasses within a factor of two of the values for the balanced ECOPATH model (Table 7.6). However for IGBEM, this is a bare majority. Eight groups were more than a factor of two from the ECOPATH values. The benthic grazer and epifaunal predator groups are nearly three times larger than the corresponding group in ECOPATH, which may be symptomatic of IGBEM’s apparent tendency to emphasise a trophic web based on primary production over the detritus-based web. The biomass of detritus is a factor of three lower for IGBEM.

Although it would be possible to tune IGBEM to more closely match more of the biomasses in PPB, this would entail moving a number of the clearance, growth and mortality parameters for the benthic groups beyond the biological limits currently recorded in the literature. This suggests that there may be an additional mechanism or property at work in the bay that IGBEM does not include or cannot capture. It may also suggest that the loss of epifauna to the ‘static fish’ (a constant loss term in IGBEM) may not be quite high enough (chapter 1). The macrophyte groups (seagrass and macroalgae) are much higher than the ECOPATH values and this is undoubtedly due to the spatial structure in IGBEM, which allows for a roughly five year cycle of “macrophyte-barrens” formation and recovery as nutrient availability interacts with the density of benthic grazers. A linked cycle of this form has not been recorded for PPB and this may be a model artefact. At the other extreme, the other benthic primary producer

Table 7.6: Comparison of the group data for the three models. The values are given as the relative size of the Biomass, P/B and Q/B values output by IGBEM and BM2 in relation to those in ECOPATH (value for model x / value in ECOPATH). The values are only given for those groups shared by all three models. To allow for this comparison, the plankton groups in the nutrient models are aggregated to the levels of ECOPATH; the benthic invertebrate groups in ECOPATH are aggregated to the level of trophic groups; the demersal fish of the nutrient models are equated with the flatfish in ECOPATH; and the herbivorous fish were equated with the mullets.

Group	IGBEM			BM2		
	B	P/B	Q/B	B	P/B	Q/B
Phytoplankton	1.09	1.42	-	1.21	1.12	-
Small zooplankton	1.37	0.70	0.69	1.40	0.68	0.67
Large zooplankton	1.14	1.03	1.02	0.77	1.09	1.24
Deposit feeders	0.80	2.12	0.80	1.20	1.89	0.59
Filter feeders	0.96	0.42	0.27	0.96	1.05	1.16
Infaunal predators	1.38	1.98	0.54	1.11	2.13	0.39
Epifaunal predators	2.92	1.13	0.55	0.64	1.08	0.57
Benthic grazers	2.49	0.27	0.08	0.85	2.34	0.70
Microphytobenthos	0.10	1.10	-	0.13	1.00	-
Seagrass	4.10	1.00	-	2.23	0.17	-
Macroalgae	2.01	0.25	-	1.01	0.74	-
Clupeoids	1.92	1.92	0.27	1.66	1.90	0.25
Juvenile flatfish	1.05	1.26	1.03	1.56	1.49	1.20
Flatfish	0.80	0.92	0.43	1.17	1.01	0.68
Juvenile piscivores	2.22	1.40	0.91	0.29	0.75	1.42
Piscivores	1.24	0.41	0.37	1.16	0.80	0.73
Juvenile mullet	0.64	3.50	1.15	0.92	2.90	1.05
Mullet	0.40	1.42	0.49	2.32	1.67	0.85
Detritus	0.30	-	-	1.34	-	-

(microphytobenthos) is only a tenth of the ECOPATH value. This is probably due to the competitive exclusion of the group by the large macrophyte groups and a poor calibration of a relatively new part of the final IGBEM model (chapter 1). This is an unfortunate consequence of the fact that the nutrient models were developed in another context prior to this particular investigation.

Within the fish groups, the biomass of juvenile piscivores in IGBEM is more than double that of the ECOPATH model, suggesting the survival of 0+ to 2+

individuals is probably too high in IGBEM. While the same groups prey on the juvenile piscivores in all three models, the mortality imposed by the static predators in IGBEM (sharks, birds and mammals) is tuned based on older age classes and total longevity of the fish group. As a result it does not seem to be high enough for the younger, more vulnerable, age classes. Further, as the elevated biomass is damped with the aging of cohorts into the older age classes, it is possible that the mortality rate of the oldest age classes is a little too high. Thus, while the mix of age-specific and general pressures on the fish groups serves to represent the dynamics of the intermediate age classes of the piscivorous fish group well, it does not perform as satisfactorily for the younger and older age classes. In contrast to the juvenile piscivores, the adult mullet biomass is two and half times lower in IGBEM than ECOPATH. This is due to additional mortality resulting from the starvation of mullet in IGBEM during those periods when the macrophytes are in a “barrens” state. Lastly, the biomass of the detritus in IGBEM was only a third of the estimate used in the ECOPATH model. This is a result of a combination of factors in IGBEM: the point source detrital input is about two thirds of what it should have been, the assimilation by the detritus feeders is too efficient and detrital burial is too fast.

The comparison of the biomass values in the BM2 and ECOPATH models shows that these two are much closer. Between the two, only the microphytobenthos, seagrass, juvenile piscivore and mullet groups differ by more than twofold. Microphytobenthos is once again only a tenth of that in the ECOPATH model, whereas seagrass is nearly three times higher in BM2 than in ECOPATH. These results suggest that more time should have been spent tuning the macrophytes and investigating whether or not it was possible to suppress their competitive exclusion of the microphytobenthos. The dynamics of the competitive exclusion and its relation to a “macrophyte-barrens” cycle, which is also displayed by BM2, are only obvious once the

full spatial models are considered (chapter 2) and will not be discussed here. It is worth noting that the “macrophyte-barrens” cycle in BM2 has a shorter period and is not of the same amplitude or as spatially widespread as that in IGBEM (chapter 2) and as a result has a smaller impact on the participating groups.

With regard to the fish, it is noteworthy that the same groups stand out in comparison to ECOPATH for both IGBEM and BM2, but that the direction of difference is reversed between the two nutrient models. Apparently the feeding and migration regimes in BM2 are different enough to allow for the dynamic predators to target the juvenile piscivores more effectively, while the adult mullet escape starvation.

Comparison of P/B and Q/B ratios

The comparison of P/B and Q/B ratios amongst the three models proves to be insightful. The P/B ratios produced by the nutrient models are generally close to those in the ECOPATH model, within a factor of two.

In IGBEM the P/B ratios for the benthic deposit feeders and juvenile mullet are more than twice as high as those in ECOPATH, while those for filter feeders, benthic grazers, macroalgae and piscivores are less than half. It may be that, with so many explicit processes to be parameterised in IGBEM, their cumulative effect can result in inappropriate levels of productivity. This would seem to be supported by the fact that all of the consumer groups with low P/B ratios also had low Q/B ratios. It is the low P/B ratio for macroalgae that cannot be easily explained.

For BM2, the P/B ratios for the infaunal predators, benthic grazers and juvenile mullet are all more than double those of ECOPATH, whereas the ratio for seagrass is less than half. The general form of the equations and the specific conversion efficiencies used for groups on diets of low nutritive value in BM2 may allow for P/Bs for consumers that are too high. Resetting the efficiencies of these groups to lower levels

could correct this problem. The mechanism causing the low levels of seagrass productivity is much harder to identify.

The Q/B ratios are much more easily explained. In general, the values produced by the nutrient models are often too low in comparison to those in ECOPATH . This suggests that overall the assimilative processes have the potential to be much more efficient in the nutrient models and may be too high, at least for some groups.

System-level indices

Another useful way to compare models is to use system or flow indices (Finn 1976, Ulanowicz 1986, Christensen 1995). A selection of such indices calculated for each of the models is given in Table 7.7. It is obvious that the models do not match as well under all the indices presented. Relative Ascendancy, Mean Trophic Level, Mean Trophic Level of the Catch, Primary Production / Biomass and Total Biomass / Total Throughput all suggest that the model systems are close in form and system maturity. In contrast, Finn's Mean Path Length, Net Primary Production and Total Biomass indicate

Table 7.7: A comparison of ten system-level indices for the “base case” runs of the three models. Finn's Recycling Index and Relative Ascendancy are given as proportions not percentages here. The bracketed figure for Mean Trophic Level of the Catch for ECOSIM and BM2 is the value when only finfish are considered as in IGBEM.

System index	ECOSIM	BM2	IGBEM
Finn's Mean Path length	4.0	4.9	3.6
Finn's Recycling Index	0.2	0.24	0.07
Mean Trophic Level	1.98	2.16	2.05
Mean Trophic Level Catch	2.8 (3.19)	2.42 (3.2)	3.28
Net Primary Productivity	3070.99	4860.84	4131.38
Primary Production/Biomass	14.12	13.56	13.74
Relative Ascendancy	0.32	0.30	0.33
System Omnivory Index	0.18	0.06	0.18
Total Biomass (excluding detritus)	217.45	358.0	300.69
Total Biomass / Total Throughput	0.016	0.015	0.023

large differences between the model systems. Finn's Recycling Index and the System Omnivory Index are intermediate, suggesting that one or other of the nutrient models is close to the ECOPATH model while the other is not. In combination these indices suggest that the gross form of the model systems are quite similar, while the internal details producing this form vary in some crucial respects.

Temporal Dynamics

Moving from the static to the dynamic, the patterns of biomass dynamics for the ECOSIM model are in good agreement with the temporal dynamics of the nutrient models, ignoring those dynamics caused by the spatial structure of the trophic models – such as the “macrophyte-barrens” cycles. Once forced by the time series of primary production, even the interannual variability in the ECOSIM model mirrors that of the nutrient models, especially for the dominant benthic deposit feeder group (Figure 7.4). Given that phytoplankton are commonly used as an indicator group in monitoring studies (via chlorophyll *a* measurements), it is worth noting that there is not a good match between the three models for the phytoplankton (Figure 7.5). This resulted from a lack of a seasonal and tidal cycle in the ECOSIM implementation, and a differential in the sensitivity to forcing in the nutrient models. The clupeoids (or planktivorous fish) are another group that show some notable differences in temporal pattern between ECOSIM and the nutrient models. This group is more responsive in ECOSIM than in the nutrient models (Figure 7.6). This is contrary to expectations based upon the dynamics of their prey species. Of the three models, the planktonic groups in ECOSIM show the least response to forcing. Thus, a reversal of responsiveness higher in the trophic web is not anticipated.

Figure 7.4: Overlay of a section of the time series of the benthic deposit feeder group in all three models. The period shown represents one cycle of the forcing function file.

Light grey line is ECOSIM, dark grey line is BM2 and black is IGBEM.

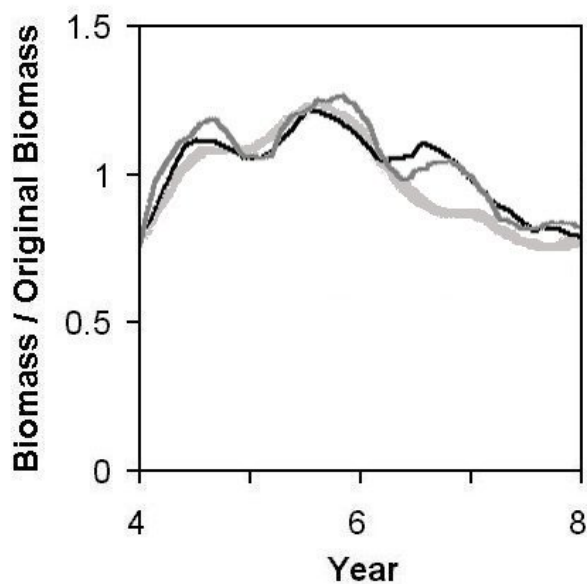


Figure 7.5: Overlay of a section of the time series of total phytoplankton for the three models. The period shown represents one cycle of the forcing function file. Light grey line is ECOSIM, dark grey line is BM2 and black is IGBEM.

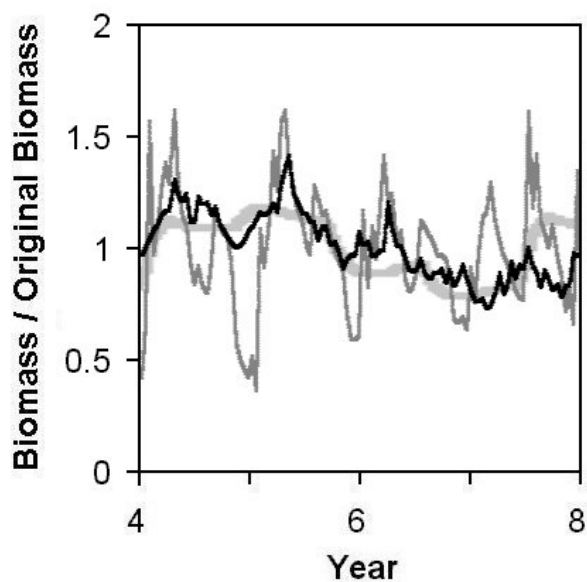
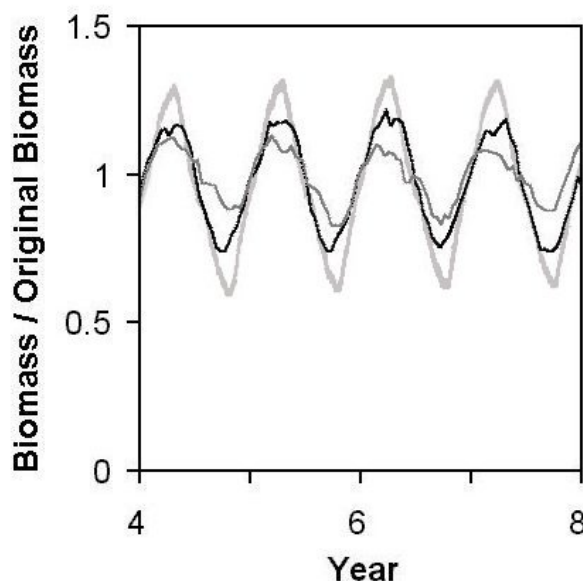


Figure 7.6: Overlay of a section of the time series of the planktivorous fish group (clupeoids) in all three models. The period shown represents one cycle of the forcing function file. Light grey line is ECOSIM, dark grey line is BM2 and black is IGBEM.



Alternative forcing scenarios

While there is relatively good agreement between models in the “base case” runs, greater differences between the models become apparent under changing conditions (fishing pressure or nutrient loading).

The period of transient dynamics in response to new ambient conditions is similar in all models. The most responsive (short lived, high turnover) groups reach their new densities within 2 years and the slower groups within 5 years. However some of the slower groups in ECOSIM appear to track their faster food groups too quickly, in comparison with the nutrient models.

The general patterns of biomass change seen in the three models under changing conditions are given in Table 7.8. On the whole, the qualitative changes seen in the groups as conditions changed do correspond – all the models agreeing (i.e. the direction of change matched) in 65% of cases. On an individual basis qualitative agreement is

even higher, ECOSIM agreed with BM2 in 72% of cases and with IGBEM in 78% of cases. In the majority of cases where the models do not agree, ECOSIM recorded no change whereas the nutrient models record some change. There is only direct disagreement (direction of change opposite) between ECOSIM and the nutrient models in less than 5% of cases. However, the magnitude of change often differs substantially between the various models. The magnitude of change observed in ECOSIM differs from that seen in BM2 by more than a factor of 2 in 27% of cases (in over half of those the magnitude of change differs by more than a factor of 5). In comparison, the magnitude of change seen in ECOSIM and IGBEM differs by more than a factor of 2 in only 19% of cases (less than half of which are cases where they differ by more than a factor 5). It is noteworthy that the cause of the divergences in the magnitude of change is not the same across all the scenarios. In those scenarios where it was the nutrient levels that change, it is usually the higher trophic groups of ECOSIM that show the most divergence, suggesting less buffering up the trophic web in ECOSIM than in the nutrient models. Conversely, in the scenarios where F changes with time it is the biomass of the groups (particularly the fish) within the nutrient models that undergo the greatest changes. The formulation of the consumption and closure terms in the various models is probably at least one of the explanations for these effects. The forage arena formulation in ECOSIM allows for compensatory dynamics, especially under stock declines. This is something the feeding formulations in the nutrient models do not capture nearly as well, if at all. Moreover, the dynamic top predator groups included in ECOSIM not only track their prey but are often bycatch groups themselves. As a consequence, when there are changes in F they undergo direct and indirect impacts on their population size and so react in a more complex fashion than the simple decline that the closure terms of the nutrient models allow. The end result is that the formulation of the food web interactions for the fish in the nutrient models cannot adjust for the effects

Table 7.8: The relative change in biomasses (biomass after change / biomass before change) for each of the three models under the test scenarios. As only qualitative comparisons are possible, the values presented are rounded in most cases. The lumping and association of groups noted for Table 7.6 also applies here. Due to their value as a potential indicator group, sharks are also given for ECOSIM.

(a) nutrient change scenarios

Group	Nutrient input rises			Nutrient input drops		
	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Phytoplankton	6	2	10	0.15	0.5	0.2
Small zooplankton	6	2	5	0.15	0.5	0.1
Large zooplankton	10	1.2	4.5	0.1	0.75	0.15
Deposit feeders	2	2.2	3	0.5	0.1	0.3
Filter feeders	8	1.4	2	0.02	0.8	0.3
Infaunal predators	4	1.5	2	0.2	1	0.25
Epifaunal predators	8	1	4	0.2	1	0.3
Benthic grazers	0.5	0.8	0.3	1.3	2	1.5
Microphytobenthos	20	100	1.5	0.05	0.5	0.25
Seagrass	0.05	0.08	0.07	10	1.2	20
Macroalgae	4	5	0.1	0.05	0.75	1.2
Clupeoids	7	1.1	1.2	0.03	0.8	0.75
Flatfish	5	1	1	0.1	1	1
Piscivores	2	1.2	1.2	0.1	0.8	0.75
Mullet	0.05	0.75	0.75	4	1	0.75
Sharks	9	-	-	0.25	-	-
Detritus	2	2	2.5	0.3	0.8	0.5

(b) change in F scenarios

Group	Fishing pressure rises			Fishing pressure drops		
	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Phytoplankton	1	1.1	1	1	0.1	1
Small zooplankton	0.75	0.05	1.2	1.2	2	1.2
Large zooplankton	1.5	2	1.5	0.75	0.01	0.75
Deposit feeders	1	1	1	1	1	1
Filter feeders	0.5	0.001	0.8	1.2	1.1	1.1
Infaunal predators	1	1	1	1	1	1
Epifaunal predators	1.2	0.005	1.2	0.9	1	0.9
Benthic grazers	0.75	0.001	0.75	1.2	4	1.2
Microphytobenthos	1	1	1	1	0.5	1
Seagrass	1.2	1.2	1.2	0.9	0.9	1
Macroalgae	1	1	1	1	1.5	1
Clupeoids	0.3	0.1	0.2	1.5	2	1.5
Flatfish	0.5	0.05	0.12	1.2	2.5	2
Piscivores	0.8	0.2	0.15	1.1	2.5	2
Mullet	0.12	0.1	0.3	1.5	2	1.5
Sharks	0.25	-	-	1.5	-	-
Detritus	1	0.4	1	1	1	1

of top-down fishing-imposed pressures as effectively as the formulation implemented in ECOSIM.

Beyond these general observations, one of the most notable points is that microphytobenthos increases by two orders of magnitude in BM2 under rising nutrients, while macroalgae declines to negligible levels in IGBEM. Both of these responses can be traced back to the size of the phytoplankton blooms produced by each model as nutrient input rises. IGBEM produces quite intense blooms, which starve the underlying macrophytes of light and so even the macroalgae cannot compete. This is not the case in BM2 where only moderate blooms occur. In this case enough light reaches the sediment, but epibenthic fouling kills off the seagrass allowing microphytobenthos to take its place and spread from that foothold. Both of these patterns have been identified in eutrophied systems (Conley 1999, Herbert 1999, Lotze et al. 1999), though the pattern produced by IGBEM is more common.

Consideration of the system indices (Table 7.9) in conjunction with the patterns of biomass change again shows that consideration of multiple indicators is necessary for the formation of a complete picture. Between them, the Mean Trophic Level indices and the Total Biomass do capture the fact that the model systems shift in much the same way, just to different extents. However, they don't reflect the changes in the levels of biomass that occur in some groups.

Across all the measures used to compare the models, it is noteworthy that there is no consistent grouping of the models. Given their similar origins and internal premises one might expect that IGBEM and BM2 would behave in a similar way. This is true for the groups in the nutrient models that are modelled in a similar fashion, such as the fish groups. It is not the case for many of the other groups where the formulations are quite different between IGBEM and BM2 (the differences noted in Table 7.1 are strongest in the non-fish groups). For instance the water column groups in BM2 do not

Table 7.9: System-level indices for all the simulations. The bracketed figure for Mean Trophic Level Catch for ECOSIM and BM2 is the value when only finfish are considered as in IGBEM.

STATUS QUO SIMULATION						
System Index	ECOSIM	BM2	IGBEM			
Mean Trophic Level	1.98	2.16	2.05			
Mean Trophic Level Catch	2.8 (3.19)	2.41 (3.2)	3.28			
Total Biomass	217.45	359.0	300.69			
BASE CASE SCENARIOS						
Nutrient input rises			Nutrient input drops			
System Index	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Mean Trophic Level	2.09	2.21	2.37	2.0	2.22	1.8
Mean Trophic Level Catch	2.73 (3.21)	2.74 (3.08)	3.27	1.97 (2.26)	2.84 (3.09)	3.32
Total Biomass	1731.65	629.57	646.22	23.13	159.83	196.21
Fishing pressure rises			Fishing pressure drops			
System Index	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Mean Trophic Level	1.99	2.26	2.01	2.01	1.98	2.15
Mean Trophic Level Catch	2.98 (3.23)	2.76 (3.12)	3.37	2.68 (3.2)	2.91 (3.04)	3.26
Total Biomass	218.48	331.95	289.48	220.21	245.40	314.65
FISHERIES POLICY RUNS						
Economic Strategy			Ecological Strategy			
System Index	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Mean Trophic Level	1.97	2.1	2.28	2.01	2.24	2.49
Mean Trophic Level Catch	2.87 (3.2)	2.86 (3.42)	3.48	2.72 (3.21)	2.29 (3.04)	3.64
Total Biomass	215.3	134.62	259.86	220.42	381.72	316.25
Nutrients Rise						
Split Economic			Split Ecological			
System Index	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Mean Trophic Level	2.00 – 1.95	2.24 – 2.22	2.41 – 2.46	2.00 – 2.03	2.20 – 2.23	2.38 - 2.44
Mean Trophic Level Catch	2.76 – 2.6 (3.20 - 3.20)	2.82 – 2.89 (3.06 - 3.08)	3.52 – 3.55	2.80 – 2.69 (3.21 - 3.20)	2.84 – 2.91 (3.04 - 3.09)	3.46 - 3.54
Total Biomass	217.44 – 1401.16	387.44 – 805.4	393.84 – 396.47	217.31 – 1366.65	316.89 – 833.56	383.05 – 388.64
Nutrients drop						
Split Economic			Split Ecological			
System Index	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Mean Trophic Level	1.98 – 2.04	2.2 – 2.31	2.39 – 2.41	2.01 - 2.06	2.21 – 2.28	2.61 – 2.43
Mean Trophic Level Catch	2.88 – 2.02 (3.16 - 2.20)	2.84 – 2.90 (3.04 - 3.09)	3.51 – 3.52	2.86 - 1.93 (3.21 - 2.74)	2.52 – 2.13 (3.03 - 3.05)	3.66 – 3.56
Total Biomass	216.68 – 39.5	316.1 – 271.29	258.01 – 232.93	220.18 – 39.98	349.42 – 308.6	301.59 – 226.64

react as strongly as those in IGBEM, which respond similarly to those in ECOSIM (despite the omission of light and nutrient limitation from the ECOSIM formulation).

Summary of conclusions for “base case” results

The history of each model’s development is quite telling and the nutrient models may have benefited from more tuning. In spite of this a number of general conclusions can be drawn. The ECOSIM model is not as buffered against nutrient related changes as the nutrient models are, but it is more buffered against changes in the fisheries. The spatial structure of the nutrient models also allow for the expression of some more complex model behaviour. In general the behaviour of the low to middle trophic groups in the nutrient models is probably more realistic than that in ECOSIM, but the higher trophic groups react more sensibly in ECOSIM than the nutrient models. This is probably another consequence of their respective focus and development histories.

7.3.B Fishing policy analysis

The open and closed loop policy optimisation features of the ECOSIM software are used to explore “optimal” fishing strategies (F_s for each fishery) given a range of economically and ecologically weighted objective functions.

The main strategies suggested by the policy analysis are then implemented in the nutrient models to see if the outcomes agreed with those predicted by ECOSIM. For ease of understanding, the ECOSIM optimisations will be presented and then the outcomes of the nutrient models will be discussed. This two step presentation will then be repeated for the optimisation done when nutrient forcing changes during the period of the simulation. As with the comparisons discussed above, there is usually qualitative agreement between the models, though specific responses can vary widely.

ECOSIM, with no change in nutrients

The results of the ECOSIM policy analyses under constant environmental conditions are summarised in Table 7.10. A full description is given below, but in short, over the entire range of objective function weightings trialed, only three results are produced: an economically based strategy, an ecologically based strategy, and a compromise between the two. These outcomes also persisted, with very little change, across a range of refuge parameter settings used in a sensitivity analysis performed on the model and optimisation process.

Table 7.10: Results of the policy analyses under constant environmental conditions. The Fs given are those found by the ECOSIM open loop analysis (with a *Status Quo* entry included for comparative value) and the summary statistics are for the ECOSIM output under the suggested Fs. The Overall Average Total Catch and Total Value indicate the cumulative totals over the entire run. The “Ratio End/Start Values” are the ratios of the instantaneous total catch, economic value and biomass at the beginning and end of the run.

Strategy					
	<i>Status Quo</i>	Economic	Ecological (no mand.)	Ecological (mand)	Compromise
Weighting					
Economic	-	1	0.0001	0.0001	0.5
Social	-	0.0001	0.0001	0.0001	0.5
Mandated rebuilding	-	0.0	0.0	10	1
Ecosystem	-	0.0001	1	1	1
Estimated Relative Fs					
Purse seine	1	1.9	0.9	0.4	1.7
Scallop dredge	1	2.0	0.2	0.2	3.0
Haul seine	1	0.5	0.005	0.02	0.2
Longline	1	1.2	0.04	0.06	0.5
Mesh nets	1	20.1	0.3	0.2	1.1
Dive	1	1.3	2.6	0.4	1.2
Pots	1	0.75	0.5	0.2	0.6
Aquaculture	1	1	1	1	1
(omitted from search)					
Overall Average					
Total Catch (t/km ²)	880	1420	570	376	1161
Total Value (x10 ⁶)	1.2	1.9	0.7	0.5	1.4
Ratio End/Start Values					
Total Catch	1	0.56	0.97	1.15	0.76
Total Value	1	0.77	0.4	1.46	0.89
Total Biomass	1	0.79	1.05	1.1	0.91

The “economic” strategy sees the F s increase, relative to current levels, for all but the haul seine and pot fisheries. The mesh net fishery undergoes the most extreme change as the optimisation suggests increasing its F to 20x current levels (Table 7.10). The resulting biomasses (Figure 7.7) indicate that under the proposed set of F s, the sharks and flatfish are effectively extirpated. By reducing these predatory groups, their higher value prey species can flourish, thus improving the economic performance to about 1.5x *status quo* levels. Among the invertebrate fisheries, the increase in F produces a moderate decline in the biomass of the high value species, abalone and scallops. Within the bycatch species the most important effects are that, under this economically optimal set of F s, the big piscivorous finfish and the marine mammals decline to about half of their *status quo* levels. The drop in these groups furthers the release from predation of the high value prey species (King George whiting and southern rock lobster, see Figure 7.7), the mechanism apparently motivating the very large increase in F for the meshnet fishery. In spite of the overall increase in total catch and value relative to *status quo*, the widespread depletion of so many groups causes the annual catch and value to drop by nearly a quarter and biomass by almost half from the beginning to the end of the simulation.

The “economic” strategy shows the least sensitivity to changes in the refuge parameters, though the species currently considered most heavily impacted by fishing (snapper, King George whiting and southern rock lobster) do vary a little from case to case. Overall, the only significant change is when all v 's are set to 0.7, which sees a flip to a result more closely resembling the standard “ecological” strategy described below.

A variety of weightings of the ecological objective function all produce largely similar results (Figure 7.8), with lower F s for the majority, if not all, the fisheries (Table 7.10). The one fishery that is sensitive to the ecological objective function and criteria chosen is the dive fishery for abalone. When mandatory population restoration is given

Figure 7.7: Plot of ECOSIM biomass trajectories under the Fs that result from an economically oriented objective function. Note that, in the interests of plot clarity, interannual variation in forcing has been removed here.

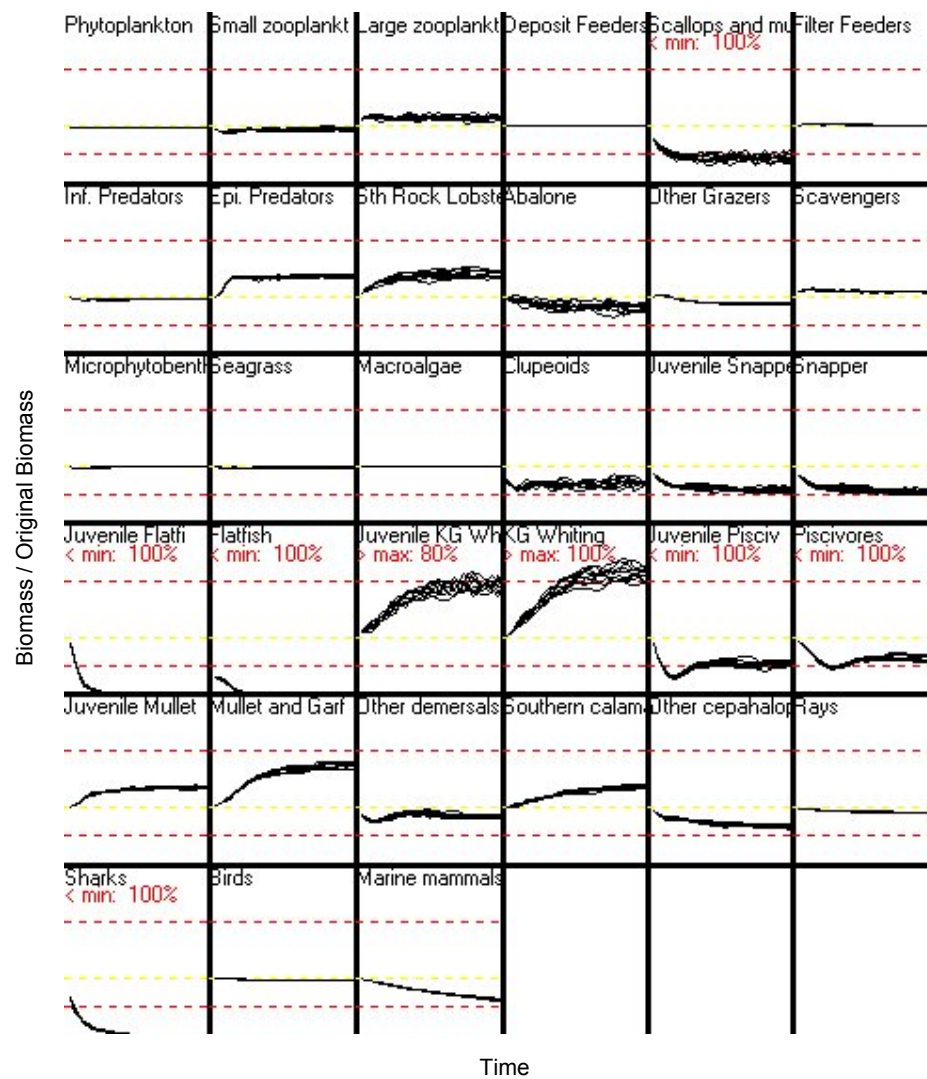
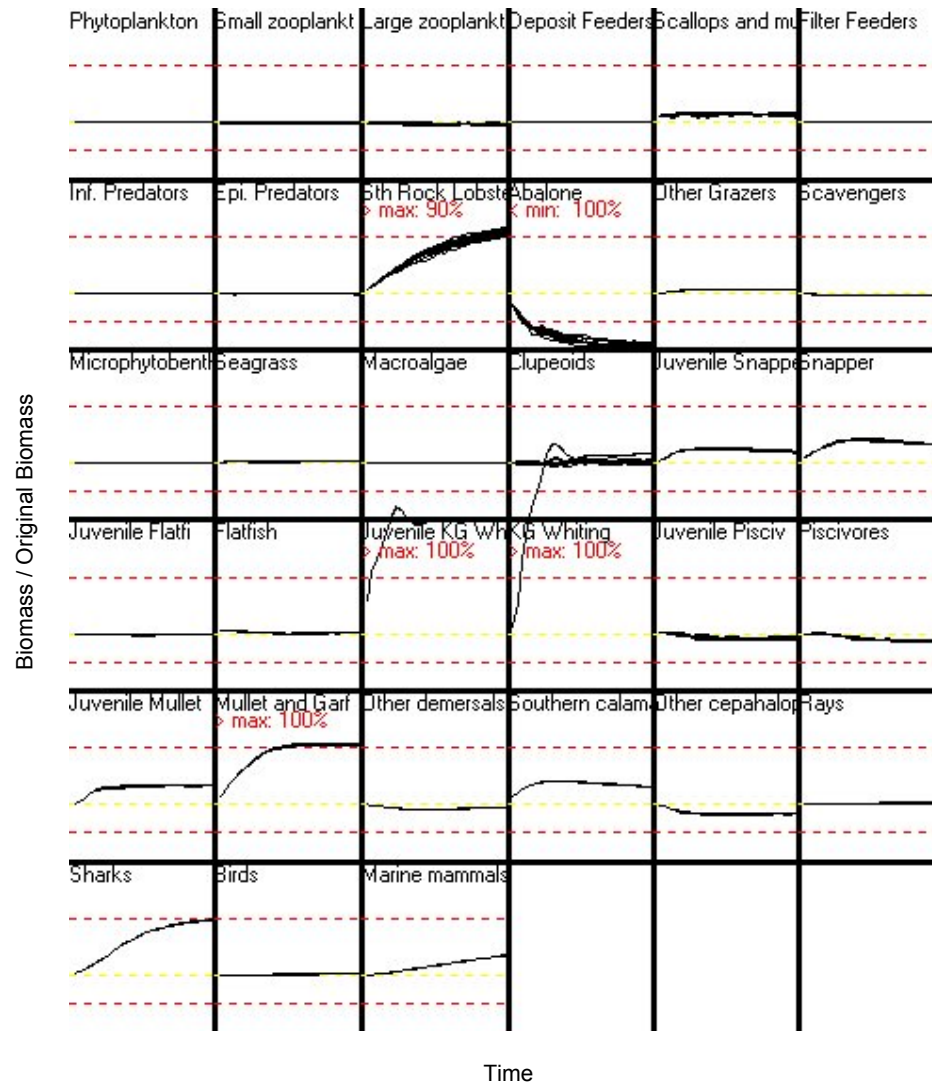
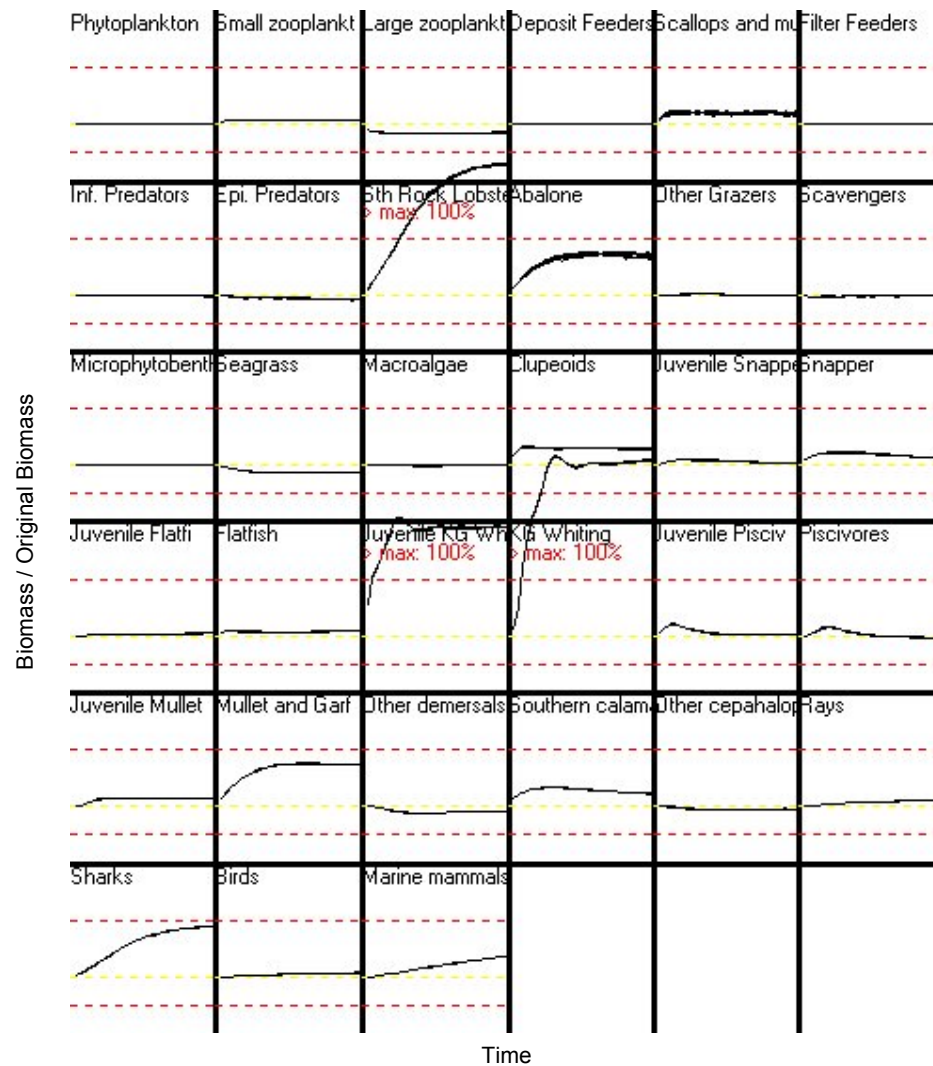


Figure 7.8: Plot of ECOSIM biomass trajectories under the Fs that result from an ecologically oriented objective function.

(a) with no mandatory stock restoration implemented,



(b) with some mandatory stock restoration implemented and ecological weighting given to all species not just those of immediate social interest. Note that, in the interests of plot clarity, interannual variation in forcing has been removed here.



little or no weight, the F for abalone fishery actually increases. This is probably due to its competition with the mullets (a fish identified as a group to be conserved or restored) and its very high market price (which is still of some importance given the small but non-negligible economic weight in the objective function). This results in the abalone biomass being depressed to the level of commercial extinction (Figure 7.8a) and the annual total value, for that run, falling by more than half. In contrast, when mandatory population restoration and maintenance are given at least some weight, the F for the dive fishery either remains at current levels or falls with the other fisheries. This indicated how critical the ecological criteria can be to certain species. If ecological concern is focused only on the charismatic or favoured recreational species then other components of the ecosystem, such as the abalone, continued to be quite intensively exploited (Figure 7.8a). When ecological criteria emphasise at least some conservation of all groups a much more balanced, mature system results (Figure 7.8b). However, this balance comes at the expense of the landed catch, with the total catch falling to about a quarter of that taken under the economic objective, while the total value is down by more than two thirds. The low F s do allow the target groups to grow steadily though and the annual catch, value and biomass all rise from the beginning to the end of the period. Even with a low exploitation rate, the value of the pot fishery more than doubles, increasing the average total value of all fisheries.

The “ecological” strategy is more heavily impacted by changes in the refuge parameters, with the greatest variation in behaviour shown in the high value epifaunal groups, as well as in the cephalopods and the snapper groups. Nevertheless, there is little qualitative change in the predictions. The most significant variations are observed at low refuge parameter settings ($v=0.2$), where a more “even” ecosystem (one not so heavily skewed toward the charismatic species) results regardless of the explicit ecological criteria specified.

Attempts to find a compromise between the ecologically and economically based optimisations do not identify a smooth transition from one to the other. Rather, as the weightings are gradually changed from economically to ecologically dominated, the “economic” strategy given above persists as the optimal outcome for a wide range of weightings. It is not displaced as the optimal policy outcome until the point where the economic:ecological weight equalled 0.71:1. At this point there is a flip-point, which sees the optimal outcome returned as either the “economic” or “compromise” strategy depending on the initial start point of the search routine. The “compromise” strategy (Figure 7.9 and Table 7.10) closely resembles the “economic” strategy and is apparently heavily influenced by the economic contribution to the objective function. This is obvious not only from the pattern of biomass change, but also from a majority of the F settings, the total catch and value of the entire run and the instantaneous catch, value and biomass results. A direct comparison of the important statistics for the three main optimisation solutions can be found in Table 7.11. Many of the values for the “compromise” strategy are very similar to those for the “economic” strategy. However, the ecosystem contribution is still present and showed up in the values of the predatory species, which are depressed under the “economic” strategy. These groups do not suffer the excessive depletion under the compromise strategy. The biomass of the snapper groups reverses the pattern of change seen under the “economic” strategy and increases rather than decreases; while that of the sharks and marine mammals remain at *status quo* levels and the flatfish are not depleted to the point of extinction. As the objective function weightings are moved still further, this “compromise” strategy is the only outcome returned until the point where the economic:ecological weight equalled 0.5:1. From this point on, as the economic weighting is reduced still further, the optimal strategy is the “ecological” strategy.

Figure 7.9: Plot of ECOSIM biomass trajectories under the Fs that result from an objective function that is weighted for a compromise of the ecological and economic strategies. Note that, in the interests of plot clarity, interannual variation in forcing has been removed here.

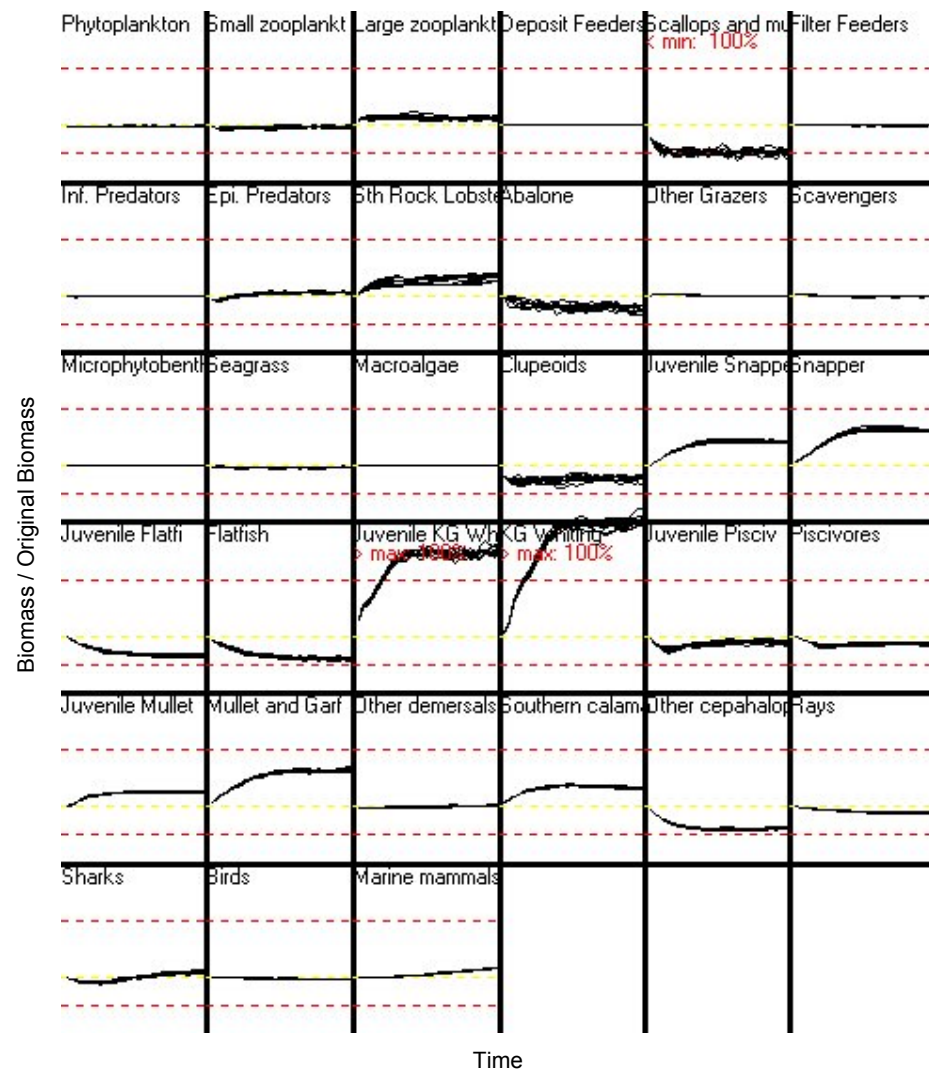


Table 7.11: Comparison of the important “economic”, “ecological” and “compromise” strategies. The “ecological” strategy refers to the case where there is mandatory restoration of groups. The relative change in biomass indicates the ratio of the biomass that results from the implementation of the fisheries policy: the *status quo* biomass for the group. Only groups that demonstrate significantly different biomass trajectories between the various strategies are given. For flatfish and snapper the juvenile and adult groups act in the same way, so only the values for the adult group are provided.

	Strategy		
	Economic	Compromise	Ecological
Estimated Relative Fs			
Purse seine	1.9	1.7	0.4
Scallop dredge	2.0	3.0	0.2
Haul seine	0.5	0.2	0.02
Longline	1.2	0.5	0.06
Mesh nets	20.1	1.1	0.2
Dive	1.3	1.2	0.4
Pots	0.75	0.6	0.2
Relative change in biomass			
Large zooplankton	1.1	1.1	0.9
Scallops and mussels	0.7	0.7	1.2
Epifaunal predators	1.4	1.0	0.9
Southern rock lobster	1.35	1.4	3.0
Abalone	0.8	0.9	1.5
Seagrass	0.95	0.95	0.9
Clupeoids	0.9	0.9	1.1
Snapper	0.9	1.4	1.2
Flatfish	<0.01	0.8	1.1
Other cephalopods	0.75	0.6	0.9
Rays	0.95	0.9	1.1
Sharks	<0.01	1.5	1.9
Mammals	0.75	1.2	1.4

Table 7.12: The relative change in biomasses (biomass under new policy / “base case” biomass) for each of the three models under the ecological and economically based strategies. The lumping and association of groups noted for Table 7.6 also apply here and sharks are given for ECOSIM.

Group	Economic Strategy			Ecological Strategy		
	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Phytoplankton	1	2	2	1	1	1.5
Small zooplankton	1	1	3	1	0.9	4
Large zooplankton	1	4	1.2	1	1.2	0.9
Deposit feeders	1	5	2	1	0.75	0.75
Filter feeders	0.5	0.01	1.5	1.2	1.2	2
Infaunal predators	1	1	1.5	1	1	1.5
Epifaunal predators	1.5	0.1	1.5	1.7	3	1.5
Benthic grazers	0.9	2	0.5	1.5	1.5	0.3
Microphytobenthos	1	2	1	1	1	1
Seagrass	1	1	1	0.75	1.1	20
Macroalgae	1	1	2	1	1	3
Clupeoids	0.8	0.5	0.5	1	1	1
Flatfish	0.1	0.01	0.02	1.5	3	1.5
Piscivores	0.5	0.75	0.75	0.75	5	2
Mullet	1.5	2	1.2	2	2.5	2
Sharks	0.05	-	-	2	-	-
Detritus	1	0.5	2	1	1.5	1.5

“Optimal” policies implemented in nutrient models, with no change in nutrients

The sets of Fs associated with the two strategies (“ecological” and “economic”) identified by the ECOSIM policy optimisation routines are tested in IGBEM and BM2.

The relative changes in biomass for all three models are given in Table 7.12.

Under the ecologically oriented set of Fs there is very good qualitative agreement between the three models with regard to the biomass trajectories of the various groups. There are some minor differences between IGBEM and the other two models, due to the fact that IGBEM does not allow for the fishing of invertebrates. Attempts to mimic fishing mortality in these groups using general background (natural) mortality did not succeed. The majority of the differences seen between IGBEM and the other two models, in this case, are due to this failure and its cascade effects. The more

important divergence between the ECOSIM and nutrient model predictions is in the piscivorous groups. The nutrient models predicted that the piscivorous fish would increase, whereas ECOSIM suggests that the group would decline marginally. This difference is almost certainly due to the fact that the highest trophic levels (sharks, birds and marine mammals) are only static not dynamic components of IGBEM and BM2. In ECOSIM these groups are free to change with the system, and impact the piscivorous group accordingly.

When the economically driven set of Fs are applied in IGBEM and BM2, there is, again, wide qualitative agreement between the time-series produced by the models. In this case all the models are in agreement over the impact of the proposed fishing mortalities on the fish groups. The same does not hold true for the invertebrate groups. As with the ecological case, IGBEM predicts an increase in the filter feeders, while the other models do not. BM2 actually predicts a collapse in the population of filter feeders, leading to a significant drop in the epifaunal predators (despite the drop in F for the pot fishery). This in turn releases the benthic grazers and allows that biomass to grow (despite the increase in F for the dive fishery). Thus, due to food web interactions, two of the three harvested invertebrate groups respond contrary to what would be expected from a simple consideration of the change in F of the fisheries targeting them.

Another important difference between the models is in the detritus, where there is no concurrence between any of the models – ECOSIM predicts no change, BM2 a twofold drop and IGBEM a twofold increase. This appears to be due to the dynamics of the major detritus producers and consumers in each model. The collapse of the flatfish and epifaunal predators in BM2 allows the deposit feeders in that model to increase by fivefold and this in turn depletes the standing stock of available detritus. In IGBEM the major producers of detritus (the primary producer groups) all increase, leading to an increase in the amount of detritus. The biomass of deposit feeders in IGBEM does

increase a little, but not to the extent seen in BM2 as the epifaunal carnivores also increase and kept the deposit feeders in check. Consequently, the detritus remains slightly higher than the *status quo* level. In ECOSIM there is no change in the biomass of the major producers or consumers of biomass so the detritus remains unchanged.

The system indices (Table 7.9) do capture the general shifts in the model systems and the fact that the shift differs between the models. The much larger response by the harvested groups in the nutrient models is reflected in the much greater changes in Total Biomass, relative to the *status quo* value (Table 7.9), displayed by these models. Further, the three indices taken as a whole also highlight the fact that overall ECOSIM does not respond as strongly over as many groups as the nutrient models do. However, these system-level indices do not capture the very large responses in a few of the groups.

Change in F scenarios vs the results of the optimisations

At face value, the “economic” strategy could be equated with the scenario where there is a general increase in F (Table 7.8b). Similarly, the “ecological” strategy could be equated to the scenario where there is a general decrease in F (Table 7.8b). Comparing these results, a few very important differences can be observed between the patterns of change in the two cases.

For ECOSIM, the most notable contrast is that a general (across the board) rise in F results in a decline in the mullet, whereas this group increases under the “economic” strategy. Several differences are apparent for the nutrient models. In IGBEM, the response of two groups differs in comparing the “ecological” strategy with the general decrease in F. The benthic grazers fall under the “ecological” strategy (Table 7.11), while they do not change at all under the general decrease in F (Table 7.8b). In turn this contributes to a sizeable increase in the biomass of seagrass. Comparing results

from the four strategies for BM2, many more groups show very different responses in biomass. Comparing the “ecological” strategy with the “drop in F”, the greatest contrast occurs with the phytoplankton and large zooplankton. They remain relatively steady under the “ecological” strategy, but both groups decline severely in the scenario where there is a general decrease in F. Comparing the “economic” strategy with the “rise in F”, a number of groups show different responses. The small zooplankton, deposit feeders, benthic grazers, epifaunal predators and mullet are all much higher under the “economic” strategy than under the general increase in F.

These differences appear to be due to at least two proximate factors. First, the changes in F under the optimal policies are not all of the same size (as is the case under the blanket change in the scenarios) and some fall or rise by much more (or less) than fivefold. Secondly, under the “economic” strategy the Fs do not increase for all groups (the Fs for the Haul seine and Pot fisheries actually fall). Both these factors can clearly lead to differences in biomass response between the fished groups. However the differences in the non-fished groups are not directly attributable to changes in F. Several groups that are predators, prey or competitors of the harvested groups show very different responses between the scenarios of broad changes in F and the specific sets of F suggested by the optimisations. This is something single species models would obviously not be able to predict.

Fishing policy analysis by ECOSIM, with a change in nutrients

A summary of the outcomes of the ECOSIM policy analyses under changing nutrients and ecological or economic criteria is given in Table 7.13.

For an economically oriented objective function, there is little difference in Fs between the cases of constant and decreasing nutrients. In contrast, Fs under

Table 7.13: Results of the policy analyses under changing environmental conditions. The Fs given are those found by the ECOSIM open loop analysis and the summary statistics are for the ECOSIM output under the suggested Fs. The Overall Average Total Catch and Total Value indicate the cumulative totals over the entire run. The “Ratio End/Start Values” are the ratios of the instantaneous total catch, economic value and biomass at the beginning and end of the run.

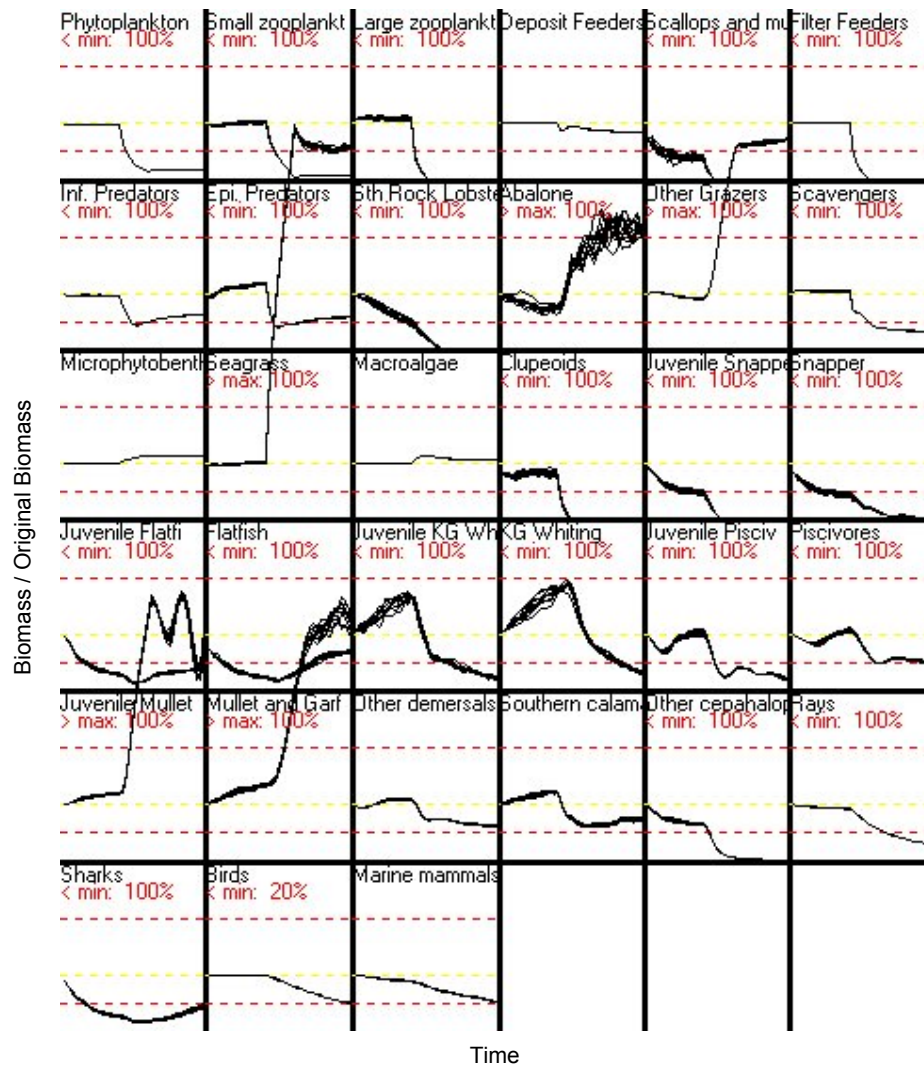
	Nutrients Rise						Nutrients drop					
	Single			Split			Single			Split		
	Economic	Ecological	Single	Economic	Ecological	Split	Economic	Ecological	Single	Economic	Ecological	Split
Estimated Relative Fs												
Purse seine	1.3					1.1 – 1.15	0.7	1.6 – 1.1	1.8	1.75 – 1.3	0.4	0.1 – 1.1
Scallop dredge	2.1					0.8 – 1.5	1.5	1 – 1.2	2.45	2 – 0.5	0.2	0.08 – 0.3
Haul seine	1.25					0.8 – 1.1	0.5	0.6 – 1	0.6	0.5 – 1	0.02	0.02 – 0.02
Longline	0.85					1.2 – 1.3	0.9	0.6 – 1.1	1.9	2.5 – 0.75	0.06	0.03 – 0.06
Mesh nets	0.65					1.4 – 1.1	1.1	0.7 – 1.9	15.0	11.9 – 0.7	0.1	0.01 – 0.3
Dive	0.9					1.2 – 0.7	1.3	0.5 – 1.2	1.9	1.2 – 2.4	0.45	0.25 – 0.6
Pots	1.5					1.1 – 2.1	0.9	1.7 – 1.4	1.1	0.9 – 1.2	0.1	0.1 – 0.1
Aquaculture	1					1 – 1	1	1 – 1	1	1 – 1	1	1 – 1
(omitted from the search)												
Overall Average												
Total Catch (t/km ²)	5267					4325	4030	4028	931	813	221	169
Total Value (x10 ⁶)	5.6					5.5	5.0	5.1	1.4	1.2	0.5	0.4
Ratio End/Start Values												
Total Catch	6.59					6.86	6.6	4.77	0.02	0.06	0.01	0.04
Total Value	6.15					6.03	5.77	4.56	<0.01	<0.01	<0.01	<0.01
Total Biomass	2.35					2.3	2.29	2.28	0.72	0.73	0.73	0.73

increasing nutrients diverge from those found to be optimal with no change in nutrients for all but the scallop dredge and purse seine fisheries. The optimal policy in this case is more a mix of the “economic” and “ecological” strategies when there is no change in nutrients. The Fs for the purse seine, longline and dive fisheries are basically averages of the “economic” and “ecological” strategies for those fisheries. The Fs for the scallop dredge, haul seine and pot fisheries are the same size as, or even larger than, those for the “economic” strategy baseline, while the meshnet fishery is almost as low as in the “ecological” strategy when there is no change in nutrients.

With an economically oriented objective function, the set of two policies (split-policies) chosen under a fall in productivity was intuitively reasonable. The policy begins by following the path of the “economic” strategy suggested when there is no change in nutrients. Once the change in productivity occurs the split policy changes from one resembling the “economic” strategy to the “ecological” one as the policy optimisation attempts to correct for the declines initiated by the drop in productivity (Figure 7.10). For rising nutrient inputs the response is somewhat different. Depending on the basal food group (phytoplankton or detritus vs macrophyte), some sub-webs increase substantially while others collapse. As a consequence, the objective function surface seems to have become highly complex and the best result that can be found is to stay fairly close to current Fs both before and after the change in conditions.

Under an ecological objective, the solutions produced under constant conditions and the single solution when productivity falls are substantially similar. However, in this case the Fs suggested for the meshnet and pot fisheries are about half of those under constant conditions – as would be expected given the reduction in the target groups of these fisheries with the fall in productivity. As with the economic objective, the single F spanning a rise in productivity is different from that for the constant conditions. The result is much closer to the “economic” than the “ecological” strategy obtained when

Figure 7.10: Plot of ECOSIM biomass trajectories under the split Fs that result from an economically oriented objective function applied when there is a drop in productivity. Note that, in the interests of plot clarity, interannual variation in forcing has been removed here.



there is no change in nutrients. All but the purse seine and meshnet fisheries have Fs at or just below the level of those for the constant “economic” strategy. For the two exceptions (purse seine and meshnet) there are substantial increases in F in comparison with constant conditions (rising by two and six-fold respectively). Despite this, these two fisheries are still closer to the standard “ecological” strategy.

When a split policy is implemented with an ecologically weighted objective

function, results under both increasing and decreasing productivity appear to be intuitively reasonable, though some of the details are surprising. In the case where productivity rises over time, the solution is as expected. It begins by paralleling the constant “ecological” strategy and then the F s generally rise after the increase in productivity takes effect. The F s for the longline, dive and pot fisheries reach the levels of those in the “economic” strategy. Under a fall in productivity the solution once again begins by tracking the “ecological” strategy. Interestingly, even though the absolute F values continued to remain low after the conditions changed (all but the dive fishery remaining closer to the “ecological” than the “economic” strategy), they do increase in relative terms (rising by between two and tenfold). It is unclear exactly why this occurs. However, one suggestion is that in finding the F s for the second half of the split-policy, the optimisation is responding to the increase in biomasses that occur as a result of the low F s set in the first half of the policy and raises F s accordingly.

One clear result emerges from comparison of the overall catch and objective function values for each of the runs, as well as the ratios of instantaneous catch, value and biomass at the beginning and end of each run. The effects of the change in productivity all but overwhelm any due to the fishing strategies implemented. If productivity rises, then all these summary statistics rise too, regardless of the strategy. Similarly, if productivity falls then all the summary statistics fall whether an “economic” or “ecological” strategy is implemented. However, the effects of the policies are not completely subsumed by the effects due to the changes in nutrients. The “economic” strategies continue to produce total catches and values, for the entire period, that are appreciably higher than those produced by the “ecological” strategies. The disparities in the values are much more striking for the case where there is a drop in productivity. The “ecological” total catches are about a quarter and the total values are about a third of the “economic” ones. Under a rise in productivity the “ecological” total

catches are only 5 – 25% smaller, and the total value only 5 – 10% smaller, than those from the “economic” strategies. The values of the ratios of the instantaneous measures are not so neatly partitioned along policy lines. When there is a rise in productivity the three measures are similar for both the single solution policies and the split “economic” strategy, though the single “ecological” strategy is marginally lower. In comparison to these three policy outcomes, the split “ecological” strategy produces much lower values for all three indices. The similarity among the values is not surprising given the very similar Fs in each case. Moreover, the split “ecological” strategy actually results in the greatest general increase in Fs over the course of the run. Consequently, it yields the smallest increase in the three measures as the higher Fs mitigate, to a limited extent, the effects of the higher productivity. In those cases where the productivity drops, the effects of the change in productivity almost completely overwhelm the effects of the policies. The only values, for any of the three relative measures, which differ amongst the various policies are those for the ratio of catches. Both the split policies have a higher ratio of catch than the single policy cases.

“Optimal” policies implemented in nutrient models, with a change in nutrients

The split-policy Fs for both the nutrient change scenarios (rise and fall), identified in the ECOSIM policy analysis, are applied in BM2 and IGBEM. The biomass trajectories in IGBEM (Table 7.14a and b) do largely concur with those of ECOSIM, at least at a qualitative level, though ECOSIM tends to result in more extreme changes in biomasses (Table 7.14a). This is true for both the ecological and economically driven Fs for both of the nutrient cases. As with the comparisons for no change in nutrients, there are some differences in the biomasses predicted for a few of the groups that are treated differently in the various models.

The dynamics of the piscivores in the nutrient models do not closely match those

in ECOSIM, particularly for the “ecological” strategies under falling nutrients and the “economic” strategies under rising nutrients. In both of these cases ECOSIM predicts a large fall in biomass while the other models suggest a small to negligible rise. Once again, this appears to be due to the static representation of the birds, sharks and mammals in IGBEM and BM2. This combined with the changing fishing pressures helps to force all of the fish groups in a different direction to that of ECOSIM, with its dynamic handling of all groups. The effect is amplified by the differing stock recruitment assumptions used in the models. For instance, under either policy, the case of falling nutrients sees ECOSIM predict large falls in flatfish, whereas IGBEM and BM2 predicted modest rises in the long-term as their benthic prey groups only fell marginally and the final F s are lower than current levels. The impact of the initially high F under the “economic” strategy soon disappears from the nutrient models due to their constant recruitment term. In contrast, for the herbivorous and planktivorous fish the same mechanism prevents the nutrient models from allowing the large increases in biomass that ECOSIM predicts, as the constant recruitment in the nutrient models does not allow for the positive feedback between stock size and recruitment. The models also differed with regard to the dynamics of the abalone biomass. While all the models produced similar outcomes in the case of the ecologically based F s under decreasing productivity, this is not the case any of the other scenarios. This cannot be explained by the failure of the ‘invertebrate fishing fix’ employed in IGBEM, but apparently arises from the differences observed in the dynamics of the macrophyte groups. ECOSIM consistently predicts a much smaller change in these groups with changes in nutrients, which could be traced to a lack of a shading effect by phytoplankton and other suspended material in the ECOSIM model. It is conceivable that a mediation or forcing function could be built into ECOSIM to mimic this, but this was not attempted. These long-term predictions do not take into consideration interim stock dynamics due to the

Table 7.14: Changes in biomass that result from the implementation of the suggested optimal fisheries policies.

(a) The relative change in end-point biomasses (end biomass under new policies and nutrient change / end biomass in “base case”), for each of the three models under the ecological and economically based strategies.

Group	Nutrients Rise						Nutrients drop					
	Split Economic			Split Ecological			Split Economic			Split Ecological		
	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Phytoplankton	6	0.2	3	5	1.5	5.5	0.15	1	0.5	0.2	0.8	0.5
Small zooplankton	7	2	2.5	7	2	3	0.1	0.5	0.5	0.1	0.9	0.75
Large zooplankton	8	0.75	1.6	6	1.5	2	0.1	1.5	0.9	0.1	0.8	0.9
Deposit feeders	4	2	2.5	4	2	2	0.5	0.5	0.75	0.5	0.75	0.75
Filter feeders	8	1.3	2	8	1.5	2	0.13	1.1	0.75	0.13	0.9	0.9
Infaunal predators	4	1	3	8	1	3	0.2	1	0.75	0.3	1	0.9
Epifaunal predators	10	1.2	2	10	1.2	2	0.1	0.15	0.2	0.1	0.75	0.3
Benthic grazers	0.5	4	0.1	10	0.3	0.1	0.7	3	1.2	2	2	1.3
Microphytobenthos	8	4	0.6	10	100	1	0.1	0.75	2	0.1	1.2	1.1
Seagrass	0.1	0.1	0.3	0.1	0.01	0.2	7	1.4	1.1	10	0.8	1.2
Macroalgae	2	1.5	5	10	1.5	3	0.1	0.2	0.01	0.1	0.01	0.01
Clupeoids	6	1.2	1.2	8	0.5	1	0.1	1.5	0.8	0.1	1	1
Flatfish	4	0.75	0.8	2.5	1.5	0.6	0.1	1.3	1.2	0.1	2	2
Piscivores	3	0.8	1	5	2	0.9	0.1	1.5	0.75	0.1	2	1.5
Mullet	1.5	2	1.5	4	1.1	1.2	5	1	1	10	4	1.5
Sharks	9	-	-	9	-	-	0.05	-	-	1.5	-	-
Detritus	2.5	2	2	3	2	2	0.5	0.9	1.5	0.5	0.13	0.75

(b) The direction of biomass change for the group once the nutrient and fishing policy changes have occurred (‘/’ stands for increase, ‘\’ indicates a decrease and ‘–’ represents no significant change). The lumping and association of groups noted for table 7.6 also apply here and sharks are given for ECOSIM.

Group	Nutrients Rise						Nutrients drop					
	Split Economic			Split Ecological			Split Economic			Split Ecological		
	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM	ECOSIM	BM2	IGBEM
Phytoplankton	/	\	/	/	/	/	\	-	\	\	/	/
Small zooplankton	/	/	/	/	/	/	\	\	\	\	\	\
Large zooplankton	/	\	/	/	/	/	\	\	\	\	/	\
Deposit feeders	/	/	/	/	/	/	\	\	\	\	\	\
Filter feeders	/	/	/	/	/	/	\	/	\	\	/	\
Infaunal predators	/	/	/	/	-	/	\	\	\	\	-	\
Epifaunal predators	/	/	/	/	/	/	\	\	\	\	\	\
Benthic grazers	\	\	\	\	\	\	\	\	-	/	\	\
Microphytobenthos	/	/	-	/	/	-	\	\	-	\	/	/
Seagrass	\	\	\	\	\	\	/	/	-	/	-	/
Macroalgae	\	/	/	/	/	/	\	\	\	\	\	\
Clupeoids	/	/	/	/	/	/	\	/	-	\	\	\
Flatfish	/	/	/	/	-	/	\	/	/	\	-	\
Piscivores	/	/	/	/	-	\	\	/	/	\	\	\
Mullet	-	/	-	/	-	-	/	-	-	/	-	/
Sharks	/	NA	NA	/	NA	NA	\	NA	NA	/	NA	NA
Detritus	/	/	/	/	/	/	\	\	\	\	\	\

split policy structure.

When the patterns of biomass change are considered (Table 7.14b) rather than the change in final biomasses, the observed biomass trajectories are similar under the various policies and nutrient conditions for the three models. The biomass trajectories corresponded for more than 91% of cases under ecologically based strategies, and for more than 96% of cases under economically based strategies. Of the three models, BM2 is the one that most often produces a different direction of change in biomass to the other two models.

The system indices (Table 7.9), in particular Total Biomass, reinforce the impression that ECOSIM is much more responsive than the other two models. The Mean Trophic Level indicated that as a whole the system usually shifts in the same way, for each of the models, as the policy and productivity changes came into effect. However, the bycatch groups that are taken in ECOSIM see its Mean Trophic Level for Catch drop slightly rather than rise slightly, as is the case for the other models.

Nutrient scenarios vs the results of the optimisations with a change in nutrients

Given the overwhelming impact of the productivity on the total values and catch, this section provides a comparison of the scenarios where there is a change in nutrients but no change in policy, with scenarios where the optimal policies under changing nutrients are found. Under each of the split policies, the mullets were less intensively exploited than in the “base case” (as the Haul seine had an $F < 1$). This allows mullet biomass to increase during the first twenty years of the simulation. This resulted in higher biomass when the productivity changes, which in turn impacts on the benthic primary producers and their biomasses. There are also impacts on other groups such as a few of the planktonic groups, which are not directly affected by the change in Fs. An interaction between the change in productivity and the Fs is possible, but the

exact mechanism is unclear.

Conclusions from fishing policy analyses

Analysis of the fisheries strategies reinforces the impression that ECOSIM is more sensitive to changes in nutrients, and that the nutrient models are more sensitive to changes in F. The analysis also highlights the fact that trophic models predict unexpected changes in species not directly impacted by fishing. Such effects cannot, of course, be predicted from single species models.

7.4 Discussion

It is now widely accepted that human activity has had a significant impact on biogeochemical cycles at local, regional and even global scales (Ver et al. 1999). One response to this realisation has been the call for an ecosystem perspective in assessing and managing human impacts such as fishing and nutrient discharge. Developing the tools to meet this challenge has proved demanding for scientists. Until recently, limitations in knowledge and technological limits in computing have constrained attempts to model marine ecosystems. Despite earlier attempts (such as Andersen and Ursin 1977) and derived methods (for instance MSVPA (Sparre 1991)), comprehensive marine ecosystem models have not been widely available until the last decade, and their utility as predictive tools is still questioned by many. This paper sheds light on at least one aspect of the “robustness” of ecosystem models, by comparing different models developed for the same marine ecosystem, and examining some of their policy implications.

Four general conclusions appear to emerge from the analyses reported in this paper (they have been summarised in Table 7.15).

Table 7.15: Summary of the major conclusions and supporting results from the three ecosystem models considered here.

Major Supporting Evidence	
ECOSIM	IGBEM
Summary of Conclusions	BM2
Multi-species effects: Changes in biomass and productivity that single species models would not identify as they are due to interspecies interactions.	Best multispecies example from each model: The direct (bycatch) effect of fishing on top predators is compounded by indirect (prey removal) effects.
Sensitivity to nutrients rather than fishing: Changes in nutrients have a much larger, and more widespread, impact than changes in fishing pressure	Benthic grazers rise despite an increase in fishing pressure (economic strategy). An increase in Chl a as a result of increased fishing pressure on clupeoids.
Choice of indicator groups: The dynamics of groups such as seagrass, sharks and chlorophyll a (as a proxy for phytoplankton) consistently characterise the behaviour of many other groups and may summarise wider system responses and changes.	Ratio nutrient/fishing pressure induced change in biomass for each model: Approximately 1.5 to 60. 1.5 to >1000. Approximately 1.5 to 20.
Choice of indicator groups: The dynamics of groups such as seagrass, sharks and chlorophyll a (as a proxy for phytoplankton) consistently characterise the behaviour of many other groups and may summarise wider system responses and changes.	Linked pairs of Groups – Indicator reflecting the behaviour of those groups in each model: Planktonic – Phytoplankton Benthic Herbivores – Seagrass Herbivorous Fish – Seagrass +sharks Other Fish – Sharks Harvested groups – Sharks Other Benthos – Phytoplankton Top predators – Sharks Planktonic – Chlorophyll a Benthic Herbivores – Seagrass Herbivorous Fish – Seagrass + piscivores Other Fish – Piscivores Harvested groups – Piscivores Other Benthos – Chlorophyll a + detritus Planktonic – Chlorophyll a Benthic Herbivores – Seagrass Herbivorous Fish – Seagrass + piscivores Other Fish – Piscivores Other Benthos – Chlorophyll a

Table 7.15: Continued

Major Supporting Evidence		IGBEM	
Summary of Conclusions	ECOSIM	BM2	IGBEM
Robustness to model formulation: Many results showed qualitative agreement across models, but there were a few important differences. These differences usually occurred when one or the other of the nutrient models produced results that differed from results that held across the other two models.	Groups for which the models do not all agree in each scenario* Nutrients change: No conflict.	Nutrients change: No conflict.	Nutrients change: Change in macroalgae opposite to other models.
	F changes: No conflict.	F changes: When F rises detritus drops only in this model.	F changes: No conflict.
	Economic F: Detritus unchanged.	Economic F: Detritus drops.	Economic F: Detritus rises.
	Ecological F: Zooplankton unchanged, piscivores and seagrass drop.	Ecological F: Small zooplankton drop and large zooplankton rise.	Ecological F: Small zooplankton rise and large zooplankton drop.
	Split econ F and N rises: No conflict.	Split econ F and N rises: Phytoplankton and large zooplankton drop.	Split econ F and N rises: No conflict.
	Split econ F and N drops: Mullet rise only in this model.	Split econ F and N drops: Large zooplankton, clupeoids and piscivores rise.	Split econ F and N drops: Detritus rise sand benthic grazers drop.
	Split ecol F and N rises: Benthic grazers rise.	Split ecol F and N rises: Clupeoids drop.	Split ecol F and N rises: Flatfish and piscivores drop.
	Split ecol F and N drops: Microphytobenthos, clupeoids, flatfish and piscivores drop and seagrass rises.	Split ecol F and N drops: No conflict.	Split ecol F and N drops: No conflict.

* “No conflict” indicates that the model agrees with at least one other model for all groups.

Multi-species effects

One possible value of ecosystem models is in identifying impacts that other methods would miss. For instance, single species models cannot predict the potential fall in epifaunal predators and rise in benthic grazers that BM2 predicts would result from the “economic” strategy. This effect arises from a combination of direct and indirect multi-species interactions, and is in direct contradiction to what single species models would predict, given the drop in the direct harvesting of epifaunal predators and the rise in the targeting of benthic grazers under the suggested policy.

Sensitivity to nutrients rather than fishing

In looking for conclusions about the real system (Port Phillip Bay) that appear to be robust to model uncertainty, one result stands out. Given the enclosed nature of the bay and the stock structure for most of the fished species (few entirely “local” stocks), the bay is much more likely to respond strongly to the effects of eutrophication than to those of fishing. For all models, all system-level indices are more sensitive to a change in nutrients than to a change in F (Table 7.9). The bay’s geography (large and shallow, with restricted oceanic access) is such that if nutrient inputs change significantly - especially if they reach the level where nitrogen disposal, by flushing and denitrification, is overwhelmed (Murray and Parslow 1997) - the entire system can be quite heavily impacted. This scenario occurs when nutrients are increased by fivefold, though the increase in the biomasses of the higher trophic levels is probably overstated in ECOSIM. Extensive impacts as a result of large changes in nutrients have been found to be true of other models of Port Phillip Bay (Murray and Parslow 1999a), not just the ones discussed here. Such impacts also have the potential to be quite long-lived (chapter 2). This is due to positive feedbacks involving denitrification (Murray and Parslow 1997, 1999a), the immense nitrogen reservoir in the bay’s sediments, and the apparent

dominance of a detritus based, rather than a more traditionally recognised primary production based, food web (Harris et al. 1996, chapter 2).

Turning to impacts of fishing, many of the finfish species recruit, at least in part, from externally based reproductive stocks (Gunthorpe et al. 1997). This means that at least some of the effects of fishing within the bay may be limited, so long as the external stocks are healthy. Although all three models showed less sensitivity to fishing than to nutrient impacts, there are notable differences between the nutrient models and ECOSIM that are attributable to model structure and assumptions. The damping of responses to productivity by the constant recruitment term in the nutrient models does suggest that it may play a significant role in compensating for changes in F . Despite this stabilizing feature in the nutrient models, substantial shifts in the F s still result in corresponding shifts in the local population sizes of the target species, their competitors, predators and prey. For example, such impacts can be seen in the change in the biomasses of the flatfish, epifaunal carnivores, sharks and deposit feeders across the three models under the “economic” strategy. In general, impacts of fishing are likely to be greater on species that recruit locally, such as snapper (Coutin 1997, Gunthorpe et al. 1997), than on species that recruit principally from outside the Bay (for example King George whiting (Gunthorpe et al. 1997, Smith and MacDonald 1997)).

This phenomenon, whereby anthropogenically induced changes in nutrient status dominate over the effects from fishing, is not unique to Port Phillip Bay. Worldwide, many semi-enclosed bays have seen nutrient impacts overwhelm or modify the impacts from fishing (Caddy 2000). As Caddy points out, these nutrient impacts on marine catchment basins make integrated management essential, as fisheries management alone cannot address the problems.

Choice of indicator groups

Another general result of interest is that, across the range of management objectives and fishing and nutrient pressures imposed upon the ECOSIM model of Port Phillip Bay, only three qualitatively different system states arose. Moreover these system states can be characterized by the status of a few key groups. Given the increasing management interest in identification of indicator species, it is notable that a small group of species is sufficient to broadly characterise these three states. Using the sharks as an indicator group, the three states are characterized by:

- The biomass of sharks declining to negligible levels. This characterizes a fishing policy heavily influenced by economic objectives and/or a drop in the system's productivity (nutrient input).
- Sharks persisting at about the current levels of biomass. This characterizes a system free from recent extensive productivity shifts. It also arises from a fishing policy which attempts to find a compromise between ecological and economic objectives, although it is not clear that a deliberative strategy of this sort is the cause of the current state of the resources in PPB itself.
- The biomass of sharks increasing. This reflects a fishing policy guided primarily, or entirely, by ecological objectives and/or a wholesale rise in the system's productivity (though not to the point where the system is highly eutrophied as ECOSIM is currently unable to capture the indirect effects of changes in the nutrient and denitrification based system dynamics)

The persistence of the three system states across a range of refuge parameter settings, management objectives and environmental scenarios, in conjunction with the lack of response to fishing by many ancillary or lower trophic groups, does seem to result from the nature of the bay as discussed above. It is perhaps not surprising that a bay with as many in-built buffers as PPB has only a few “managed system” states where it is stable,

requiring quite large perturbations to push it from one state to another.

Despite the generalization noted above, the sharks alone do not summarize the entire state of the system under all policies and environmental changes. However a set of indicators comprising chlorophyll a, seagrass and sharks captures most of the effects, and changes in these indicators do seem to provide a warning that larger changes in the state of the system may have begun. Chlorophyll a is a much stronger indicator of the effects of changes in system productivity than any of the other groups and should be included in a set of indicators for that purpose. In contrast, the biomass of the larger fish, particularly sharks within ECOSIM, are good indicators of fishery based impacts on the system, though they also respond to changes in productivity. Seagrass is included in the list as it is particularly sensitive to changes in nutrient conditions, but is also impacted upon by the effects of fishing (either through habitat destruction or change in predation pressure). Thus between the three, they gave a good indication of what force is having the largest effect on the system and what overall state the system is in. It should be noted that this conclusion is limited to a “perfect information” case for a modelled system. In monitoring real systems in the field, a larger set of indicator groups may be more appropriate as they not only indicate change, but may also suggest some explanation of the cause. However it is encouraging that groups frequently referred to as being sensitive to change in nature are those which stand out as indicator species within the models. The loss of large shark species and the incursions of dogfish mark regime shifts in systems such as the Gulf of Alaska (Wright and Hulbert 2000) and the Grand Banks (Fogarty and Murawski 1998), while the loss of seagrass has been noted in many studies of impacted estuaries (Nienhuis 1983, Walker and McComb 1992). Chlorophyll a (as an easily measurable index of phytoplankton) is already used in monitoring for the effects of eutrophication (Harris et al. 1996) and is increasingly the subject of correlative studies with fisheries production (Lima and Costello 1995, Parsons and Chen

1994). Identifying species, or groups, which are persistently identified as being sensitive to changes in ambient environmental conditions and anthropogenic activities may be a first step to finding a set of useful indicators.

Robustness to model formulation

In spite of the qualitative agreement between the models at the broadest level, and the consistent form of the policy solutions found by ECOSIM, the differences in detail within and between the models also serve as a warning. For example, some of the policy conclusions from ECOSIM are found to be sensitive to the refuge parameters used. This underlines the requirement for analysis of parameter sensitivity regardless of the complexity of ecosystem models. Since completely inclusive, systematic sensitivity analyses are not possible for such models, a good understanding of likely key parameters is essential (chapter 1).

The results from this study also serve to underline the broader sensitivity to model formulation and approach. Formulation of multiple models, or at least multiple formulations of crucial mechanisms and processes, can identify groups that are sensitive to key assumptions. Substantial differences in some key groups have been demonstrated in this study in all three models examined. For example with the “economic” strategy and no change in nutrients, the interaction of the impact of fishing and the food web dynamics in the three models leads to three different patterns of change in the detritus. Given the long-term storage of nutrients that detritus represents in PPB, such a range of outcomes is a crucial result. While different models may agree qualitatively at the overall system level, management objectives are often concerned with particular groups and species, and conclusions about impacts on these can vary widely between model formulations (e.g. abalone).

However, a cautionary note must be sounded about the assumption that

conclusions that match across models are robust. While such an assumption is sound with regard to the assumptions that differ between models it cannot guard against the situation where the models share similarities in ecosystem structure and do not allow for processes which result in fundamental change to that structure. This is potentially one of the greatest problems facing “ecosystem” models. Where should the line be drawn, beyond which it is declared that model uncertainties and lack of process knowledge are too great to dare to use process-based models to extrapolate system state at the level of the entire system? This is a question that remains to be answered. For instance, the biogeochemical models employed here do a reasonable job of capturing the effects of mild to moderate eutrophication, but may well fail under conditions of extreme eutrophication (chapters 1 and 2).

7.5 Conclusions

Three further general points arising from this study are worth noting.

First, care needs to be taken in specifying objective functions for policy analysis. Clearly and not surprisingly, emphasis on economic objectives alone can result in systems that are very different from pristine. Perhaps more importantly, even where ecological criteria are included in the objective function, emphasis on particular groups (particularly the “charismatic megafauna” beloved of some conservation groups) can also result in systems that are far from pristine. The results in this paper suggest that focusing attention on the conservation of higher trophic groups does not produce a balanced system. A system populated only by those sub-webs featuring marine mammals is no more inherently balanced and representative than a system where they do not feature at all. Balanced objectives are not only required across sectors of human interest (say, economic and ecological), but also across the trophic groups that make up the systems being impacted.

Second, this study draws attention to an ongoing need to identify reliable means of synthesizing the often complex and voluminous information that can be produced by ecosystem models. A wide range of output indices have been presented in this study, including indices for individual groups as well as a few system-level indicators. None of these alone captures the key differences across policy options and forcing scenarios. On the other hand, a small set of key indices that can be intuitively and quickly grasped will be needed to communicate results to policy makers and decision makers.

Finally, while ecosystem models are emerging as key tools for investigating options for managing marine ecosystems, there is still much to learn about these tools. In this context, comparisons of predictions across alternative model formulations for the same system can be informative. The results from this study suggest that while some conclusions may be robust to model uncertainty, others clearly are not. While the use of ecosystem models for investigating management of marine systems has increased substantially in the last few years, there are as yet very few instances of multiple implementations for the same system. This study suggests that this comparative approach may lead to better understanding of key processes and assumptions in the use of these tools for ecosystem management.