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**Towards Ecosystem-Based Fishery
Management in New South Wales:**

**Proceedings of the Experts and
Data Workshop, December 8-10,
2003, Cronulla, Australia**

**TOWARDS ECOSYSTEM-BASED FISHERY
MANAGEMENT IN NEW SOUTH WALES:
PROCEEDINGS OF THE EXPERTS AND DATA
WORKSHOP, DECEMBER 8-10, 2003,
CRONULLA, AUSTRALIA**

edited by

Robyn E. Forrest, James P. Scandol and Tony J. Pitcher

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DIRECTOR'S FOREWORD

As director of the Fisheries Centre, I am pleased to present these proceedings, the result of a workshop held in Cronulla, Australia in December 2003. Over the years, the Fisheries Centre has specialised in building strong collaborative relationships with partners and institutions all over the world and these proceedings are an output of one such collaboration. In 2002, the Fisheries Centre signed a Memorandum of Agreement with the New South Wales Department of Primary Industries (then NSW Fisheries), in order to produce simulation models to aid transition towards ecosystem-based management of the fisheries of New South Wales.

At the Fisheries Centre, we specialise in developing models of many types and in synthesising all kinds of data in novel and useful ways. But the real experts in any ecosystem are the people who work with its many components every day, who understand the nuances of the data, the subtleties of the behaviour of its organisms and the complexity of its fisheries. Without input from such experts, the modeller is lost. Therefore in December 2003, the 'Experts and Data Workshop' was convened, at the southernmost edge of Sydney, in the waterfront laboratories of the New South Wales Department of Primary Industries, Cronulla.

The workshop provided a depth and breadth of knowledge of the fisheries and coastal environment of New South Wales that had never previously been brought together in one meeting. Individually, the papers presented here provide useful and detailed information as well as important references. Viewed as a whole, however, these proceedings provide a picture of an ecosystem that is cherished, despite many challenges presented by lack of long-term data, natural variability and enormous complexity in terms of both behaviour of its organisms and in the fisheries themselves.

I hope that in reading this volume, you come to appreciate not only the challenges involved with modelling complex marine environments, but that you also come to recognise the importance of collaboration: both at the level of individual scientists, managers, students and others who work in the field as well at the institutional level, where bringing together of expertise in many different fields can truly help to find solutions to the complex problems facing fisheries today.

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PREFACE & EXECUTIVE SUMMARY

In December 2003, NSW Department of Primary Industries (then NSW Fisheries) and the UBC Fisheries Centre at the University of British Columbia, Canada, held the 'Experts and Data Workshop' as part of a joint project 'Towards Ecosystem-Based Fishery Management in New South Wales Using Spatial Ecosystem Simulation'. The workshop had three broad objectives: (1) gather data and estimates of parameters for input into ecosystem models; (2) bring together researchers with expertise in different taxa and fields, to provide a broad picture of the current state of knowledge of the living marine environment of NSW; and (3) to provide perspective on the current status of ecosystem-based fishery management in NSW and Australia.

More than 60 scientists and students attended the workshop giving 39 presentations and contributing 21 papers and extended abstracts to this Fisheries Centre Research Report. The workshop provided a unique opportunity for scientists in NSW and elsewhere in Australia to learn about and discuss ecosystem-based fisheries management. Commercial fisheries in NSW had recently undergone a rigorous process of environmental assessment, and many in the audience were keen to understand what additional issues are involved with ecosystem-based fisheries management. Furthermore, there is intense curiosity about the potential role of ecosystem models to broaden our understanding, and improve our management, of the aquatic ecosystems that underpin our commercial and recreational fisheries.

This report documents many of the presentations given at the workshop. Participants came from Australia (60), New Zealand (1) and Canada (2) and were invited to speak on their specialised topic under one of three main themes: Links between science, management and modelling in New South Wales and region; Fisheries of New South Wales; and Marine ecosystem of New South Wales. The proceedings are divided into these three sections.

The NSW Department of Primary Industries and the UBC Fisheries Centre would like to thank the workshop participants for their contributions and the valuable debate that ensued. Data that have been painstakingly collected and expert understanding that has taken years to attain will both play a fundamental role in the sustainable management of living aquatic resources. The workshop and proceedings were made possible by the generous sponsorship of the NSW Department of Primary Industries and by the many research institutions of Australia and New Zealand from which the workshop participants came.

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Editors

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WORKSHOP PROGRAMME

Monday 8th December, 2003

Speaker (Affiliation)	Title
Steve Kennelly (NSW DPI)	Welcome Address
James Scandol (NSW DPI)	Aims of the workshop & introductions
Tony Pitcher (UBC)	What does ecosystem management look like?
	<i>Coffee Break</i>
Beth Fulton (CSIRO)	CSIRO's virtual oceans
Robyn Forrest (UBC)	Mass-balance models of NSW estuaries and coast
Cathy Bulman (CSIRO)	Preliminary trophic models of the South East Fishery and North West Shelf
Neil Gribble (QLD DPI)	Ecosystem management of the Great Barrier Reef World Heritage Area: is it possible?
	<i>Lunch break</i>
Janet Grieve (NIWA-NZ)	Pilot trophic model for subantarctic water over the Southern Plateau, New Zealand
Mark Baird (UNSW)	An East Australian Current configuration of a coupled physical – biological ocean model
David Rissik (DIPNR)	Data support and modelling output for effective coastal zone management in NSW
Stephanie Moore (UNSW)	The interaction between nutrients and hydrography in the development of estuarine phytoplankton blooms following rain events
	<i>Coffee Break</i>
Philip Gibbs (NSW DPI)	The role of ecological science within Environmental Impact Assessments in NSW
Jeffrey Dambacher (CSIRO)	Qualitative analysis of model ecosystems

Acronyms

CSIRO: Commonwealth Scientific and Industrial Research Organisation; **DIPNR:** Department of Planning, Infrastructure and Natural Resources; **NIWA-NZ:** National Institute of Water and Atmospheric Research- New Zealand; **NSW DPI:** New South Wales Department of Primary Industries (formerly NSW Fisheries); **QLD DPI:** Queensland Department of Primary Industries; **UBC:** University of British Columbia, Canada; **UNSW:** University of New South Wales.

Tuesday 9th December, 2003

Speaker (Affiliation)	Title
Tony Underwood (University of Sydney)	Ecological research and environmental management
Dan Breen (NSW DPI)	Modelling marine biodiversity for marine protected areas in NSW
Gee Chapman (University of Sydney)	Spatial variability in marine invertebrates in NSW
Bronwyn Gillanders (University of Adelaide)	Connectivity between juvenile fish in estuaries and adult fish on open coasts
Bruce Pease (NSW DPI)	Some examples of large-scale movement and migration through the estuaries and inshore coastal zone of New South Wales by diadromous and marine fishes
	<i>Coffee Break</i>
Dianna Watkins (NSW DPI)	NSW Fisheries: commercial fisheries management
Bryan van der Walt (NSW DPI)	Recreational fishing in NSW
Phil Duncan (NSW DPI)	NSW Indigenous Fisheries Strategy
Kylie Pitt (Griffith University)	Interactions between jellyfish and fisheries in New South Wales
Gretta Pecl (University of Tasmania)	Squid life history and population dynamics: a challenge for annually-averaged ecosystem models
	<i>Lunch</i>
Keith Martin-Smith (Project Seahorse, UBC; University of Tasmania)	Role of syngnathids in shallow coastal ecosystems of southeastern Australia
Doug Hoese (Australian Museum)	Small benthic fishes
Nick Otway (NSW DPI)	Sharks off NSW – some “science”, the remainder educated guesswork
Jock Young (CSIRO)	Large pelagic species in the ETBF - from single species research to study of the pelagic ecosystem
	<i>Coffee Break</i>
Steve Montgomery (NSW DPI)	Coastal prawns
Geoff Liggins (NSW DPI)	Rock lobsters and abalone

Wednesday 10 th December, 2003

Speaker (Affiliation)	Title
Dennis Reid (NSW DPI)	NSW estimates of fishing activity from the National Recreational Fishing Survey 2000/01
Robert Williams (NSW DPI)	Estuarine structure, function and biodiversity
Charles Gray (NSW DPI)	Estuarine fish and fisheries in NSW: some considerations for ecosystem-based modelling and management
Doug Ferrell (NSW DPI)	Snapper (<i>Pagrus auratus</i>) and Kingfish (<i>Seriola lalandi</i>)
John Stewart (NSW DPI)	Garfish (Hemiramphidae) in New South Wales, Australia: aspects of their distributions, fisheries, life-histories and ecology
	<i>Coffee Break</i>
Kevin Rowling (NSW DPI)	Fishery-induced changes in abundance and size structure of demersal trawl fish species off New South Wales
Ken Graham (NSW DPI)	Demersal diversity, diets and demise of deepwater dogsharks off New South Wales
Carla Ganassin (NSW DPI)	The occurrence and importance of birds in the marine and estuarine waters of NSW
David Pollard (NSW DPI)	Threatened species
	<i>Lunch</i>
Karen Astles (NSW DPI)	Filling chasms and building bridges - the science/management interface in fisheries
James Scandol (NSW DPI)	Simulation modelling and ecosystem-based fisheries management
Facilitator: James Scandol	Wrap-up discussion: linking ecology, models and management

SECTION 1

LINKS BETWEEN SCIENCE, MANAGEMENT AND MODELLING IN NEW SOUTH WALES AND REGION

ECOSYSTEM-BASED FISHERIES MANAGEMENT AND MODELLING THE MARINE ECOSYSTEM OF NEW SOUTH WALES: A BACKGROUND

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ABSTRACT

Ecosystem-based fisheries management (EBFM) is a set of principles aimed at providing a more holistic approach to managing fisheries. In many countries, including Australia, a number of legislative and policy instruments exist to implement EBFM, or the related ecologically sustainable development (ESD). EBFM policy goals are often broadly-stated (e.g., “to maintain ecological processes” or “to limit impacts on the ecosystem”) and may mask costly data needs and possible trade-offs that need to be considered. Computer simulation models can, variously, be used to clarify policy goals, identify policies that are likely to be robust, illustrate important trade-offs, highlight data-gaps, and help to prioritise data-collection programmes. The current collaboration between NSW Department of Primary Industries and the UBC Fisheries Centre aims to develop simulation models that will meet some of these needs as NSW moves towards a greater focus on EBFM. This paper provides a brief background to progress towards EBFM in Australia and NSW, an overview of the use of simulation models as tools, a very brief introduction to the fisheries and marine ecosystem of NSW, and a description of the overall aims of the project. These topics, except for the last, are presented and discussed in much greater detail in the papers that follow.

INTRODUCTION

Australian fisheries management agencies are under increasing pressure to demonstrate that they are practicing Ecosystem-Based Fisheries Management (EBFM) (e.g., Commonwealth of Australia 1998; Ward *et al.* 2003; see Scandol this volume; Scandol *et al.* 2005). EBFM is variously defined as a set of concepts or ‘principles’ that encapsulate the need to manage fisheries in ways that recognise their potential to alter ecological and human systems. The FAO’s Technical Guidelines for Responsible Fisheries (FAO 2003) states that under an Ecosystem Approach to Fisheries, fisheries management should respect the following principles:

- fisheries should be managed to limit their impact on the ecosystem to the extent possible;
- ecological relationships between harvested, dependent and associated species should be maintained;
- management measures should be compatible across the entire distribution of the resource (across jurisdictions and management plans);
- the precautionary approach should be applied because the knowledge on ecosystems is incomplete; and
- governance should ensure both human and ecosystem well-being and equity.

These principles have been expressed in a variety of ways (e.g., Pitcher 2000; Ward *et al.* 2002; Pikitch *et al.* 2004). The last point is sometimes expanded to more explicitly describe human considerations. For example, the Australian Worldwide Fund for Nature’s policy document states as one of its principles that “A successful ecosystem-based management system will recognise economic, social and cultural interests as factors that may affect resource management” (Ward *et al.* 2002 p. 5). Goals that reflect human interests may, however, conflict with ecological goals. Maintaining biomasses of marine species above certain levels, especially species that have low productivity, may be incompatible with maintenance of economically viable fisheries. Proponents of EBFM have, in general, been slow to acknowledge that EBFM needs to include explicit recognition of inherent trade-offs such as this (but see Walters and Martell 2004; Christensen and Walters 2004a; Hilborn *et al.*

2004; and Mote Symposium papers in *Bulletin of Marine Science* Vol. 74 (3), 2004). Commercial species eat each other – harvesting one species affects harvests of others; some species are less resilient to fishing than others even though they are harvested simultaneously; different sectors of society have conflicting objectives. In many cases, it will simply not be possible to meet multiple objectives at the same time and explicit recognition of this and evaluation of resulting trade-offs should be part of EBFM management plans (Hilborn *et al.* 2004; Walters and Martell 2004).

Many of the principles of EBFM are being incorporated into Australian fisheries policy through various policy documents and legislation at both State and Commonwealth (Federal) level. Fisheries managers throughout Australia are now faced with the difficult question of what they must do differently to meet the requirements of EBFM. A step towards EBFM at the Commonwealth level has been development of Australia's Oceans Policy (AOP: Commonwealth of Australia 1998; see Wescott 2000; Alder and Ward 2001), which has been established as an "integrated and comprehensive" approach to address Australia's management and conservation obligations under the Third United Nations Convention on the Law of the Sea (UNCLOS III: United Nations 1983). Australia's Oceans Policy was drafted after extensive consultation with State and local governments and a large network of stakeholder groups, including industry, community groups and research organisations (Wescott 2000; Alder and Ward 2001). One of the stated aims of EBFM, as laid out in AOP, is to: "Maintain ecological processes in all ocean areas, including water and nutrient flows, community structures and food webs, and ecosystem links ... [and] ... Maintain marine biological diversity, including the capacity for evolutionary change and viable populations of all native marine species in functioning biological communities." At the practical level, however, little is currently known of marine ecological processes in Australia, especially at large scales. There have been some Commonwealth initiatives to address this lack of information. Key steps have been the establishment of the National Oceans Office (www.oceans.gov.au/oceans.jsp) and the drafting of a set of comprehensive Regional Marine Plans specifically aimed at implementing Australia's Oceans Policy (e.g., National Oceans Office 2004). Substantial funding has also been provided to survey and map Commonwealth fishing grounds, and to conduct research into fields such as trophic ecology, habitat-use and productivity off the southeast coast (e.g., Bax and Williams 2000; and papers in *Marine and Freshwater Research* Volume 52 (4), 2001).

At the State level, fewer resources have been directed towards gathering data to support EBFM. In general, Australian fisheries management agencies have not invested heavily in assessment of commercial fish stocks, with the status of the majority of commercial species unknown (Phillips *et al.* 2001). The reason for this is that many of Australia's fisheries are relatively low in value and are managed in association with policies on cost-recovery (e.g. McColl and Stephens 1997; NSW Fisheries 2001). Data-intensive stock assessment tends to be the exception rather than the rule, generally being applied only to the most valuable fisheries (only abalone and rock lobster in New South Wales). For the many multi-species, small to medium-scale fisheries operating in Australia's states, there is little ecological information with which to frame EBFM strategies.

This is not to say that there has been no progress in introducing EBFM at the State level, even if it is under a different name. Since the 1990s, state and Commonwealth governments have been committed to the implementation of ecologically sustainable development (ESD) via a number of legislative and policy instruments (e.g., Commonwealth Environment Protection and Biodiversity Conservation Act 1999; NSW Fisheries Management Act 1994; Council of Australian Governments 1992). Some authors have concluded that the principles of EBFM are consistent with ESD (Fletcher 2003; Scandol *et al.* 2005) and that therefore initiatives that were designed to meet ESD requirements (e.g., Fletcher *et al.* 2002; see Gibbs this volume) also meet the needs of EBFM. In New South Wales, the past five years have seen considerable effort directed towards producing Environmental Impact Assessments (EIAs) for all state-controlled fisheries and related activities (see Gibbs this volume). The EIAs provide assessment of the current activity of each fishery, and identify links with other parts of the human and ecological environment (e.g., NSW Fisheries 2001; 2002a, b; see www.fisheries.nsw.gov.au/commercial/management-strategies). While the Environmental Impact Assessments make use of existing information about the fisheries of New South Wales and highlight likely interactions among components of the ecosystem, no new ecological research was done to produce them and all so far point out large gaps in understanding of ecosystem processes and

the nature of fisheries impacts on these processes, e.g., “The draft Fishery Management Strategy has revealed substantial knowledge gaps that affect the management of the Estuary General Fishery. The knowledge gaps cover four main areas – stock assessments of all retained species, bycatch, accuracy and precision of effort data and ecological interactions among retained species. [...] There is little understanding of how fishing pressure affects fish stocks in the Estuary General Fishery. [...] Whilst there is some basic knowledge about the general biology of species in the Estuary General Fishery there is little knowledge about how the species interact” (NSW Fisheries 2001, pp. E-250-251).

The costs associated with collecting data to address the above knowledge-gaps are likely to exceed the funds available for research in the near future. Innovative approaches, such as the development of assessment methods that rely on more-easily collected data (e.g., “life-history” information, such as age, growth and reproductive data) may need to be developed. This is currently an active area of research in other parts of the world (e.g., Jennings *et al.* 1998; McAllister *et al.* 2001; Rose *et al.* 2001; Cortés 2002; Dulvy *et al.* 2004; Cheung *et al.* 2005; Goodwin *et al.* 2006).

MODELS AS SUPPORT TOOLS

There is often a mismatch between the spatial and temporal scale of experiments designed to answer ecological questions and the scale of information required by natural resource managers. This is due partly to the patchy nature of ecological processes, which typically vary over small scales; partly to funding and logistical limitations, which restrict the possibilities for large-scale experiments; and largely to differences in the questions being addressed by ecological researchers and managers (Underwood 1998). More than ever, managers are required to consider the effects of policy on components of complex systems about which they have little or no information. Inherent unpredictability due to indirect interactions in complex ecosystems (e.g., May *et al.* 1979; Yodzis 1994; 2001; Punt and Butterworth 1995; Walters and Kitchell 2001) and natural variability further reduce managers’ ability to predict the outcomes of their decisions. Moreover, the complexity of marine ecosystems can obscure the identification of appropriate policy objectives.

The complexity, variability and lack of knowledge about marine systems does not, however, necessarily preclude making good policy decisions. Scientists and managers do not need detailed knowledge of all system processes to be able to predict that one policy is preferable to another over a wide range of possible states of nature and futures (Walters and Martell 2004). Policies that consistently outperform others under a range of uncertainty in a simulation framework can be considered relatively robust and worthy of further exploration. Predictive computer simulation models can be indispensable for comparing the effects of different policies over a range of uncertainty (see, for example, Mangel 1985; Hilborn and Walters 1992; Hilborn and Mangel 1997). Models can help to identify processes most likely to be important to predicting the effects of policy and help to focus research programmes (Walters 1993). Importantly, models can also help to clarify policy questions and expose trade-offs that need to be considered in setting and achieving policy objectives.

Over the past six decades, different classes of stock assessment models have been used by fisheries scientists to predict the impacts of fishing on fish stocks. These include surplus production models (e.g., Schaeffer 1957); dynamic pool models (Beverton and Holt 1957); and fully age-structured biomass dynamic models (see Megrey 1989), which can be collapsed into simpler delay difference models (Deriso 1980; Schnute 1985, 1987). Most of these types of models have been applied to single species, although some have been extended to a multispecies context (e.g., Murawski 1984; Pope 1991; reviews by Bax 1998; Whipple *et al.* 2000). In recent years, whole ecosystem models have been developed to help scientists and managers focus on ecosystem-scale policy questions. Probably the most widely used of these is the *Ecopath* family of mass balance ecosystem models (Polovina 1984; Christensen and Pauly 1992; Walters *et al.* 1997, 1999) but see also Bax (1998); Whipple *et al.* (2000); Latour (2003); Fulton (this volume). Description and examples of some applications of ecosystem models built using the *Ecopath* with *Ecosim* (EwE) software can be found in Bulman (this volume); Bradford-Grieve (this volume); Gribble (this volume); and in a recent special volume of *Ecological Modelling* (Vol. 172 (2-4), 2004). See Christensen and Walters (2004b) for discussion of the capabilities and limitations of these models. See Fulton *et al.* (2003) and Fulton and Smith (2004) for

comparison of the performance of three structurally different ecosystem models.

While ecosystem models are unlikely to reach the stage where they can quantitatively and accurately predict all ecosystem dynamics, we do not necessarily need them to do so. If they can help to identify robust management strategies, expose trade-offs and clarify policy objectives, ecosystem models can be useful. When well-constructed and calibrated with data, EwE models can be good at capturing long-term qualitative dynamics and can be used to compare the performance of different management strategies over a range of uncertainty (Christensen and Walters 2004b). Uncertainty can be dealt with in a number of ways, including 'tuning' to data to constrain dynamics; use of Monte Carlo simulations over a range of variability in parameter estimates; and through use of stochastic closed-loop simulations, which dynamically simulate managerial responses to changes in the value of different reference points and which include 'submodels' for generating observation and implementation error (Walters *et al.* 2002). Ecosystem models can also facilitate the gathering and synthesis of existing data about the system and can help to identify knowledge gaps.

FISHERIES OFF THE COAST OF NEW SOUTH WALES

Important coastal marine habitats off the coast of New South Wales include more than 130 estuaries (90 of which have been fished commercially: Pease 1999; see Williams this volume), rocky reefs, mangroves, seagrass beds and kelp forests, which support a large diversity of fish and invertebrates. The continental shelf is relatively narrow (generally around 20-40 km offshore), and supports invertebrate and finfish fisheries. The continental slope has also supported fisheries since the 1970's (Tilzey 1994; Grieve and Richardson 2001; Tilzey and Rowling 2001). Waters tend to be oligotrophic, due to lack of upwelling and to tropical water transported south by the East Australia Current.

There are seven commercial marine fisheries operated wholly by the state of New South Wales: Estuary General, Estuary Prawn Trawl, Ocean Trawl, Ocean Haul, Ocean Trap and Line, Rock Lobster and Abalone. Effort in state commercial fisheries is controlled using a share management scheme. Limited shares are allocated according to historical participation in the fishery and are transferable. Except for the valuable abalone and rock lobster fisheries, which are managed by yearly quotas, all fisheries are managed by input (i.e., effort) controls. Management measures include a complex set of gear and mesh size restrictions, seasonal temporal and areal closures and minimum legal lengths of fish (see NSW Fisheries 2001; Gray this volume). All fisheries except for abalone and rock lobster are multispecies and most are multigear, with different operators being endorsed to use various types of gear. Within these management restrictions, there is no other limit to the quantity of fish that can be landed. Each fishery has a Management Advisory Committee (MAC) that meets regularly and contributes to management decisions. MAC members include commercial fishers and scientists, conservationists, indigenous representatives and scientists from other related disciplines.

All recreational fisheries operating out of NSW ports, regardless of distance offshore, are also state-managed. Recreational fisheries in New South Wales are significant. The recent national survey of recreational fishing estimated that in the financial year 2000-2001, there were approximately 7.7 million recreational fishing 'events' in New South Wales (Henry and Lyle 2003; see Reid this volume). Catches of several important commercial species were found to exceed commercial catches, a finding common to previous surveys of recreational fishing in Australia (e.g., Pollock 1980; West and Gordon 1994). Prior to the National Survey of Recreational and Indigenous Fishing (Henry and Lyle 2003), recreational fishing surveys were done on a local basis using various methodologies (e.g., State Pollution Control Commission 1981; West 1993; West and Gordon 1994; Isaacson 2000; Steffe and Macbeth 2002a,b; Murphy *et al.* 2002; reviewed by Henry unpubl. manuscript). The National Survey of Recreational and Indigenous Fishing is the first comprehensive survey enabling comparison of activity across the whole state and country.

The NSW government has recently closed several entire estuaries and many individual areas within other estuaries to commercial fishing. These areas (named Recreational Fishing Havens or RFHs) have been created by buying out commercial licences using funds raised from the new saltwater recreational fishing fees, introduced in 2002. The declaration of the Recreational Fishing Havens, in

conjunction with the introduction of recreational fishing fees, is intended to have the following outcomes: i) better management and knowledge of a possibly significant source of fishing mortality, about which there has previously been very little quantitative information; ii) a better angling experience for the state's growing population of recreational fishers; iii) reduction of damage to fish habitat by bottom trawlers in the RFHs; and iv) retirement of a largely unprofitable and inefficient sector of the commercial fishing industry (Dominion Consulting Pty Ltd 2002). Several species of fish that are targeted by commercial and recreational fishers are known to migrate among estuaries during their adult life (see Pease this volume). Many also undergo ontogenetic migrations, spending a part or all of their juvenile phase in estuaries before moving to coastal and offshore adult habitats. A common characteristic of the life-history of many fish and crustaceans in New South Wales is to undergo a northward spawning migration (e.g., Montgomery 1990; West and Gordon 1994; Gillanders 1997; Gillanders *et al.* 2000; Gillanders 2002). Eggs, larvae or juveniles then swim or are carried on the southward-flowing East Australia Current back into coastal and estuarine waters to continue their life-cycle. These behaviours, coupled with complex fishing effort dynamics, provide a challenge to modelling the likely effects of the recent estuarine closures.

Offshore, the Commonwealth Trawl Sector (part of the Southern and Eastern Scalefish and Shark Fishery; formerly the South East Fishery) is the largest Commonwealth-managed fishery operating off the coast of NSW and the one most likely to interact with state-managed fisheries due to its size, proximity to the coast and overlap with the southern part of the state Ocean Trawl fishery. The Commonwealth Trawl Sector's (CTS) fishing grounds extend from Sydney to Cape Jervis (south of Adelaide), and from 3 nautical miles off the coast to 200 nautical miles offshore. Within the CTS region, the shelf-edge and upper continental slope are generally the most productive (C.M. Bulman, CSIRO, pers. comm). Therefore, while the CTS extends to the edge of the Exclusive Economic Zone, most of the catch occurs on the continental shelf and slope from approximately 200 to 600 metres in depth (Tilzey and Rowling 2001). In 1992, Individual Transferable Quotas were introduced into the fishery, marking a transition from an 'input' to an 'output' controlled fishery. There are sixteen species under quota in the fishery, although more than 100 species are commercially harvested (see Kailola *et al.* 1993; Williams and Bax 2001).

Scientific research into Commonwealth fisheries is now done by the Commonwealth Scientific and Industrial Research Organization (CSIRO), although, from the mid-1960s to the mid-1980s, responsibility for offshore fisheries was mostly assumed by New South Wales (Tilzey and Rowling 2001). A significant part of this research was a set of surveys of the upper slope trawl grounds by the NSW State-owned Fisheries Research Vessel *Kapala* (Gorman and Graham 1976, 1977, 1979, 1980a,b, 1981; Graham *et al.* 1997). The initial, exploratory upper slope surveys were done in 1976-7 and were fully replicated twenty years later in 1996-7, allowing for some striking comparisons of the abundance of many species, before the advent of large-scale fishing and then after twenty years of fishing (Andrew *et al.* 1997; Graham *et al.* 1997; Graham *et al.* 2001; Graham this volume). In particular there have been significant declines in the abundance of many demersal sharks, particularly deepwater dogsharks, skates and some species of fish. One of the most significant declines in abundance of bony fishes has been that of gemfish (*Rexea solandri*), which was shown to suffer severe recruitment failure in the early 1980s (Rowling 1990, 1997, this volume).

PROJECT AIMS

The aim of this collaborative project between NSW DPI and the University of British Columbia Fisheries Centre is to provide simulation models in support of EBFM in New South Wales. Obviously, only a limited set of research questions can be addressed in a single project and these will be partly determined by available data and information. Two themes are of particular interest: 1) alternative approaches for assessment of data-limited fisheries; and 2) trade-offs in non-selective multi-species fisheries. The former theme will include analysis of relationships between life-history and productivity parameters in single-species simulation models; subsequent analysis of Australian multispecies fisheries; and development of a simple ecosystem model, using the EwE software. Single species models and EwE will be used to reconstruct likely historical events on the continental shelf and slope of NSW, using data from the *Kapala* trawl surveys, in order to illustrate trade-offs

associated with a range of hypothetical historical and future fishery policies (for example “weakest stock” management versus maximising system yield, Hilborn *et al.* 2004). Deepwater dogsharks are probably among the least productive species caught in slope trawl fisheries (R. E. Forrest, unpublished work) and will be used as a case study to explore trade-offs associated with protection and recovery plans for highly vulnerable species.

The papers provided in the following pages have been submitted by experts on different components of the fisheries and marine ecosystem of NSW. Wherever possible, the information provided by them is being incorporated into the models, which will be published elsewhere.

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ECOLOGY AND ENVIRONMENTAL ASSESSMENT OF FISHING ACTIVITIES

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BACKGROUND

The evolution of the New South Wales environmental assessment process for commercial fisheries and other designated activities (fish stocking, recreational and charter boat fisheries, beach safety [shark meshing] program) results largely from a decision of the New South Wales Land and Environment Court in January 2000. The court decided that the issue of an individual commercial fishing licence had to meet the requirements of the *Environmental Planning and Assessment Act 1979* (EP&A Act). This requires that the environmental impacts of any authorised activities have to be assessed at the time the licence is issued or renewed.

In addition the Commonwealth Environment Minister, amended Schedule 4 of the *Wildlife Protection (Regulation of Exports and Imports) Act 1982*. This amendment removed the blanket exemption from export control that has historically applied to marine species. By December 2004 the Commonwealth Minister for Environment and Heritage had to assess all Australian fisheries that export harvested species. Fisheries that fail to be assessed as 'demonstrably sustainable' by that date may not be able to export their product. The Wildlife Protection Act was repealed in January 2002 and the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) was amended to incorporate the repealed provisions. The new Part 13A of the EPBC Act regulates export control for marine species.

Further the EPBC Act makes it an offence for a person to undertake an action that has the potential to significantly impact on a matter of 'national environmental significance', without first obtaining a permit from the Commonwealth Minister for Environment and Heritage. Matters of national environmental significance include: declared World Heritage areas; declared Ramsar wetlands; listed threatened species and ecological communities; listed migratory species; listed marine species; nuclear actions; and the environment of Commonwealth marine areas.

NSW ENVIRONMENTAL ASSESSMENT: A STRATEGIC APPROACH

To achieve the requirements of New South Wales state legislation a fishery management strategy (FMS) must be developed for each managed fishery. This strategy outlines the management goals, objectives, controls and other measures for achieving the objectives, as well as describing performance indicators and monitoring programs that will apply to a particular fishing activity. Such strategies are being developed for all commercial and non-commercial fisheries. The FMS contains the 'management tools' applying to the fishery, as well as data collection protocols and triggers for the review of the strategy.

The Environmental Assessment of each fishery is an important tool to assess the current activity and highlight those aspects that need to be addressed in the FMS. It details likely impacts and allows for alternative management and mitigation measures when formulating management strategies to ensure conservation of coastal biodiversity and ecological processes. The environmental assessment enables the community to review and contribute to the proposed strategy and management regimes. It also informs decision makers of the likely costs and benefits of the strategy and the need for mitigation measures.

The term 'environment' includes biological, biophysical, economic and social aspects. This definition has a much broader scope than simply the assessment of the target species and requires a multitude

of issues to be considered in the environmental assessment. For example, within each fishery over twenty components are considered including, retained species, bycatch, bait, biodiversity, habitat damage, protected and threatened species, trophic structure, translocation, fish disease, water quality, noise, light, air quality, energy, external factors, economic viability, employment and safety, as well as issues associated with Indigenous and European heritage.

The environmental assessment tests the sustainability of the proposed level of fishing activity authorised under the proposed fishery management strategy. The assessment considers the cumulative implications of issuing approvals for the designated fishing activity along with interactions and impacts of other fisheries and activities on the resources under assessment.

Clearly, in achieving the statutory requirements of New South Wales' environmental assessments, the expectations of the Commonwealth assessments will also be met. This includes the sustainability of fish stocks and harvest regimes, the protection of listed marine species, threatened species and ecological communities, Commonwealth marine areas etc, and incorporation of the principles of ecologically sustainable development (ESD) under the EPBC Act.

THE ROLE OF ECOLOGY

Ecological science, simply defined as "The study of the interactions of organisms with their environment and with each other" can also be thought of as the study of the patterns of distribution and abundance and the processes that create the patterns. Input from the ecological sciences is essential if the sustainability of resources is to be demonstrated for the eight major ecological components to be considered in the EIS (retained species, bycatch species, bait species, threatened and protected species, species assemblages, species diversity, ecological (meaning 'ecosystem') processes and aquatic habitats).

Studying the ecological processes and patterns in the context of the environmental impact of a fishing activity defines three levels of ecological process with each level cascading to the next. At the level of the individual organism, processes such as competition and predation occur. A group of individual organisms interact at the assemblage level and examples of the ecological processes at this level are recruitment, growth and mortality. The assemblage level cascades to the ecosystem level where processes such as primary production, nutrient cycling, food webs and energy flow predominate. The complexity of, and difficulty in observing, this third ecosystem level is such that it is often represented using models. For the predictive requirement of EIS's, fisheries based ecosystem models are an appropriate tool to explore ecosystem level interactions.

In the EIS's completed for commercial fisheries in NSW, it is clear that biological, descriptive and some process data exists at the individual, assemblage and occasionally ecosystem level for retained species, bycatch species, bait resources threatened and protected species and for some elements of the aquatic habitats. There is, however, limited or inadequate knowledge about species assemblages, species diversity, ecosystem processes and offshore habitats often at all three ecological process levels.

Historically, in fisheries science the sustainability of a fish stock has been based on traditional stock assessment modelling and focused around the commercial harvest. Recently the broader impacts on ecological processes at the ecosystem level are being included in the stock assessments. Ecological processes shape the fish community and are equally important as traditional stock assessments in predicting the sustainability of the fishery. This is especially true when the management outputs of the sustainability assessment are considered within an ESD framework. The critical importance of ecology to environmental impact assessment as discussed above cannot be overstated. Therefore it is critical to supplement traditional stock assessments, which focus on the harvested species with ecosystem-based management practices.

Ecosystem based fisheries management (EBFM) as incorporated in the EIS's of NSW fisheries acknowledges the effects of fishing on the harvested species in addition to the ecological system, the

consequent impact on the harvested species by disruption of ecological processes, the dynamic nature of the system, integrates human use and values and is a science-based management framework which interfaces with policy and politics.

There are significant knowledge gaps hampering the complete incorporation of EBFM in the EIS's. These gaps are primarily related to a lack of ecosystem level process studies. Models to represent the ecosystem processes in the NSW estuarine and inshore coastal commercial fisheries are needed. However, the models are only one element and not the final solution to the questions fisheries managers ask in developing sustainable management strategies for harvest fisheries. The system description from the model has to be linked with the management actions. This can be achieved by the development of effective indicators and performance measures.

This presentation presents part of the background needed for, and development of, ecosystem based fisheries models for the NSW coast including estuaries and near shore shelf waters. Other papers in this workshop present expert advice about the structure and function of the ecological components in the model and relevant data on functional groups of species used to populate the trophic mass balance models.

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THE ROLE OF ECOSYSTEM SIMULATION IN ECOSYSTEM BASED FISHERIES MANAGEMENT

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ECOSYSTEM BASED FISHERIES MANAGEMENT

There has been widespread recognition that fisheries management must now consider the broader ecosystem impacts of harvesting. This recognition can be found within international conventions (such as the UN Fish Stocks Agreement), as well as national and state/provincial legislation. Parallel to this development has been an improved understanding of the social-economic dimensions of fisheries, and in particular, the value-based nature of environmental issues. Many of these contemporary directions in fisheries management have now been captured within the concept of “ecosystem-based fisheries management” (Ward *et al.* 2002; FAO 2003).

When the complexities of aquatic ecosystems are compounded with the multi-faceted nature of socio-economic systems, there is a demand for tools and concepts to simplify the system to one that is, in a word, comprehensible. Such tools aim to assist decision-makers to develop and implement effective public-policy for the (ecologically) sustainable development of fishery resources. These decision-support tools, or models, have a very wide definition and scope and include simple paper-based maps of fishing-grounds to elaborate computer models that capture ecological and social processes. All of these types of tools will contribute to ecosystem-based fisheries management (EBFM) and are discussed in more detail below.

ENVIRONMENTAL PLANNING AND ASSESSMENT

It is important to recognise that most of the attributes of EBFM are not particularly new within environmental management. In particular, the socio-economic dimensions of environmental decision-making have played a crucial role within discipline and practice of town-planning for at least a century. Decisions about where to locate industry, commerce and residents so as to minimise conflicts over pollution, amenity and transport have been taken ever since humans developed cities. In NSW, Part 4 of the *Environmental Planning and Assessment Act* (1979), was essentially derived from the development control system used to plan the use of land (Farrier *et al.* 1999).

Indeed it could be argued that fisheries and forestry are perhaps the last human activities that are being forced to comply with standards of environmental planning and assessment that have been applied to urban development, industrial development and mining for decades. There was, therefore, almost an inevitability that the legislative machinery of planning and assessment law would be directed at forestry and fisheries. In NSW, the former occurred in the 1980's (Farrier *et al.* 1999) whilst the latter occurred at the end of the 1990's (Hurrell and Jardim 2000).

Planning and assessment procedures are highly applicable to fisheries and marine resource management. These methods have a lengthy pedigree in terrestrial land management and marine park management (such as the Great Barrier Reef Marine Park, see Lawrence, Kenchington *et al.* 2002). Regardless of their limitations, planning processes have evolved through the sedimentary accumulation of legislation and case law and therefore represent *the* mechanism of how many of these complex decisions are actually taken.

Planning and assessment institutions such as environmental impact assessment, even though they

have been criticised (see Fairweather 1989, 1994), are here to stay. The inherently predictive demand of environmental impact assessment does not sit easily with experimental methodologies that are informative after the completion of an experiment. There will thus be an ongoing demand for tools that attempt to predict or forecast the state of a system before a managerial change has been imposed.

DECISION-SUPPORT METHODS AND TOOLS FOR EBFM

When attempting to understand the role of ecosystem simulation models within EBFM it is important to place this within the context of other computer-based decision-support tools (Lein 1997). The central new challenge for the implementation of EBFM is the development of effective processes to link the management of socio-economic systems with the underlying ecosystems. Professionals working in the environmental field need to be pragmatic about what tools actually work and provide benefit over cost.

Underlying many decision-support systems are robust data management systems and simple descriptive statistical models, such as percentages and averages. These tools probably provide the most widely used decision-support methods for environmental management. Tools such as geographic information systems (GIS), that overlay human uses with characteristics of the underlying ecosystems, are clearly important for EBFM. Where there are established physical and chemical laws (such as with the dispersion of pollutants) then these predictive models have also found their way into contemporary environmental management.

Application of complex ecosystem simulation models to environmental management is in its infancy. The expectation from environmental and planning methods for predictive results may, however, be beyond the capability of these models. The interconnected nature of these systems and lack of contrasting and cost-effective data to calibrate models may make predictions unreliable and untrustworthy. A valid scientific criticism is that many of these predictions will be untestable within logistic constraints. In such situations, the most appropriate role of simulated systems may be to evaluate potential management strategies. Good management strategies will be those that generate the desirable outcomes regardless of uncertainties in system processes or observations.

EVALUATION OF EMPIRICAL INDICATORS

Within the evaluation of management strategies, perhaps the most immediate requirement for contemporary fishery managers is the development and implementation of indicators to measure the performance of EBFM. The standard managerial strategy of identifying objectives and then collecting data to measure progress towards achieving those objectives, is now accepted in contemporary fisheries management. Unfortunately the indicators and reference points developed for single species fisheries management are unlikely to be sufficient for EBFM. A valuable role of ecological simulation models would be to assist the definition and evaluation of measurable indicators that provide a metric for the state of ecosystem structure and function. Clearly there will not be a single indicator or variable for such a complex role. Consideration should be given to the application of multivariate statistical methods to distil empirical observations to a small set of interpretable dimensions. If such indicators do not exist then ecological simulation models would be ideal vehicles to explore such a claim. It maybe that simplistic notions such as the "percentage of habitat open to fishing" are valuable indicators within EBFM.

FINAL COMMENTS

Contemporary environmental management, whether given the title EBFM or otherwise, is the subtle process of managing the complex demands of human societies within the constraints of our

biophysical environment. Various institutions have been developed to support this process. Science should provide insights and tools to enable these institutions to make more informed decisions. EBFM involves inherently spatial considerations so the primary computer models used within decision-making will be geographic information systems and the supporting data-base management systems. Various types of simple statistical methodology will also play a continual role in decision-making. When dynamic representations of ecosystems are required then ecosystem simulation models will play a crucial role. The predictive limitations of these systems will, however, have to be recognised. A more valuable role of these systems is for management strategy evaluation and, in the short term, development of empirical indicators and reference points for management.

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DATA SUPPORT AND MODELLING OUTPUT FOR EFFECTIVE COASTAL ZONE MANAGEMENT IN NSW

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INTRODUCTION

The New South Wales Government's Estuary Management Program assists local government to prepare and implement estuary plans for the states 130 significant estuaries. Estuary committees are supported through 50% funding and through the provision of important and often expensive data.

The Estuaries Unit of the Department of Infrastructure, Planning and Natural Resources (DIPNR) has traditionally had a strong background in the physical sciences area with a focus on collecting hydrodynamic information. This information has been used as the basis for undertaking hydrodynamic modelling but also to underpin much of the works focussed actions which have been implemented in the past. Much of this data have been collected through the engagement of Manly Hydraulics Laboratory, who ensure high data capture rates and quality control of data, as well as providing a well-organised database.

With an increased awareness in the importance of integrating physical, biological and chemical information to develop a better understanding of estuarine processes, the data and often the modelling outcomes are becoming increasingly important.

The aim of this paper is to outline the types of physical data that have been collected in some of the NSW estuaries and to provide an indication of how these data can be used to model estuarine hydrodynamics and to point towards the biological relevance of these models.

DATA COLLECTION ACTIVITIES

A variety of data have been collected and continue to be collected from selected estuaries in the state. These are generally collected to support the preparation of estuary management plans where hydrodynamic models can underpin various management initiatives. These data have been traditionally funded by the NSW government as they are considered to be too expensive for individual councils to collect. The data can also be maintained through a central agency.

Bathymetry

Using the latest aerial photography available, a base map is produced by photogrammetrically digitising the features (eg scarps, shoals, roads and buildings). Several survey control marks are to be established to calibrate the photogrammetric model. Below water level, sounding is carried out using high precision survey echo sounders with position provided by differential GPS. Above water level spot heights are determined using Real Time Kinetics (RTK, a high accuracy Global Positioning System). Water level recorders are positioned throughout the estuary to provide tidal water level information, which are then used to standardise height data. The accuracy of the photogrammetrically derived base plan and the hydrographic survey data is less than one metre. Plans of hydrosurvey are produced in Australian Height Datum (AHD) and to the Map Grid of Australia (MGA). Calibration / verification procedures are followed in the collection of the hydrographic survey data. All plans are reviewed by accredited hydrographic surveyors (certified by the Institute of Surveyors).

Water level data

Water levels are collected in many estuaries and coastal lakes in NSW. The primary aim of these exercise is to provide a comprehensive data set defining tidal variations or variations due to

incoming freshwater flows or evaporation. These data are collected for different periods of time but are often for long periods when flooding is an issue of concern. They are also collected for short periods when hydrographic surveys are undertaken. Water level is an important component to consider when undertaking hydrodynamic modelling of the system. Data are collected by deploying automatic water level recorders at selected sites. These water level recorders are surveyed and related to known height benchmarks. Data are used to calculate tidal planes and to determine various tidal constituents.

Salinity data

Detailed salinity depth profiles are collected at each of low and high tide slack from a number of stations along the main channel of the estuary. Data are collected using a high quality conductivity, temperature, depth recorder, usually a Seabird 19 or 23 which has been calibrated both pre and post collection.

Salinity data are also occasionally collected from fixed instruments located at fixed sites and depths in estuaries (see long term water quality data, below). These instruments are changed and calibrated on a monthly basis.

Recently salinity data collected by Manly Hydraulics Laboratory on behalf of DIPNR from over 100 sites were reanalysed to determine the sensitivity of various estuaries to upstream freshwater extraction. Data were collected on a long-term (a few years) and short-term (1 to several months) basis. Several variables were generally collected. This study focussed on the use of salinity data as salinity is an indicator of the extent that marine water has intruded into the estuary (van Senden and Rissik 2004).

Tidal gauging

The estuary being studied is assessed and gauging stations are selected according to what is assumed to be a major flow point or an area at which flows will vary from that of the main channel. Water movement at each selected cross section is then ascertained over a full tidal cycle every half-hour, at selected cross sections using Acoustic Doppler Current Profilers, mounted on small boats. These surveys are undertaken at times when there is greatest variability between low and high tides. Outputs include three dimensional information on incoming and outgoing tidal velocities of the full depth range across the channel. These data can be used to determine discharge data (tidal prisms).

Long term water quality data

DIPNR have funded a network of water quality stations in various locations in estuaries around NSW for various time periods. These stations have collected data using Datasonde water quality meters. Data have included depth, temperature, salinity, turbidity, pH, dissolved oxygen. Information from these gauges is often made available on a real time basis through telemetry. Information from these gauges has assisted with management of acid sulfate soils, with responses to eutrophication and to turbidity. Changes to salinity and temperature following rainfall have also been elucidated.

Short term water quality data

In many estuaries, activities aimed at assessing the differences between salinity, temperature and density at high and low tides have been conducted. At high tide slack and again at low tide slack, a water quality profile has been undertaken at regular intervals until the tidal limit. These enable the differences between the long channel salinity and density structures at each of high and low tide slack to be identified.

Similar data collection exercises were undertaken following the recent period of extended drought to determine the influence of drought on salinity intrusion into the estuary.

ANALYSIS AND MODELLING

Data collected during the various field investigations can be analysed or used in models to determine a

variety of useful variables that can feed into the planning or management process. Models can assess dispersion within systems, can assess flushing and residence times. These transport and advection factors can then be used to interpret biological or chemical variability.

Estimated flushing times

Flushing times for a number of NSW estuaries have been determined using the following methods. This work was conducted in conjunction with Patterson Britton as part of the comprehensive Coastal Assessment.

Tidal rivers are relatively narrow and shallow and have strong tidal velocities which produce good local mixing. As a consequence, salinity varies very little across estuary width and only slightly with depth. In a tide averaged sense, the salinity structure of a tidal river is one dimensional and can be well characterised by the depth averaged, longitudinal variation of salinity.

The tide average location of the limit of salt intrusion in a tidal river varies in response to the variation in freshwater flows. Strong freshwater flows force the limit of salt intrusion downstream towards the mouth. Conversely, during low freshwater flows the limit of salt intrusion moves upstream. During protracted drought conditions, the limit of salt intrusion can extend to the limit of tidal influence. This has occurred in the majority of NSW estuaries during the recent severe drought conditions.

When the freshwater flow is relatively constant for an extended period (i.e. steady state conditions), the limit of salt intrusion is established at an equilibrium position. This location is established relatively quickly (i.e. approximately two weeks). Even if disturbed temporarily by a flood, the equilibrium position of the limit of salt intrusion is re-established in a period of 2-4 weeks depending upon the size of the flood and the length of the estuary.

The flushing characteristics of each “intermittently closing and open lake and lagoon” (ICOLL) were modelled for a range of entrance states using a two dimensional, finite element, hydrodynamics/dispersion model (RMA, 1973). A numerical model of each ICOLL was established using hydrographic survey data and each model was calibrated so that it reproduced the measured ratio of the range of mean spring tide in the ocean to that in the main body of the ICOLL.

a)

b)

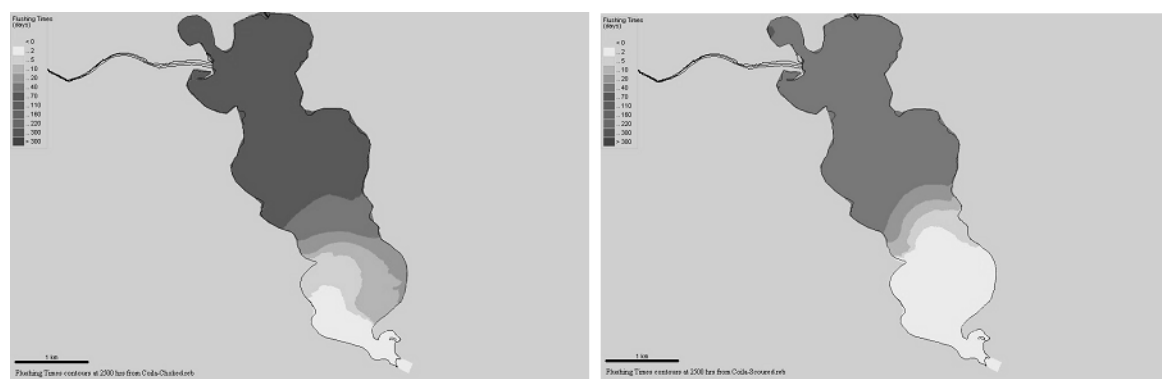


Figure 1. Flushing times for Coila Lake with a constricted entrance (a) and a scoured entrance (b) (Rissik et al. 2003).

Detailed hydrodynamic modelling

Detailed hydrodynamic modelling has been undertaken to determine aspects such as flushing times but can also be used to trace particles within the estuarine environment providing information about dispersion and distribution of particles throughout the estuary. This is an important aspect for estuarine consideration as tidal movement has the potential to move particles into the estuary as well as moving them outwards. This form of modelling can also indicate where there may be cross sectional differences in the behaviour of particles.

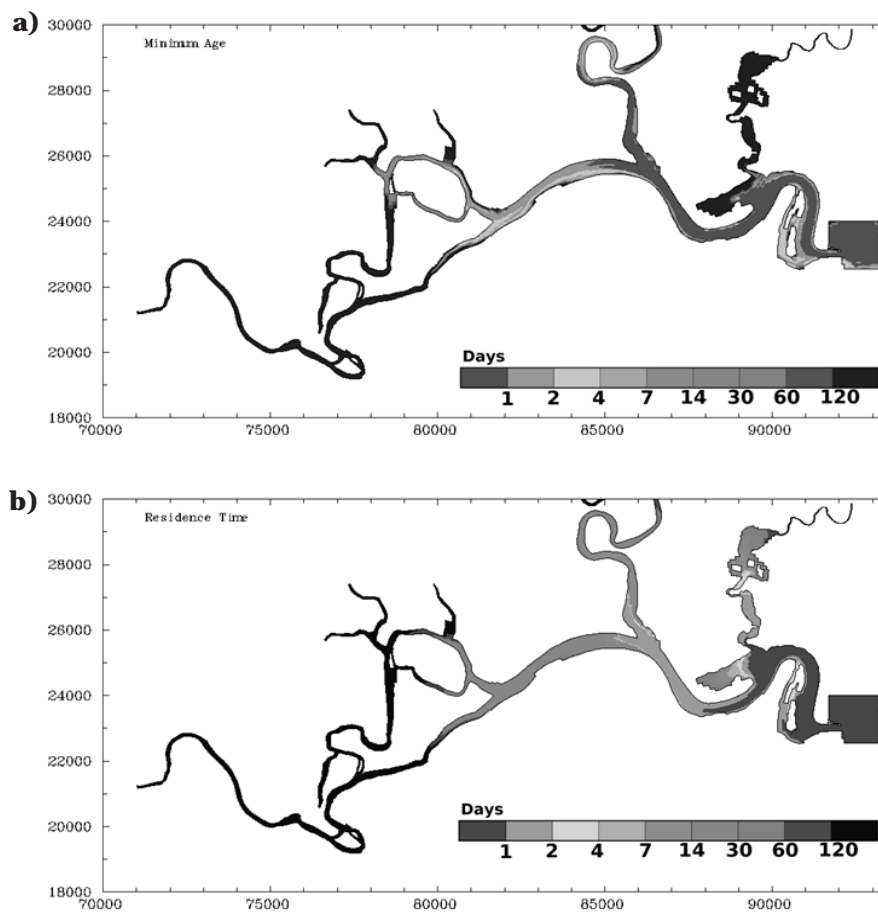


Figure 2. Dispersion of particles within the Hastings Estuary. (Unpublished modelling output from John Floyd DIPNR).

Biological relevance of flushing time

An example of the influence that flushing times can have on biological processes can be seen in a study of the drowned river valley Berowra Estuary in Sydney. Studies have reported high algal production, high algal biomass and high zooplankton biomasses in a reach of the estuary with a prolonged flushing time compared with lower levels at more rapidly flushed upstream and downstream sites (Rissik *et al.* in prep). Flushing time was greatest in an area where the estuary was deep (up to 17 metres) and wide and would be very difficult and prohibitively expensive to influence. It was considered that the only realistic way to reduce the potential for algal blooms to occur was through reduction of nutrient input from the catchment.

Conclusions

A long program of physical water quality data has been undertaken in NSW. This data has been analysed to a limited extent, but the data sets provide effective information for interpreting biological and chemical data. Specific modelling exercises have been conducted in conjunction with local government in many of the estuaries with a general focus on flood mitigation and entrance management. More recently effort is being made to address different objectives in modelling exercises.

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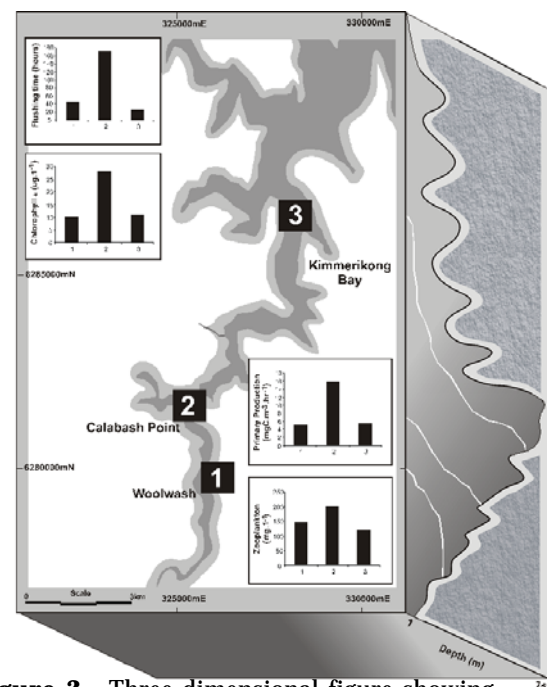


Figure 3. Three dimensional figure showing Berowra estuary and its bathymetry, with flushing times, primary production rates, chlorophyll a biomass and zooplankton biomass from three locations within the estuary (Rissik *et al.* in prep.)

CSIRO's VIRTUAL OCEANS

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The state of marine ecosystems and the management of the exploitation of marine resources has become the focus of national and international interest and legislation. Ecosystem models are one tool used to consider these issues. The CSIRO Division of Marine Research has developed a number of ecosystem models using different underlying premises in an effort address a number of management issues. Two of the models are discussed here - Atlantis and InVitro.

Atlantis is a biogeochemical marine ecosystem model that tracks the nitrogen and silica flow through the main functional groups (benthic and pelagic, invertebrate and vertebrate, sedentary and highly mobile) and detritus types (labile detritus, refractory detritus and carrion) found in temperate marine ecosystems. The primary producer and invertebrate consumer groups are represented by aggregate biomass pools, while the vertebrate groups are represented using age-structured models. The main processes considered in Atlantis are production, consumption, waste production, migration, recruitment, habitat dependency, and mortality (both natural and due to human activities). The human activities accounted for include nutrient pollution, coastal habitat degradation due to development, and fisheries. Of these three submodels, the harvesting submodel is the most elaborated, allowing for multiple fleets - each with differing gear selectivity, habitat association, target, byproduct and bycatch groups, effort dynamics and management structures. All of these anthropogenic and biological components are implemented in a three dimensional spatially-resolved geography. Polygons are used to represent the major geographical features of the simulated marine system, with the size of each polygon reflecting the extent of spatial homogeneity in the physical variables accounted for in the model (depth, seabed type (reef or flat), canyon coverage, porosity, bottom stress, erosion rate, salinity, light and temperature). Movement through the model space is dealt with explicitly (for the migration of higher trophic levels and the movement of fleets), or by a simple transport model (for advective transfer).

While Atlantis is a deterministic model, InVitro is a hybrid model, combining deterministic components with agent-based (i-state configuration) components. InVitro is three-dimensional and uses a number of submodels to represent the natural biophysical system, each of the important sectors of human activity (petroleum exploration and extraction, conservation, fisheries and coastal development), and the monitoring activities and overall decision-process associated with management of each sector. The combined dynamics of all of these submodels determine how the system might respond to natural events (e.g. cyclones) and any human activities (e.g. fishing, dredging, land-clearing or discharging of contaminants). The biophysical submodel reproduces the main physical and biological features of the natural marine ecosystem (e.g. bathymetry, currents, waves, seabed types, habitat defining flora and fauna, and local and migratory populations of marine animals). This submodel also includes a representation of the impact of natural forces and any activities by the various human sectors. The sector submodels produce goods and services, and lobby the relevant management agencies for their desired policy outcomes. The management submodels observe the system produced by the biophysical model (imperfectly) and make decisions about the location and magnitude of the sector activities (to try and achieve specific environmental and social objectives). All management decisions are then implemented by the sectors, which must constrain their activities (and resulting impacts) accordingly. The 'currency' and 'resolution' of each agent in InVitro can be defined in many ways so that they best match the requirements for the system being modelled. For instance, for the North West Shelf implementation InVitro some agents are represented at the individual level (turtles and trawl or trap fishers, individual oil and gas wells or outfall pipes), while others are aggregates (sub-populations of fin-fish, schools of sharks, prawn boils, entire seagrass beds or reefs, whole port towns, and sector management bodies). Functional and physical attributes are

detailed for each of these agents and rules are specified for a range of appropriate behaviours (e.g. growth and movement of fauna, harvesting by fisheries etc). Decisions and responses by agents are made dependent on the current state of an agent's environment - characterised based on appropriate attributes like the bathymetry, currents, temperature, light intensity, chemical concentrations, habitat type, resident communities, zoning (for the sectors) and so on. A scheduler handles the timing of the agents' activities (and any interactions among the agents). The scheduler functions in much the same way as a multi-tasking operating system, assigning priorities to agents and splitting available time to give the illusion of concurrency. All environmental agents are dealt with first, as any change in the environment must be handled before the other agents can function; then the scheduler randomly handles requests by all non-environment agents at each level in the priority queue. This method of handling agents avoids any potential for systematic advantage of a particular agent (or agent-type) due to internal ordering of processes. The scheduler also ensures temporal consistency and synchronicity, preventing the "subjective" time of an agent straying far from that of its neighbours.

Both of these models can be readily applied to systems of varying scales (from small bays to large proportions of Australia's marine area). To date Atlantis has been applied to Port Phillip Bay and the southeast temperate and subtropical waters of Australia, while InVitro has been applied to the tropical shelf waters of the Northwest Shelf. There is immense interest amongst the developers of the models to apply them in many more locations (particularly those where other ecosystem models have been applied), both to see if "they work as is", but also to gain insight into common or contrasting ecosystem features from a number of locations and models.

PRELIMINARY TROPHIC MODELS OF THE SOUTH EAST FISHERY AND NORTH WEST SHELF

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INTRODUCTION

In this paper I present some general descriptions and very preliminary results of the trophic ecosystem models we have been developing for two communities in the Australian EEZ: the East Bass Strait (EBS) portion of the South East Fishery (SEF) and the North West Shelf (NWS). These two communities are not only at diametrically opposite corners of the continent, and therefore influenced by different oceanographic and climatic regimes, but they have also been subject to very different management regimes and scientific investigations. Consequently, both the model objectives and structures differ. The trophic and circulation models for the EBS are being developed to investigate management issues such as the impact of increasing seal populations, changing discarding practices and environmental variability on fisheries production. The NWS model was developed to describe the trophic interactions explicitly and to allow comparison with a broader ecosystem model being developed within the NWS Joint Environmental Management Study, a joint initiative by CSIRO and the Western Australian Government.

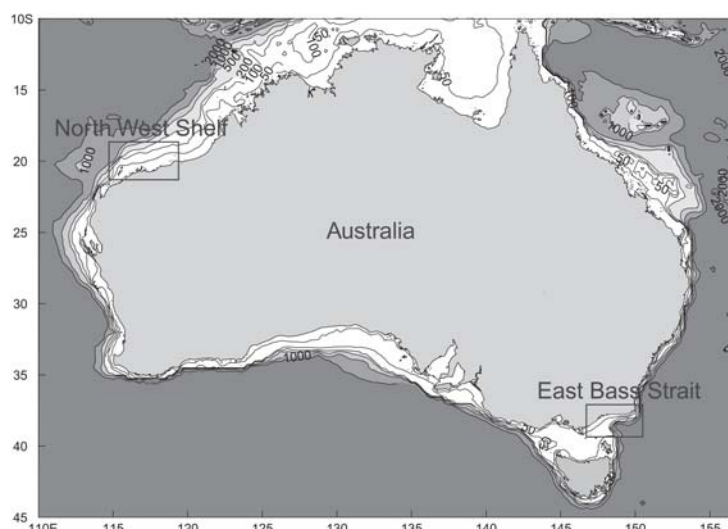


Figure 1. Location of the North West Shelf model area and the East Bass Strait model area.

EAST BASS STRAIT

The East Bass Strait (EBS) study area is situated between 36 and 39°S on the southeast corner of mainland Australia (Fig 1), within the Twofold Shelf marine area (Interim Marine and Coastal Regionalisation for Australia (IMCRA) Technical Group 1998). The ecosystem model being developed for this area covers the shelf and the slope to about 700 m, where there is a major change in community composition (CSIRO Marine Research 2001). The water influences are mostly from Bass Strait, although incursions from the East Australian Current occur seasonally as do spring and summer upwellings on the outer shelf and slope areas (Condie and Dunn in prep). The surface water temperatures average around 13°C, but are significantly cooler on the slope in winter and warmer on the shelf in summer. The shelf area consists of soft and hard grounds interspersed with reefy outcrops and is described in detail in Bax and Williams (2000, 2001) and Williams and Bax (2001). The invertebrate communities are highly diverse and show quite high endemism (Williams *et al.* 2001, National Oceans Office 2002). However, *Maoricolpus roseus*, the New Zealand screw shell now dominates the biomass of several of the inshore habitats and its impacts could be severe (Williams *et al.* 2000).

The fisheries of the SEF have been operating since the early 1900s but the most dramatic changes

have occurred much more recently. Up to the 1970s the fishery operated on the shelf of New South Wales and north-eastern Victoria with little formal management or co-ordinated research (Tilzey and Rowling 2001). Steam trawlers and Danish seiners were the main fishing methods used and tiger flathead was the main target species. During late 1960s and early 1970s, diesel-powered otter trawlers allowed the rapid expansion of the fishery into the upper- and mid-slopes and further afield. By the early 1980s the blue grenadier and orange roughy were major target species in the slope waters particularly around Tasmania. Non-trawl fisheries such as gillnet, line and trap also developed. Management was largely under State jurisdiction until the late 1970s when the Offshore Constitutional Settlement assigned federal, state or joint jurisdictions based on species rather than geography (Tilzey and Rowling 2001). Continuing development of the fishery during the 1980s prompted the establishment of the Australian Fisheries Management Authority in 1992 to manage the federal fisheries. The introduction of the Individual Transferable Quota system at this time also changed the focus of research from single-species studies to stock assessment. Many of the SEF quota species have now been subject to periodic stock assessments, which have highlighted specific research needs. These stock assessment procedures are now extending to risk assessment, harvest strategies and multi-species management strategy evaluations (Tilzey and Rowling 2001). In the development of the EBS model, focussed data on the commercial fishery have been combined with CSIRO scientific data collected during 1994-96.

NORTH WEST SHELF STUDY AREA

The North West Shelf (NWS) study area is situated between 18 and 21° S on the north-west coast of Australia (Fig. 1), and is contained within the Pilbara offshore region and the southern-most part of the North West Shelf region (IMCRA Technical Group 1998). The oceanographic environment is very dynamic being subject to several tropical cyclones every year, large tidal ranges, seasonal monsoons and interannual variability from El Niños (Condie *et al.* 2003). The warm, subtropical waters average about 25°C (Godfrey and Mansbridge 2000) and are derived largely from the Indonesian through-flow. The model is of the shelf to the 200m isobath and the ground comprises a variety of types from soft bottom to coral reefs, with high levels of biodiversity (Condie and Dunn in prep).

On the North West Shelf, foreign fishing dated back to 1935 but was effectively ended by the late 1980s by the progressive closure to foreign vessels. Prior to the declaration of the Australian Fishing Zone (AFZ) in 1979, foreign fishing had been unregulated. Following the declaration, the Australian Fisheries Service managed the northern demersal fishery under a system of total allowable catch quota. Since there was little activity from domestic fisheries, this quota was wholly allocated to the foreign fleets. Although the full quotas were usually not caught, the effort was concentrated on the North West Shelf, putting the ecosystem under considerable pressure. This pressure was increased from 1984, when domestic fishing fleets became increasingly active. A trap fishery for lethrinids and lutjanids began at this time followed by a drop line fishery in the Timor Sea in 1987 and by a trawl fishery in 1988. By this time, separate quotas were allocated to the North West Shelf, Arafura and Timor Seas (Jernakoff and Sainsbury 1990) and the areas in which foreign fishing fleets were allowed to operate were also restricted.

The major commercial species that were fished were generally those aimed at the Asian market, some of which were of considerable value to the domestic fisheries and some that were not (Jernakoff and Sainsbury 1990). The lethrinids (emperors and sea-breams), were particularly targeted, while lutjanids (sea-perches and snappers), serranids (rock-cods and coral trout), nemipterids (threadfin breams), saurids (lizardfishes), sparids (sea bream), haemulids (sweetlips) and carangids (trevallies, scads and queenfish) were also of interest. The fishery was managed to maximise the total multi-species yield, resulting in under- or over-fishing of the individual species (Jernakoff and Sainsbury 1990).

CSIRO's involvement began in 1979 with comprehensive assessments of the stocks on the North West Shelf. The area was surveyed from 1978 to 1997 for various purposes and at varying frequency. The survey data that we used in developing this model come from a series of surveys in 1982 and 1983, a sequence of annual surveys from 1986 to 1991, and annual surveys in 1995 and 1997. The period that this model represents is the late 1980s after the decline of the foreign fishing effort and at the onset of

the growth of the domestic trawl fishery.

THE MODELS

Models were built for each ecosystem for the period of the late 1980s using the Ecopath with Ecosim software (Christensen and Pauly 1992; Walters *et al.* 1997, 1999, 2000; Christensen and Walters 2003). They were both fishery-based but the data availability and quality varied widely between them. This is hardly surprising considering that the domestic fisheries were focussed on the eastern seaboard for more than half a century longer than on the NWS.

Understandably, the NWS data sets were smaller and the data more spurious because data were collected *ad hoc* for specific investigations rather than stock assessment. In contrast, fisheries statistics for the SEF were collected routinely and directed

towards fishery assessment. An observer monitoring program in the SEF initiated in the mid 1990s added valuable information about the commercial and non-commercial catch. However, the CSIRO survey data for the NWS provided good data for estimating initial and time series biomass. Similarly, the EBS data initial biomasses were based largely on CSIRO surveys although the temporal coverage was not as extensive.

The trophic data for the NWS model was limited to a few species from the NWS (Sainsbury and Jones unpub. ms., Sainsbury and Whitelaw 1984, unpub. ms., CSIRO unpublished data) and was supplemented by studies of the same and other species in the Gulf of Carpentaria (Salini *et al.* 1992; 1994; Brewer *et al.* 1995), or from other parts of the world (Blaber *et al.* 1990; Fishbase: Froese and Pauly 2003). On the other hand, a large amount of data including a trophic study of over 70 species was available for the EBS area (Parry *et al.* 1990; Bulman *et al.* 2001) and several other large studies in the SEF provided data for the deeper species (Blaber and Bulman 1986; Bulman and Blaber 1987; Bulman and Koslow 1992).

The EBS model consists of 58 model groups covering the shelf and slope (Fig. 2a, Appendix 1), whereas the NWS model consists of 37 groups and covers only the shelf (Fig. 2b, Appendix 2). Not all groups in the models are explicit in the food web: size groups in the food web were split to account for different feeding preferences, e.g., those that were piscivorous or those that were largely invertebrate feeders. The scientific surveys identified more than 200 fish species on the EBS compared to more than 500 fish species on the NWS (excluding large sharks and small pelagic fishes which were not targeted by the demersal trawl surveys). The fish species were different between the areas: threadfin breams

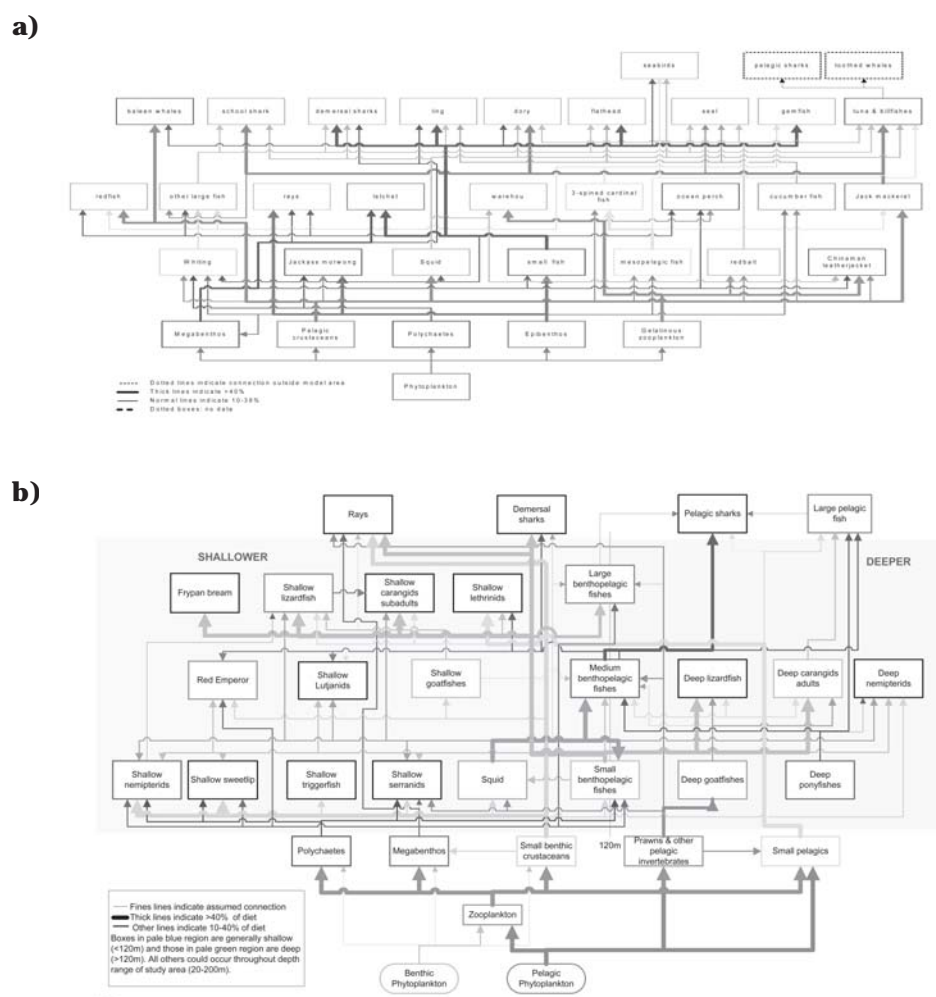


Figure 2. Food web of the (a) Eastern Bass Strait study area of the South East Fishery and (b) North West Shelf on which the EwE models are largely based. Not all groups represented in the models are shown in the food web.

(Nemipteridae), goatfishes (Mullidae), emperors and sea-perches (Lutjanidae and Lethrinidae), rock-cods (Serranidae) and trevallies (Carangidae) were the commercial species abundant on the NWS, as well as the lizardfishes (Synodontidae), whereas in the EBS the sharks and rays were the most abundant groups, followed by species such as jack mackerel (Carangidae), barracouta (Gempylidae), whiptails (Macrouridae), cardinal fish (Apogonidae), redfish (Berycidae), cucumberfish (Chlorophthalmidae) and leatherjackets (Monacanthidae).

BALANCING THE MODELS

Initial balancing of the NWS model highlighted major problems with the original estimates of swept-area abundance. Highly piscivorous species such as the lizardfish and carangids required far more fish to support their biomass than was estimated from surveys. It was clear from inspection of length frequency data of species caught in the trawl surveys, that the small fish component was grossly under-represented suggesting that small fish were not retained by the net, thus giving us a biased sample. This was corrected by accounting for the size selectivity of the trawl net mesh. We derived mesh-selectivity coefficients from available length-frequency data for small species under about 30 cm, and estimated the proportions of the populations not caught by the net. The coefficients were extrapolated to other species that we did not have length data for, but that were similar in size and morphology. This substantially increased the biomass estimates for these smaller species. This correction process was not applied generally to species larger than 30cm. However, this revision process also allowed for the significant decrease in availability of adult carangids beyond maturation to trawl nets.

Despite these adjustments for selectivity, the red emperor stock assessment estimate for the area was over twice our swept-area abundance estimate so, because we assumed greater confidence about the stock assessment estimate, we assumed that the difference was a factor of availability of the fish to the net. The availability factor calculated for red emperor was then applied generally to all species' abundance estimates, further raising the estimates. The NWS model was then balanced relatively simply by small adjustments to proportions in the dietary matrix and to biomasses of some species.

In the final balanced model, the total fish biomass was 33 t/km², with most of the increase from selectivity adjustments being in the small pelagic and demersal fish groups. Although the increase in fish biomass from the original estimates seemed large, the final fish biomass compared well to the West Florida Shelf, with an estimated biomass of 28 t/km² (Okey and Mahmoudi 2002). Total biomass estimates of other systems vary from low to high: ~ 6 t/km² on the Mexican shelf (Arreguín-Sánchez *et al.* 1993); ~17 t/km² (which may be an underestimate) on the Great Barrier Reef of Australia (Gribble 2001); 44 t/km² on the Venezuelan shelf (of which small pelagic fish = 33 t/km² (Mendoza 1993) and 246 t/km² on a Caribbean reef (Optiz 1993).

The Ecosim module was run forced by a time series of catches, catch per unit effort for some gear types and calculated fishing mortalities. The fit of the model to the time series observations is currently poor. Improvement in the fit requires further refinements to tune it, such as:

- inclusion of biomass accumulation rates to allow growth in the system
- exploration of predator-prey vulnerabilities to find better-fitting combinations
- review of production parameters of some species
- investigation of primary production anomalies
- adjustment of the diets of juveniles, possibly requiring additional stanzas.

In conclusion, we learned from the development of the NWS model the importance of considering selectivity and availability factors when using trawl survey data for biomass estimation. Consequently, for species in the EBS model, we are deriving selectivity coefficients from escapement and selectivity experiments (Bax and Knuckey submitted report; I. Knuckey, unpublished data) to develop a more realistic model. An historical model that might reflect the state of the EBS prior to commercial fishing is also under construction. Circulation & primary productivity models are being developed to investigate and describe seasonal variability at the base of the food web. Although only preliminary, we will use these EBS models to investigate some issues of concern to the management of the fishery and identify potential risks or areas of concern for further directed research.

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APPENDIX 1

Trophic groups of the EBS model and some representatives of the group

Group name	Representative species in group
Baleen whales	Baleen whales
Toothed whales	Toothed whales
Seals	<i>Arctocephalus pusillus doriferus</i>
Seabirds	Seabirds
Penguins	Penguins
Tunas	Tunas
Pelagic sharks	Pelagic sharks
Demersal sharks	<i>Galeorhinus galeus</i>
	<i>Cephaloscyllium laticeps</i>
	<i>Mustelus antarcticus</i>
	<i>Squatina australis</i>
	<i>Squalus megalops</i>
Rays	<i>Urolophus paucimaculatus</i>
	<i>Urolophus cruciatus</i>
	<i>Raja australis</i>
	<i>Urolophus</i> sp. A
	<i>Urolophus viridis</i>
	<i>Raja</i> sp. A
	<i>Narcine tasmaniensis</i>
Warehous	<i>Seriolella punctata</i>
	<i>Seriolella brama</i>
Redbait	<i>Emmelichthys nitidus</i>
Redfish	<i>Centroberyx affinis</i>
Ling	<i>Genypterus blacodes</i>
Dories	<i>Zenopsis nebulosus</i>
	<i>Zeus faber</i>
	<i>Cyttus australis</i>
Jack mackerel	<i>Trachurus declivis</i>
Jackass morwong	<i>Nemadactylus macropterus</i>
Flathead	<i>Platycephalus bassensis</i>
	<i>Neoplatycephalus richardsoni</i>
Gemfish	<i>Rexea solandri</i>
Latchet	<i>Pterygotrigla polyommata</i>
Shelf ocean perch	<i>Helicolenus percoides</i>
Chinaman's leatherjacket	<i>Nelusetta ayraudi</i>
Cucumberfish	<i>Chlorophthalmus nigripinnis</i>
School whiting	<i>Sillago flindersi</i>
Cardinal fish	<i>Apogonops anomalus</i>
Shelf small invertebrate feeder	<i>Azygopus pinnifasciatus</i>
	<i>Caelorinchus fasciatus</i>
	<i>Cyttus novaezelandiae</i>
	<i>Pempheris multiradiatus</i>
	<i>Parma microlepis</i>
	<i>Synchiropus calauropomus</i>
	<i>Pseudolabrus psittaculus</i>
	<i>Diodon nictemerus</i>
	<i>Arothron firmamentum</i>
	<i>Paramonacanthus filicauda</i>
	<i>Parequula melbournensis</i>
	<i>Macroramphosus scolopax</i>
	<i>Lepidotrigla modesta</i>
	<i>Notolabrus tetricus</i>
	<i>Caelorinchus parvifasciatus</i>

APPENDIX 1 CONTINUED

Trophic groups of the EBS model and some representatives of the group

Group name	Representative species in group
Slope medium invertebrate feeder	<i>Caelorinchus australis</i>
Slope large predator	<i>Trachipterus jacksonii</i> <i>Polyprion oxygenois</i> <i>Ruvettus pretiosus</i> <i>Lepidopus caudatus</i> <i>Diastobranchus capensis</i>
Pelagic small invertebrate feeder	<i>Sardinops neopilchardus</i>
Pelagic small predator	<i>Arripis georgianus</i>
Pelagic medium invertebrate feeder	<i>Tubbia tasmanica</i> <i>Seriolella caerulea</i> <i>Arripis truttaceus</i> <i>Trachurus novaezelandiae</i>
Pelagic medium predator	<i>Scomber australasicus</i>
Pelagic large predator	<i>Pomatomus saltatrix</i> <i>Rachycentron canadum</i> <i>Sphyrnaena novaehollandiae</i>
Mesopelagic fish	<i>Idiacanthus</i> sp. <i>Phosichthys argenteus</i> <i>Myctophidae</i> species <i>Melanostomias</i> sp.
Prawns	Pelagic prawns
Macrobenthos	Includes sessile, sedentary or small invertebrates such as echinoderms, benthic ascidians, sponges, bryozoans, gastropods and bivalves
Megabenthos	Includes large and mobile benthic prawns, crabs & molluscs such as benthic cephalopods (cuttlefish, four squid and eight octopus species) and larger gastropods, bivalves and <i>Maoricolpus</i>
Polychaeta	
Squid	six squid and three pelagic octopus species, including southern calamari and Gould's squid
Gelatinous zooplankton	pyrosomes & salps
Large zooplankton	>20mm, generally carnivorous: Euphausiids, mysids, copepods, amphipods, pelagic tunicates, chaetognaths and cnidarians.
Small zooplankton	<20mm, generally herbivorous: calanoid & cyclopoid copepods, pteropods, ostracods, cladocerans
Phytoplankton	
Detritus	
Discards	

APPENDIX 2

Trophic groups in the North West Shelf model

Group name	Representative species	Common Name
Coastal sharks	<i>Sphyrna mokarran</i> <i>Galeocerdo cuvieri</i> <i>Carcharhinus plumbeus</i> <i>Carcharhinus sorrah</i> <i>Hemigaleus microstoma</i> <i>Loxodon macrorhinus</i>	Great hammerhead Tiger shark Sandbar shark Spot-tail shark Weasel shark Sliteye shark
Rays	Dasyatidae <i>Dasyatis thetidis</i> <i>Himantura toshi</i> <i>Himantura uarnak</i> <i>Rhynchobatus djiddensis</i> <i>Taeniura meyeni</i> <i>Aetobatus narinari</i>	Rays Black stingray Back-spotted whipray Reticulate whipray White-spotted guitarfish Blotched fantail ray White-spotted eagle ray
Small tunas	<i>Thunnus obesus</i> <i>Thunnus albacares</i> <i>Scomberomorus commerson</i> <i>Euthynnus affinis</i> <i>Katsuwonus pelamis</i>	Bigeye tuna Yellowfin tuna Narrowbanded Spanish mackerel Bonito Skipjack tuna
Shallow lethrinids	<i>Lethrinus</i> sp. 1 <i>Lethrinus nebulosus</i>	Lesser spangled emperor Spangled emperor
Red Emperor	<i>Lutjanus sebae</i>	Red emperor
Shallow lutjanids	<i>Lutjanus malabaricus</i> <i>Lutjanus vittus</i> <i>Lutjanus erythropterus</i> <i>Pristipomoides multidens</i> <i>Pristipomoides typus</i>	Saddle tail seaperch Brownband seaperch Crimson seaperch Jobfish Jobfish
Shallow nemipterids	<i>Nemipterus furcosus</i> <i>Nemipterus celebicus</i> <i>Scolopsis monogramma</i>	Rosy threadfin bream Striped threadfin bream Rainbow monocle bream
Deep nemipterids	<i>Nemipterus bathybius</i> <i>Nemipterus virgatus</i>	Yellow belly threadfin bream Yellow-lipped threadfin bream
Shallow Serranids	<i>Epinephalus multinotatus</i>	Rankins cod
Frypan bream	<i>Argyrops spinifer</i>	Frypan bream
Juvenile/small carangidae	<i>Carangoides caeruleopinnatus</i> <i>Carangoides chrysophrys</i> <i>Carangoides gymnostethus</i> <i>Seriolina nigrofasciata</i> <i>Carangoides malabaricus</i>	Coastal trevally Longnose trevally Bludger Black-banded kingfish
Adult/large carangidae	<i>Carangoides caeruleopinnatus</i> <i>Carangoides chrysophrys</i> <i>Carangoides gymnostethus</i> <i>Seriolina nigrofasciata</i> <i>Carangoides malabaricus</i> <i>Carangoides equula</i>	Coastal trevally Longnose trevally Bludger Black-banded kingfish
Small pelagic fishes	<i>Sardinella albella</i> <i>Herklotsichthys koningsbergeri</i> <i>Decapterus russelli</i> <i>Auxis thazard</i>	gold-stripe sardine large-spotted herring Indian scad
Shallow lizardfish	<i>Saurida undosquamis</i>	Brushtooth Lizardfish
Deep lizardfish	<i>Saurida filamentosa</i>	White-spot lizardfish
Shallow mullidae	<i>Parupeneus heptacanthus</i>	Red spot goatfish
Deep mullidae	<i>Upeneus moluccensis</i>	Gold-band goatfish
Triggerfish	<i>Abalistes stellaris</i>	Triggerfish

APPENDIX 2 CONTINUED

Trophic groups in the North West Shelf model

Group name	Representative species	Common Name
Sweetlip	<i>Diagramma labiosum</i>	Painted sweetlip
Ponyfish	<i>Leiognathus bindus</i>	Orange tipped ponyfish
Shallow small fish	small fish (<30 cm)	
Deep small fish	small fish (<30 cm)	
Shallow medium fish	medium fish (30-50 cm)	
Deep medium fish	medium fish (30-50 cm)	
Shallow large fish	large fish (>50 cm)	
Deep large fish	large fish (>50 cm)	
Sessile epibenthos	Sessile epibenthos sponges, corals	
Megabenthos	Large mobile invertebrates incl. octopus, bivalves, crabs, prawns) >20mm	
Macrofauna	Infauna (polychaetes) & small epifauna <20mm (benthic copepods) & meiofauna	
Prawns	Commercial prawns: bananas, tiger, king	
Cephalopods	Squids	
Large zooplankton	Zooplankton>20mm, carnivorous jellies, ichthyoplankton	
Small zooplankton	Zooplankton <20mm including pelagic copepods	
Pelagic phytoplankton		
Benthic phytoplankton		
Microphytobenthos		
Detritus		

PILOT TROPHIC MODEL FOR SUBANTARCTIC WATER OVER THE SOUTHERN PLATEAU, NEW ZEALAND: A LOW BIOMASS, HIGH TRANSFER EFFICIENCY SYSTEM

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ABSTRACT (BRADFORD-GRIEVE ET AL. 2003)

The Southern Plateau subantarctic region, southeast of New Zealand, is an important feeding area for birds, seals and fish, and a fishing ground for commercially significant species. The Southern Plateau is a major morphometric feature, covering approximately 433,620 km² with average depth of 615 m. The region is noted for its relatively low levels of phytoplankton biomass and primary production that is iron-limited. In order to evaluate the implications of these attributes for the functioning of this ecosystem, a steady-state, 19-compartment model (Fig. 1) was constructed using Ecopath with Ecosim software of Christensen et al. [www.ecopath.org (2000)]. The system is driven by primary production that is primarily governed by the supply of iron and light. The total system biomass of 6.28 g C m⁻² is very low compared with systems so far modelled with a total system throughput of 1136 g C m⁻² y⁻¹. In the model the Southern Plateau retains 69% of the biomass in the pelagic system and 99% of total production. Although fish are caught demersally most of their food is part of production in the pelagic system. Mean transfer efficiencies between trophic levels II and IV of 23% are at the high end of the range reported in the literature and are partly an artefact of the detail with which the beginning of the food web has been portrayed. In the model, adult fish production is almost completely accounted for by the fisheries take (32%), consumption by seals (7%), toothed whales (21%), other adult fish (13%), and squid (20%). Fish and squid catches are at the trophic levels of 4.8 and 5.0 respectively. The gross efficiency of the fishery is 0.018% (catch/primary production) which shows that most of the system's production is not harvested but is going to sustain the system as a whole. Although not all data come from direct knowledge of this system, the model reflects its general characteristics, namely a low biomass and primary production system dominated by the microbial loop, low sedimentation to the seafloor, high transfer efficiencies, a long food web and supporting high-level predators. Given that this system appears to be tightly coupled, it is expected that interannual changes in primary production would be quickly transferred to the rest of the system. We might expect fish and other vertebrates to be impacted by a lowering of their production/biomass ratios, by changes to their fecundity and breeding success or fitness to migrate to breed in the case of hoki, changes to their diets.

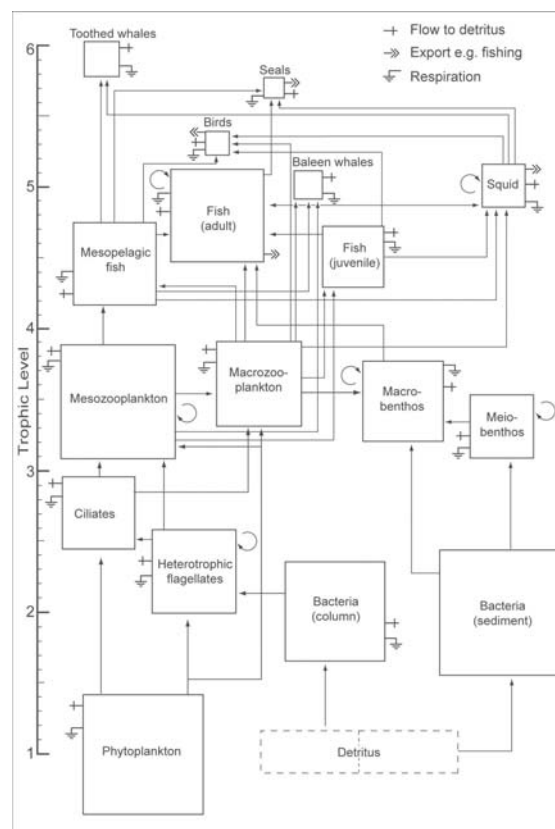


Figure 1. Trophic model for the Southern Plateau, New Zealand. The box size is proportional to the square root of the compartment biomass (from Bradford-Grieve et al. 2003).

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ECOSYSTEM MANAGEMENT OF THE GREAT BARRIER REEF WORLD HERITAGE AREA: IS IT POSSIBLE?

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BACKGROUND

The Australian Great Barrier Reef Marine Park (GBRMP) is a designated “multi-use” world heritage area, which by its charter must balance the needs of traditional owners, commercial and recreational fishing practices, and the conservation requirements of the park’s world heritage status. Management of fishing in all its forms, is seen as a major challenge for the Great Barrier Reef Marine Park Authority (GBRMPA), as this activity is likely to have the greatest anthropogenic impact on the highly complex and diverse ecosystem of the reef and inter-reefal areas. The ecosystem effects of fishing in general were reviewed and summarised by the ICES Marine Symposia in Montpellier, 1999, (Anon 2000) and this forum emphasised the need to consider the effects of fishing on the whole ecosystem, not just the economically important target species.

Ecosystem approaches to fisheries management were reviewed at the 16th Lowell Wakefield Fisheries Symposium (Anon 1999). A number of participants noted that broad ecosystem properties, as opposed to those of fish assemblages, are relatively robust and are not appropriate indicators of fishing pressure (Tyler 1999; Bax *et al.* 1999). The Experimental Lakes project in Canada (Schindler *et al.* 1985) provides one such example. Systematic acidification of the lakes changed the abundance, composition, and dominance of species present, but did not affect genuine ecosystem properties such as productivity and nutrient cycling. A similar conclusion can be drawn from changes in the continental shelf ecosystem of the U.S. Northeast, where large-scale shifts in species composition resulted from fishing (Sissenwine 1986), while comparable changes in system level processes have not been reported (Bax *et al.* 1999).

As noted by Tyler (1999) the assemblage is a more tractable level to try to monitor and manage, in terms of stress applied by fishing effort, rather than the alternative to monitoring the system level processes. Gribble (2000) developed a trophic mass-balance ecosystem model of the northern GBR, which concentrated on predator-prey and dominance relationships of the mainly inter-reefal assemblages. Data for the simulations were from a joint study by the Queensland Department of Primary Industries (QDPI) and the Commonwealth Science and Industry Research Organisation (CSIRO) to investigate the effects of trawling in the northern GBR. Cross-shelf surveys were completed in a 10,000 sq nautical mile cross-reef shelf transect, collecting data on over 1000 taxa (Poiner *et al.* 1998).

The original model focussed on the impact of the industrial prawn trawl fleet (Gribble 2000) but included the facility to monitor the effects of a second fishing “fleet”, comprised of commercial line fishing and indigenous subsistence fishing. It is proposed in this study to expand the Gribble (2000) model and use it as a tool to explore the possibility of applying the principle of ecosystem management to the GBR Marine Park. Of particular interest is scenario modelling the interaction of traditionally independent management plans for the various overlapping fisheries, to explore potential ecological conflicts or synergisms. This has not been attempted previously, even for multi species fisheries let alone where multiple multi-species fisheries overlap.

OBJECTIVES

- (1) To expand the 25 compartment mass-balance trophic model, constructed for the northern GBR

ecosystem (Gribble 2000), which focused primarily on the ecosystem effects prawn trawling, into a 30 compartment model that can be used to explore:

- the effects of line fishing on targeted *lutjanids* and *serranids*, and its downstream ecosystem effects;
- the inclusion of more recent coastal river survey data to explore estuarine-reef connectivity implications of coastal and estuarine fishing (commercial gillnet, recreational and indigenous subsistence harvest).

(2) Using the latest ECOPATH EwE software to include time-series of logbook catch statistics in the modelling. The most recent fisheries data, collated for the QDPI 1995-2001 Fisheries Resource Condition and Trend Report, for trawl, reef line, and coastal and estuarine fishing will be used to tune the expanded model and increase the robustness of its output.

(3) To apply these models to different sets of linked marine ecosystems; coastal mangrove/stream, reef lagoon and inter-reef, and coral reef. Of particular interest are the biomass and species flows or “connectivity” between the coastal mangrove/stream and reefal systems.

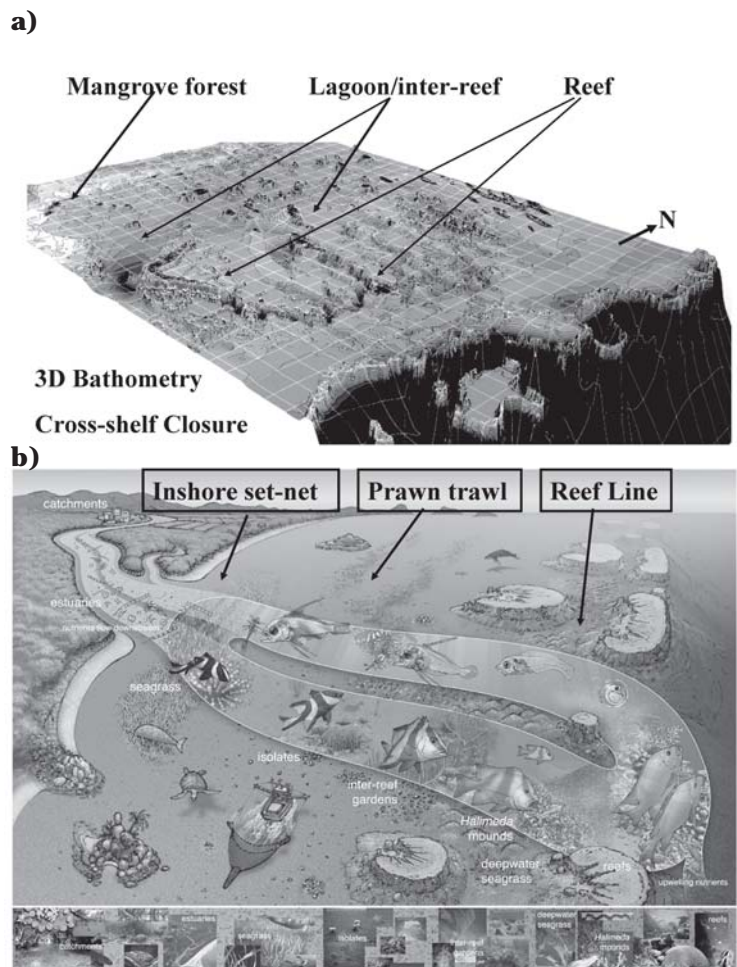


Figure 1. a) Cross section of the Australian continental shelf and Great Barrier Reef study area. *Source:* CSIRO. b) Schematic diagram of linked habitats and associated fisheries. *Source:* Cappo and Kelly 2000.

From a fisheries management perspective, the application of such a linked model would be to determine the ecosystem effects and interactions of proposed management plans in each of the component fisheries. Because all such plans have to meet Ecologically Sustainable Development (ESD) requirements, the problem is actually ecological rather than of conventional single-species optimal harvest. Hence the need to widen the traditional fisheries management perspective and information pool.

(4) Provide simulations of a variety of management scenarios to explore the interaction of management initiatives in the reef line and coastal fisheries with those of the trawl fishery and vice-versa. It is possible for synergisms to be generated but it is also possible that “good” management practices in one fishery may have negative effects on another. It is also possible that the cumulative ecosystem effects of a number of fisheries, while not harmful in isolation, may have unforeseen negative impacts.

RESULTS

Main characteristics of the ECOPATH model

The ecosystem simulations of the northern Great Barrier Reef World Heritage Area (GBR-WHA) were implemented using ECOPATH EwE (version 5 beta) software (Christensen and Pauly 1992) using the ECOSIM and ECOSPACE routines for temporal and spatial simulations respectively (Christensen *et al.* 2000). More detail on the structure and underlying equations of ECOPATH EwE, and of the base

“GBR-prawn” model, are presented on the ECOPATH website www.ecopath.org and in Christensen *et al.* (2000) and Gribble (2000).

Trophic structure of the GBR World Heritage Area Model

The base ecosystem model is an equilibrium trophic hierarchy, with the biomass flows balanced such that there are not more predators than prey to feed them, nor conversely are there “wasted” prey with insufficient predators to exploit the resource.

Data sources/Parameter estimates

Estimates of species composition and biomass of the major avian, reptile, fish, mollusc, and crustacean assemblages (including the harvested species and discards), as well as diet, consumption and production were calculated from:

- two annual cross-shelf prawn-trawl surveys in the 10,000 sq km far-northern GBR study area (Fig 1). Biomass of fish and non-fish taxa were based on parallel fish-trawling and benthic dredge samples taken at the time of the prawn surveys, (Poiner *et al.* 1998);
- the literature on prawn predation (Brewer *et al.* 1991; Salini *et al.* 1990, 1992, 1998; Haywood *et al.* 1998; Randall *et al.* 1990; Roman *et al.* 1990);
- Diet and life-history information in FISHBASE (Froese and Pauly, 2002) fish database;
- Previously published Ecopath models; (a) the trophic interactions in Caribbean coral reefs, Opitz (1993, 1996), (b) for the shrimp fishery in the Southwest Gulf of Mexico (Sherry Manickchand-Heileman, University British Columbia Fisheries Centre, 1999, pers. com.), and, (c) the mangrove forest of Darwin Harbour, Northern Australia, (Julie Martin, Northern Territory University, 2002 pers com); and
- Additional information on the species composition and biomass of inshore mangrove associated biota (see Fig. 1) gathered by field survey in the far northern GBR in 2000 (Sheppard *et al.* 2001) This habitat had not been sampled extensively in the original surveys of the lagoon, inter-reef and reef (see Poiner *et al.* 1998). Fisheries data for the coastal gillnet fishery that operates in this habitat came from the QFISH compulsory logbook database. Logbook records had been partially validated by research fisheries observers in 1999 (Gribble, unpublished data).

All data, not derived from the GBR-WHA surveys, were taken from tropical prawn grounds with similar general characteristics. The expanded “GBR-WHA” model deals mainly with the inner lagoon and inter-reef trawl grounds as this habitat complex represents 80-90% of the World Heritage Area, (Poiner *et al.* 1998). The coral-reef proper was included, as was the reef line fishery, but the model represents a simplification and generalisation of the fractal-like complexity of this ecosystem. Similarly only a simplification of the full mangrove forest/swamp ecosystem, along with the coastal gillnet fishery, was included in the model.

Combined with the original parameter estimates in “GBRprawn”, the new survey data increased the species used to estimate trophic guilds to over 500 fish and 900 non-fish species. New trophic guilds of “inshore finfish” and “Dugong” (a protected inshore species) were added. The species mix of “small schooling fish” and “small fish omnivores” trophic guilds changed due to the addition of the mangrove and estuary species, hence the proportion of these guilds taken as trawl bycatch also changed because of a reduction in vulnerability to capture in the inshore areas. The biomass estimates for both guilds were also increased appropriately, in line with the combined survey estimates.

The cross-shelf connectivity or linkage between component ecosystems was introduced via the split-pool facility within ECOPATH, where juveniles with a particular diet can be linked with adults whose diet has changed. For example herbivorous plankton feeding in juveniles of the “small schooling fish” trophic guild, changes to a more carnivorous diet in adults. This guild includes species commonly referred to as “baitfish”, which are commercially harvested from inshore shallow waters, as well as being significant prey items for mid-shelf and offshore fish species. The latter are high value targets for commercial reef-line and gill-net fisheries. The “small schooling fish” trophic guild is also a component of the lagoon and inter-reef bycatch from prawn trawlers and, partly as a consequence of this, is a component of the diet of seabirds.

Estimates of biomass, consumption, production, and diet matrices (see Gribble 2000) represent the underlying assumptions of the model and a different set of assumptions may also produce a balanced model. As with all models the aim was to capture the major biomass dynamics and flows of the much more complex, “real” system.

Note: Heuristic validation of the basic model using historic logbook data has been reported separately in Gribble (2000) and Gribble (in prep.).

Spatial simulations and speed of movement

The “GBR-WHA” model was made spatially explicit by mapping five broad habitat types, (fringing mangrove swamp, inner reef lagoon, inshore reefs/islands, reef/shoals, and outer reef lagoon) onto a virtual landscape and moving the trophic guilds across them. The land and islands were mapped as “no-movement areas” and the trophic guilds distributed around rather than across them. Movement rates were set at biologically reasonable speeds for typical species within each guild.

Fishery parameter estimates

The fishery component of the model was divided into three fleets, with the recreational fishery harvest not treated separately but included as a component of each:

- The inshore gillnet fishery (FLEET 1), which tends to take large predatory coastal fish, or through the baitfish fishery, to take small schooling fish, bycatch for this fishery is mainly small omnivorous fish. species. There also is a very small bycatch of the inshore seagrass herbivores, Dugong and Green turtles (Gribble *et al.* 1998);
- The reef line fishery for large reef/inter-reef carnivores, both schooling and non-schooling fish, which was combined with the Indigenous harvest of turtles (FLEET 2); and
- The prawn trawl fishery for penaeid prawns (FLEET 3), which produces the highest proportion of discarded by-catch, mainly small fish, conservatively estimated at a ratio of 6: 1 by weight of by-catch to retained catch (Poiner *et al.* 1998).

Estimates for the recreational fishery were derived from two overlapping sources; a national survey of non-commercial fishing in 2001, and, the Queensland RFISH database which is based on telephone and logbook surveys carried in 1999 and 2001 (Jim Higgs, QFS RFISH program, 2002 pers com). The list of targeted species is extensive and fish are caught from the inshore mangroves, lagoon, inter-reef and reef. Icon recreational species are Coral trout (Reef), Mackerel (Lagoon), and Barramundi (Inshore Mangroves).

The gillnet fishery was restricted to the fringing mangroves and inshore lagoon. The trawl fleet could fish in both the inshore lagoon and the inter-reef but the cost of fishing increased by 10% to 40% further offshore into the inter-reef habitat. The reef line fishery fleet was restricted to the reef-shoal and inter-reef habitats. Again it was made slightly more “costly” to line fish in the offshore sections of these habitats rather than in the more accessible inshore edge of the reef-shoal and inter-reef. The rationale for these increasing costs were the increased fuel required to travel further offshore, increasing loss of fishing gear in the rougher terrain, and an increased risk of boat damage in the poorly charted offshore reef-shoal zone.

Further out, the offshore lagoon habitat was not fished in this simulation because of its exposed position, very rough ocean floor (extensive plate coral), and to reflect refugia for turtles and seabirds around nest-site islands and shoals. This scenario broadly matched the known fishing behaviour of trawlers and line fishers in the far northern GBR (Gribble and Robertson 1998; Poiner *et al.* 1998).

SUMMARY OF PROGRESS

Two versions of a GBR-WHA ecosystem model were developed. The first is an extension of the original GBR-prawn model (Gribble 2000, 2003) with additional species and trophic guilds added. The second is a more profound change from the original model where all parameters have been re-defined and the spatial area modelled has been increased significantly (to simulate a “virtual GBR”).

In both versions three sub-systems, inshore/mangrove, lagoon/inter-reef, and coral-reef were modelled, running within the overall ecosystem model. Some trophic groups such as sharks were common linking elements but for each sub-system, species within them tended to eat others within that subsystem. Balancing each sub-system then balancing the whole ecosystem was more difficult than originally envisaged. Optimal balances in the trophic flows were delicate and it was possible to move from one stable state to another, with one or more sub-systems sub-optimally balanced but the overall ecosystem stable.

This is possibly a manifestation of the “multiple stable state” theory in ecology but is somewhat counter intuitive as it suggests that the GBR ecosystem is less robust than my observations had led me to believe. It is possible that my models so far have not included a “buffering” trophic relationship, which would increase the stability of an optimal set of trophic flows. Alternatively the situation could be a dynamic one where at different times/seasons and under different levels of harvest stress the ecosystem could move backwards and forwards through a series of stable states.

It is also possible that homeostatic mechanisms operate over different time-scales such that ultimately a optimal balance will be restored but on an evolutionary rather than “human” scale. This may be true for the northern hemisphere cod, which appears to have been replaced by arctic cod in over-fished fisheries, but tropical reef fish stocks are subject to high levels of natural disturbance and could be expected to have evolved fairly rapid compensatory strategies.

In this regard I have negotiated with Prof. Carl Walters, UBC Fisheries Centre, to critically review the ongoing development of GBR models and hopefully to suggest appropriate ways to test and refine them.

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SECTION 2

FISHERIES AND MARINE ECOSYSTEM OF NEW SOUTH WALES

SECTION 2A

FISHERIES OF NEW SOUTH WALES

ESTUARINE FISH AND FISHERIES IN NSW: SOME CONSIDERATIONS FOR ECOSYSTEM-BASED MODELLING AND MANAGEMENT

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OVERVIEW - NSW ESTUARIES AND FISH

Various types of estuaries of differing sizes occur along the coast of NSW, including drowned river valleys, barrier estuaries and intermittently closed and open lagoons (Roy *et al.* 2001). These different types of estuaries are not distributed equally along the coast and they have different geomorphic and hydrographic characteristics. Consequently, the flora and fauna and the fisheries they support can vary among estuary types (Pease 1999). Many estuaries have been significantly altered and habitats degraded by anthropogenic activities, and they are also subject to much impact from surrounding land uses.

More than 250 species of fish have been identified from NSW estuaries, of which about 120 are harvested (Pease 1999). The effects of fishing on these fish can vary depending on the life history characteristics and strategies of each particular species. Information on life history characteristics is adequate for the most common and valuable species, but there is a dearth of information for the majority of species. Many species are estuarine dependent (e.g. River Garfish, Estuary Catfish, Estuary Perch, Australian Bass, Black Bream, *Ambassis* spp.), whereas others are estuary opportunists (e.g. Yellowfin Bream, Sea Mullet, Luderick), as they also inhabit coastal waters. Several of these species are known to move great distances along the coast and between distant estuaries (see Pease this volume). The different life history stages of some fishes occur in different environments. Several species spawn at sea and spend their early juvenile stages in estuaries. Consequently for many fish species, there is a significant linkage between coastal waters and estuaries.

Within an estuary, there exist spatial and temporal differences in the distribution of ichthyofauna depending on hydrography (e.g. salinity gradients), distance from estuary mouth and habitat distribution. The habitat relationships of estuarine fish are relatively well described for many species (Bell *et al.* 1984; 1992; Burchmore *et al.* 1984, 1988; Middleton *et al.* 1984; Ferrell and Bell 1991; Ferrell *et al.* 1993; Gray *et al.* 1990, 1996; West and Walford 2000; West and King 1996). Trophic interactions among species are less understood (but see SPCC 1981).

COMMERCIAL FISHERY

Fish have been harvested commercially from NSW estuaries since the early 1800s. In recent years more than 800 commercial fishers have operated annually in the estuary fishery, producing in 2002/03 approximately 3200 t of finfish valued at around \$AUD 10 million. The fishery is relatively small-scale on a global scale, and could be considered artisanal. It is, however, important on a regional scale, and has a large interaction with other regional coastal fisheries, including recreational. Most of the important species of fish harvested commercially from estuaries in NSW are also highly sought after by recreational fishers and, in some estuaries for some species, the recreational catch is very large and exceeds that of the commercial sector (West and Gordon 1995; Steffe and Chapman 2003; Henry and Lyle 2003).

The estuarine commercial fishery is the most complex multi-method and multi-species fishery in

NSW. The fishery is currently managed by a complex set of gear and fishing time specifications, area and seasonal closures, and minimum fish length restrictions. Typically, many fishers operate on a single basis and are adept at switching from one fishing method to another, and from targeting one suite of species to another. Commercial fishers use approximately 20 different fishing methods, including both towed and static gears, the main fishing gears being gillnets and beach-seine nets for finfish and seine, stow and trap nets for prawns. Different combinations of gears are used in different estuaries to target different suites of organisms. The potential environmental impacts of fishing gears on other flora and fauna can be gear-specific and can also vary among estuaries (Gray 2001, 2002; Gray and Kennelly 2003; Gray et al. 2003 a,b, 2004).

Although commercial fishing takes place in over 80 estuaries, about 70% of landings are taken from the top 10 estuaries (Gray *et al.* 2000). The Clarence River in northern NSW supports the largest estuarine finfish fishery, accounting for approximately 20% of total reported estuarine production. This fishery, however, is not typical of all estuaries. The other high producing estuaries include the large barrier estuaries in central NSW.

Annual commercial production of finfish from NSW estuaries has fluctuated over the past 40 years. From the mid 1950's to the late 1970's annual finfish production generally fluctuated between 3000 and 4000 t, after which it increased to peak at about 5600 t in 1986/87, but has since declined to about 3200 t. Although more than 120 species of fish are caught in the estuarine fishery, the top 10 species generally contribute around 85% of total finfish landings from estuaries. The principal species (by weight and monetary value) comprising the estuarine fishery are sea mullet (*Mugil cephalus*), luderick (*Girella tricuspidata*), yellowfin bream (*Acanthopagrus australis*), sand mullet (*Liza argentea*), dusky flathead (*Platycephalus fuscus*) and sand whiting (*Sillago ciliata*).

Reported catch data is available for most species since the mid 1900's. These data can provide information on spatial and temporal trends in reported production. Catch per unit of effort (CPUE) data are available since the late 1970's. These data, however, are relatively crude (fisher-days) as they are pooled across all methods. More refined CPUE based on individual methods is available since the mid 1990's. No biomass estimates exist for estuarine fish in NSW.

Sex, length and age composition of retained catches is available for several species (e.g. sea mullet, bream, luderick, dusky flathead, sand whiting, river eel, silver biddy: Gray *et al.* 2000, 2002; Smith and Deguara 2002). Data are available for varying times and estuaries and, for some species, data on ocean catches are available. Length composition data of catches are available for some species since the 1950's.

Observer-based surveys of retained and discarded catches have been done on several gear types (Gray *et al.* 1990; Andrew *et al.* 1996; Liggins *et al.* 1996; Liggins and Kennelly 1996; Gray 2001; 2002; Gray *et al.* 2003 a,b, 2005a, b; Gray and Kennelly 2002; 2003; Gray *et al.* 2006). These studies document that the species discarded and discarding rates vary among gear types and estuaries. Thus any impacts of discarding may vary depending on the species, gear-type and estuary. Several studies have been initiated to develop and test discard-reducing gears, most notably in the prawn gears (see Broadhurst 2000; Broadhurst *et al.* 2004; Macbeth *et al.* 2006) but also in the gillnet (Broadhurst *et al.* 2003; Gray *et al.* 2006) and beach-seine fisheries (Gray *et al.* 2000; Gray and Kennelly 2001; Kennelly and Gray 2000).

RECREATIONAL FISHERY

Large recreational fisheries occur in estuaries in NSW. Although no estimates of total weight of harvest is available, the recent national survey estimated that more than 5.9 million fish were retained by recreational fishers in NSW estuaries per annum. As with the commercial fishery, the catch composition of the recreational fishery varies between different estuaries, and also between boat and shore-based fishing. No long-term assessments of recreational fishing exist. Rather, several creel surveys of recreational fishing in various estuaries have been done (Steffe and Chapman 2003; Steffe and Macbeth 2002 a,b). These studies provide spot estimates of catch composition and harvest size.

CONCLUSIONS

Several studies have assessed aspects of the biology and ecology of some key estuarine fish species. The retained and discarded catches from commercial fishing are relatively well documented. Aspects of retained recreational catches are also understood, but little is known of discarding in the recreational sector. The assessment of fish assemblages and stocks needs to be done across different types of estuaries that are subject to different management regimes (e.g. open and closed to commercial/recreational fishing). Appropriately designed, stratified randomised fishery-independent research surveys may be able to bridge this gap. Much process-oriented work needs to be done to understand the dynamics of fish assemblages in estuaries. Greater emphasis needs to be placed on assessing and managing assemblages, rather than individual species.

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NSW ESTIMATES OF FISHING ACTIVITY FROM THE NATIONAL RECREATIONAL FISHING SURVEY 2000/01

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A survey to provide estimates of recreational and indigenous fishing in Australia was implemented in 2000/2001 (Henry and Lyle 2003). This abstract describes some of the principal results of the survey that apply to recreational fishing in NSW. The national survey of indigenous fishing was limited to Northern Queensland, the northern part of Western Australia and the Northern Territory.

The sampling procedure consisted of two phases – a screening sample, followed by a longitudinal study, which followed the fishing activities of individual fishers for a 12 month period. Interviewers had regular telephone contact with the diarist-fishers over the sample period. The statistical design was based on a single-stage cluster sample, with the randomly selected household as the primary sampling unit, and individual fishers (5-years and older) within the household as the secondary unit. Cluster sampling is recommended where there is no frame listing, or where a frame listing is prohibitively expensive to obtain (Schaeffer *et al.* 1996). Previous studies have used only one randomly chosen fisher from each household. This makes untestable assumptions about correlation of fishing behaviour within households and also over-represents single-fisher households.

Data from the screening phase were used to provide estimates of participation rates at 1 May 2000. These data together with those from the longitudinal study and followup surveys of non-response at screening and of non-intending fishers allowed the estimation of participation rates, catch, effort and expenditure over the period 1 May 2000 to 30 April 2001. Full details of the statistical procedures are given in Henry and Lyle (2003).

THE SCREENING SAMPLE

Sample allocation

Allocation of the final target sample of 43 945 households to the six states and two territories was based on the general principle of obtaining estimates of harvest and effort at roughly comparable levels of precision for the lowest level of geographical aggregation for each state. This level was either Statistical Division (SD), as specified for the Australian Census by the Australian Bureau of Statistics (ABS), or where population size was too small at this level, a combination of Statistical Divisions. The sample allocation resulting from this

procedure was generally quite similar to the allocation based on the square root of the population of a state divided by the sum of the square roots of each of the states. The latter procedure is used to allocate interviews between counties for the National Marine Fisheries Service surveys of recreational anglers in the marine waters of the United States (van Vorhees *et al.* 2001). This method of allocation ensures an adequate level of sampling for geographical areas with relatively small populations.

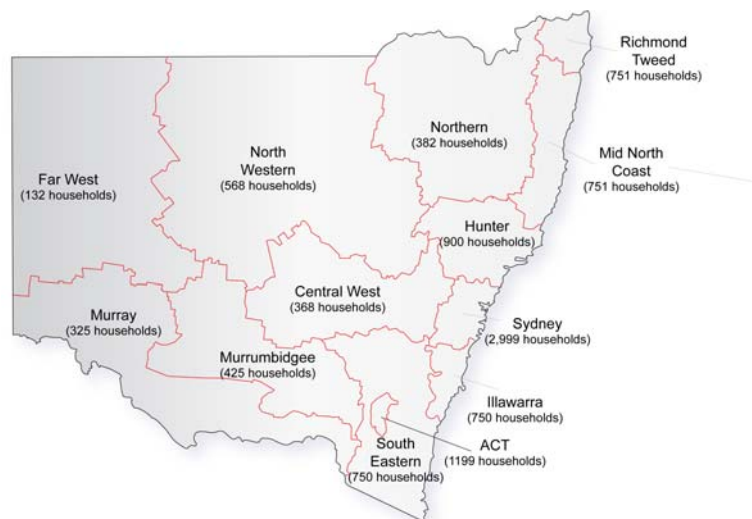


Figure 1. Population strata for the NSW component of the National Recreational Fishing Survey (NRFS).

A probability sample was drawn from electronic white-pages listing. The target screening sample comprised 43 945 households nationally (29 837 full response) and 10 300 households (7 097 full response) in total for the 13 NSW/ACT strata (Figure 1). The usable fully responding sample for the NSW/ACT longitudinal component was 3033 persons (intending fishers), and for the non-intending fishers followup survey, 786 persons.

Representation checks, population benchmarks and expansion of sample

The data obtained from the screening sample were compared to population data for persons in private dwellings at 30 June 2000. ABS Consulting provided estimates of the number of private households by Statistical Division, and the number of persons associated with these. A number of cells in the sample SD/sex/age matrix were not representative of the corresponding population cells. The most common anomalies between the sample and population benchmarks were for households comprising only one person. An integrated weighting approach was used to apply the population benchmarks to the survey sample. This is an iterative method for obtaining a single uniquely defined weight per household that is appropriate for both persons and households, rather than using heterogeneous weights for members of the one household according to gender/age characteristics, and a separately determined weight for the household. Given the cluster design of the survey, and the fact that some estimates are estimated at the person level (effort, catch, etc) and some are based on the household level (expenditure, boat ownership, etc.), this method seemed preferable to the standard post-stratification approach.

Follow-up surveys

In the screening survey, nonresponse arose as a result of refusals (partial or full) and non-contacts. Sub-samples of these two groups were followed up by telephone contact, and categorised in terms of substantive data, including age, gender and previous fishing participation. Comparisons with the response group were undertaken to determine whether adjustments were necessary for data expansion. There was a non-angler bias in households that initially refused the screening interview, with the least avid groups generally under-represented. If this factor was not taken into account, over-estimates of harvest and effort would result. The impact of persons 'unexpectedly' dropping into the recreational fishery was evaluated through interviews conducted at the end of the diary phase with a sample of respondents who expressed no intention to go fishing during the diary period. These non-intending anglers indicated whether or not they had in fact fished during the diary period. These data were set against observed participation rates amongst the intending anglers (diarists) to determine actual participation rates.

RESULTS

Participation

The number of fishers in NSW was estimated to be 998, 501 (standard error 33, 686).

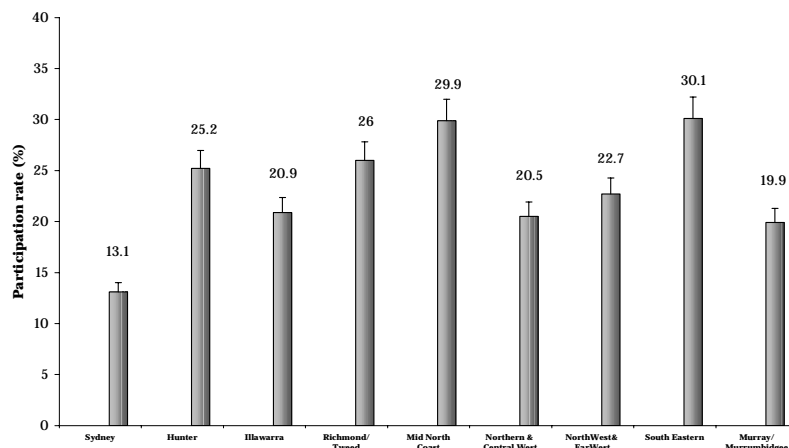


Figure 2. Participation rates by population stratum for NSW.

Effort

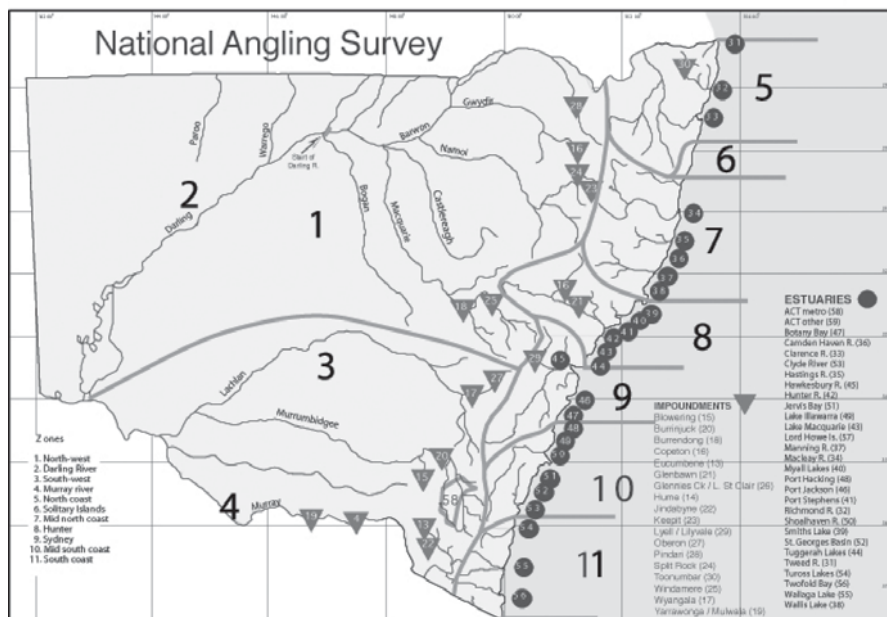


Figure 3. Effort (number of days fished) by fishing subregion.

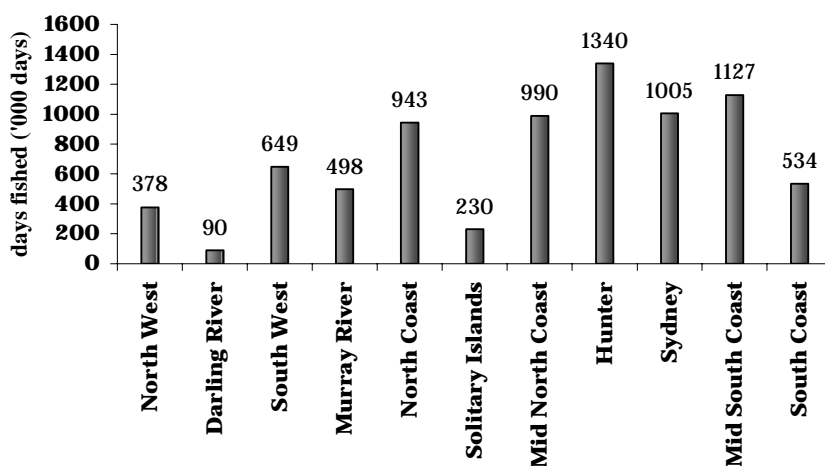


Figure 4. Fishing subregions for NSW used in the NRFS.

Harvest

In NSW, the estimated weight of retained finfish was 6 600 tonnes (s.e. 377). The estimated number of retained finfish was 14 million (s.e. 0.8 million).

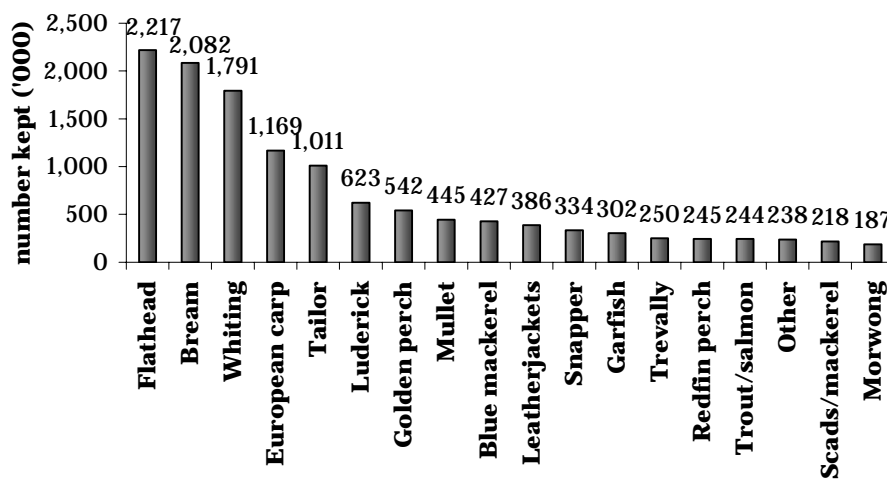


Figure 5. Estimated harvest in NSW (number of fish) for major finfish taxa.

SUMMARY

Data from the survey provide comprehensive estimates of recreational fishing activity in NSW, which may be directly compared to other states and territories, as the survey used uniform methods of sampling and analysis across all states. Data from the national survey provide intrastate and interstate estimates for each state/territory. The sample sizes are substantial at the scale of population stratum level, and allow for suitably precise estimates of catch, effort and fisheries-related expenditure for fishing subregions nominated by fisheries managers. For fisheries with relatively low participation rates (e.g. rock lobster harvesting, which is of the order of 0.1 - 0.2% participation rate for NSW), targeted surveys are required. These usually require a known sampling frame to enable a suitably sized sample to be drawn (from species specific licence frame or frame of licence endorsement etc.). Similarly estimates on a finer scale than fishing subregion, require the use of directed surveys on the appropriate scale.

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SECTION 2B

MARINE ECOSYSTEM OF NEW SOUTH WALES

ESTUARINE STRUCTURE, FUNCTION AND BIODIVERSITY

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The increase in population density along the New South Wales (NSW) coast has seen degradation of natural resources across a range of seascapes. Unfortunately, conservation and planning initiatives for sustained use of existing resources are hampered by a complex decision making process involving four tiers of players: scientific, managerial, executive, and political. While some of the inefficiencies at the first interface have been examined and suggestions made for improvement (Roux *et al.* 2006), little has been done, or perhaps little can be done, to modify operational procedures at the third and fourth tiers.

Ecosystem modelling is the most recent in a series of steps by which the scientific community has endeavoured to stimulate the decision making process. Underlying this approach is the desire to present credible “what if?” scenarios to focus the decision-making framework. Earlier approaches reliant on matters of abstract principles (such as “intergenerational equity”) and simple conceptual models have had little success in getting appropriate executive and political response.

Irrespective of campaigns to improve decision making at higher tiers, the complexity of ecosystems is such that there will continue to be a need for the construction of sound models by which to predict environmental outcomes. This is particularly so for aquatic ecosystems, given the spatial and temporal complexity of these systems. Aquatic systems of unique complexity are estuaries.

In spite of the fact that estuaries are complex systems, one fundamental fact was recognised some time ago. The Remane principle (Remane and Schleiper 1971) indicates that the number of species in an estuary is higher at the mouth than elsewhere. Rochford (1951, 1955), in attempting to establish a hydrologic rationale for estuarine processes, also noticed that some species of plants and animals are uniquely distributed. Subsequent studies of the distribution of macrophytes (West *et al.* 1985) and fish within NSW estuaries have also found differences along a salinity gradient (Gray *et al.* 1990; West and King 1996; West and Walford 2000).

Rochford (1951) identified four hydrologic zones based on water quality characteristics: Marine, Tidal, Gradient and Freshwater. While identifying change at the seasonal scale in some estuarine characteristics, he did not cater for temporal variation at the scale of millennia, century and decade. At the millennial scale, the estuaries of NSW vary in size and shape and hence hydrological character. Shape variation is due to rise and fall of sea level of the order of 120-130m over 105,000 year intervals (Milankovic cycle). The present configuration of the NSW coast has been in place since sea level stabilised at its present height 6,000-6,500 years ago. Twenty thousand years previously, at the last glacial maxima, the NSW shoreline was 15-20 km further east, and some of the now autonomous 950 waterways that drain the coast (Williams *et al.* 1998) would have been linked. For example, the Cooks, Georges and Hacking Rivers appear to have been tributaries of a larger river system (Roy and Crawford 1981).

Sea level has risen and fallen 17 times over the last 2,500,000 million years (Pleistocene Era). Each time sea level falls, the soft sediments of estuaries are scoured out, leaving small freshwater creeks at considerably lower elevations than at present. When the ice caps melt, sea level rises and progressively refills the outer portion of the channel with marine sands. A relic of the scoured channel, in effect a deep hole, remains after sea level stabilises.

Unlike Rochford (1951), Roy (1984) recognised the temporally dynamic nature of shorelines and the impact of sea level on estuarine geomorphology, and proposed a scheme that classified the estuaries of NSW into three major types (drowned river valley, barrier estuary and intermittent estuary) and identified four geomorphic zones, common to all types, on the basis of sedimentary characteristics. These zones, progressing from the oceanic to freshwater reaches are the Marine Tidal Delta (MTD),

Central Mud Basin (CMB, the relic of the scoured channel referred to above), Fluvial Delta (FD) and Riverine Channel (RC).

At the elevated and stable sea level, during the Inter-Glacial periods, the estuaries begin to infill with sediments, albeit at different rates depending on the size of the catchment, amount of rainfall, soil type, topography and other features. As estuaries infill, they evolve. Most conspicuously, the Central Mud Basin progressively disappears as it collects some marine but primarily terrestrial sediment. Erosion in the catchment causes the fluvial delta to move downstream and the floodplain to expand behind it. Another consequence of infilling is a progressive increase in tidal range. For example, the maximum difference in tidal height in Lake Macquarie, a youthful barrier estuary, is of the order of 100mm, whereas along a substantial length of the mature Clarence River the full tidal height experienced along the NSW coast (2m) is felt.

Roy *et al.* (2001) speculated on the strength of the relationship between the four geomorphic zones of Roy (1984) and the four hydrologic zones of Rochford (1951), and suggested the underlying geomorphology influenced the hydrologic framework. So far the strength of the relationship between the two sets of zones has been little tested.

The floral and faunal observations of Rochford (1951, 1955) and other investigators on species distribution along the long axis of an estuary (Table 1) suggest the Remane principle is operating in NSW estuaries. It is another step to assume that zones, either hydrologic or geomorphic, exist and the distribution of biota is clearly bounded. Organisms that appear to inhabit definable locations in estuaries include:

Species of macrophytes. Kelp (e.g., *Sargassum* spp.) and some seagrasses such as *Posidonia australis* and *Halophila* spp. are located in the MTD and CMB portions of estuaries. Seagrasses in the family Zosteraceae have a more expansive distribution, located upstream in the FD as well as the MTD and CMB. The mangrove *Avicennia marina* is located in the MTD, CMB and FD, while the mangrove *Aegiceras corniculatum* is mostly found further upstream in CMB, FD and the RC.

Species of invertebrates. These include the spat of the Sydney rock oyster (*Crassostrea commercialis*) and at least one species of sea urchin (*Heliocedaris erythrogramma*). The former, caught by farmers on sticks in downstream locations, are relocated to central portions of estuaries for faster growth. The recruitment of *H. erythrogramma* to the south side of Botany Bay in the mid 1980s was noted as part of the reason for the large-scale loss of seagrass from the most downstream portion of the Georges River estuary. School prawns (*Metapenaeus macleayi*) use the brackish waters of the RC as a nursery.

Species of fish that recruit to marine fisheries. As the juveniles of the snapper (*Pagrus auratus*) do not penetrate into the low salinity waters of estuaries, it is tempting to conclude they only inhabit the MTD and CMB where salinity is at oceanic levels. (Sampling in offshore locations in the Sydney metropolitan region has located few of these juveniles.) Activities that might disturb the lifestyle of juvenile *P. auratus*, such as channel dredging, might need to be timed or otherwise planned to minimise interference with this valuable resource.

Should some species inhabit discrete zones, this could be considered the last stage in a conceptual model that integrates a number of abiotic and biotic factors:

Geomorphology influences the
 Hydrology (especially salinity) that influences the
 Benthic biota (especially macrophytes and infauna) that influence the
 Mobile fauna, especially distribution of small and juvenile fishes (e.g.,
 nursery function).

The rationale behind the model is that, at the broadest of scales, geomorphic features (such as catchment size, topography and soil type) influence water quality. As well, catchment runoff also reflects anthropogenic change to flow caused by clearing (increased hydrograph) and barriers (decreased

Table 1. Examples of zonal distribution of flora and fauna in NSW estuaries.

Rochford (1951) zones Roy (1984) zones		Freshwater Riverine Channel		Gradient Fluvial Delta	Tidal Central Mud Basin	Marine Tidal Delta
Flora	Species					References
Kelp	Sargassum spp.					West et al. (in prep.)
Seagrass	Posidonia australis				X	Pickthall et al. (2004), West et al. (in press)
Seagrass	Halophila spp.					Pickthall et al. (2004), West et al. (in press)
Seagrass	Zostera spp.				X	Rochford (1951, 1955)
Seagrass	Zostera spp.			X	X	Pickthall et al. (2004), West et al. (in press)
Mangrove	Avicennia marina			X	X	Pickthall et al. (2004), West et al. (in press)
Mangrove	Aerircas corniculatum		X	X	X	Pickthall et al. (2004), West et al. (in press)
Saltmarsh	Sarcocornia quinqueflora			X	X	Pickthall et al. (2004), West et al. (in press)
Saltmarsh	Sporobolus virginica			X	X	Pickthall et al. (2004), West et al. (in press)
Reed	Phragmites australis		X			Pickthall et al. (2004), West et al. (in press)
Fauna						
Oyster spat	Crassostrea commercialis					Rochford (1951, 1955)
Oyster adult	Crassostrea commercialis				X	Rochford (1951, 1955)
Sea urchin	Haliocidaris erythrogramma					Shepherd et al. (1989)
School prawn	Metapenaeus macleayi		X			Ruello (1973)
Juvenile Snapper	Pagrus auratus				X	Trnski (2002)
Adult Mulloway	Argyrosomus japonicus				X	Steffe (pers. comm. 2003)

hydrograph). Hydrology, and particularly salinity, in turn mediates the distribution and abundance of some benthic biota. Species of large plants and/or infauna assume conspicuous, although not necessarily ideal, locations in estuaries but their distributions are modified during extended wet or dry intervals. Anthropogenic events that mimic natural circumstances, such as effluent discharges that manifest themselves as extended rainfall events, or freshwater extractions that give the appearance of extended dry weather, are likely to modify distribution. Put simply, estuarine macrophytes (seagrass, mangrove and saltmarsh) as well as estuarine infauna will change in distribution and extent due to changes in sediment character and water quality.

The relevance of these putative zones to modellers is that estuarine processes are not ubiquitous (e.g. algal blooms, fish distribution are constrained) and need to be partitioned spatially. Spatial partitioning is not readily done in hydrologic terms as long-term water quality studies of any one estuary in NSW are rare. The sampling of mobile fauna to determine zonal distributions is in its infancy and is also expensive given the need for unconfounded sampling designs. Therefore, on the basis that hydrologic and faunal assessments are rare, modellers might choose to use a geomorphic approach. This short circuit has a number of advantages for the creation of complex mathematical models:

1. Geomorphic zones within estuaries are readily differentiated via inspection of topographic maps, aerial photographs and bathymetric contours. Further, the area of the zones can easily be calculated. The latter has obvious benefit for modellers in terms of establishing grid size and scale. Recent work (Mesley, unpublished 2003) with GIS techniques shows promise as an accurate way to identify the least readily identified boundary, the one between the Fluvial Delta and the Central Mud Basin.
2. Substrata of interest (e.g. seagrass beds) can be located within and/or partitioned across zones.
3. Processes of interest (e.g., seagrass beds as nurseries for species of economic importance, or deep anoxic holes as sources of nutrients) can be located within and/or partitioned across zones.
4. More specifically, production of commercial species of fishes, be they prey or foraging species, can be inferred from sample or inferred densities and the aerial extent of habitat types.

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SOME EXAMPLES OF LARGE-SCALE MOVEMENT AND MIGRATION THROUGH THE ESTUARIES AND INSHORE COASTAL ZONE OF NEW SOUTH WALES BY DIADROMOUS, ESTUARINE AND MARINE FISHES, WITH SPECIAL REFERENCE TO THE AUSTRALIAN LONGFINNED EEL, *ANGUILLA REINHARDTII*

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INTRODUCTION

I was initially asked to present a summary of biological and fisheries information on longfinned river eels relevant to the New South Wales (NSW) ecosystem simulation project. The spatial frame for this project is the estuarine and continental shelf regions of NSW. In the course of examining the eel information, it became obvious that many of the significant life history events for such a highly migratory fish species occur in freshwater and oceanic regions outside the spatial frame of the ecosystem project. Many other highly mobile and migratory fish species occur within the spatial frame of the project during some phase of their life history but also occur outside the frame during others. Therefore, I decided to present some information about eels, in conjunction with a more general look at the issue of movement and migration of fish in NSW.

It is important to understand and consider immigration and emigration of different life history stages from the spatial frame in a mass balance ecosystem model. These processes may be ignored if they are assumed to be in equilibrium, but they should not be ignored until they are examined holistically for key species or functional groups within each trophic level. Over a range of temporal scales, these processes may not be at equilibrium.

The primary objective of this paper was to develop a simple compartmental model for summarising the movement of key functional groups of fishes through the spatial frame of the ecosystem simulation project. Functional groups of fishes associated with each compartment were selected as examples to demonstrate movement between compartments in the model. These fish groups were: 1) diadromous species that typically spend some part of their life history in fresh water, 2) non-diadromous, estuarine species that spawn in the ocean then use estuaries as nurseries, 3) marine tropical reef species that spawn in the ocean then use estuaries as nurseries and 4) oceanic pelagic species. A secondary objective within this general framework was to provide detailed information on catadromous longfinned river eels for use as a key species in the ecosystem simulation project. Finally, some factors that may cause uncertainty and non-equilibrium in mass balance ecosystem models are presented.

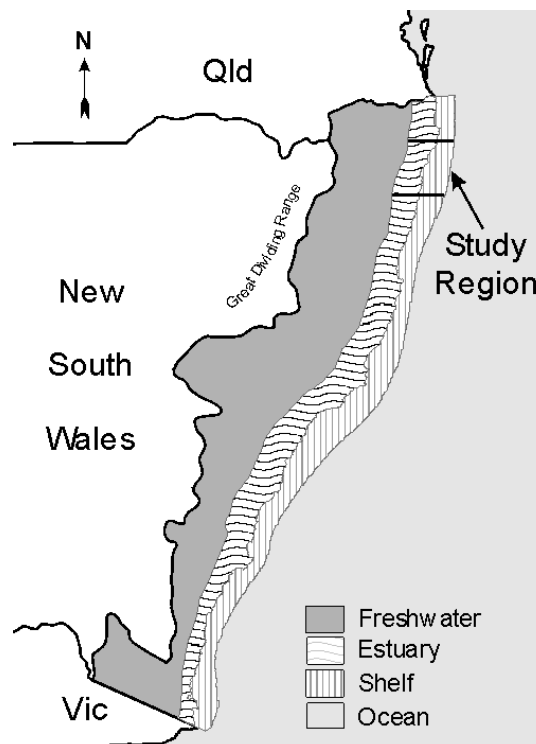


Figure 1. Map of the New South Wales coastal region showing the approximate geographic boundaries of the freshwater, estuary, shelf and ocean compartments. The western boundary of the freshwater compartment is the altitude contour connecting the maximum altitudes of the Great Dividing Range. The western boundary of the estuary compartment is a line connecting the upper tidal boundary of each estuary. The eastern boundary of the shelf compartment is the 200 m isobath. The northern and southern boundaries are constrained by the boundaries defined for the ecosystem simulation model.

METHODS

The basis of the conceptual model is a set of four spatial compartments (Fig. 1). The freshwater compartment consists of the freshwaters within the coastal catchments of NSW. It is bounded by the Great Dividing Range to the west, the state boundaries on the north and south and the coastal estuaries to the east. It is composed of the flowing freshwater rivers and their tributaries as well as the lacustrine waters within natural and man-made impoundments.

The estuary compartment consists of the marine and brackish tidal waters of the estuaries within the coastal catchments of NSW. It is bounded by the freshwater compartment to the west and estuary mouths along the coastline to the east. The northern and southern boundaries are constrained by the spatial frame of the ecosystem simulation model and may range from the state boundaries in the most extensive case to a single estuary in the least extensive case.

The shelf compartment consists of the marine waters between the coastline and the edge of the continental shelf (200 m isobath). The northern and southern boundaries are the same as those of the estuary compartment. The ocean compartment consists of all marine waters outside of the northern, southern and eastern boundaries of the shelf compartment, as constrained by the spatial frame of the ecosystem model.

Most of the information about life history and movement of each functional group of fishes was

Table 1. Summary of information about tagging projects that provided data for this study.

Investigators	Tagging period	Species	Estuaries	Sampling methods	Tag type
John Virgona	1980-1981	Yellowfin bream	Lake Macquarie	Beach seine	Metal opercular
Robert Kearney Ron West Glen Cuthbert Trudy Walford	1988-1991	Yellowfin bream Sand whiting Luderick Dusky Flathead	Richmond River Clarence River Bellinger River Kalang River Nambucca River Macleay River	Beach seine Trawl	Plastic dart
Robert Kearney	1989-1990	Dusky Flathead	Botany Bay Port Hacking	Line fishing	Plastic dart
Robert Kearney Aldo Steffe David Foster	1992-1995	Dusky Flathead Sand whiting Luderick Dusky Flathead	Shoalhaven R. St Georges Basin Lake Conjola Burrill Lake	Beach seine	Plastic dart
Michael Howland	1994-1996	Sand whiting	Richmond River	Beach seine	Plastic dart

Table 2. Number of fish tagged during all projects by estuary and species, with number and percent recaptured for each species.

	Yellowfin bream	Sand Whiting	Luderick	Dusky	Flathead	Grand total
Richmond River	5790	2351	845	289		9275
Clarence River	8016	2050	907	177		11150
Bellinger/Kalang River	553	1981	597	24		3155
Nambucca River	1826	1849	1115	19		4809
Macleay River	976	1490	742	24		3232
Lake Macquarie	2921					2921
Botany Bay					716	716
Port Hacking					10	10
Shoalhaven River	1805	2417	811	53		5086
St George's Basin	295	147	389	146		977
Lake Conjola	2048	629	763	209		3649
Burrill Lake	639	285	702	16		1642
Total tagged	24869	13199	6871	1683		46622
Total recaptured	2035	1530	1051	212		4828
Percent recaptured	8	12	15	13		10

compiled from a variety of published sources. However, the information about non-diadromous, ocean spawning, estuarine species that use estuaries as nurseries was compiled from an unpublished paper presented at the annual meeting of the Australian Society for Fish Biology in 1999 (Pease 1999). Information on yellowfin bream (*Acanthopagrus australis*), sand whiting (*Sillago ciliata*), luderick (*Girella tricuspidata*), and dusky flathead (*Platycephalus fuscus*) was compiled from six separate tagging projects that were conducted by NSW Fisheries (Table 1). The tag and recapture data from these projects are summarised in Table 2.

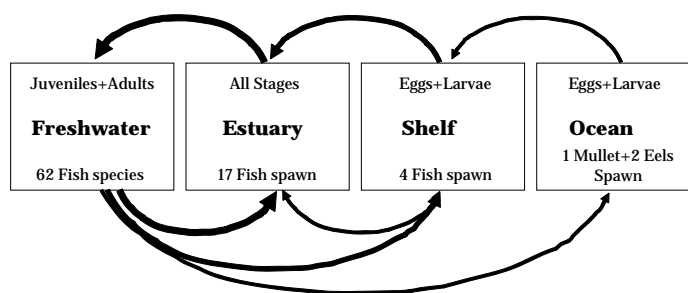
Conceptual models are illustrated with a series of box diagrams. Spatial compartments are represented by boxes. Early life history stages that are found within each compartment are listed at the top of each box, while adult life history stages and spawning activities that occur within the compartment are listed at the bottom of the box. Movement of a life history stage from one compartment to another is represented with an arrow. The relative proportion of all fish of the specified functional group and life history stage that move between compartments is indicated by the thickness of the arrow. The diagrams are not meant to be quantitative. They demonstrate simple conceptual principles.

RESULTS AND DISCUSSION

Diadromous species

Diadromous fish are defined as those that typically move between fresh and estuarine or marine waters at well-defined life history stages (McDowall 1987). There are 62 species of fish that are typically found in the fresh waters of NSW during at least one stage in their life history (McDowall 1996; Harris and Gehrke 1997; Fig. 2). Twenty-two of these species are thought to spawn in the estuary, four are thought to spawn on the shelf and three in the ocean (Harris 1984; McDowall 1996). All of these species are found in the freshwater compartment as juveniles. Adults of most of these catadromous (live in fresh water and migrate to marine waters to spawn; McDowall 1987) and amphidromous (migrate between fresh and marine waters at some regular phase of the life cycle but not for the purpose of spawning; McDowall 1987) species also return to the freshwater compartment after spawning in other compartments. Only three anadromous (live in marine waters and migrate to fresh water to spawn; McDowall 1987) species spawn in the freshwater compartment then use estuarine and shelf nursery areas before returning to the freshwater compartment.

Diadromous and Catadromous Fish Species



Harris, J.H. 1984. Aust. Zool. 21(3):235-250

Harris, J.H. and Gehrke, J.H. (Eds.).1997. Fish and Rivers in Stress. The NSW Rivers Survey. NSW Fisheries
McDowall, R. (Ed.). 1996. Freshwater Fishes of South-eastern Australia. Reed Books.

Figure 2. Summary of movement through the four compartments by the 62 species of fish which spend at least part of their life-history in fresh water.

Diadromous species of most commercial significance are the mullet (*Myxus petardi*, *Myxus elongatus*, *Liza argentea*, *Aldrichetta forsteri*, *Valamugil georgii* and *Mugil cephalus*) and river eels (*Anguilla reinhardtii* and *A. australis*). The sea mullet (*M. cephalus*) has a well documented life history demonstrating extensive northward migration to oceanic spawning grounds (Kesteven 1953; Virgona *et al.* 1998). However, catadromous river eels illustrate the most extreme migration strategy among compartments of any fish species in NSW. During the sexually immature stages of their life history they live in a wide range of habitats encompassing both the freshwater and estuary compartments within coastal catchments, from montane streams and impoundments (Gehrke *et al.* 1999; Gehrke and Harris 2000) to marine dominated habitats near the mouths of estuaries (Walsh *et al.* 2003). When they become sexually mature they migrate to the deep ocean to spawn once before dying, i.e. they have a semelparous reproductive strategy (Walsh *et al.* 2003).

Two species of catadromous anguillid eels occur in abundance and are harvested commercially on

the east coast of Australia. *Anguilla reinhardtii* (Steindachner 1867), known locally as the long-finned eel and hereafter referred to simply as longfin, is a predominantly tropical species found in the coastal catchments of eastern Australia from Cape York to Tasmania (Beumer 1996). It is also known to occur in New Guinea, Solomon Islands, New Caledonia, Lord Howe Island and New Zealand (Schmidt 1928; Ege 1939; Allen 1991; Jellyman *et al.* 1996). *Anguilla australis* (Richardson 1841), known locally as the short-finned eel and hereafter referred to as shortfin, is a predominantly temperate species found in the coastal catchments of eastern Australia from southern Queensland (Caboolture River) to Tasmania (Beumer 1996). This species also occurs in New Caledonia, Norfolk Island, Lord Howe Island, and New Zealand (Schmidt 1928; Ege 1939; Dijkstra and Jellyman 1999).

Both species are believed to spawn in deep tropical waters of the south-western Pacific Ocean north of New Caledonia (Aoyama *et al.* 1999). The leptocephalus larvae are carried to the east coast of Australia by the East Australian Current (Jespersen 1942; Castle 1963; Jellyman 1987; Beumer and Sloane 1990), where they metamorphose into post-larval glass eels which recruit to eastern Australian estuaries. In Australia, juvenile and adult longfins are most abundant in the coastal catchments of Queensland and New South Wales (NSW), while shortfins are most abundant in Victoria and Tasmania. Both species are relatively abundant in NSW, where their distributions overlap.

Large numbers of both species recruit to the estuaries of NSW as glass eels annually (McKinnon *et al.* 2002).

Longfin glass eels recruit primarily from January through May, while shortfins recruit primarily from May through August (Pease *et al.* 2003). Within the coastal catchments glass eels start to develop pigmentation and metamorphose into yellow stage, or sexually differentiated, immature eels. Yellow eels remain in the coastal catchments for 10 to 50 years (Beumer 1996; Pease *et al.* 2004) until they become sexually mature silver eels that migrate back to the oceanic spawning grounds in autumn. The conceptual model of eel migration is illustrated in Figure 3.

Shortfins move to the freshwater

compartment within the first year or two of residence in coastal catchments, where they generally remain until they become silver eels (Beumer 1996). A high proportion of the female longfins move to the freshwater compartment as glass eels and young yellow eels, while some of the females and most of the males remain in the estuary compartment until they become silver eels (Walsh 2003; Walsh *et al.* 2003; Walsh *et al.* 2004).

There is a significant trap fishery for eels in the estuary and a limited permit trap fishery in freshwater impoundments. The total eel catches from these fisheries are summarised in Figure 4. Prior to 1980 a high proportion of the catch may have been comprised of shortfins but since that time it is believed that they comprise less than 20% of the total annual catch (Walford and Pease 2000). Therefore, our studies of yellow eels in NSW have concentrated primarily on longfins.

The majority of eels that are commercially landed in NSW are harvested by the estuarine trap fishery for large yellow eels. This fishery has operated since at least 1970 and the number of fishers has remained stable since at least 1984. Annual landings and catch per fisher-month increased through the 1980's to a peak in the early 1990's when the high-value export market developed. Fishing effort, as measured by catch per fisher-month, has remained high since then but annual landings and CPUE declined in the mid 1990's, then levelled off. This temporal pattern in CPUE indicates that harvest

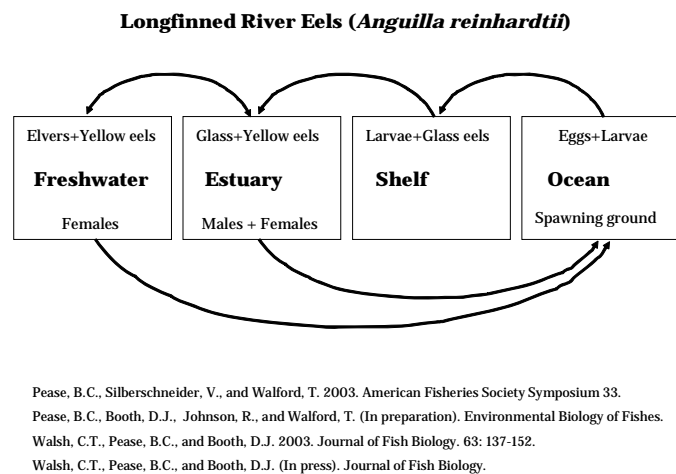


Figure 3. Summary of movement through the four compartments by the two species of river eels (*Anguilla reinhardtii* and *A. australis*).

prior to the early 1990's was having little impact on eel stocks, but increased effort in the early to mid 1990's rapidly reduced the level of available surplus production. The stable nature of annual landings (estimated at 150 to 170 thousand eels in recent years) and catch per unit of effort since the mid 1990's, indicates that this fishery is operating within sustainable limits. Using the simple "Key facts" technique for estimating life history parameters in Fishbase (Froese and Pauly 2000), natural mortality of longfin eels was estimated to be 0.13 per year (standard error = 0.2 - 0.9) with a fishing mortality (assuming minimum market size of 58 cm) of 0.31 per year (no s.e. estimate available) in the Clarence River, which has historically supported the largest eel fishery in NSW.

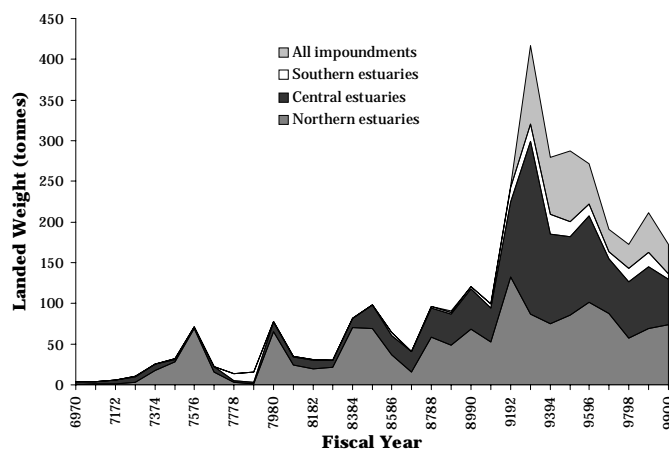


Figure 4. Landed weight of yellow eels from all estuarine and freshwater impoundment trap fisheries from Walford and Pease (2000) with catches updated through fiscal year 1999/2000. Estuarine landings divided into the three estuarine bioregions defined by Pease (1999).

Longfins should also be considered as a key species in the ecosystem simulation project because: 1) all of their life history stages, except eggs and larvae, are found in all of the estuaries of NSW (Pease *et al.* 2003; Walsh *et al.* 2004); 2) they support a significant fishery in the estuary compartment (Walford and Pease 2000); and 3) they also play a significant role in the food web within coastal catchments (Beumer 1979). Yellow phase longfins are high level predators because they attain a large size (up to 1.65 m in length and 22 kg in weight; Beumer 1996) and eels > 30 cm in length are principally piscivorous (Beumer 1979; Sloane 1984).

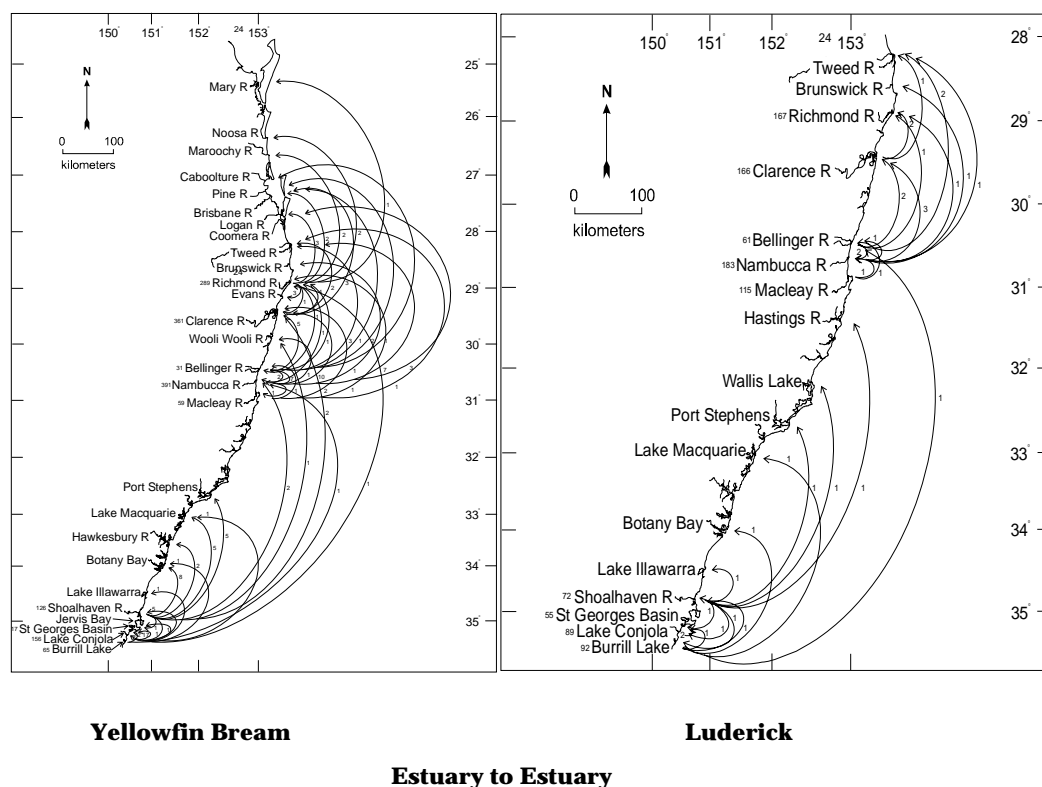


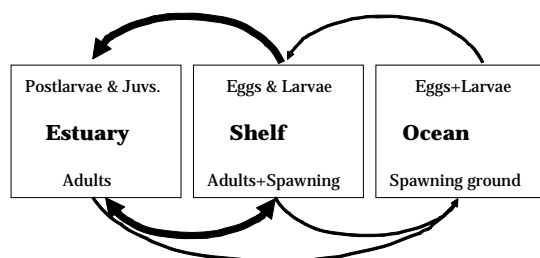
Figure 5. Movement of tagged yellowfin bream (*Acanthopagrus australis*) and luderick (*Girella tricuspidata*), which were recaptured in an estuary other than the one they were originally tagged in. The origin of each arrow shows the estuary that fish were tagged in. Each arrowhead shows the location of each recapture. Numbers next to arrows indicate the number of fish with identical origin and recapture sites. Numbers on the west side of estuaries indicate the number of fish tagged in those estuaries.

Table 3. Summary of information about tagged fish that emigrated from the estuaries they were tagged in. Percent of emigrated recaptures has been calculated using only fish that had been at large for more than 90 days in order to reduce the non-movement bias associated with tagging stress and increased sampling effort associated with the tagging operation. Other tabled values are based on all emigrated recaptures.

Species	Emigrated from tag estuary			Travelled north	
	Percentage of recaptures	Total number	Percentage of emigrants	Mean distance (km)	Max. Distance (km)
Yellowfin bream	20	250	94	201	850
Luderick	6	37	86	163	440
Sand whiting	7	59	86	110	450
Dusky flathead	12	17	94	85	280

Longfins < 30 cm are more omnivorous and eat a wide range of crustaceans as well as fish. Glass eels of both species are probably an important component of lower trophic levels because they are relatively small (< 65 mm in length; Pease *et al.* 2003), recruit annually in high densities (McKinnon *et al.* 2002), and are probably eaten by most piscivorous fish, including eels (Beumer 1979; Sloane 1984).

Commercially Important Coastal/Estuarine Fish Species



Bell, J.D., Pollard, D.A., Burchmore, J.J., Pease, B.C., and Middleton, M.J. 1984. AJMFR. 35: 33-46.
Hannan, J.C., and Williams, R.J. 1998. Estuaries. 21(1): 29-51.
Steffe, A.S. 1991. PhD Dissertation. Macquarie University.
Pease, B.C., Bell, J.D., Burchmore, J.D., Middleton, M.J., and Pollard, D.A. 1981. SPCC. Report BBS23.
Pease, B.C. and Walford, T.R. (In preparation). J. Fish. Biol.

Figure 6. Summary of movement through the marine and estuarine compartments by some of the commercially and recreationally important fish species that spawn in the ocean then recruit to estuarine nursery habitats as juveniles.

bream that were recaptured outside of the estuary that they were tagged in. It can be seen that a number of tagged fish from all of the study estuaries emigrated and many of these fish travelled relatively great distances, mostly in a northerly direction. Emigration characteristics of all four species are summarised in Table 3. Total emigration rates (excluding recaptures within 90 days of tagging to minimise potential fishing effort and behavioural biases associated with the tagging operation) varied from 6 to 20 percent. These values cannot be used as direct estimates of population emigration rates because there are no accurate estimates of fishing effort associated with the recaptures. However, they may provide relative order of magnitude estimates for the different species. The majority (86 to 94 percent) of all emigrants travelled north and all species demonstrated the capability of moving hundreds of kilometres.

It is assumed that the primary reason for emigration of these ocean-spawning species from the estuaries is spawning. All of the yellowfin bream, sand whiting and dusky flathead, as well as all but 5 of the luderick that emigrated were large enough to be sexually mature. Each time they spawn the emigrants that travel north to spawn may either remain in the ocean or return to an estuary further north. In this way they may travel further up the coast each year. The south setting EAC brings the

Non-diadromous, ocean spawning, estuarine species that use estuaries as nurseries

Recent tagging studies (Pease 1999) have shown that four of the top seven fish species in the estuary general fishery ranked by weight (Tanner and Liggins 2001) are ocean spawning species that may travel relatively great distances northward over the course of their lives. These four species are yellowfin bream, luderick, sand whiting and dusky flathead. Interestingly, the other three species (sea mullet, sand mullet and longfinned eels) are all diadromous species discussed in the previous section.

To illustrate the movement patterns of these tagged fish, Figure 5 summarises the movement of tagged yellowfin

larvae back to estuaries south of the spawning location. The proposed conceptual model for movement of these species is shown in Figure 6. Movement into and out of the ocean compartment represents movement across the northern boundary of the shelf compartment.

Marine tropical reef species that spawn in the ocean then use estuaries as nurseries

Pease and Herbert (2002) observed that 18% of the fish species recorded from Port Hacking during the 95 year period between 1903 and 1998 were tropical transients which probably came south with the EAC. Some of the larger species may swim south from more tropical waters as large juveniles and adults but approximately 13% of the reported species are small tropical reef species (primarily in the families chaetodontidae, pomacentridae and acanthuridae) that are transported south as larvae by the EAC. Most of these species do not survive through their first winter in the colder temperate waters (Hutchins 1991).

Therefore, the conceptual model

depicts a one way flow of larvae and small juveniles from the ocean compartment (north of the ecosystem spatial frame) through the shelf compartment to the estuary (Figure 7).

Oceanic pelagic species

Many of the fish species in the shelf compartment are oceanic species that move very large distances and pass through this compartment during particular seasons when oceanographic conditions are suitable. The large tunas (scombridae) and billfish (istiophoridae) are known for their extensive trans-oceanic and even inter-oceanic movements and migrations (Kailola *et al.*

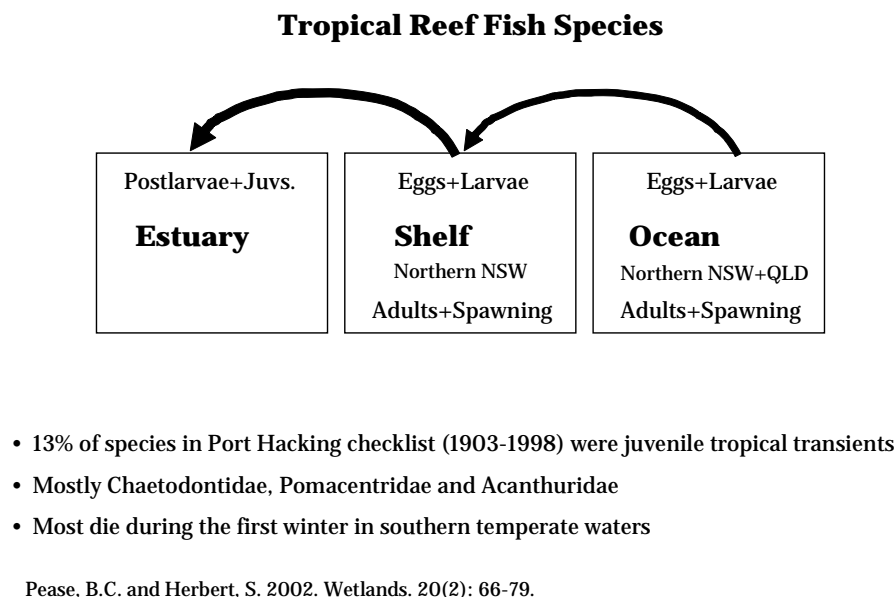


Figure 7. Summary of movement through the marine and estuarine compartments by marine tropical reef fish species that recruit to estuaries from the East Australian Current.

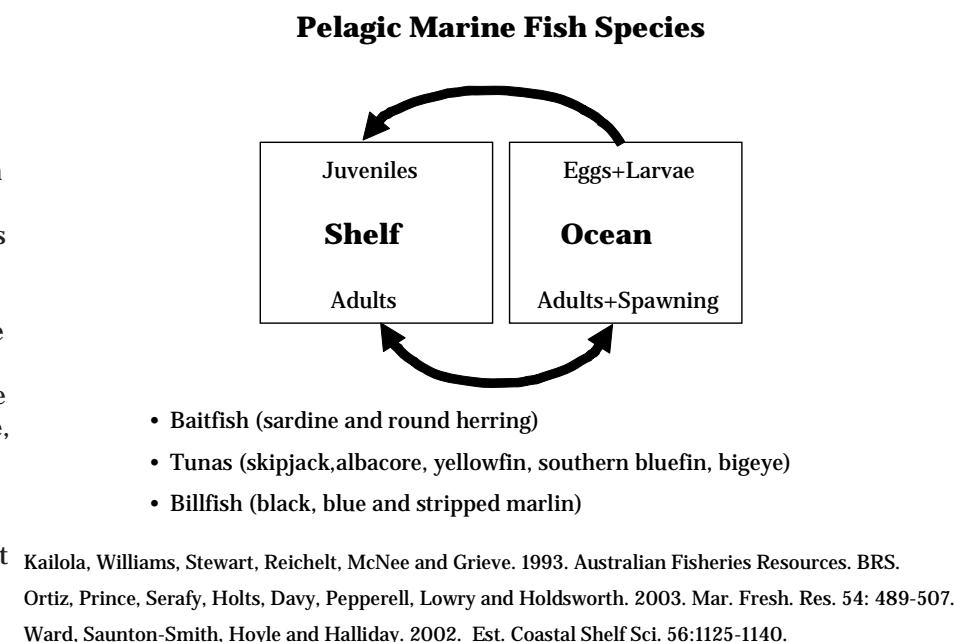


Figure 8. Summary of movement through the marine compartments by oceanic pelagic fish species.

1993; Ortiz *et al.* 2003). Most of these species spawn in tropical waters north of NSW. The juveniles and adults are typically found in or near the EAC, feeding at oceanic fronts between the warmer and colder waters at its edge. These oceanic fronts seasonally move north and south and pelagic fish species often move with them. Ward *et al.* (2003) shows that even smaller, inshore pelagic fish species, such as tailor *Pomatomus saltatrix*, sardine *Sardinops sagax*, round herring *Etrumeus teres*, and Australian anchovy *Engraulis australis* may migrate north from the temperate waters of NSW to spawn in the ocean off southern Queensland. The early life history stages are then transported southward to NSW shelf compartment by the EAC. Figure 8 shows the conceptual model of movement of pelagic fish species between the shelf and oceanic compartments.

SUMMARY

There is extensive movement of fish into and out of the estuary and shelf compartments. Adult fish move in all directions

through all four compartments, primarily as a result of spawning migrations (Figure 9). There is no clear indication of net movement by adults in any particular direction. However, the early life history stages (eggs, larvae and small juveniles) move primarily into the estuarine and shelf compartments from the freshwater (spawning events) and ocean compartments (transport by the EAC). These early life history stages, not only maintain recruitment to populations of these fish species, but also provide a significant source of food for larger juvenile and adult fish at higher trophic levels.

The movement of some fish species between compartments may be at equilibrium at some times, but it is doubtful that the movements of all fish species between compartments are at equilibrium over inter-annual temporal scales. ENSO oscillations cause inter-annual variability within the freshwater and estuary compartments through flood/drought cycles and similar levels of variability within the shelf and ocean compartments through changes in the EAC (Figure 10). Within this larger frame of natural variability there is also uncertainty about the life history stages and biomass of many fish

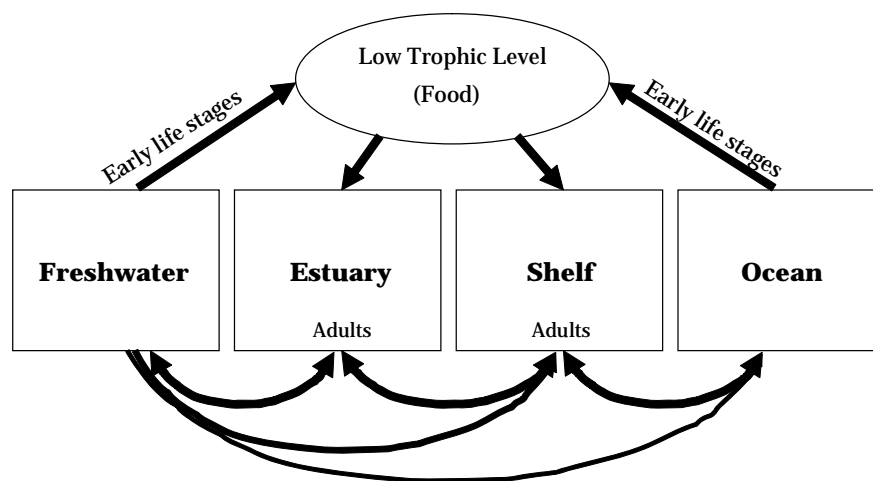


Figure 9. Summary of movement through the four compartments by all migratory and highly mobile fish species, with primary regard for the estuary and shelf study regions.

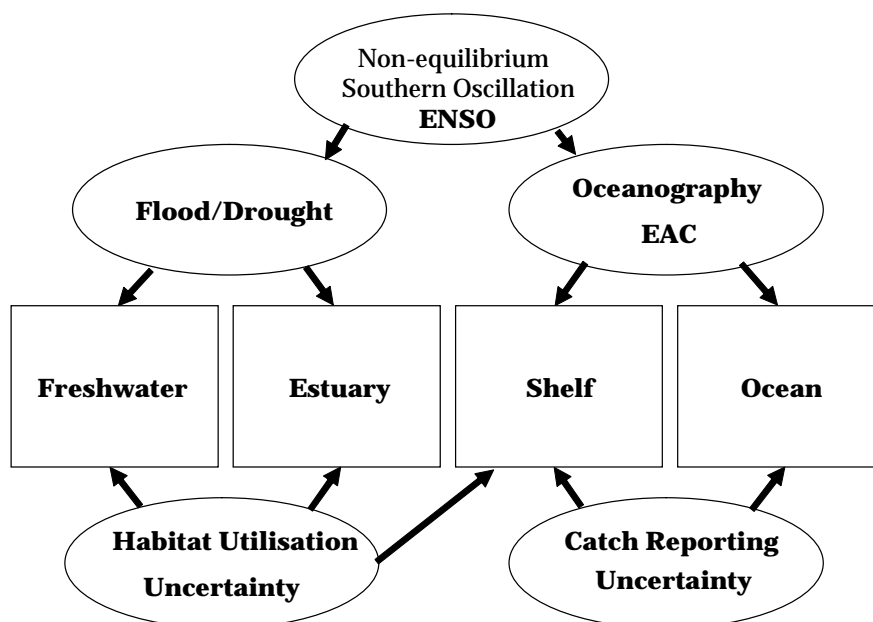


Figure 10. Factors which may lead to non-equilibrium conditions (top of figure) and uncertainty (bottom of figure) in the patterns of fish movement between the four compartments.

species in these compartments. We do not fully understand habitat requirements of many fish species in the freshwater, estuary and shelf compartments or how barriers affect movements of different life history stages between habitats and compartments. Uncertainty in the spatial resolution of catch reporting for commercial and recreational ocean fisheries is significant and may result in inaccurate biomass estimates for the shelf compartment.

Movement of fish into and out of the spatial frame of the NSW ecosystem simulation project is significant. It is hoped that basic movement patterns, such as those discussed above, can be incorporated into the trophic mass balance model of the NSW ecosystem.

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THREATENED SPECIES

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A comprehensive Conservation Overview and Action Plan for Australian Threatened and Potentially Threatened Marine and Estuarine Fishes was prepared for the Natural Heritage Division of Environment Australia by scientists from the NSW Fisheries Research Institute and the Australian Museum (Pogonoski *et al.* 2002). This document, which provides the basis for this task, reviewed the biological characteristics and conservation status of 114 species of threatened and potentially threatened Australian marine and estuarine fishes, and outlined some of the constraints encountered in carrying out the task. A specialist workshop was held on this topic in conjunction with the Australian Society for Fish Biology in September 1999, which brought together approximately 40 experts from government and non-government organisations, private industry and academic institutions in Australia, New Zealand and the USA. The main aims of the workshop were to discuss the proposed Australian conservation status of as many of the identified species as possible, and to attempt to reach consensus on a conservation status for each species. Information from the workshop discussions has been incorporated into the species synopses presented in this overview document where appropriate. Comments and advice were also sought from a wide range of individuals and organisations with expertise in fishes throughout the duration of the project.

The Overview and Action Plan analysed in considerable detail 114 species of the approximately 4,100 marine and estuarine fish species known to occur in Australian waters. Of these:

- No taxa were listed as Extinct (EX)
- 3 taxa were listed as Critically Endangered (CR)
- 6 taxa were listed as Endangered (EN)
- 8 taxa were listed as Vulnerable (VU)
- 16 taxa were listed as Lower Risk, conservation dependent (LR cd)¹
- 14 taxa were listed as Lower Risk, near threatened (LR nt)
- 53 taxa were listed as Data Deficient (DD)²
- 15 taxa were listed as Lower Risk, least concern (LR lc)

The conservation status for one commercial species (i.e. eastern gemfish, *Rexea solandri*) remained contentious and a final decision on it was postponed until the Australian Fisheries Management Authority (AFMA) finalised decisions on the future management of this species (note that the NSW Fisheries Scientific Committee has subsequently recommended that eastern gemfish be listed as Endangered in NSW). Species synopses were included for all species listed above as CR, EN, VU, LR (cd), LR (nt), DD and LR (lc) in Australian waters. An additional 19 taxa of sharks and rays which will be included by the IUCN Species Survival Commission's Shark Specialist Group (SSC SSG) in its forthcoming publication on the status of chondrichthyan fishes (Fowler *et al.* in press, but see also Cavanagh *et al.* 2003) were also considered. All of these (19 taxa) were provisionally assigned to the Lower Risk (least concern) category in Australian waters, but no species conservation synopses were included. Although a number of these sharks and rays have been flagged as having conservation problems in other parts of the world (e.g. North American or European waters), most are widely distributed species and not heavily fished or otherwise significantly threatened in Australian waters. However, in any future assessments of the conservation status of Australian fishes, a more detailed examination of these species will be warranted due to their life-history characteristics and/or their potential susceptibility to capture by fishing operations.

1. One species, the eastern gemfish *Rexea solandri*, has been provisionally listed in this category, but may be listed as Vulnerable in the future depending on future management decisions made by AFMA.

2. The western stock of gemfish *Rexea solandri* that is listed as Data Deficient is included here.

The Overview and Action Plan highlighted the main conservation concerns for some of the marine and estuarine fish species in Australia. The main causes of population declines in marine and estuarine fishes were overfishing (of both target and non-target or bycatch species), habitat degradation (from urban development and related activities, trawling, dredging, water pollution, etc.) and, to a lesser extent, exotic species introductions.

This Overview and Action Plan, through both the individual species conservation synopses presented in it and the synthesis of these findings, thus attempted to address the main problems affecting some of the more threatened and potentially threatened species of marine and estuarine fishes in Australian waters. It was the first attempt at broadly assessing the conservation status of the roughly 4100 marine and estuarine fish species that are known to occur in Australian waters, and will hopefully provide the impetus for more research into threatened Australian marine and estuarine fishes in the future. Information on those species present in NSW waters should be considered in any ecosystem analysis for this area.

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INTERACTIONS BETWEEN JELLYFISH AND FISHERIES IN NEW SOUTH WALES

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ABSTRACT

Extraordinary numbers of jellyfish are often observed in the estuaries and coastal lagoons of NSW. Medusae are voracious predators of zooplankton and are one of the major processors of nutrients in estuaries. Here I review the life history and population dynamics of jellyfishes, their influence on trophodynamics and nutrient cycling in estuaries and their possible interactions with fisheries. Jellyfish may benefit fisheries by providing a source of prey or habitat (particularly for juvenile stages) for commercially valuable species but may be detrimental by preying on larvae and competing with them for zooplanktivorous prey.

INTRODUCTION

Scyphozoan jellyfish are the most conspicuous macrofauna in the coastal lagoons and estuaries of New South Wales. Frequently they form spectacular population blooms and at such times they are likely to represent the greatest biomass of any species of animal in the estuaries. Three species of Scyphozoa are found commonly; *Aurelia aurita* (Semaestomeae), *Catostylus mosaicus* (Rhizostomeae) and *Phyllorhiza punctata* (Rhizostomeae). Although the ubiquitous *A. aurita* is probably the most well studied scyphozoan worldwide, little is known about its ecology in New South Wales and more is known about *C. mosaicus* and *P. punctata*.

LIFE HISTORY

Scyphozoan jellyfishes have complex life histories consisting of a benthic polypoid stage and a pelagic medusoid stage (e.g. (Pitt 2000)). Medusae are gonochoristic and planula larvae are brooded by the females before settling on the benthos and metamorphosing into polyps. Polyps may reproduce asexually by budding or undergoing fission and in favourable conditions ephyrae (juvenile medusae) bud from the polyp in a process known as strobilation. In NSW, polyps have only been found on settlement plates (Pitt unpubl. data) and the natural location of polyps is not yet known. Studies in aquaria, however, indicate that polyps may settle on a variety of hard surfaces (Pitt 2000). Although strobilation has never been observed in the field, the timing of strobilation can be inferred from the timing of the appearance of juvenile medusae. For *C. mosaicus* strobilation predominantly occurs between early summer and early winter although very small numbers of juvenile medusae may appear at any time of year (Pitt and Kingsford 2003). The stochastic nature of recruitment within this six-month window suggests that strobilation is not simply triggered by temperature, and it is likely that multiple factors are involved. Factors influencing the timing and magnitude of strobilation events require further research and currently represents a major limitation in our understanding about the population dynamics of jellyfishes, not just in Australia, but worldwide.

ABUNDANCES OF JELLYFISH

Populations of jellyfish are characterised by having large and rapid fluctuations in abundances. For example, at Lake Illawarra in 1998, there was a 30-fold increase in abundances of *C. mosaicus* over just four weeks (Pitt and Kingsford 2000). The appearance of "blooms" of jellyfish is partly explained by the very rapid growth rates and relatively short longevity of medusae. For example, medusae of *C. mosaicus* can grow from 40mm to 190mm diameter in just two months and have been recorded to

survive for a maximum of 13 months although most cohorts persist for shorter periods (Pitt and Kingsford 2003). In contrast, recruits of *P. punctata*, appear during late autumn or early summer, grow rapidly and seem to disappear by winter (Pitt, pers. obs.). Abundances of *A. aurita* have not been recorded but seem to be most common during summer (Pitt, pers. obs.). The timing and magnitude of population blooms is likely to be determined by variations in recruitment, survival and fecundity of polyps and survival of ephyrae and small medusae. Factors influencing survival and fecundity of the different stages of the life history have not been investigated.

SYMBIOSES

Scyphozoan jellyfish differ to most other gelatinous zooplankters in that some species contain dense concentrations of symbiotic algae (zooxanthellae) in their tissues. Like corals, zooxanthellate medusae derive some of their nutrition from the translocation of photosynthetic products from their endosymbionts but most still prey to some extent on zooplankton (McCloskey *et al.* 1994). Hence the presence of zooxanthellae is likely to influence heterotrophic feeding rates. Zooxanthellae also typically utilise the inorganic excretory products of their hosts and, therefore, may influence excretion rates (Miller and Yellowlees 1989). Unlike *A. aurita* and *C. mosaicus*, *P. punctata* contains dense concentrations of zooxanthellae in its tissues (R. Hill, unpubl. data). The presence of zooxanthellae in *P. punctata* suggests that it may have different influences than *C. mosaicus* and *A. aurita* on nutrient cycling and trophodynamics.

FISHERIES FOR JELLYFISH

Jellyfish have been harvested in Asia for over 1000 years (Omori 1981). Since the 1970s, however, the catch has increased, almost exponentially and in the late 1990s the global harvest exceeded 500 thousand tons (FAO). Increased demand for jellyfish has seen new fisheries being established in countries such as the United Kingdom and United States and in 1995 an experimental fishery for *C. mosaicus* was established in the northern half of NSW (Kingsford *et al.* 2000). Catches were small, however (<40t yr⁻¹; (Kingsford *et al.* 2000) and the fishery has been discontinued. Interest in developing fisheries for *C. mosaicus* continues in Queensland and the Northern Territory.

INFLUENCE OF JELLYFISH ON PLANKTON DYNAMICS

Jellyfish are voracious predators of zooplankton and during population blooms, are likely to be the major planktivores in NSW estuaries. A recent study at Lake Illawarra, using *in situ* mesocosms, found that concentrations of zooplankton decreased (Fig. 1) and concentrations of chlorophyll increased (Fig. 2) in the presence of *C. mosaicus* (Pitt *et al.* 2004). Such results suggest *C. mosaicus* exerts a top-down influence whereby removal of zooplankton by jellyfish, reduces the grazing pressure of zooplankton on phytoplankton, resulting in increased phytoplankton concentrations. Although rates of predation have not yet been measured for *P. punctata*, the amount of prey captured by the oral arms (feeding appendages) of similar-sized *C. mosaicus* and *P. punctata* has been measured (Peach and Pitt, unpubl. data). Both species predominantly captured copepod nauplii and gastropod and bivalve larvae but *C. mosaicus* typically caught twice as many zooplankters as *P. punctata* (Fig. 3) probably because it derives only part of its nutrition heterotrophically and because it has fewer nematocysts than *C. mosaicus* (Peach and Pitt, unpubl. data). So although *P. punctata* probably also exerts some top-down

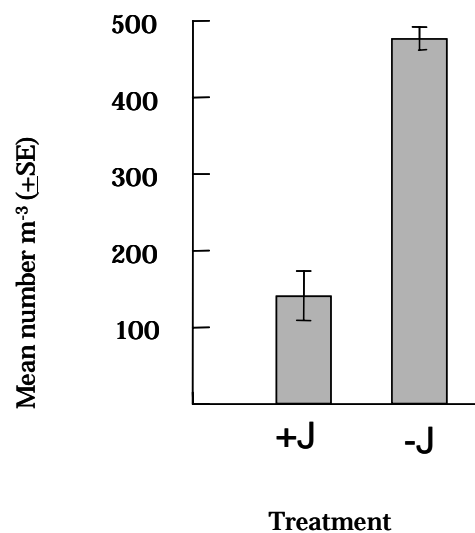


Figure 1. Mean concentration of polychaete larvae in mesocosms with jellyfish (+J) and without jellyfish (-J) (n=3). Larvae were sampled five days after the experiment commenced.

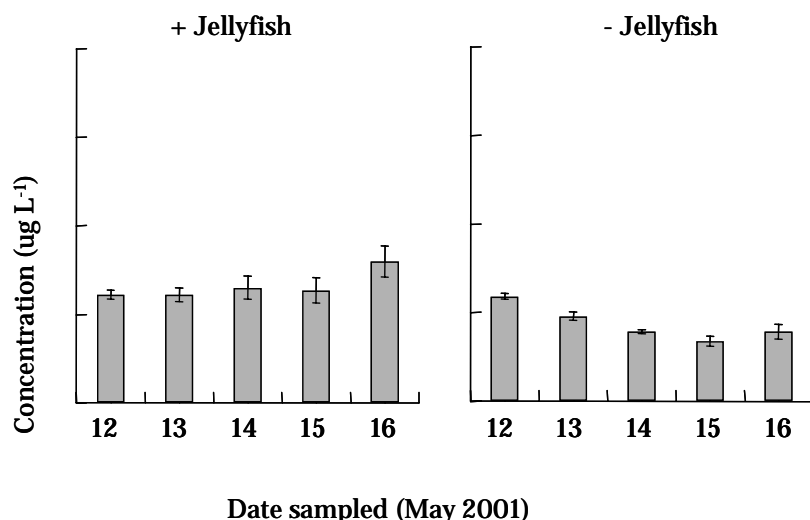


Figure 2. Temporal changes in the concentration of chlorophyll in mesocosms with and without jellyfish.

ably because its excretory products are utilised by its endosymbiotic zooxanthellae (Pitt *et al.* 2005; Fig. 4).

Recycling of inorganic nutrients to the water column by *C. mosaicus* may stimulate primary production. Indeed, during peak population blooms, excretion of ammonia by *C. mosaicus* is estimated to meet 8% of the inorganic nitrogen required to support phytoplankton production (Pitt *et al.* 2005). *C. mosaicus*, therefore, appears to regulate phytoplankton production by a combination of top-down and bottom-up effects. In contrast, since *P. punctata* has no net excretion of ammonia, it appears to have no bottom-up influence. Consequently the inorganic nutrients in zooplankters consumed by *C. mosaicus* are likely to be rapidly recycled to the water column but those consumed by *P. punctata* are likely to be bound within the medusa and only released when the medusa decomposes. The influence of *C. mosaicus* and *P. punctata* on the nutrient and trophodynamics of coastal systems appears to be very different.

INTERACTIONS BETWEEN JELLYFISH AND FISHERIES

Commercial fishers in NSW estuaries target both finfish and invertebrates such as prawns, crabs and, in some places, squid, so it is important to understand how jellyfish populations interact with both vertebrate and invertebrate species. Reviews have been published about interactions between jellyfish and fish (e.g., Arai 1988; Purcell and Arai 2001), but interactions with commercially valuable invertebrates are less well known.

JELLYFISH AS A SOURCE OF PREY FOR FISH AND INVERTEBRATES

The caloric content of jellyfish is poor relative to other types of prey (Arai 1997) and gelatinous zooplankton are rarely found in the guts of fish, so most studies of pelagic food webs have considered

influence on plankton communities, the magnitude of its influence is likely to be substantially less than that of *C. mosaicus*.

In addition to the top-down predatory effect, jellyfish may influence phytoplankton production by excretion of inorganic nutrients. Measurements of the excretion rates of *C. mosaicus* and *P. punctata* indicate that *C. mosaicus* excretes relatively large amounts of ammonia (1505mg g⁻¹ (wet weight) hr⁻¹) whereas *P. punctata* exhibits no net excretion of ammonia, prob-

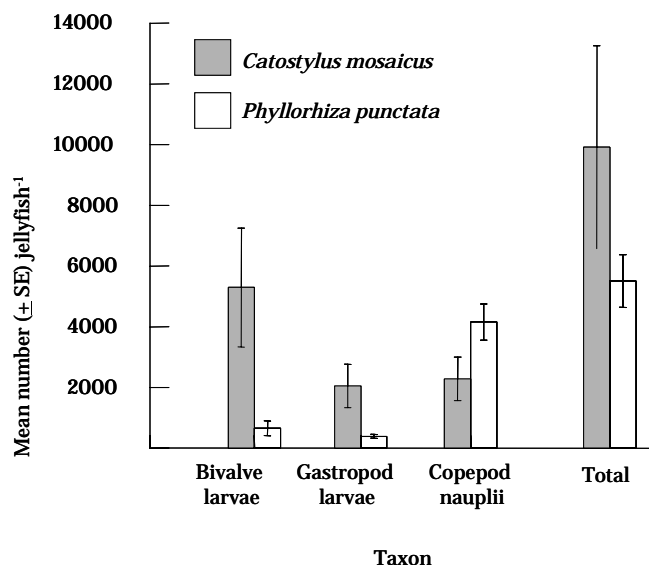


Figure 3. Mean number of the major zooplankton taxa found on the oral arms of *Catostylus mosaicus* and *Phyllorhiza punctata* at Smiths Lake, May 2003 (n=4).

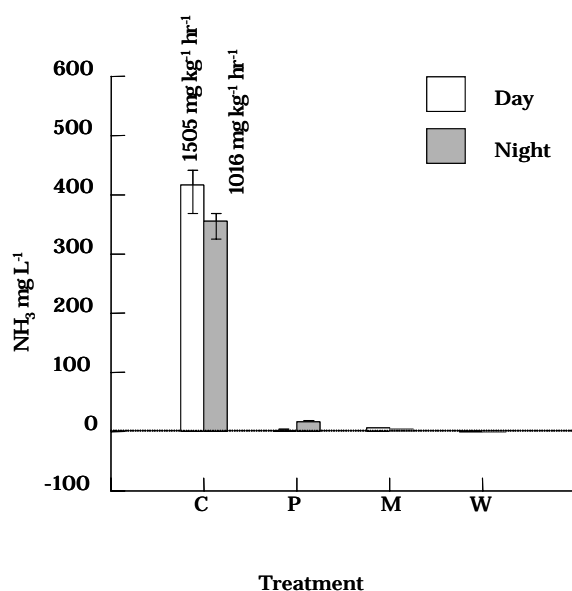


Figure 4. Variation in change in NH₃ concentrations (i.e. difference between start and finish of 6hr incubations) among treatments for experiments conducted during the day (white bars) and night (grey bars). Numbers above the bars indicate the average (± 1 SE) amount of NH₃ excreted or taken up ($\mu\text{g kg}^{-1}$ wet weight hr^{-1}). Positive values indicate excretion, negative values indicate uptake of nutrients. C = *C. mosaicus*, P = *P. punctata*, M = Mucus control, W = Water control.

jellyfish to be trophic “dead ends”. There are two reasons to suggest, however, that the importance of jellyfish as a source of prey has been underestimated. First, jellyfish swim slowly, are large and often occur in concentrated aggregations, which makes them an easily accessible source of food. So although the energy yielded by jellyfish may be less than other types of prey, the reduced energy spent in locating and capturing jellyfish may result in a net energetic gain for predators (Arai 1988). Second, gelatinous zooplankters are very difficult to identify in the guts of fish since they are rapidly digested and have few hard structures that persist in the gut and enable identification. Hence gelatinous material is probably underrepresented in gut content analyses. Recently, however, concerted efforts have been made to identify gelatinous material in the guts of fish. Mianzan *et al.* (1996) undertook microscopic examination of the gut contents of 25,000 fish from 69 species in the coastal waters of Argentina. They used the presence of nematocysts or ctenes plates (from ctenophores) as an index that the fish had preyed on gelatinous zooplankton. During spring, when gelatinous zooplankton was most abundant, 28% of all fish sampled and 35% of the species contained some gelatinous zooplankton in their gut. This study suggests that gelatinous zooplankton is utilised as

a source of food by a wide range of species and that jellyfish are a very important source of food for fish.

No studies have examined whether fish in estuaries in NSW prey extensively on jellyfish although leatherjackets (Monacanthidae) have been observed feeding on *C. mosaicus* (Pitt, pers. obs). The absence of jellyfish from estuaries at certain times, however, suggests that they are unlikely to be a critical source of food for fish, rather that fish may prey on them opportunistically.

The major invertebrate predators of medusae are other gelatinous zooplankters (Strand and Hamner 1988) and, in some places, anemones (Fautin and Fitt 1991). The invertebrates targeted in NSW estuaries (e.g. prawns, crabs and squid) have not been reported to feed on jellyfish, although crabs might prey on decomposing medusae.

INTERACTIONS BETWEEN JELLYFISH AND LARVAL FISH AND INVERTEBRATES

Jellyfish and the planktotrophic larvae of fish and invertebrates are voracious predators of plankton and, given that their distributions overlap, there is considerable scope for interactions between these groups (Arai 1988; Purcell and Arai 2001). Negative interactions for larvae may include direct predation by jellyfish and competition for planktonic prey. Yet jellyfish may also enhance survival of the larvae and pelagic juveniles of some species by providing a source of food and habitat.

PREDATION BY JELLYFISH ON LARVAL FISH

Gelatinous zooplankters are considered to be major predators of fish larvae and eggs (Purcell 1985; Purcell *et al.* 1994). Indeed in the Kiel Fjord, Germany, predation by *Aurelia aurita* on larval herring was thought to be the major factor driving the interannual fluctuations in abundances of herring

(Moller 1984). In Port Phillip Bay, Victoria, the scyphozoans *Cyanea capillata* and *Pseudorhiza haeckeli* appear to have a very small influence on the overall abundance of ichthyoplankton in the bay, although aggregations of medusae may deplete ichthyoplankton locally (Fancett and Jenkins 1988). In contrast, no larval fish and very few fish eggs have been found on the oral arms of *C. mosaicus* (Browne and Kingsford in press); Peach and Pitt, unpubl. data) or *Phyllorhiza punctata* (Peach and Pitt, unpubl. data) in NSW but *P. punctata* was sampled at only one place and time and more extensive spatial and temporal sampling is required before concluding *P. punctata* has no impact. In the Gulf of Mexico, fish eggs made up 15% of the gut contents of *P. punctata* (Graham *et al.* 2003) suggesting that this species may impact upon fish populations at other places or times. Nonetheless, the limited data gathered to date suggest that *C. mosaicus* and *P. punctata* have minimal predatory impact on larval fish and eggs in NSW. Diets of *A. aurita* have not been examined in Australia but, given that this species feeds on ichthyoplankton elsewhere (Moller 1984), it is likely that this species may impact on stocks of larval fish in Australia.

PREDATION ON INVERTEBRATE LARVAE

Brachyuran zoea are captured by the oral arms of *C. mosaicus* (Browne and Kingsford in press) but have not been found on the oral arms of *P. punctata*, although sampling of *P. punctata* has been very limited (Peach and Pitt, unpubl. data). An average of 4.6 zoea were found on *C. mosaicus* at Smiths Lake in 2003 but the species of zoea were not identified and measures of abundances and production rates of zoea are needed to determine whether predation by medusae is likely to influence adult crab populations.

COMPETITION BETWEEN JELLYFISH AND FISH

Competition between jellyfish and other zooplanktivores has never been demonstrated conclusively, but several studies have found overlap between the diets of fish and jellyfish, suggesting that, if food were limiting, competition could occur (Purcell and Sturdevant 2001; Brodeur *et al.* 2002). No studies have concurrently sampled the gut contents of larval fish and jellyfishes in Australia but studies done at different times and places indicate similarities in their diets. For example, in Smiths Lake, *C. mosaicus* and *P. punctata* predominantly captured copepod nauplii, gastropods and bivalves (Peach and Pitt, unpubl. data.) and in Tasmania (Young and Davis 1992) and New South Wales (Rissik and Suthers 1996) copepod nauplii were also found in the guts of larval fish. Given the dietary overlap between these groups, if food were limiting, competition between these groups may occur.

The release from competition with fish for zooplanktivorous prey as a result of climate-related changes in productivity (Brodeur *et al.* 2002) or overfishing (Parsons 1992) has been suggested as a possible mechanism behind the general increase in jellyfish seen globally (Mills 2001). Trophic interactions between jellyfish, fish and zooplankton, however, are likely to be complex and populations of jellyfish may be regulated by mechanisms other than competition for food (Purcell and Arai 2001). Clearly more research into competition between fish and jellyfish and the mechanisms regulating productivity of jellyfish are required.

JELLYFISH AS HABITAT

A great diversity of larval fish aggregate around drifting objects, including jellyfish (Kingsford 1993). Medusae may provide shelter from predators or be a source of food for some fish and may, therefore, enhance survivorship of some species (Hay *et al.* 1990). Larval carangids, such as *Trachurus* sp. frequently associate with *C. mosaicus* and up to 80 fish have been observed around a single medusa (Kingsford 1993). Juvenile monacanthids (up to approximately 30mm TL) also sometimes associate with *C. mosaicus* although they are observed less frequently than *Trachurus* and usually only one individual is observed with a medusa (Pitt, pers. obs). Invertebrates such as copepods and decapods also occur on medusae of *C. mosaicus* (pers. obs; (Browne and Kingsford in press) but no commercially important invertebrates have been observed.

INTERACTIONS BETWEEN JELLYFISH AND FISHING GEAR

Population blooms of jellyfish are problematic for commercial fishers. Frequently they are caught as by-catch in trawls and at some places (e.g. the Clarence River) jellyfish exclusion devices (known as "blubber chutes") are used to exclude medusae (Broadhurst and Kenelly 1996). Since the jellyfish will sting other organisms caught in the nets, capturing large numbers of jellyfish in nets may spoil the catch of valuable species and will probably decrease survival rates of other by-catch. Jellyfish are likely to have less of an impact on static gear such as gill nets, although jellyfish may still accumulate against nets.

CONCLUSION

Jellyfish occur in extraordinary abundances in the coastal waters of NSW and at times are likely to be the major processors of nutrients in these systems. Although progress is being made in understanding the role of medusae in trophodynamics and nutrient cycling, empirical data regarding interactions between jellyfish and other commercially harvested species in NSW are limited. More research is required, therefore, to determine how fluctuations in populations of jellyfish (and possible commercial harvesting of jellyfish) might influence other valuable fish and invertebrates and, conversely, how commercial harvesting of other species might influence jellyfish populations.

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GARFISH (HEMIRAMPHIDAE) IN NEW SOUTH WALES, AUSTRALIA: ASPECTS OF THEIR DISTRIBUTIONS, FISHERIES, LIFE-HISTORIES AND ECOLOGY

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Until recently very little was known about garfish in New South Wales (NSW). These gaps in our knowledge are currently being addressed as part of a study funded by NSW Fisheries and the Fisheries Research & Development Corporation (FRDC), see Stewart *et al.* 2005. There are three main species of garfish found in the waters of NSW. The Eastern Sea garfish (*Hyporhamphus australis*) and the river garfish (*H. regularis*) are distributed along the entire coast of NSW, whereas the snub-nose garfish (*Arrhamphus sclerolepis*) is restricted to the north of the state. These three main species inhabit different waters, the eastern sea garfish is totally marine, the river garfish is generally confined to brackish estuarine systems and the snub-nose garfish is mainly a freshwater fish. Two other Hemiramphids are occasionally found in commercial catches of eastern sea garfish, they are the southern sea garfish (*H. melanochir*) in the south of the state and the robust garfish (*Hemiramphus robustus*) in the northern half of the state.

These three species of garfish are fished commercially using nets that fish the upper layers. The fisheries tend to be seasonal, with the major catches of eastern sea garfish being made during the summer months (when the fish are abundant in inshore waters) and the major catches of river garfish during the winter months when the fish aggregate in shallow water to spawn. These species are also taken using hook & line by recreational fishers for bait and food. Commercial landings of eastern sea garfish have declined dramatically since 1992/93 and landings are currently less than 10% of these historical levels. An index of CPUE (catch per fisher) shows a similar dramatic decline. As a result of this apparent decline eastern sea garfish have been listed as being over-fished and are the subject of a recovery program. Landings of river and snub-nose garfish have been relatively stable during this period and there are no major concerns for their sustainability.

Results to date indicate that all three species have protracted spawning periods between August and December, although there is evidence that some eastern sea garfish may spawn at any time of the year. Reproduction is typified by multiple batch spawning, low fecundity (500-3000 eggs per batch), and large eggs (3mm diameter). The eggs are demersal and are covered in long hairs (up to 8mm) that anchor them during incubation. Eggs take two to three weeks to hatch (depending on water temperature) and the larvae are around 6mm long when hatched.

Garfish in NSW are fast growing. Preliminary estimates of ages from sectioned otoliths show maximum ages of 3 for eastern sea garfish and 7 for river and snub-nose garfish. All three species probably mature sexually and enter the commercial fishery during their second year of life. Eastern sea garfish reach a maximum size (fork length) of around 36 cm, river garfish 30 cm and snub-nose garfish 29 cm.

Garfish consume seagrass, but have a relatively short, simple straight gut that is atypical of herbivores. This suggests that the seagrass itself may not be the main source of nutrition. Sea garfish are omnivorous and have been shown to consume copepods, crustaceans, amphipods, algae and seagrass. River garfish are mainly herbivorous, with seagrass making up 65% of their diet, algae 28% and diatoms and insects around 5%. Garfish are preyed upon by many large predatory fish (including tailor *Pomatomus saltatrix*, Mulloway *Argyrosomus japonicus* and Australian salmon *Arripis trutta*) as well as water birds.

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ROLE OF SYNGNATHIDS IN SHALLOW COASTAL ECOSYSTEMS OF SOUTHEASTERN AUSTRALIA

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ABSTRACT

I examined the importance of syngnathids in southeastern Australian coastal ecosystems through review of relevant literature and unpublished data. Syngnathid species diversity and endemism are very high with approximately 46 species (32 Australian endemics) recorded from coastal waters of New South Wales, Tasmania and Victoria. In most habitats – macroalgae, reef, sand, soft sediments, and continental slope – syngnathids have been found in low numbers and at low biomass. However, in seagrass habitats, syngnathids are often numerically dominant, comprising >80% of individuals in some studies. Greater numerical dominance of syngnathids was found in *Heterozostera* and *Zostera* meadows than in *Posidonia* although this may be an artifact of the sampling gear used in different studies. Three species of pipefish, *Stigmatopora argus*, *S. nigra* and *Vanacampus phillipi*, have been recorded at densities in excess of 10 000 individuals ha⁻¹ of seagrass habitat with peak densities of 60 000-80 000 individuals ha⁻¹ recorded for *S. argus* in three studies. Abundance generally peaked in autumn and winter or winter and spring following an extended summer breeding period. Growth rates of pipefishes appeared to be greater than other species of similar size. Reproductive output was low. Syngnathids have only been recorded feeding on small crustaceans, primarily copepods, amphipods and mysids, in some cases restricted to two or three species. In turn, syngnathids have been recorded in the diets of a number of fish and bird species including Australian salmon, yellow-eye mullet and fairy penguins, although they did not appear to be dominant food items. Given their high levels of abundance, syngnathids may provide an important trophic link between benthic habitats and open-water ecosystems. Identified knowledge gaps include quantification of age and growth rates; larval and adult dispersal; drivers of abundance.

INTRODUCTION

Given that the family Syngnathidae (seahorses, pipefishes, seadragons and their relatives) is only one of many found in southeastern Australian waters, why should we consider it separately from other families of small coastal fishes? Syngnathids have a number of attributes that make them prominent and important members of fish communities:

- The family is highly speciose with over 300 species in 55 genera globally (Kuitert 2000). Syngnathids are found in almost all shallow coastal habitats from temperate zones to the tropics.
- Australia has a high diversity of syngnathids (>100 species) with a considerable number of endemic genera and species, particularly in temperate waters.
- They are charismatic, highly recognizable flagship species for marine conservation. Seadragons are state marine emblems in both South Australia and Victoria.
- All syngnathids are protected species in New South Wales, Tasmania and Victoria state waters (<3 nautical miles from the coast) and in all Australian Commonwealth waters (3 nautical miles from the coast to the edge of Australian territorial waters).
- Syngnathids have highly specialized biology including narrow diet breadth, specific habitat requirements and low reproductive output with obligate male brooding.

These attributes may render them more susceptible to habitat loss, overexploitation or incidental bycatch.

Despite the interest in syngnathids as conservation icons, remarkably little is known about most species. In this paper I have attempted to synthesize available information on the distribution, abundance (current and historical), diet, predators and importance of the family in coastal ecosystems of southeastern Australia. Emphasis has been given to data relating to trophic interactions in line with the overall objectives of working towards ecosystem-based management of fisheries in New South Wales. Finally, I have identified gaps in our knowledge and I present suggestions for priority research.

METHODS

The majority of the information in this paper has been synthesized from published work: papers in peer-reviewed journals, books, student theses and 'grey' literature reports (n=34 in total). Where appropriate I have supplemented this material with unpublished data from my own or other syngnathid researchers' works.

There are few growth rate data for syngnathids but a number of studies have provided length-frequency data allowing the calculation of growth rates from modal progression analysis (King 1995). I have calculated von Bertalanffy growth coefficients (K) for *Stigmatopora argus* in Tasmania, *S. argus* in New South Wales and *Urocampus carinirostris* and *Vanacampus phillipi* in Victoria from the data given in Jordan *et al.* (1998), Duque-Portugal (1989) and Howard and Koehn (1985) respectively, estimating maximum length (L_∞) from the ten largest individuals in each population (King 1995). I have compared these data and growth coefficients provided by Takahashi *et al.* (2003) to other species of fish using the auximetric graph function provided in FishBase (Froese & Pauly 2004).

RESULTS

Diversity, distribution and habitats

Up to 47 species of syngnathid have been recorded from the waters of southeastern Australia (Table 1) including 33 from NSW, 24 from Tasmania and 28 from Victoria. These species are representative of all syngnathid subfamilies and include seahorses (10 species), pygmy pipehorses (1), pipehorses & seadragons (7), flagtail pipefish (3) and pipefishes (26). However, some of these putative species may represent synonyms e.g. cytochrome b sequences suggest that *Hippocampus bleekeri* is a junior synonym of *Hippocampus abdominalis* (Armstrong 2001). Conversely, taxonomic studies on some poorly studied groups may reveal cryptic species within currently recognized species e.g. *Solegnathus hardwickii*. A number of species are known from only a single or few specimens such as *Hippocampus colemani*, *H. jugumus* and *H. minotaur*. There is a high degree of endemism in the syngnathid fauna with ~70% of the species recorded only from Australia. Species from Tasmania and Victoria are representative of a southern temperate fauna whereas New South Wales represents a transition zone between temperate and tropical species (Kuitert 1996; 2000).

Distribution records for many species are probably incomplete due to the small size and cryptic nature of syngnathids. A full range of distribution patterns has been found for syngnathids from extremely widespread in *Syngnathoides biaculeatus* and *Trachyrhamphus bicoarctatus* (Red Sea to South Africa, Japan to Australia and throughout the Indo-Pacific) to Australia-wide distributions for *Filicampus tigris*, *Stigmatopora argus*, *S. nigra* and *Urocampus carinirostris* to extremely restricted distributions for *Heraldia nocturna*, *Hippocampus colemani* and *Idiotropiscus* sp. (all endemic to New South Wales) (Dawson 1985; Kuitert 2000).

The majority of syngnathid species are small (<300 mm TL) and found in shallow (<30 m), inshore habitats such as macroalgal beds, rocky reefs, seagrass and soft-bottom estuarine habitats (Table 1). Only a few species are found in deeper, offshore areas including the commercially important pipehorses *Solegnathus* spp. which are taken as trawl bycatch, dried and sold for traditional Chinese medicine. Many species appear to have moderate to high degrees of habitat specialization, particularly those found in seagrass.

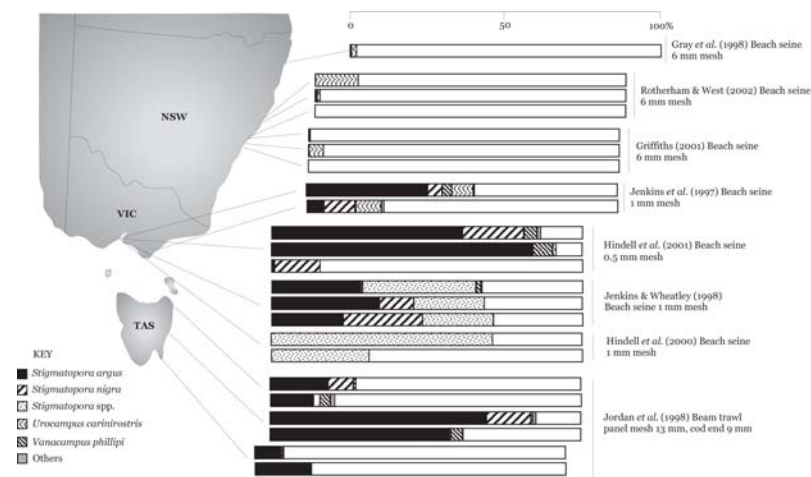


Figure 1. Numerical contribution of syngnathids to fish communities in *Zostera* and *Heterozostera* seagrass in southeastern Australia. Shaded bars represent different syngnathid taxa, white bar all other species. Authors of each study and the sampling gear used are given on the far right.

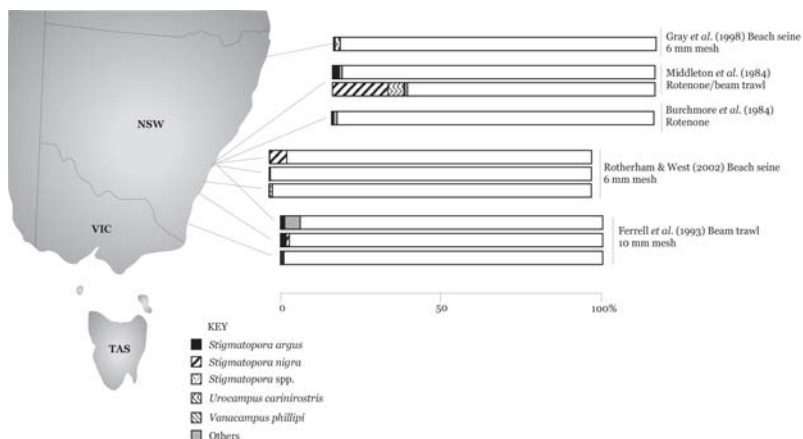


Figure 2. Numerical contribution of syngnathids to fish communities in *Posidonia* seagrass in southeastern Australia. Shaded bars represent different syngnathid taxa, white bar all other species. Authors of each study and the sampling gear used are given on the far right.

Heterozostera and *Zostera* meadows and constituted as many as 90% of individuals at some sites (Figure 1). *Stigmatopora* spp. were often the most abundant species collected although *U. carinirostris* and *V. phillipi* were also important. Due to the small size of these pipefish, their numerical dominance was generally only found when small-meshed sampling gear was used; this suggests that syngnathid abundance may be underestimated during surveys for commercial species where larger gears are used. Syngnathids appeared to be less dominant in *Posidonia* habitats (Figure 2) but direct comparisons were often not possible because of differences in sampling gear. Duque-Portugal (1989) sampled both *Posidonia* and *Zostera* contemporaneously and found lower abundance of both *S. argus*, *S. nigra* and *U. carinirostris* in the former.

Absolute abundance of individual syngnathid species in seagrass has been estimated to be in the range of 60-200 individuals ha^{-1} for rarer species such as *Phycodurus eques* (Connolly *et al.* 2002) and *V. margaritifera* (Gray *et al.* 1996) while peak abundances for *S. argus* in natural seagrass were as greater as 60 000 individuals ha^{-1} (Duque-Portugal 1989; Jenkins & Sutherland 1997) (Table 2). Considerable variation in abundance was observed both spatially (Pollard 1994; Gray *et al.* 1996) and temporally across the year (Duque-Portugal 1989; Jordan *et al.* 1998). Peak abundance for most species was observed in autumn/ winter or winter/spring (Table 3).

Abundance

Syngnathids are generally found in fairly low abundance in most habitats. For example, *H. abdominalis* populations in Tasmania were recorded at mean densities of 56-110 individuals ha^{-1} whilst *Solegnathus spinosissimus* were found at densities of only 1-2 individuals ha^{-1} (K. Martin-Smith & M. Davey unpub. data). Similar patterns have been reported for other seahorse species (Bell *et al.* 2003, Foster and Vincent in press).

However, in seagrass habitats syngnathids are often the dominant fish taxa. Pollard (1984) determined an index of community dominance based on a multiple of numerical abundance and number of species and found that syngnathids had a rank order ranging from 2-7 in *Zostera* in New South Wales and 2-6 in *Posidonia* in New South Wales out of 48 fish families. In a number of quantitative studies conducted in southeastern Australia, syngnathids made up more than 50% of individuals collected from

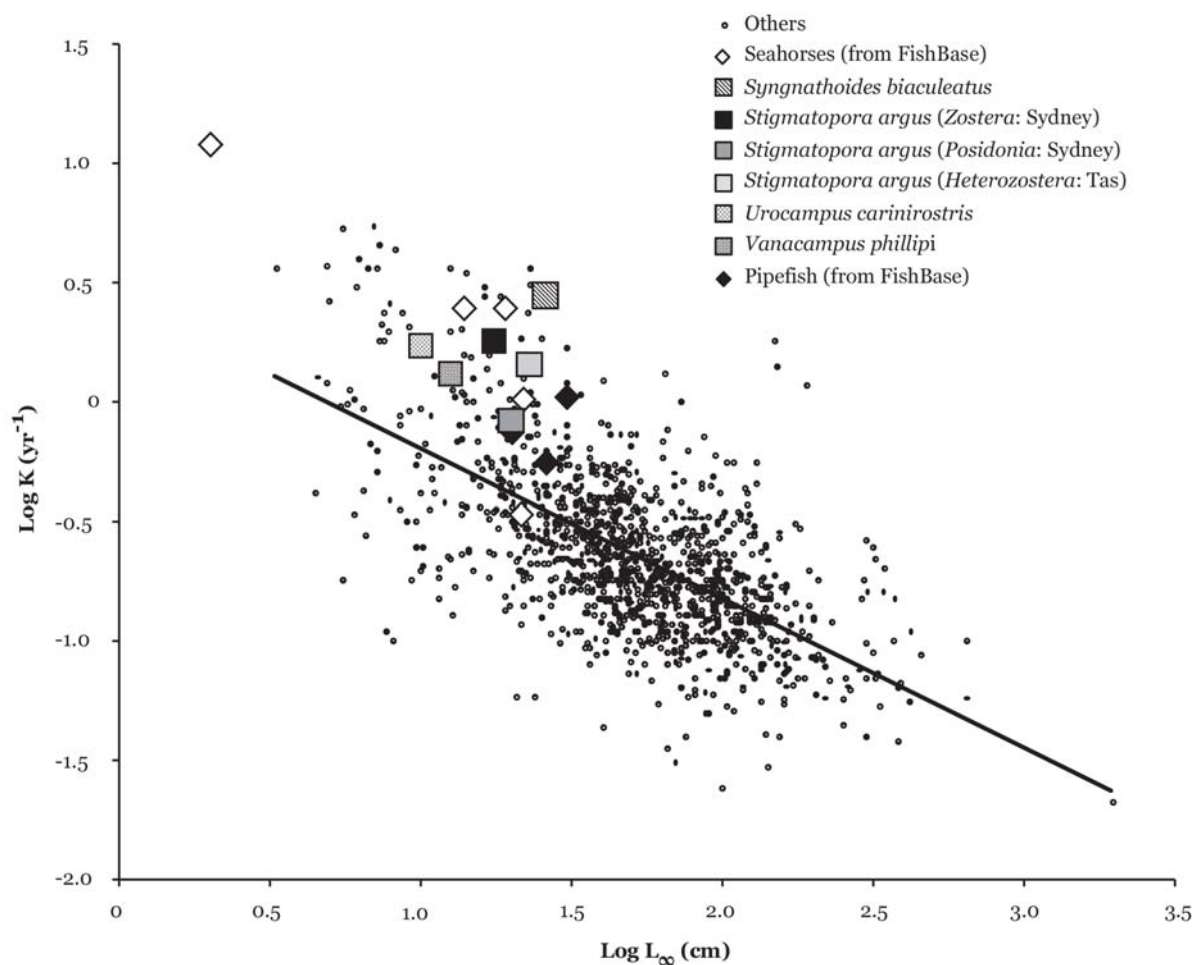


Figure 3. Auximetric plot of von Bertalanffy growth coefficient (K) against maximum size (L_{∞}) for syngnathids and other fish species. All data are from FishBase except for *Stigmatopora argus*, *Syngnathoides biaculeatus*, *Urocampus carinirostris* and *Vanacampus phillipi* (see Methods for details).

Similarly, some species showed significant differences in abundance on shorter temporal scales, *U. carinirostris* being caught in greater numbers at night for example (Gray *et al.* 1998; Guest *et al.* 2003).

Growth and longevity

The few growth and longevity data that have been obtained for syngnathids suggest rapid growth and short lifespans. The von Bertalanffy growth coefficient for all pipefish and most seahorses is considerably greater than the mean for similar-sized species (Figure 3) and, in the case of *Syngnathoides biaculeatus*, among the highest recorded for any fish (Takahashi *et al.* 2003). Howard and Koehn (1985) suggested annual life cycles for *U. carinirostris* and *V. phillipi* in Victoria as did modal progression analysis of *S. argus* and *S. nigra* in New South Wales (Duque-Portugal 1989).

Reproduction

All syngnathids have highly specialized modes of reproduction with males providing sole parental care through incubation of eggs in or on the body. Eggs are embedded in the skin under the tail of pipehorses and seadragons, while pipefishes and seahorses have specialized brooding structures ranging from ventral flaps to fully enclosed pouches. Consequently, brood sizes are low: 48 ± 3.3 (se) for *U. carinirostris*, 25 ± 2.3 (se) for *V. phillipi* (Howard and Koehn 1985), up to 100 for *H. nocturna* and *Stipicampus cristatus*, 250-300 for *P. eques* and *P. taeniolatus* (Kuiter 2000), 60-200 for *S. biaculeatus* (Takahashi *et al.* 2003) and 300-1200 for *H. abdominalis* (K. Martin-Smith, unpub. data).

Diet

Syngnathids feed almost exclusively on small crustaceans although small amounts of other items have occasionally been found in their gut contents (Table 4). There was an ontogenetic shift from copepods to peracarids with increasing size for *M. semistriatus*, *S. argus* and *V. phillipi*, (Edgar and Shaw 1995b) and seasonal shifts in percentage of different food items for *H. abdominalis* (Woods 2002). Hindell *et al.* (1999) used stable isotope analysis and found that *Stigmatopora* spp., feeding on planktonic calanoids, had similar $\delta^{15}\text{N}$ to a piscivorous fish. They concluded “our data suggest that benthic fishes, such as *Stigmatopora* sp., potentially provide a strong coupling between open-water ecosystems and benthic habitats...”

Predators

A number of species of fish and bird predators have been found with syngnathids in their gut contents (Table 5). However, their importance as food items has not been fully investigated. In an enclosure experiment, syngnathids in the stomach contents of *Arripis truttacea* comprised <5% by mass and 7% by occurrence (Hindell *et al.* 2000). Similarly syngnathids (all *S. nigra*) only comprised 6% of fish consumed by *A. truttacea* in Western Port, Victoria (Edgar and Shaw 1995b). However, syngnathids represented 33% of fish consumed by *Platycephalus laevigatus* and 100% of the fish consumed by *Aldrichetta forsteri* and *Haletta semifasciata* although these latter two results only represented three individuals (Edgar and Shaw 1995b).

DISCUSSION AND CONCLUSIONS

Importance in coastal ecosystems in southeastern Australia

Syngnathids are clearly important in the context of coastal biodiversity with a considerable number of species and a high degree of endemism in southeastern Australian waters. As most species are small and cryptic they may be under-represented in collections, particularly if the sampling gear used is designed to catch larger, commercial species. New species of syngnathid continue to be discovered, even from areas where the fish fauna is considered to be well known (e.g., Kuitert 2003).

In terms of abundance, syngnathids do not appear to be a major ecosystem component except in seagrass habitats. Available evidence suggests that most species of syngnathid are found in low numbers, perhaps as a consequence of their low reproductive output (Foster and Vincent in press). Small body size means that standing biomass will also be low and, despite fast growth rates, the contribution of syngnathids to energy flow in non-seagrass systems will be low.

In seagrass, syngnathids have been found in very high numbers (Duque-Portugal 1989; Jenkins and Sutherland 1997; Jenkins and Wheatley 1998; Jordan *et al.* 1998; see Table 2). I have not been able to find length-weight relationships for the species in question but using values derived for other syngnathids (Koutrakis and Tsikliras 2003) the standing stock of syngnathids may be in the order of 10-50 kg ha⁻¹ for the studies mentioned above. Edgar and Shaw (1995a) provided biomass estimates of 167-258 kg ha⁻¹ in seagrass habitats in Victoria, so it can be seen that syngnathids may represent a significant proportion of standing stock.

All estimates of syngnathid growth rates suggest that they grow rapidly and have short generation times (Figure 3). Thus, there may be considerable energy flux through any putative ‘syngnathid pathway’. The abundant species in seagrass, such as *S. argus* and *S. nigra*, feed primarily on planktonic crustaceans (Edgar and Shaw 1995b; Kendrick 2002) and a stable isotope study has demonstrated that they have a signature similar to some piscivorous fish (Hindell *et al.* 1999). Syngnathids may thus provide a trophic link between distinct, spatially separated habitats.

The fate of energy incorporated into syngnathid biomass is not well understood. Fish and bird predators of syngnathids have been identified but there are few data on the importance of syngnathids in their diet. Edgar and Shaw (1995b) suggest that syngnathids are primarily eaten by a guild of benthic fish.

Table 1. List of syngnathid species recorded from South East Australia (states of New South Wales, Tasmania and Victoria) with maximum recorded Total Length (TL_{max}), Total Length at maturity (TL_{maturity}) and habitat preferences.

Species	Endemic	NSW	Tas	Vic	TL _{max} ¹	TL _{maturity}	Habitat	Ref. ²
Cosmocampus howensis		+			12.5	7.2	vegetated habitats	1, 3
Festucalex cinctus	E	+			130-160	9.9	rubble, dredge in 8-31m	1, 3
Filicampus tigris	E	+			30.0	17.5	mud, sand, rubble 2-27m	1, 3
Halicampus boothae		+			160-175	9.4	rocky reefs	1, 3
Heraldia nocturna	E	+	+		80-100	6.3	rocky reefs	1, 3
Heraldia sp.	E		+		10.0		coastal bays/low reefs	3
Hippichthys heptagonus		+			15.0	7.5	lower reaches rivers/estuaries	1
Hippichthys penicillatus		+			160-180	10.0	lower reaches rivers/estuaries, mangroves, Zostera	1, 3
Hippocampus abdominalis		+	+		32.0		algae, seagrass, rocky reefs, jetties	2, 3, 4
Hippocampus bleekeri		+	+		35.0		algae, seagrass, estuaries, rocky reefs, jetties	3, 4
Hippocampus breviceps	E		+		80-100		seagrass, Sargassum, weed	2, 3, 4
Hippocampus colemani	E	+			2.5		seagrass	5
Hippocampus fisheri ?		+			8.0		single specimen from Lord Howe Island	2
Hippocampus jugumus ?	E	+			8.0		ditto – may be synonym of H. fisheri	4
Hippocampus kelloggi		+			25.0		deep water	2
Hippocampus minotaur	E	+		+	5.0		trawled from 64-100 m, fine sand or hard bottom	2, 3, 4
Hippocampus tristis	E	+			22.0		trawled 18-53 m	3, 4
Hippocampus whitei	E	+			13.0		Zostera, sponge, shark nets	2, 3, 4
Histiogamphelus briggsii	E	+	+		240-250	12.3	sand bottom - loose seaweed & Zostera	1, 3
Hypsognathus rostratus	E	+	+		305-400		sand flats	1, 3
Idiotropiscis sp.1	E	+			5.5		red algae	3
Kaupus costatus	E		+	+	14.0	86.5	seagrass, algae	1, 3
Kimblaesus bassensis	E	+	+		160-200		rubble	1, 3
Leptoichthys fistularius	E	+	+	?	65.0	20.0	Zostera	1, 3
Lissocampus caudalis	E	+	+		10.0	7.0	Sargassum, Zostera	1, 3
Lissocampus runa	E	+	+	+	10.0	6.9	Zostera, algae, rubble reefs	1, 3
Maroubra perserrata	E	+	+	+	8.0	5.6	rocky reefs	1, 3
Mitotichthys mollisoni	E	+	+	+	22.0		brown weed	1, 3
Mitotichthys semistriatus	E		+	+	250-268	14.0	Zostera	1
Mitotichthys tuckeri	E	+	+	?	20.0	12.2	kelp, Sargassum	1, 3
Notiocampus ruber	E	+	+	?	170-180		red algae	1, 3
Phycodurus eques	E		+	+	35.0		Ecklonia reef	3
Phyllopteryx taeniolatus	E	+	+	+	432-450	21.4	kelp, algal reef	1, 3
Pugnaso curtirostris	E		+	+	150-200	13.9	Posidonia and Zostera	1, 3
Solegnathus dunckeri	E	+			50.0	33.7	trawled from 75-140 m	1, 3
Solegnathus sp.1	E	+			50.0		trawled from ~100 m	3
Solegnathus hardwickii		+			50.0	29.6	trawl collections	1
Solegnathus spinosissimus		+	+	+	50.0	30.0	trawled from 30-250 m	1, 3 ¹

Table 1 continued.

Species	Endemic	NSW	Tas	Vic	TL ¹ _{max}	TL _{maturity}	Habitat	Ref. ²
Stigmatopora argus	E	+	+	+	270-280	112	Zostera, algae	1, 3
Stigmatopora nigra	+	+	+	+	175-280	6.5	bays, estuaries	1, 3
Stipecampus cristatus	E	+	+	+	240-250	190	brown & red reef algae, sparse seagrass	1, 3
Syngnathoides biaculeatus	+	+	+	+	280-300	180	seagrass	1, 3
Trachyrhamphus bicoarctatus	+	+	+	+	400	260	seagrass, sand, rubble, reef	1, 3
Urocampus carinrostris	+	+	+	+	100	48	Zostera, algae, brackish estuaries	1, 3
Vanacampus magaritifera	E	+	+	+	165-200	129	weed. Zostera, algal reef	1, 3
Vanacampus phillipi	E	+	+	+	195-200	94	weed. Zostera, estuaries	1, 3
Vanacampus poecilloaemus	E	+	+	+	280-300	170	weed. Zostera	1, 3

¹ Total length is not usually used for seahorses (Hippocampus spp.) and refers to height (tip of coronet to tip of tail) or standard length (see Lourie et al. 1999 for details). Where a range has been given this represents different figures from different authors.

² References: 1. Dawson (1985), 2. Lourie et al. (1999), 3. Kuiter (2000), 4. Kuiter (2001), 5. Kuiter (2003).

Table 2. Mean abundance of syngnathids in seagrass habitats from southeastern Australia. Where a range of densities is given, this indicates that the study was conducted at multiple sites. Peak abundance is given if the sampling was temporally replicated.

Species	Mean Density (ha ⁻¹)	Habitat	Reference
<i>Hippocampus breviceps</i>	500	Wide-leaved art. seagrass	Jenkins and Sutherland (1997)
<i>Hippocampus whitei</i>	~200	<i>Posidonia australis</i>	Burchmore <i>et al.</i> (1984)
<i>Mitotichthys semistriatus</i>	200-500	<i>Zostera muelleri</i> & <i>Heterozostera tasmanica</i>	Howard and Koehn (1985)
	4000	<i>H. tasmanica</i>	Edgar and Shaw (1995a)
<i>Phycodurus eques</i>	60	Mixed macroalgal reef & seagrass meadows	Connolly <i>et al.</i> (2002)
<i>Stigmatopora argus</i>	~100	<i>P. australis</i>	Burchmore <i>et al.</i> (1984)
	1500-15 000	<i>P. australis</i>	Duque-Portugal (1989)
	1500-60 000	<i>Z. capricorni</i>	Duque-Portugal (1989)
	1000-5000	<i>P. australis</i>	Ferrell <i>et al.</i> (1993)
	4000	<i>Z. muelleri</i> & <i>H. tasmanica</i>	Connolly (1994)
	1500-5000	Wide-leaved artificial seagrass	Jenkins and Sutherland (1997)
	52 000-84 000	Narrow-leaved artificial seagrass	Jenkins and Sutherland (1997)
	~25 000-56 000	<i>H. tasmanica</i>	Jenkins and Wheatley (1998)
	20 000	<i>H. tasmanica</i>	Jordan <i>et al.</i> (1998)
<i>Stigmatopora nigra</i>	1500-37 000	<i>Z. capricorni</i>	Duque-Portugal (1989)
	8000	<i>Z. muelleri</i> & <i>H. tasmanica</i>	Edgar and Shaw (1995a)
	100-600	Seagrass (unspecified)	Gray <i>et al.</i> (1996)
	1500-2000	Wide-leaved artificial seagrass	Jenkins and Sutherland (1997)
	4000-4500	Narrow-leaved artificial seagrass	Jenkins and Sutherland (1997)
	~400-36 000	<i>H. tasmanica</i>	Jenkins and Wheatley (1998)
<i>Urocampus carinirostris</i>	200-4400	<i>H. tasmanica</i>	Howard and Koehn (1985)
	900-10 000	<i>Z. capricorni</i>	Duque-Portugal (1989)
	2000	<i>Z. muelleri</i> & <i>H. tasmanica</i>	Edgar and Shaw (1995a)
	300-2000	Seagrass (unspecified)	Gray <i>et al.</i> (1996)
	400	<i>H. tasmanica</i>	Jenkins and Wheatley (1998)
<i>Vanacampus margaritifera</i>	~100	<i>P. australis</i>	Burchmore <i>et al.</i> (1984)
	50-200	Seagrass (unspecified)	Gray <i>et al.</i> (1996)
<i>Vanacampus phillipi</i>	100-3000	<i>H. tasmanica</i>	Howard and Koehn (1985)
	10 700	<i>Z. muelleri</i> & <i>H. tasmanica</i>	Edgar and Shaw (1995a)
	100-1000	<i>H. tasmanica</i>	Jenkins and Wheatley (1998)

Table 3. Seasonal patterns of abundance for seagrass-associated syngnathid species.

Location	Seagrass	Syngnathid	Season of Peak Abundance				Study
			Spr.	Sum.	Aut.	Win.	
Norfolk Bay, Tas	<i>Heterozostera tasmanica</i>	<i>Stigmatopora argus</i>	+			+	Jordan <i>et al.</i> (1998)
Georges Bay, Tas	<i>H. tasmanica</i>	<i>S. argus</i>			+		Jordan <i>et al.</i> (1998)
		<i>S. nigra</i>			+		
Westernport, Vic	<i>H. tasmanica</i>	<i>Urocampus carinirostris</i>			+	+	Howard and Koehn (1985)
		<i>Vanacampus phillipi</i>			+	+	
Port Phillip Bay, Vic	<i>H. tasmanica</i>	<i>S. argus</i>			+	+	Jenkins <i>et al.</i> (1998)
Botany Bay, NSW	<i>Zostera capricorni</i>	<i>S. argus</i>	+			+	Duque-Portugal (1989)
		<i>S. nigra</i>	+				
		<i>U. carinirostris</i>	+	+	+		
Pittwater, NSW	<i>Posidonia australis</i>	<i>S. argus</i>				+	
	<i>Zostera capricorni</i>	<i>S. argus</i>			+	+	Duque-Portugal (1989)
		<i>U. carinirostris</i>			+		

Table 4. Major diet items of southeastern Australian syngnathids. All percentages given are for volumetric contribution to gut contents except for Edgar & Shaw (1995b) which is given by weight.

Species	Major Diet Items	Reference
<i>Filicampus tigris</i>	Gammarids (68%), caprellids (14%)	Kendrick (2002)
<i>Hippocampus abdominalis</i>	Carid shrimps (28%), amphipods (23%), mysids (8%)	Woods (2002)
<i>Hippocampus breviceps</i>	Gammarids (61%), caprellids (14%/)	Kendrick (2002)
<i>Hippocampus whitei</i>	Amphipods (59%), mysids (31%)	Burchmore <i>et al.</i> (1984)
<i>Lissocampus caudalis</i>	Harpacticoids (47%), amphipods (35%), calanoids (13%)	Kendrick (2002)
<i>Mitotichthys semistriatus</i>	Calanoids (28-80%), gammarids (20-28%), other copepods (0-39%)	Howard and Koehn (1985)
	Peracarids (80-90%), copepods (10-20%)	Edgar and Shaw (1995b)
<i>Phyllopteryx taeniolatus</i>	Mysids (80%), carid shrimps (8%), <i>Lucifer</i> (6%)	Kendrick (2002)
<i>Pugnaso curtirostris</i>	Gammarids (42%), harpacticoids (25%), other crustaceans (24%)	Kendrick (2002)
<i>Stigmatopora argus</i>	Algae (50%), misc. crustaceans (30%), copepods (20%)	Burchmore <i>et al.</i> (1984)
	Copepods (94%)	Steffe <i>et al.</i> (1989)
	Copepods (60-100%), peracarids (0-40%)	Edgar and Shaw (1995b)
	Calanoids (90%), harpacticoids (10%)	Jenkins and Sutherland (1997)
	Calanoids (68%), cyclopoids (22%)	Kendrick (2002)
	Copepods (62%), decapod larvae (22%), amphipods (13%)	Sakabe (2003)
<i>Stigmatopora nigra</i>	Copepods (97%)	Steffe <i>et al.</i> (1989)
	Copepods (90-100%)	Edgar and Shaw (1995b)
	Calanoids (70%), cyclopoids (21%)	Kendrick (2002)
<i>Urocampus carinirostris</i>	Cyclopoids (45-60%), calanoids (19-24%), harpacticoids (15-19%)	Howard and Koehn (1985)
	Copepods (30-100%), peracarids (0-70%)	Edgar and Shaw (1995b)
<i>Vanacampus poecilolaemus</i>	Mysids (68%), carid shrimps (22%)	Kendrick (2002)
<i>Vanacampus phillipi</i>	Calanoids (23-32%), gammarids (28%), copepods (7-14%)	Howard and Koehn (1985)
	Peracarids (20-90%), copepods (10-80%)	Edgar and Shaw (1995b)
	Gammarids (45%), harpacticoids (13%/), mysids (7%)	Kendrick (2002)

Table 5. Known predators of syngnathids in southeastern Australia.

Species	Common name	Species of syngnathid consumed	Reference
FISH			
<i>Aldrichetta forsteri</i>	Yellow-eye mullet	<i>Stigmatopora nigra</i>	Edgar and Shaw (1995b)
<i>Antennarius striatus</i>	Striped anglerfish	<i>Hippocampus abdominalis</i> , <i>H. breviceps</i>	Kuiter (2000)
			Kuiter (2000)
<i>Arripis truttacea</i>	Australian salmon	<i>S. nigra</i>	Edgar & Shaw (1995b)
		Syngnathids (unidentified)	Hindell <i>et al.</i> (2000)
<i>Cristiceps australis</i>	Crested weedfish	Syngnathids (unidentified)	Hindell <i>et al.</i> (2000)
<i>Haletta semifasciata</i>	Blue weed-whiting	<i>Urocampus carinirostris</i> , <i>Vanacampus phillipi</i>	Edgar and Shaw (1995b)
			Edgar and Shaw (1995b)
<i>Platycephalus</i> spp.	Flatheads	<i>H. breviceps</i>	Kuiter (2000)
<i>P. laevigatus</i>	Rock flathead	<i>Mitotichthys semistriatus</i> , <i>U. carinirostris</i> , <i>V. phillipi</i>	Edgar and Shaw (1995b)
			Edgar and Shaw (1995b)
BIRDS			
<i>Eudyptula minor</i>	Fairy penguin	<i>H. abdominalis</i> , <i>H. breviceps</i>	Kuiter (2000), K. Martin-Smith (unpub. data)
<i>Phalacrocorax</i> spp.	Cormorants	<i>H. abdominalis</i>	K. Martin-Smith (unpub. data)

Gaps in our knowledge

It appears clear from the discussion above that we do not have a good understanding of the ecological role that syngnathids play in coastal ecosystems. Many of the data presented in this paper are indirect estimates from studies focusing on commercial species. For other population parameters such as dispersal I could find no data at all. Syngnathids may be a very important but neglected component of seagrass ecosystems both as predators and as prey. As a priority for the future, I suggest that the following areas of syngnathid research should be considered:

- Improved and validated estimates of population density over large spatial scales.
- Direct measurements of growth and age.
- Estimates of food uptake and energy conversion.
- Dispersal of juveniles and adults over small and medium spatial-scale.
- Investigation of factors driving the population dynamics of syngnathids – is it top-down control from predators or bottom-up limitation by prey availability?

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BLUE GROPER

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Blue groper (*Achoerodus viridis*; Family Labridae) are the largest reef fish in temperate Australia. They are common members of rocky reef assemblages in New South Wales with their distribution extending from Caloundra (Queensland) to Wilsons Promontory (Victoria) (Kuitert 1993). A brief summary of the distribution and abundance of the species in seagrass and reef habitat, sustainability of reef populations, the reproductive biology, feeding ecology and age and growth of the species is provided below.

DISTRIBUTION AND ABUNDANCE

Small juvenile blue groper are most commonly found in seagrasses or weed habitats in estuaries or sheltered reefs, but they occur in any sheltered habitat with some form of physical structure (Gillanders and Kingsford 1993). Peak recruitment to seagrass habitats occurs in September and October (spring), but small fish (<10 mm) can be found as early as June (winter) (Gillanders 1997b). Juveniles in seagrasses grow to a length of approximately 50 mm over three to four months before they are thought to move to deeper beds of seagrass or directly to coastal reefs (Bell and Worthington 1993; Gillanders and Kingsford 1993, 1996). Patterns of abundance of blue groper on reefs in New South Wales have been documented around Sydney, as well as along the coast of New South Wales (Gillanders and Kingsford 1993; Gillanders 1997b). Juvenile blue groper occur in their greatest abundances in shallow areas on reefs inside estuaries. Adult blue groper, especially large fish, are more common on the exposed coastal reefs than reefs inside estuaries. When growth of fish was investigated, there was no suggestion that fish from reefs inside estuaries grew more slowly than those from reefs outside estuaries (Gillanders 1997a). Thus, the patterns of abundance are likely to reflect movement from inshore reefs to exposed coastal reefs with an increase in size.

REPRODUCTIVE BIOLOGY

Population structure and gonadal structure of groper from two sites near Sydney showed that blue groper was protogynous and monandric (i.e. no primary males were found) (Gillanders 1995b). Males dominated the larger size and age classes, whereas females predominated in the smaller size and age classes. Females first mature at a size of 250 mm (0.3 kg), although the majority do not mature until 295-350 mm (0.6-0.9 kg). The sex ratio of mature fish and the whole population was biased in favour of females. Individuals at both sites matured at 1-2 years of age. Fish functioned as females for between 8 and 18 years before changing sex at sizes between 480 and 580 mm standard length. Spawning occurred between July and October.

FEEDING ECOLOGY

Achoerodus viridis is a benthic carnivore and consumes a wide variety of prey organisms (see Table 2 in Gillanders 1995a). Size-specific differences in diet exist. The diet of recruits (17-26 mm SL) collected in seagrass environments was dominated by tanaids, whereas that of rocky reef recruits was dominated by harpacticoid copepods. The diet of rocky reef fish then shifted to gammarid amphipods and other crustaceans (fish 50-150 mm SL) and to mussels and urchins (fish >300 mm SL). Juvenile fish on rocky reefs (<150 mm SL) foraged in shallow fringing habitat, whereas adult fish (>200 mm SL) foraged in deeper turf and barrens habitats, reflecting the depth distribution of the species. Feeding rate (bites on substratum per unit time) of larger fish tended to be less than that of smaller

fish. Variations in diet and feeding rate were also detected over the year but the patterns were not consistent between years. Among sites, there were differences in diet but these were not related to the position of sites inside an estuary or on the open coast.

AGE AND GROWTH

The maximum age of blue groper was 35 years. Analyses of size and age data shows that an average fish in its second year of life will be 230 mm long (0.26 kg). By the time blue groper are 10 years of age they will be 480 mm long (2.4 kg), at 20 years of age they will be 620 mm long (5.3 kg) and at 30 years they will be 725 mm long (8.4 kg).

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FISHERY-INDUCED CHANGES IN ABUNDANCE AND SIZE STRUCTURE OF DEMERSAL TRAWL FISH SPECIES OFF NEW SOUTH WALES

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INTRODUCTION

This paper summarises changes in the abundance and size composition of some of the important fish species caught by demersal trawlers off New South Wales (NSW), Australia. The information presented has been amalgamated from a number of monitoring and research programs undertaken by NSW Fisheries and other agencies, mostly since the development of the modern demersal trawl fishery in the 1970's (see Tilzey and Rowling 2001; Knuckey *et al.* 2001 and Smith and Wayte 2002 for more detailed descriptions of these programs and assessments of the fishery).

BRIEF HISTORY OF THE FISHERY AND SCOPE OF THIS PAPER

Demersal trawling in ocean waters off NSW commenced just after the First World War, when the NSW Government commissioned three steam trawlers to develop a commercial fishery on continental shelf grounds discovered by the research vessel *Endeavour*. The fishery developed rapidly following the sale of the vessels to commercial interests in 1923, with the principal target species being tiger flathead (*Neoplatycephalus richardsoni*). The steam trawl fleet expanded to a maximum of seventeen vessels in 1929, when more than 6,500 t of fish were landed.

During the late 1930's, smaller diesel powered vessels, which used Danish seine nets, entered the fishery, and by the mid 1940's the tiger flathead stock was showing signs of being overfished. Secondary species, such as redfish (*Centroberyx affinis*) began to be landed in increasing quantities. Minimum cod end mesh size regulations were introduced in the 1950's, however the tiger flathead stock was slow to recover and in the 1960's the last of the steam trawlers ceased operating.

Modern diesel powered trawlers entered the fishery in the 1970's and fishing was extended to deeper waters on the upper continental slope, where the main species targeted were gemfish (*Rexea solandri*), redfish, ling (*Genypterus blacodes*), and 'offshore' ocean perch (*Helicolenus barathri*) – the 'inshore' ocean perch (*H. percoides*) had been caught incidentally on continental shelf grounds since the commencement of the fishery. Target fishing for silver trevally (*Pseudocaranx dentex*) on continental shelf grounds also commenced about this time, following the development of markets for this species. By 1980 about 130 fish trawlers were operating from NSW ports, with just six boats using the Danish seine method (Graham *et al.* 1982). Annual landings by the fishery in the mid 1980's were around 10-12,000 t.

Since the early 1990's there has been a significant decline in both the number of fish trawlers operating and annual landings of the fishery off NSW. The following trends have been observed for each of the important species:

Tiger flathead

Tiger flathead are relatively fast growing demersal fish, which mature at 3 to 5 years of age (25 to 35 cm LCF) and reach a maximum size of about 60 cm at about 15 years of age. Annual yields have been in the range 2,500 t to 3,500 t since the mid 1980's, generally in excess of previous estimates of maximum sustainable yield of around 2,500 t. The modal size of landed tiger flathead has declined from 35 cm in the 1980's to about 30 cm recently – due in some part to the realisation by trawl fishers that the NSW minimum legal length of 33 cm does not apply to fish taken under Commonwealth jurisdiction. A population model is currently being developed by CSIRO researchers, but has

yet to be finalised. Some concern has been expressed that reduced abundance of larger tiger flathead off NSW might be due to high fishing pressure over a protracted period (Smith and Wayte 2002).

Redfish

Redfish are relatively small, slow growing demersal and mid-water fish, which mature at about 4 to 6 years of age (18 to 20 cm LCF) and reach a maximum size of about 30 cm at about 30 years of age. Annual landings have declined from around 1,500 t in the early 1990's to about 800 t recently. The modal size of landed redfish has declined from 23 cm LCF in the 1970's to about 17 cm in 2002. Discarding of small fish has been very significant throughout the history of the fishery, with an estimated 895 t being discarded during 2002. These discarded fish had a modal length of about 13 cm LCF. The redfish stock was classified as 'growth overfished' in the most recent series of stock assessments for the fishery (Caton 2003).

Silver trevally

Silver trevally is a relatively slow growing demersal species. Trevally mature at about 4 to 8 years of age (20 to 30 cm LCF) and reach a maximum size of about 60 cm LCF and a maximum age of about 25 years. Annual landings have consistently declined from around 1,500 t in the mid 1980's to around 400 t recently, but unlike redfish very few small silver trevally are discarded. The decline in catch has been accompanied by a very significant decline in modal length of the catch, down from 35 cm LCF in the 1980's to 20 - 25 cm in recent years. The silver trevally stock was also classified as 'growth overfished' by Caton 2003.

Ocean perch

Demersal trawlers off NSW catch two species of ocean perch – 'inshore' ocean perch occur mainly on the continental shelf, and while they reach a maximum size of about 38 cm, the majority of fish caught are between 15 and 20 cm in length and they are mostly (75%) discarded. The 'offshore' ocean perch occurs on the upper continental slope, reaches a maximum length of 45 cm, and the majority of fish caught are 25 to 35 cm in length (so discarding, while it still occurs (15%), is a less significant issue for the offshore species). Both species are thought to be long-lived, to 30 or 40 years of age, are widely distributed in their preferred depth ranges, and the inshore form is known to be internally fertilized, with the initial larval stages occurring within the body of the female.

Annual landings of the two species of ocean perch combined have been relatively stable at around 300 - 350 t since the mid 1980's. The offshore species comprises about 90% of landings. Size composition data for landings of the offshore form showed a decline in modal length from 35 cm in the 1970's to a broad mode 25 to 30 cm by the late 1980's, with little further change apparent during the 1990's. It is thought the lack of change in size composition over this period may be due to a significant proportion of offshore ocean perch population being found on untrawlable bottom for a large proportion of the time, therefore providing a buffer to the effects of increased fishing. Historical size composition data are not available for inshore ocean perch, however as the species has been fished since the commencement of the steam trawl fishery it is likely that any significant changes in population size composition may have occurred well before the period of relevance to this paper.

Ling

Ling is a large, relatively slow growing species, which matures at 60 to 70 cm in length (about 4 to 6 years of age), and reaches a maximum length of about 120 cm and maximum age of 30 years. Annual landings of ling in the NSW area increased from about 200 t in 1980 to around 800 t in the late 1990's, but landings have since declined. Size composition data for the NSW demersal trawl fishery show a very distinct trend. In the early years of the fishery the modal length of ling was around 65 - 70 cm and a considerable proportion of the catch was greater than 90 cm in length. In recent years, the modal length has been around 50 - 55 cm, and few fish greater than 90 cm appear in trawl catches. Although the status of the ling stock was recently assessed as "uncertain, but probably fully fished east of Bass Strait" (Caton 2003), there are indications that the stock is growth-overfished off NSW.

Gemfish

Gemfish are relatively fast growing semi-pelagic or mid-water fish, which occur mainly at depths of 200 to 600 m on the continental slope. Gemfish mature at 4 to 6 years of age (60 to 75 cm LCF), and reach a maximum length of 120 cm and a maximum age of about 15 years. Annual landings of gemfish in NSW declined from 3,000 - 4,000 t per annum in the 1980s to around 100 t in recent years.

The eastern stock of gemfish (which occurs off NSW) suffered a recruitment collapse in the late 1980's and has failed to show any significant recovery (the last cohort of 'near-normal' abundance was spawned in 1991). Fishers believe there is evidence that the stock is recovering, however the size composition of the spawning population in 2003 showed the majority of fish to be between 35 and 75 cm LCF (mostly 2 to 6 year old fish), whereas a 'recovered' population would be expected to have the majority of mature fish between 65 and 95 cm LCF (approx. 5 to 10 year old fish).

BIOMASS TRENDS

Of the species listed above, population models for estimating biomass have been developed for gemfish (Punt 1998) and redfish (Thomson 2001), and the abundance of these species has been estimated to have declined by 85 – 95% since the 1970's. Size composition data suggest that very significant declines in abundance have occurred for silver trevally and ling, while 'offshore' ocean perch and tiger flathead also appear to have declined in abundance since the modern trawl fishery commenced in the 1970's. The available evidence suggests that the total biomass of the important demersal species listed above may have declined by around 75% over the past three decades.

OBSERVATIONS ON TROPHIC RELATIONSHIPS

Very little is known of the trophic relationships for fish communities in the ocean waters off NSW, with the exception of the research reported by Bax and Williams (2000), which studied the diets of 70 species (including all the important species listed above) from a series of research cruises off southern NSW between 1993 and 1996. The potential complexity of the trophic relationships for the main commercial species is apparent from the results presented by Bax and Williams (2000). Most of the species listed above ate benthic or benthopelagic fish, while pelagic invertebrates were also important in the diet of redfish, and the ocean perch species were found to be more omnivorous, eating a range of fish, pyrosomes, crabs, cephalopods and crustaceans. Ontogenetic shifts in diet were potentially important for ling, gemfish, silver trevally and the ocean perches, although the sample size for adult ling (greater than 70 cm) was very small (N=6). The study did not identify any forage fish species as being particularly important in the diets of the species listed above, and curiously, the study failed to identify any "top predators" amongst the 70 species examined, prompting a remark that "selective harvesting of species such as tiger flathead may already have removed the top predators" (Bax and Williams 2000, p. 366).

CONCLUSION

I suggest that one question that should be addressed as part of this workshop is:
"Can Ecopath with Ecosim provide information directly usable for policy analysis in the NSW demersal trawl fisheries?"

Given the apparent significant decline in the abundance and size composition of the predatory species listed above, it could be concluded that the demersal trawl fishery has had a very significant impact on the ecology of continental shelf and slope waters off NSW. However, as there are only very limited data on the trophic relationships of these species, and there are no qualitative or quantitative data on trends in the populations of the prey species in the area of the fishery, I conclude that the answer to the question posed above is probably "highly unlikely".

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DEMERSAL DIVERSITY, DIETS AND DEMISE OF DEEPWATER DOGSHARKS OFF NEW SOUTH WALES

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INTRODUCTION

NSW Fisheries conducted several offshore exploratory and stock assessment surveys with FRV *Kapala* between 1970 and 1997. Core data collected were the distributions, relative abundances and size structures (length distributions) of commercial (or potentially commercial) species from shallow water prawn grounds to midslope depths greater than 1000 m. Also recorded were associated data on the diversity, abundance and distribution of bycatch species, and biological information for about 20 species of demersal sharks. Operational and raw catch data were routinely recorded in *Kapala* Cruise Reports (Nos 1-117; NSW Fisheries publications). Some survey results and associated data have been summarised and analysed in other publications referenced here.

A stratified trawling survey and results of other systematic sampling for deepwater prawns are summarised in Gorman and Graham (1975) and Graham and Gorman (1985). Initial surveys to assess fish stocks on the upper slope (220-650 m) were completed in 1976-77 and 1979-81. A comparable survey was then conducted in 1996-97 with results summarised in Graham *et al.* (1997), and analysed in Andrew *et al.* (1997). The relative abundances of several species, particularly for sharks and rays, changed significantly over time (see below). The NSW midslope (600-1200 m depth) between Crowdy Head and Gabo Island was explored and surveyed by trawl during the 1980s. The relative abundances of the main midslope species were assessed by stratified trawling in 1989 (Graham 1990). Continental shelf trawling grounds (< 200 m depth) were surveyed for prawns in 1990-92 and 1995-96, and for fish in 1993-94; operational and catch data are in *Kapala* Cruise Reports 110-116.

The diversity of the fish, crustacean and molluscan fauna recorded during *Kapala* surveys is summarised. More specific information on deepwater squaloid sharks, including dietary information and observed changes in abundance of some species, is then presented.

FAUNAL DIVERSITY

During *Kapala* surveys and other trawling projects since 1976, species lists were compiled for over 4000 demersal trawls distributed between southern Queensland and northeastern Victoria across a depth range of 10-1260 m. However, the depth and geographical distribution of stations was not evenly distributed along the NSW coast (Table 1), being influenced by the nature of the seabed and the objectives of the various surveys. Comparatively few trawls were done on the outer shelf or slope off northern NSW because most of that area was found to be untrawlable and, where trawling was possible, potential fishery resources appeared limited. Similarly, only restricted areas of trawl bottom were found on the midslope, mostly between Crowdy Head and Batemans Bay.

Species lists were initially of fish, but latterly included molluscs and crustaceans. All these data (presence and index of abundance) are contained in a single database on Cronulla Fisheries Research Centre data servers, and for the surveys after 1990, species lists (with frequency of occurrence) were documented in *Kapala* Cruise Reports 110-117. These data are not synthesised into a single document, but the information has contributed to bioregionalisation studies by NSW Fisheries and CSIRO Marine Research, and to some more specific studies (e.g., Iwamoto and Graham 2001).

Approximately 1000 species of fishes, 130 molluscs, and 160 crustaceans were recorded from *Kapala* trawls. Of the 800 species of demersal fish, 115 were chondrichthyans and 685 were teleosts. By depth, 435 species were caught only on the shelf (< 200 m), 98 species in outer shelf-upper slope depths (100-400 m), 133 species on the upper slope (200-700 m), and 143 species on the midslope (700-1250 m).

Invertebrate data are less comprehensive. Non-commercial species of molluscs and crustaceans were progressively included in the catch records from 1990. Of the molluscs identified, there were about 50 species each of cephalopods and gastropods, and 25 species of bivalves. Decapod and stomatopod crustaceans were recorded and included over 80 species of crabs, 40 species of prawns and 15 species of lobsters and bugs.

Large collections of fishes and invertebrates were deposited in the Australian Museum, Sydney. Smaller collections of fishes, particularly sharks and rays, were sent to CSIRO Marine Research, Hobart, to assist with taxonomic studies.

DOGSHARKS

Squaliform sharks (dogfishes) are an important component of the slope fauna with over 20 species found off NSW. Although some species also inhabit outer shelf depths (100-200 m), all are found in depths greater than 200 m, and were a major part of the fish biomass on the upper and mid slopes. During the initial 1976-77 upper slope survey, dogsharks formed about 35% of the overall fish biomass; these were mainly "Endeavour" sharks (*Centrophorus* spp.) contributing about 20%, and spiky dogsharks (*Squalus* spp.) about 12%. On the midslope trawling grounds, sampled in 1987-89, about 70% of the total catch was dogsharks, principally *Deania calcea* (55%).

Dogshark diets

Across genera, dogsharks are similar in appearance and size with the adult sizes of most species between 60 and 120 cm total length. However, analyses of their diets revealed differences in feeding behaviour and prey, even within genera.

In summary, the diets of three species of *Squalus* comprised small teleosts, cephalopods and crustaceans in varying proportions (Table 2). However, the fish components of the diets were markedly different with *S. megalops* and *S. cf. mitsukurii* eating mainly demersal species, whereas the stomachs of *Squalus* sp. F contained mostly small mesopelagic fishes. Similar differences were found in the stomach contents of the three species of *Centrophorus* examined. Again, the diets of all three included fish, cephalopods and crustaceans (Table 2) but *C. harrissoni* and *C. cf. moluccensis* preferred mesopelagic species while *C. cf. uyato* ate mainly demersal species.

The main dietary items in midslope dogsharks were lanternfishes and cephalopods (Table 2). The diets of *Deania* spp. were almost exclusively lanternfish, and for *Etmopterus* sp. B, mainly squids. Preferred prey items across three species of *Centroscymnus* varied. *Centroscymnus owstoni* ate mainly squids and the smaller *C. crepidater* preyed upon mesopelagic lanternfish, cephalopods and crustaceans. In contrast, stomachs of *C. coelolepis* contained large chunks of fish, squid, and cetacean blubber, suggesting carcass scavenging or near surface feeding.

Changes in relative abundance

The 1976-77, 1979-81 and 1996-97 stratified random surveys of the NSW upper slope revealed marked changes in the relative abundances of several species of teleosts and chondrichthyans after 20 years of commercial exploitation (Andrew *et al.* 1997). Across all survey areas and depths, the 1996-97 mean catch rate of total fish was about 70% lower than in 1976-77; the mean catch rate for sharks and rays (combined) was about 80% lower.

Results for sharks and rays are analysed and summarised in Graham *et al.* (2001). The 1996-97 mean catch rate of sharks and rays, pooled for the main 15 species (or species groups), was about 20% of the 1976-77 mean. Individual catch rates were substantially lower in 1996-97 surveys for 13 of the

able 1. Geographic and depth distribution of Kapala and associated demersal trawl stations for which species lists were compiled. Data are numbers of trawls at latitude x depth interval.

Depth (m)	10 -49	50 -99	100 -149	150 -199	200 -299	300 -399	400 -499	500 -599	600 -699	700 -799	800 -899	900 -999	1000 -1099	1100 -1199	Total
Lat °S															
27			3	1		1									5
28	129	72	17	6		3	6	2		1					241
29	286	222	10	6	5	9	14	3							558
30	15		9	2	8	1									35
31	9	105	5	1	3	1		1			1	1	2		129
32	404	245	101	1	9	10	13	9	1	4	10	15	31		853
33	115	44	152	35	45	68	96	33	14	16	23	32	25	6	704
34	9	13	191	13	10	14	43	10	7	7	9	20	13	4	363
35	65	8	102	18	54	65	55	34	11	7	12	25	15	3	474
36	64	53	97	6	21	49	35	11							336
37	2	14	72	5	42	69	45	22	6	1	3	3	2		286
38			1	4	10	10	6	3	1		1	1			37
Total	1098	776	760	98	215	300	313	128	40	36	59	97	88	13	4021

15 species or species groups. The greatest decline was observed for 'Endeavour' dogsharks (*Centrophorus* spp.), which contributed about 30% of the 1976-77 total catch but were rarely caught 20 years later. The overall mean catch rate of *C. harrissoni* and *C. cf uyato* (combined) was 126 kg/h in 1976-77 but only 0.4 kg/h in 1996-97. Similarly, the 1996-97 mean catch rates of *C. cf. moluccensis*, *Squalus* sp. F and *S. cf. mitsukurii* (combined), and the silver ghost shark (*Hydrolagus ogilbyi*) were all less than 5% of the initial surveys. In contrast, 1996-97 catch rates of spiky dogshark (*Squalus megalops*) and whitfin swell shark (*Cephaloscyllium* sp. A) were similar to those in 1976-77.

Trawling during 1979-81 on the same grounds provided intermediate and comparative data for 9 species. Although the 1979-81 data were not corrected for larger gear size, the pooled mean catch rate for sharks and rays in the depth range 300-525 m was about 28% of the mean recorded in 1976-77. In common with many multi-species trawl fisheries, the results suggested that the biomass of most species of sharks and rays on the NSW slope declined rapidly as the fishery developed and are now at very low levels.

Market data are consistent with these results and suggest the decline is continuing. Detailed Sydney Fish Market data are available from 1992. Since that year, annual sales of Endeavour sharks (*Centrophorus* spp.) declined from 83 t to 17 t in 2002; similarly, greeneye shark (*Squalus* spp.) sales totalled 150 t in 1992 but only 40 t in 2002. The decline in demersal dogshark stocks on the NSW upper slope trawl grounds is symptomatic of the overall decline in the trawl fishery off the southern half of NSW (south of Sydney). From a peak of 100-130 trawlers in the early 1980s, there are now fewer than 40 fish trawlers working the upper slope grounds.

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Table 2. Summary of deepwater dogshark stomach content analyses (total %s for most species exceed 100% as many stomachs contained two or more types of food items).

% of stomachs with each food category					
	Demersal teleost	M'pelagic teleost	Lanternfish	Cephalopod	Crustacean
Upper slope spp.					
<i>Squalus</i> sp. F	40	53	-	33	29
<i>S. megalops</i>	58	9	-	39	12
<i>S. cf. mitsukurii</i>	85	6	-	24	33
<i>Centrophorus</i> cf. <i>uyato</i>	40	40	-	20	15
<i>C. harrissoni</i>	10	81	-	20	1
<i>C. cf. moluccensis</i>	10	72	-	16	13
Mid slope spp.					
<i>Deania</i> spp.	1	-	85	15	-
<i>Etmopterus</i> sp. B	20	-	-	80	2
<i>Centroscymnus owstoni</i>	20	-	6	70	-
<i>C. crepidater</i>	-	-	50	35	10
<i>C. coelolepis</i>	large chunks of fish (30%), squid (50%) and 'blubber'(25%)				

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THE DIET OF WOBBERGONGS IN NSW: PRELIMINARY RESULTS

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ABSTRACT

Wobbegongs are top predators which have potentially strong impacts on lower trophic organisms. A diet study was undertaken on the three species of wobbegongs found in NSW using commercially hooked sharks. 403 sharks were dissected but only 115 stomachs had any prey item other than bait (28.5%). Teleosts were the major group found in wobbegong stomachs with unidentified fish the most prominent prey item. Other important teleosts found in wobbegong diet include Carangidae, Scrombidae and Labridae. Cephalopods (mainly octopus) were the second most common prey items and elasmobranchs including Orectolobidae, Rajiformes and Triakidae were also found in *O. ornatus* stomachs. The results obtained here accord with other accounts of wobbegong diet based on significantly smaller samples, although no crustaceans were found in this study. Most sharks were hooked in their gut or oesophagus (90.4%) and had a single prey item only in the gut (46%). Wobbegongs were calculated to be tertiary consumers with a trophic level of 4.23. Most prey items were demersal animals and reef-associated, consistent with the benthic environment preferred by wobbegongs.

INTRODUCTION

Wobbegongs (Orectolobidae) are large benthic sharks commonly found in NSW coastal waters. Three species of wobbegongs can be found in NSW (*Orectolobus maculatus*, *O. ornatus* and *O. sp.*) including a new, undescribed species. Wobbegongs have been commercially targeted using setlines for over 15 years. Their catch rate has declined from approximately 120 tons in 1991 to 60 tons in 2001 representing a decline of 50% in a decade (NSW Fisheries, unpublished data). This has resulted in wobbegong being listed as 'vulnerable' in NSW on the IUCN Red List and concern exists as to the sustainability of the fishery and to the implications of this decline (Pollard *et al.* 2003).

Sharks including wobbegongs are high trophic level predators, and at least some species have been shown to have an important role in energy exchange between upper trophic levels in marine ecosystems (Wetherbee *et al.* 1990). Therefore long-term reductions in wobbegong abundance could potentially result in significant changes to the coastal marine ecosystem.

However, although wobbegongs are known to feed on a variety of prey, a quantitative assessment of diet has not been conducted. This is a critical first step to any study of the ecosystem effects of the wobbegong fishery. Despite this clear need, only descriptive studies from limited samples have been undertaken. Cochrane (1992) described prey items from 15 stomachs from *O. ornatus* and *O. maculatus* caught around the NSW/Queensland border, and reported that over 93% of their diet consisted of various teleost fish. By contrast, while Chidlow (2001) reported that teleost were the dominant prey group in animals taken off Perth, Western Australia, they were not as predominant, with 60% occurrence from 20 *O. ornatus* stomachs and 66.7% occurrence from six *O. maculatus* stomachs. Here, we describe the diet of wobbegongs in NSW from a large sample of commercially fished wobbegongs from four different areas of the NSW coast. The relative importance of each prey items will be discussed as well as feeding behaviour and habits. The implications of the diet and the sampling method will be discussed in relation to the future of the fishery.

METHODS

Wobbeong stomachs were collected from commercial fishers between June 2003 and July 2004 at four locations in NSW (Nambucca Heads, Port Stephens, Newcastle and Sydney). Dissection of sharks and collection of stomachs occurred either whilst on fishing boats or at the fishing co-operative depending on the place where gutting and cleaning of sharks took place. Wobbeong stomachs were removed by cutting the pyloric sphincter and connective tissue as well as the oesophagus above the oesophageal sphincter. Stomachs were then placed in a labelled bag and stored at -20°C for later analysis in the laboratory. Empty stomachs or those containing only baits were discarded from the analysis.

The number of empty and regurgitated stomachs as well as stomachs with bait only was recorded and expressed as a percentage of the total number examined. Stomachs were only considered as regurgitated when evidence of inversion could be seen such as stomach found inside the wobbeong's mouth. Position of hook from setlines was also recorded and categorised as mouth, oesophagus or gut.

Table 1. Description of the different 'state of digestion' categories

Category	Description
1	Fresh prey item with most skin still present
2	Part of skin missing, prey relatively fresh
3	All skin missing, flesh partly digested
4	Flesh highly digested, some bones visible
5	Mostly only bones left

Recognisable prey items from stomach contents were identified to family. Identification was based on both intact and remaining hard items parts such as beaks, otoliths, skeletal matter and shells. In cases where a prey item was largely digested, identification was still possible by removing otoliths from the skull and using those for recognition. Contents identified as bait items by prominent hook marks and knife cuts were excluded from the analysis. Total number of prey items in each stomach was recorded discarding any potential bait.

The volume of each food item was determined by water displacement to the nearest 5 ml and wet weight measured on an electronic balance. In addition, the total length and width of each food item was taken. In addition, an arbitrary stage of digestion of each prey items were examined and recorded for all stomachs. Prey items were classed in five categories depending upon the degree of degradation (see Table 1).

The contribution of different prey items to the shark's diet were determined by the following quantitative methods:

- Numerical importance (%N, the number of items in each prey category, expressed as percentage of the total number of prey items);
- Frequency of occurrence (%F, the ratio of stomachs containing a particular prey compared to the total number of stomachs containing prey and expressed as a percentage) the sum of the values will exceed 100% because several prey types can be found simultaneously in a single stomach;
- Gravimetric importance (%W, wet weight of a prey category compared to the total weight of the stomach content expressed as a percentage); and
- The index of relative importance (IRI) (Pinkas *et al.* 1971) which incorporates the three previous indices was also calculated and expressed as a percentage (%IRI) based on Cortés' (1997) suggestion:

$$\text{IRI} = (\%N + \%W) * \%F$$

Wet weight was used in this study, instead of volume (as a measure of prey bulk) to calculate IRI. This did not change the method with wet weight simply replacing volume in calculations. Each of these

parameters provides different insight into the feeding habits of the shark.

Using the trophic levels of prey categories from Cortés (1999), the trophic level of wobbegong sharks was also calculated following:

$$TL_k = 1 + \left(\sum_{j=1}^4 P_j \times TL_j \right)$$

where TL_j is the trophic level of each prey category j , P_j is the proportion of prey category j expressed as %IRI proportions. Only the %IRI of prey items from major groups (class and subclass) were used.

RESULTS

A total of 403 wobbegong were examined. One hundred and ten (27.4%) were empty, 147 (36.5%) contained bait only, while 31 (7.6%) stomachs had obvious evidence of regurgitation, i.e. 71.5% of the stomach analysed did not have any prey items in their gut. Only 115 stomachs (28.5%) contained prey items other than just bait.

70.5% of the wobbegongs were gut hooked, 19.8% were hooked in the oesophagus whilst only 9.6% of the sharks caught were hooked in the mouth. In a few cases ($n=7$), the hook perforated the stomach and was found in the liver or firmly caught on a vertebrae.

Most stomachs contained only a single prey item (Figure 1). The number of stomachs decreased as

the number of prey items increases. Fifty-three stomachs had one prey item, less than half ($n=26$) had two prey items, four had three and only two stomachs had four prey items. However, eleven had over four prey items with the maximum number of items found in a single stomach being nineteen.

Most prey items ($n=97$) were classified as state of digestion 3 corresponding to 'skin fully digested and flesh partly digested' (Figure 2). Relatively similar amount of prey items were classified as relatively fresh (state of digestion 2) and highly digested (state of digestion 4) with 39 and 48 prey items respectively. Finally, only seven prey items were freshly ingested (state of digestion = 1).

Teleosts were the most dominant food category on a numerical (85%), wet weight (79.7%), occurrence (97.5%) and %IRI basis (97.1%) and included at least 11 families (Table 2). Within this group, unidentified fish were most prominent at 75.7% IRI followed by Carangidae, Scrombidae, Labridae

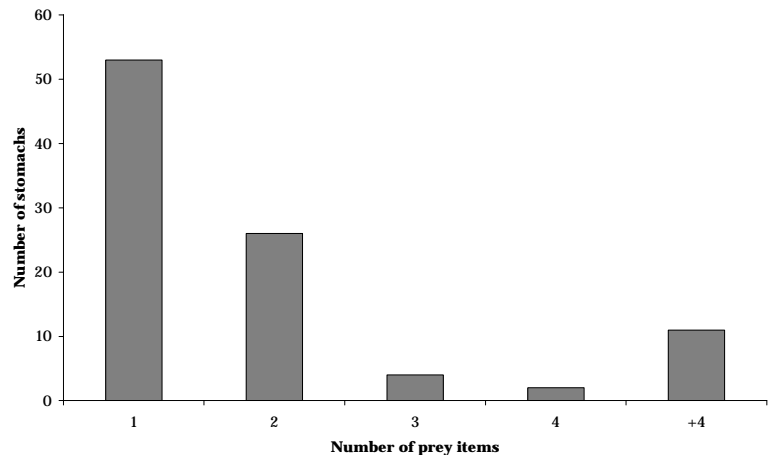


Figure 1. Number of prey items found in wobbegong stomachs.

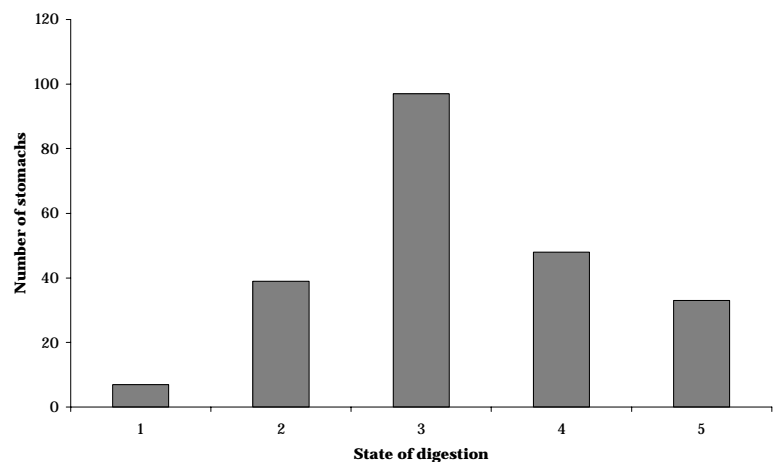


Figure 2. State of digestion of prey items found in wobbegong stomachs.

Table 2. Diet of wobbegong sharks in NSW expressed as percent by number (%N), frequency of occurrence (%F), percent weight (%W) and the index of relative importance on a percent basis (%IRI). Prey categories, demersal (D), benthopelagic (BP) or pelagic (P) are based on descriptions from www.fishbase.org.

Prey categories		%N	%F	%W	IRI	%IRI	%IRI of major groups
Algae							
Algae		4.27	10.75	0.76	54.17	1.01	0.33
Cephalopoda							
Cephalopoda	D	1.71	4.3	0.03	7.48	0.14	
Octopoda	D	5.56	13.98	8.51	196.56	3.68	
Sepiida	D	0.43	1.08	0.12	0.59	0.01	
Total cephalopoda		7.69	19.35	8.66	316.43		1.91
Teleost							
Carangidae	P	35.04	16.13	16.89	837.66	15.68	
Scombridae	P	9.4	7.53	4.64	105.68	1.98	
Labridae	D	1.28	3.23	7.56	28.53	0.53	
Mugilidae	BP	1.71	4.3	1.78	15.01	0.28	
Kyphosidae	BP	0.85	1.08	0.76	1.73	0.03	
Muraenidae	D	0.85	2.15	4.48	11.47	0.21	
Diodontidae	D	0.43	1.08	1.11	1.65	0.03	
Monacanthidae	D	0.43	1.08	0.09	0.56	0.01	
Perciforme	D	0.43	1.08	0.18	0.65	0.01	
Sparidae	D	0.43	1.08	5.05	5.89	0.11	
Sphyrnaeidae	P	0.43	1.08	1.18	1.73	0.03	
Unidentified fish		33.76	58.06	35.92	4045.94	75.72	
Total teleost		85.04	97.85	79.65	16115.01		97.13
Elasmobranch							
Elasmobranch	D	0.85	3.23	1.2	6.61	0.12	
Rajiformes	D	1.28	2.15	6.84	17.46	0.33	
Orectolobidae	D	0.43	1.08	1.41	1.98	0.04	
Triakidae	D	0.43	1.08	1.48	2.05	0.04	
Total elasmobranch		2.99	7.53	10.93	104.79		0.63

with 15.7%, 1.98%, and 0.6% IRI respectively. Other teleosts included: Mugilidae, Kyphosidae, Muraenidae, Sparidae, Diodontidae, Sphyrnaeidae, Monacanthidae and Perciformes, but these corresponded to less than 0.3% IRI each. The importance of those families was similar for %N, %W and %F.

Most cephalopods found in the stomach of wobbegongs were identified as octopus (3.7% IRI) while only one cuttlefish was found in an *O. sp.* stomach (0.01% IRI). Cephalopods were the second most important category numerically (7.7%), by wet weight (8.7%), by occurrence (19.3%) and by IRI (1.9%).

Elasmobranchs were the third and last group found in wobbegong stomachs with 3%, 10.9%, 7.5% and 0.6% in %N, %W, %F and %IRI respectively. The elasmobranch diet was composed of three Rajiformes, one *Mustelus antarcticus* or *Galeorhinus galeus* (gummy or school shark respectively), one Orectolobidae most probably a juvenile *O. ornatus* or an adult *O. sp.* found in a *O. ornatus* stomach indicating possible cannibalism and two unidentified elasmobranchs.

No crustaceans were found in any stomachs whereas algae were found in ten separate stomachs (0.3% IRI).

Most prey items found in wobbegong stomachs (including teleosts, cephalopods and elasmobranchs) were categorised as demersal. However, the two most important Teleost categories were pelagic fishes (Carangidae and Scombridae). Only two prey item categories (Mugilidae and Kyphosidae) were considered benthopelagic.

The trophic level of wobbegong sharks was calculated to be of 4.23 indicating that they are tertiary consumers.

DISCUSSION

The diet of Orectolobidae from NSW waters was dominated by teleosts, cephalopods and elasmobranchs. This generally agrees with the limited reports of dietary components found in the literature which described wobbegongs to feed on bottom reef fish and invertebrates such as crabs, rock lobsters and octopuses (Whitley 1940; Stead 1963; Last and Stevens 1994; Compagno 2001). According to FishBase's (www.fishbase.org) classification of fishes, wobbegong sharks mostly feed on demersal prey, which is consistent with their benthic habitat. However, the major Teleost prey (based on %IRI), Carangidae and Scrombidae, identified as yellowtail (*Trachurus novaezelandiae*) and slimy mackerel (*Scomber australasicus*), are pelagic fishes. Both species are schooling fish and were found in large numbers in single stomachs. For example, up to ten *T. novaezelandiae* and nine *S. australasicus* were found in one large *O. ornatus* stomach. The large %IRI in both categories is mainly due to the large number found in only a few stomachs. However, the relative importance of both Carangidae and Scrombidae is diminished when the frequency of occurrence is considered, as so few sharks consumed pelagic species. Given that both *T. novaezelandiae* and *S. australasicus* are found year round at high densities in the pelagic zone of the inshore rocky reef environment, it appears that while they are clearly on occasion consumed, they are not preferred prey.

The majority of prey species consumed were those that are closely associated with reef ecosystems and this is consistent with wobbegongs preferred habitat (Whitley 1940; Stead 1963; Last and Stevens 1994; Compagno 2001). Wobbegongs can therefore be considered to preferentially feed on demersal organisms in their preferred rocky reefs habitat.

In Western Australia, Chidlow (2001) found that teleosts were the dominant prey group with 60% and 66.7% of occurrence for *O. ornatus* and *O. maculatus* respectively. This study found a larger proportion of teleosts with 97.8% occurrence in the three species combined which is similar to Cochrane's earlier studies further north, but also on the east coast of Australia. In all three investigations, teleosts were found to be the major prey items but with a large proportion of unidentified fish.

There are reports from a number of other shark species forming part of the wobbegong's diet including cannibalistic behaviour by *O. maculatus* (Coleman 1980; Compagno 2001). Previously, only Compagno (2001) has reported consumption of rays. Cochrane did not find any evidence of elasmobranchs as prey items in her study, but both Chidlow (2001) and this study found both sharks and rays in Orectolobidae's stomachs.

In contrast to Coleman and Compagno, cannibalistic behaviour in *O. maculatus* was not found. However, an Orectolobidae was found in the stomach of an *O. ornatus*. This item was identified as either a juvenile *O. ornatus*, indicating cannibalism, or an adult *O. Sp. B*. Juvenile *O. ornatus* have rarely been caught in the same area as large *O. ornatus* indicating possible avoidance of predation by juveniles. However, large *O. ornatus* have been observed feeding upon caught *O. Sp. B* on a number of occasions (C. Huveneers, pers. obs.). One *O. ornatus* has previously been found with three *O. Sp. B* in its stomach (R. Brislane, pers. comm.). Cannibalistic behaviour in *O. ornatus* is therefore possible although in this case, the Orectolobidae found was more likely an adult *O. Sp. B*. Although the evidence is sparse, this does indicate that the potential impacts of cannibalism and inter-species predation between wobbegong species need to be included in fisheries harvest models. For example, a decrease of large wobbegong numbers induced by extensive fishing pressure could lead to an increase of juveniles if it results in reduced predation by large wobbegongs. In K-selected organisms such as wobbegong, juveniles are considered to be the most sensitive stage and with the highest elasticities (Heppell *et al.* 1999). The increase of juvenile numbers from reduced predation pressure could therefore lead to a total increase of the wobbegong population as long as the fishery does not transfer effort from diminishing large wobbegongs to more abundant juveniles.

Other elasmobranchs were only found in *O. ornatus* stomachs and were the second most common

prey item in Western Australian sharks (25% of occurrence). In NSW, elasmobranchs were the third most common prey item with 2.99% of occurrence. One Port Jackson shark (*Heterodontus jacksoni*) and four rays were found in Western Australian shark stomachs while one *Galeorhinus galeus* or *Mustelus antarcticus*, one juvenile *O. ornatus* or *O. Sp. B* and three rays were found in New South Wales.

Cephalopods were found in the two previous studies and in this investigation and range from the 19.5 found in this study to nearly 28% (Chidlow 2001). Octopus was clearly the most prominent cephalopod prey item on the east coast of Australia and this reflects the high density of large octopus in the nearshore rocky habitat (Norman and Reid 2000) and attests to the demersal, near rocky habitat feeding habit of wobbegong sharks.

No crustaceans were found in wobbegong diet despite a large sample size. This contrasts with Cochrane (1992) who reported that crustaceans were found at 6.6% of occurrence. However, careful reading of Cochrane's report indicates that this 6.6% was from a single crustacean found in only one stomach, and illustrates the caution which must be applied to interpretation of generalities from small sample sizes. While other authors (Whitley 1940; Stead 1963; Last and Stevens 1994 and Compagno 2001) have reported crustaceans as part of wobbegongs diet, neither Chidlow nor this study found any evidence of crustaceans in stomachs of wobbegongs from eastern Australia. It is possible that the one crustacean reported by Cochrane (1992) might have been secondary and from the stomach of a teleost prey item, or may have been engulfed along with other prey. Alternatively, crustaceans may be part of pup or juvenile wobbegong diet reflecting possible ontogenetic changes. We did not obtain many stomachs from sharks smaller than 90cm. Future research should include a specific goal of obtaining samples from pups and juvenile sharks, in order to identify any variation in feeding habits and potential habitat change.

The presence of algae in 10 wobbegong stomachs can be considered incidental to prey capture and reflects the largely demersal feeding habits of the wobbegongs.

The trophic level of wobbegong sharks calculated at 4.23 is similar to Chidlow's finding (4.33 and 4.23 for *O. ornatus* and *O. maculatus* respectively) and is the highest of the orectolobiformes (average 3.6, max 4.1) (Cortés 1999). The orectolobiformes trophic level estimate from Cortés (1999) did not include Orectolobidae. The high trophic level of wobbegong sharks supports that they are top predators. This high trophic level is attributed to elasmobranchs (trophic level of 3.65) found in the diet of *O. ornatus*. In Cortés (1999), no elasmobranchs were found in the diet of any orectolobiformes justifying their lower average trophic level of 3.6. Wobbegongs' trophic level is identical to that calculated for marine mammals (not including the sirenians and polar bears) but higher than those of seabirds calculated from both dietary and stable-isotope analyses (Cortés 1999). This result generally suggests that wobbegongs utilise similar resources to these other high-level marine consumers. Given wobbegongs high trophic levels, a high level of exploitation is unlikely to be suitable. Removal of those top predators may potentially have top-down effects on both their prey and other lower level consumers. In order to get a better estimate of predator trophic levels, a greater level of precision is required, for example trophic levels should be calculated separately for demersal and pelagic fish rather than just fish.

The high number of stomachs with single food items for all wobbegong species suggests that feeding is not continuous and that once a meal is consumed, no further prey is captured until that meal is digested. However, 11 stomachs were found with more than four prey items captured. Out of those, five had more than 10 items and one stomach was found with 19 different prey items. Most stomachs with over four items were composed of two species only: yellowtail (*Trachurus novaezelandiae*) and slimy mackerel (*Scomber australasicus*). The schooling behaviour of *T. novaezelandiae* and *S. australasicus* along with the presence of a large number of those two species in single stomachs suggests that wobbegongs are opportunistic feeders, feeding continuously if they have the opportunity. Large *O. ornatus* have previously been found with up to three juvenile *O. ornatus* or *O. Sp. B* confirming that wobbegongs will carry on feeding as long as potential prey can be captured (R. Brislane, pers. comm.).

In over 70% of the case, wobbegongs were hooked in the gut, and in a further 20% the hook was found deep inside the mouth in the oesophagus. This is most likely due to their feeding behaviour. Wobbegongs are considered ambush predators feeding in a similar manner to angel sharks (*Squatina australis*) although prey is taken in front of the wobbegong rather than above it as in angel sharks (Compagno 2001). The short broad mouth and large broad pharynx aid them in sucking in prey. The speed they open their jaws creates a low pressure in their mouth sucking in the water and any prey. Little chewing occurs with preys swallowed whole. With this feeding pattern, hooks are more likely to get caught in stomachs rather than wobbegong's mouth, as was clearly found in this study. Gut hooking makes the use of minimum or maximum size limit as a regulatory tool unreasonable as over 90% of the sharks released would still have hooks in their gut or oesophagus when released.

The total percentage of stomachs found empty, regurgitated or with bait only was high (71.5%) compared to and other shark species and is one of the lowest values found in the literature. It is however consistent with other wobbegong studies. Both Cochrane and Chidlow found approximately 30% of stomachs analysed contained prey items. The high percentage of empty stomachs may reflect short periods of feeding followed by periods of rapid digestion. Alternatively, the method of fishing using a setline may explain the high percentage of empty stomachs (Cortés 1997). Bait is most attractive to hungry sharks rather than sharks with full stomachs, which suggests that fish that feed to satiation have a reduced response to bait odour (Lokkeborg *et al.* 1995). However, of the wobbegongs with prey items in their stomachs, many sharks were captured with full stomachs as well as the bait showing than wobbegongs with full stomach would still be attracted and feed on the bait. One *O. ornatus* was found with nineteen teleosts and still fed on the bait. Regurgitation of stomach contents is a more likely reason for such high percentage of empty stomachs. Although only 4.1% of the stomachs were recorded to have been everted, wobbegongs might have regurgitated their stomach and reverted it back to its original position. The sharks could also have regurgitated without everting stomachs. In both cases it would have been recorded as empty rather the regurgitated stomachs. Furthermore, regurgitation was often observed when the shark was brought to the surface and onto the boat (Huveneers, pers. obs.). Regurgitated and empty data is therefore impossible to dissociate and should be looked at carefully and as indication only.

The findings of this investigation are preliminary only. More stomachs are still being collected and further identification of prey items down to species level is being undertaken. However this analysis provides a useful initial description of wobbegong sharks' diet and feeding behaviour in NSW.

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STOMACH CONTENTS OF THREE COMMON DOLPHINS (*DELPHINUS DELPHIS*) AND A STRIPED DOLPHIN (*STENELLA COERULEOALBA*) COLLECTED OFF THE COAST OF NEW SOUTH WALES

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INTRODUCTION

Diet studies of higher marine vertebrate predators assist us in understanding the ecological role of higher predators in marine ecosystems (Barros and Odell 1990; Das *et al.* 2000; Fea *et al.* 1999; Law *et al.* 2003; Silva 1999). An understanding of the dietary composition can be especially important for marine mammals as a means of determining whether there are any interactions with fisheries (Silva 1999). The consequences may lead to improvements in the management of these interactions and even help reduce incidental kills (Barros and Odell 1990).

Determining the diet of small cetaceans is particularly difficult because they are wholly aquatic animals (Hoelzel 2002). Scat and regurgitate analysis cannot be used as in studies of marine vertebrates that come ashore (e.g., sea birds and pinnipeds, Fea *et al.* 1999).

One approach to determining the diet of small cetaceans is the examination of whole stomach contents after death (Barros and Odell 1990; Silva 1999). Causes of death include strandings, incidental net capture and intentional killing (Barros and Odell 1990; Silva 1999). Caution is required in interpreting stomach content data due to possible bias. Biases may arise as necropsy data is often limited or incomplete and sex, age and cause of death are often difficult to infer (Barros and Odell 1990). Stomach contents may also present a non-representative sample of prey contents as stranded animals may not have been foraging at a normal capacity or may be starved (empty stomach) (Silva 1999). Incidental deaths through interactions with fisheries can show a bias towards the fishery target species. Whether this is because the target species is in high abundance, or the animals are scavenging from fisheries is sometimes not known (Silva 1999).

Diet studies involve separating the stomach contents and identifying the hard parts such as otoliths and cephalopod beaks (Barros and Odell 1990). The difficulty with these types of diet studies is that otoliths are often digested and there is no way of *a priori* adjusting for differential digestion (Fea and Harcourt 1997). Furthermore, otoliths found in dolphin stomachs could come from inside larger prey items themselves and should potentially be considered as secondary otoliths. Therefore only minimum prey biomass estimates can be obtained. Stomach contents may also only reveal the animal's last meal and may not be representative of the regular diet (Das *et al.* 2000).

For questions of feeding ecology, fatty-acid and stable isotope analysis can be used in conjunction with stomach contents and provide further information on the history of prey uptake by determining what has been assimilated and not just that recently ingested (Das *et al.* 2000). However, these alternative methods have their own limitations, are expensive and require validation studies. The most economical means of looking at interactions with fisheries is to examine stomach contents as this can rapidly identify prey down to a genus or species level (Barros and Odell 1990; Silva 1999). Biomass estimates can be calculated and reveal what percentage of the diet is made up of individual prey species. For example, Silva (1999) revealed that anchovy was the main fish consumed by *Delphinus delphis* in the waters off Portugal - a species that was also the main target species of the fishery. The implications for interactions with fisheries may be of some import, especially in areas where endangered animals are being incidentally killed. There have been few published studies on the diet of small cetaceans in Australia in recent years (Law *et al.* 2003).

The common dolphin *Delphinus delphis* is widely distributed throughout temperate and tropical waters (Silva 1999). *D. delphis* is known as a pelagic species (Silva 1999) although it can often be seen

inshore and even in bays (Law *et al.* 2003). The diet of *D. delphis* is of near surface pelagic, schooling fish and squid and octopus species (Law *et al.* 2003). The striped dolphin *Stenella coeruleoalba* is a cosmopolitan cetacean with a wide distribution range from tropical to temperate areas (Cebrian 1995). It is considered to be the most abundant dolphin in the Mediterranean (Evans 1987; Bompar *et al.* 1991). The diet of *S. coeruleoalba* is composed of various fish, cephalopods and sometimes crustaceans (Carwardine *et al.* 1998).

This study aims to identify the prey species eaten by *Delphinus delphis* and *Stenella coeruleoalba* in waters off the coast of New South Wales through analysis of stomachs collected from one stranded animal of each species.

METHODS AND MATERIALS

Stomach contents from three common dolphins (*Delphinus delphis*) and one striped dolphin (*Stenella coeruleoalba*) were examined. The first common dolphin (C1) was collected from entanglement in a shark control net on Coogee beach on the 16th September 2003. The striped dolphin (C2) was collected from stranding on Balmoral beach on the 17th August 2001. The second common dolphin (C3) was collected from entanglement in a shark control net on Maroubra beach on the 10th November 2003. The third common dolphin (C4) was collected from stranding on Bondi beach on the 12th November 2003.

The dolphin stomachs were stored at -20°C. The stomach and intestine contents of both stomachs were emptied using the same technique. The stomach contents were washed with gently running water through four stacked sieves with decreasing mesh sizes from 5mm to 1mm. All hard parts were collected, sorted and stored individually. Individual prey items were refrozen for later identification.

Teleost fish species were identified by comparison of otoliths with a reference book (Smale *et al.* 1995) and a reference collection from the Australian Museum in Sydney. Cephalopod beaks were identified using a key from the Museum Victoria. Left and right otoliths were counted separately as were upper and lower cephalopod beaks; the greater number of these was used to represent the minimum number of prey species consumed. Measurements of unbroken cephalopod beaks and sagittal otoliths were made with digital callipers to the nearest 0.01mm. Otoliths were only measured if they appeared intact and were not obviously eroded. Calculations of minimum prey number and minimum original prey biomass were made from the total numbers obtained as above. Determination of original prey biomass for cephalopod beaks was from regression equations from Lu and Ickeringill (2002).

Individual prey items were identified through comparison of various intact features such as jaw structure, gill covers and eye socket size.

RESULTS

Initial examination of the stomachs of C1 and C2 after thawing revealed that the stomachs were still intact and that decomposition was minimal. Initial reports on collection of the common dolphin state that the animal was found after being entangled. The duration of the entanglement before the dolphin was found is unknown. The striped dolphin was found washed up on a beach with possible evidence of old net scar injury, which might have potentially caused the death of this dolphin. Initial examination of the stomach from the common dolphin, C3 after thawing revealed that the stomach was still intact and that decomposition was minimal. Initial reports on collection of C3 state that the animal was found after being entangled for no more than a few days, as decomposition was minimal and bloating was not yet present. Initial examination of C4 stomach after thawing showed that the stomach was highly decomposed and prey items were only loosely held in a thin web of decaying tissue. Reports on collection of C4 state that the animal was found washed up on Bondi beach and may have been dead for several weeks. The animal was bloated and had what appeared to be a bullet wound in its side, which may indicate cause of death. We could not determine the sex of any of the dolphins.

The number of otoliths in the stomach from C1 was much lower than the number found in the stomach of the striped dolphin, C2 (Table 1). The greatest number of otoliths found in the stomach of C1 was from the species *Trachurus novaezelandiae* or yellowtail. However one otolith from *Centroberyx affinis*, commonly known as redfish or Nannygai was also found (Table 1). Some partially digested prey items were found in C1's stomach and could not be identified due to the stage of digestion. The total weight of those unidentified digested prey items was 13.07g. Unfortunately, most of the otoliths found in the stomach from C2 were too eroded to identify the prey items down to species. However, those otoliths were most likely to come from the Perciformes order, either Carangidae or Scorpaenidae family (Table 1). In all cases, otoliths were too eroded to measure them accurately.

The number of otoliths in the stomach from C4 was much lower than the number found in the stomach of C3 (Table 1). The greatest number of otoliths found in the stomach from C4 were from the species *Pagrus auratus* (Table 1), commonly known as snapper, or in the juvenile form as red bream. The partially digested prey items were also found to be of the same species. The average fork length of prey items was 26.3 (SD±6.506). The greatest number of otoliths found in the stomach of C3 were from the species *C. affinis* (Table 1). The half digested prey items were identified as *C. affinis* also. The average fork length of prey items was 17.5cm (SD±1.322). The *Neobythites* spp. could only be identified down to the genus as reference otoliths were only identified to the genus and no literature was available on this species for otolith identification. *Pseudophycis breviuscula* was originally identified as *Austrophycis megalops*, as taxonomic reclassification of this species had not yet been transferred to the otolith reference collection.

Table 1. Fish species and Otolith occurrence in the Stomach Contents of *Delphinus delphis* and *Stenella coeruleoalba*. Where N=total number of otoliths, P=total number of Prey (including whole prey items), MOL=mean otolith length, SE=standard error for otolith length.

Species	Common Name	N	P	MOL left	right	SE left	right
Common Dolphin C1							
<i>Trachurus novaezelandiae</i>	Yellowtail	23	13				
<i>Centroberyx affinis</i>	Nannygai	1	1				
Not identifiable		3	3				
Broken		10					
Striped Dolphin C2							
Perciformes (carangidae or scorpidae)		150-200	Approx. 80				
Common Dolphin C3							
<i>Centroberyx affinis</i>	Nannygai	27	18	7.68	7.59	0.0906	0.1189
<i>Neobythites</i> spp	Ray finned fish	1	1	7.14	-	-	-
Unidentified		6	3	2.38	2.35	0.435	0.4147
Not identifiable		4	-	-	-	-	-
Common Dolphin C4							
<i>Pagrus auratus</i>	Snapper	4	5	5.25	5.07	0.3	0.205
<i>Pseudophycis breviuscula</i>	Northern bastard codling	1	1	4.14	-	-	-

Table 2. Cephalopod biomass estimates from the stomach of *Delphinus delphis*. Where N=total number of beaks, Wt=fresh weight g, ML=mantle length cm.

Species	Common Name	N lower	upper	Wt Mean	SE	ML Mean	SE
C1							
<i>Nototodarus gouldi</i>	Gould's flying squid	2	4	108.38	5.4	18.6	4.8
C3 and C4							
<i>Eucloteuthis luminosa</i>	striped squid/luminous flying squid	2	2	0.026	-	-	-
<i>Sepioteuthis australis</i>	Southern reef squid	1	2	0.034	-	-	-

There were a few squid beaks found in the stomach of C1 with four upper and two lower beaks (Table 2). The common *Nototodarus gouldi* or Gould's flying squid was identified as the only species found in the stomach. No cephalopods were present in the striped dolphin's stomach or intestinal tract.

DISCUSSION

These results show that *Delphinus delphis* fed both on fish and cephalopods whereas *Stenella coeruleoalba* fed on teleosts only. However, the otoliths found in *S. coeruleoalba* were much smaller than those in *D. delphis* and could be secondary otoliths originating from prey items. No primary prey items or half digested remains were found in *S. coeruleoalba*'s stomach. The variation in prey species consumed by *D. delphis* was most likely due to variation in distribution and abundance of prey species, as was found by Silva (1999). The level of erosion of all otoliths was relatively strong especially in the striped dolphin making prey identification difficult. The analysis provided here should therefore be considered cautiously, keeping in mind the difficulties encountered during identification.

Both teleost species identified in the common dolphin, C1's stomach are pelagic fishes (*C. affinis* and *T. novaezelandiae*) and coincide with the pelagic habits of common dolphins. Similarly, Gould's flying squid is also known to disperse throughout the water column at night. The main fish species found in C1, *T. novaezelandiae* (yellowtail) is a pelagic fish that form large schools in marine inshore areas such as bays and estuaries. Yellowtails are traditionally used as bait for recreational and commercial fishers. Some are also sold for human consumption caught with handlines or seine nets, mainly off the coast of New South Wales and Western Australia. Unfortunately, biomass could not be calculated from otolith length or weight. Measurement of the undigested prey items was also not possible as there was no tail present on the prey items making it impossible to locate the end of body and beginning of the prey's tail.

The stomach of C3 was in very good condition and still had the intestine attached. A large number of the otoliths found in this stomach occurred in the vast number of folds within the stomach. The main fish species found in C3, *Centroberyx affinis* is a pelagic fish, with a major fishery off the coast of New South Wales. The juveniles inhabit shallower inshore waters and adults are found deeper (Morison and Rowling 2001). Although biomass could not be calculated from otolith length or weight, measurement of the undigested prey items showed that the average fork length was 17.5cm ($SD \pm 1.322$). This average length puts them in an age class of 3-5 years (Morison and Rowling 2001). *C. affinis* of this length are usually thrown back by fisherman (Morison and Rowling 2001). If *D. delphis* was found to feed on larger fish of this species in NSW coastal waters, there may be some implications for overlap with fishers. The difficulty with these length estimates is that generally there was no tail present on the prey item and the end of the vertebrae was used as a marker for where the tail began, although this may not have been the case.

The abundance of squid in the diet of *Delphinus delphis* has been reported as being high in number but accounts for a low percentage of the biomass (Silva 1999). In this study squid was not found to be numerous or high in biomass. The presence of a large amount of *T. novaezelandiae* in the stomach of C1 may indicate that there was no requirement for large amounts of squid, as *T. novaezelandiae* was so abundant. Similarly, the presence of a large amount of *C. affinis* in the stomach of C3 may indicate that there was no requirement for large amounts of squid for this individual either.

The small number of otoliths found in C4, as compared to C3, may be explained by the degradation of the stomach itself. Because the stomach of C4 was so degraded it is possible that many of the otoliths were lost as the stomach itself decomposed. The presence of reef dwelling fish (*Pagrus auratus*) and reef dwelling squid (*Sepioteuthis australis*) suggest that C4 may have been feeding in the vicinity of a reef shortly before it died, although no further inference can be made from this data.

While definitive conclusions about the interactions of *Delphinus delphis* or *Stenella coeruleoalba* with NSW fisheries would be premature, the data reported in this study illustrates the value of opportunistic studies. The cost of alternative techniques to determine the diet of potentially

important competitors with inshore fisheries is high. By maximising the information obtained from opportunistically collected animals, it is possible to gradually build a firmer picture of the role of higher predators in our coastal ecosystems. Amalgamation of this data with future studies may provide us with a clearer picture of the overall diet of *Delphinus delphis* and *Stenella coeruleoalba*.

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