Development of a Fisheries Habitat Suitability Model Utilising a Geographic Information System

David Ball, Liz Morris, Jeremy Hindell and Allister Coots

State Government Victoria
Department of Primary Industries

Australian Government
Fisheries Research and Development Corporation

Project No. 2000/157

Development of a Fisheries Habitat Suitability Model Utilising a Geographic Information System

David Ball, Liz Morris, Jeremy Hindell and Allister Coots

April 2004

Project Number 2000/157

ISBN 1 74146 210 X

Preferred way to cite:
Ball, D., Morris, L., Hindell, J and Coots, A. (2004) Development of a Fisheries Habitat Suitability Model Utilising a Geographic Information System. Final report to Fisheries Research and Development Corporation Project No. 2000/157. Primary Industries Research Victoria, Queenscliff.

Published by Primary Industries Research Victoria, Department of Primary Industries, Queenscliff, Victoria, 3225.

Formatted/designed by Primary Industries Research Victoria Queenscliff

Printed by PIRVic Queenscliff, Victoria

Table of Contents

List of Tables

List of Figures

NON-TECHNICAL SUMMARY

2000/157 Development of a Fisheries Habitat Suitability Model Utilising a Geographic Information System

Principal Investigator: **David Ball**

Co-Investigator: **Liz Morris**

Address: Primary Industries Research Victoria

Marine and Freshwater Systems

PO Box 114, Queenscliff Victoria 3225

Tel: (03) 5258 0210 Fax: (03) 5258 0270

Email: David.Ball@dpi.vic.gov.au

Objectives:

1. Develop a fisheries habitat suitability model for Victorian bays and inlets utilising a Geographic Information System (GIS).
2. Integrate a wide range of existing spatial and non-spatial data for habitat types, environmental parameters, species distribution, species life histories and habitat requirements in the GIS through a relational database.
3. Develop a customised *ArcView* GIS user interface for querying the fisheries habitat suitability model and producing habitat suitability maps.

Non Technical Summary:

The identification and protection of fish habitats are increasingly being recognised as complements to traditional fishery management approaches. This project sought to apply a Geographic Information System (GIS) to address questions about the spatial relationships between fishery species and habitats in Victorian bays and inlets. The project integrated spatial and non-spatial data on habitats, environmental parameters, species habitat requirements, species life histories and catch and sampling statistics. Spatial modelling techniques were then applied to combine multiple layers of data to produce habitat suitability maps for selected fish species.

The primary aim of this study was to model habitat suitability in Port Phillip Bay, Western Port and Corner Inlet for commercially significant species. The main species addressed by this study were: King George whiting *Sillaginodes punctata*, snapper *Pagrus auratus*, greenback flounder *Rhombosolea tapirina,* rock flathead *Platycephalus laevigatus*, sand flathead *Platycephalus bassensis,* Australian salmon *Arripis trutta* & *A truttaceus,* southern calamari *Sepioteuthis australis* and yellow-eye mullet *Aldrichetta forsteri.*

For the first stage of the project we adopted a Habitat Suitability Index (HSI) modelling approach. This is a spatial analytical technique that can be used to estimate the distribution of fish species by linking environmental data with species presence/absence. The process of HSI modelling involves deriving suitability indices (SI's) for each species that indicate a preference or affinity level for selected environmental or habitat variables. SI's were derived from existing fisheries independent data using a mix of qualitative and quantitative approaches. These datasets had been collected for a range of purposes and some habitats were either under-represented in the sampling or may not have been sampled at all. In these cases, we reviewed the literature and used expert opinion to identify habitat preferences.

For species investigated in this study, SI values were assigned to each habitat or environmental parameter according to a scale ranging from 0 (unsuitable habitat) to 1 (optimum habitat). Depth and substrate type/biota were the main spatial habitat layers used to produce the models, with a seabed sediment layer also available for Port Phillip Bay. Spatial models or maps of the habitat suitability index values were then generated in the GIS by calculating a mean SI value across the habitat layers. The resulting composite HSI values were then grouped into classes from low to high to predict zones of habitat suitability.

A customised GIS Habitat Suitability Model Interface was developed with ArcView 3.3 and ArcView Spatial Analyst. In order to simplify the HSI modelling process and allow users to generate spatial models with limited instruction, a habitat suitability modelling wizard was developed that guides the user through a step by step process on how to operate the model.

A second stage of the project investigated developing habitat suitability models using commercial fisheries catch data. This was a new approach to habitat suitability modelling and involved an analysis of fishing log books that identify the amount and type of species caught, fishing gear used and time fished by all commercial fishers. Port Phillip Bay catch statistics were extracted for the period April 1998 to June 2001 providing three complete years of data. The location of commercial catches was recorded through a system of fishing blocks.

Each commercial fishing block was characterised by depth, substrate type/biota and seabed sediment types within the GIS to determine the proportion by area of each habitat variable. A statistical analysis was then carried out to test the hypothesis that those fishing blocks where fish species were caught would differ in terms of their environmental parameters from areas where fish of that species were not caught. Where there was a significant difference, the relationship between the environmental parameters of the fishing blocks was further explored using multivariate statistical techniques. Following the analysis of fishing catch blocks, a series of simple rulings were used to determine whether environmental combinations were of 'high', 'medium' or 'low' suitability for the species in question. Once the combined environmental parameters were defined as high, medium, low or undefined for each species, the GIS layer was reclassified to create a predictive map of habitat suitability.

Outcomes Achieved

We produced a GIS application for modelling the spatial distribution of habitat for important commercial fish species in Victorian bays and inlets. Two separate modelling approaches were investigated. The first approach adapted an existing habitat suitability index methodology based on analysis of fishery independent data, and a second new approach was developed using fishery dependent (commercial catch) data. The models can be readily updated to incorporate new information and understandings about species-habitat interactions or affinities as they become known. Similarly, the models can also incorporate new spatial data representing the distribution of important environmental variables or habitats.

The spatial models produced in this study present a simplified picture of habitat suitability and do not account for many complex relationships and interactions between species and environmental variables. However, in the absence of a more complete knowledge of the nature of these relationships and the spatial scales at which they operate, the habitat suitability modelling approach presents a relatively effective method for identifying likely distributions of important fishery habitat. This ability to address the spatial aspects of fishery habitat has not been readily available to fishery and natural resource managers to date.

Keywords: GIS, Habitat Suitability, Port Phillip Bay, Western Port, Corner Inlet

Acknowledgments

This project was co-funded by the Fisheries Research and Development Corporation and Fisheries Victoria, Department of Primary Industries. The Fisheries Research and Development Corporation plans, invests in and manages fisheries research and development throughout Australia. It is a statutory authority within the portfolio of the federal Minister for Agriculture, Fisheries and Forestry, jointly funded by the Australian Government and the fishing industry.

Sean Blake at Primary Industries Research Victoria (PIRVic) Marine and Freshwater Systems assisted with the data analysis and map production.

Dr Greg Jenkins at PIRVic Marine and Freshwater Systems reviewed the draft report.

Dr Greg Parry at PIRVic Marine and Freshwater Systems assisted with accessing and interpreting data from the Port Phillip Bay Trawl Program.

The Victorian Department of Natural Resources and Environment Science Quality Unit awarded David Ball an overseas travel grant under its Enhancing Science Networks program to support this project. This grant enabled David to travel to the USA during 2001 and visit a number of organisations at the forefront of the application of GIS to the management of marine fisheries and essential fish habitat. The experience gained during this visit was invaluable to this study and the assistance of the following organisations is gratefully acknowledged: Florida Marine Research Institute, NOAA National Ocean Service – Biogeography Program and University of Miami – Rosentiel School of Marine and Atmospheric Science. The assistance of Dr Peter Rubec at the Florida Marine Research Institute was invaluable in organising this research trip.

1. Introduction

1.1. Background

The identification and protection of fish habitats are increasingly being recognised as complements to traditional harvest management approaches and as critical parts of maintaining living resources (Monaco & Christensen 1997). A recognisable shift has occurred from viewing fished species and fisheries habitats as separate unconnected entities, to viewing them as components of larger ecosystems and towards an ecosystem based management approach.

Spatial modelling of fish species-habitat links is well suited to the application of Geographic Information Systems (GIS). GIS provides the ability to integrate and analyse data from large and diverse datasets and to model spatial relationships between variables in different data types. Habitat Suitability Index (HSI) modelling is one such spatial analytical technique that can be used to estimate the distribution of fish species by linking environmental and habitat data with species presence/absence. HSI modelling with a GIS enables the spatial distribution of fish species and habitats, as well as potential effects of habitat change, to be observed and mapped (Christensen *et al*. 1997).

The process of HSI modelling involves deriving suitability indices (SI's) for a selected species (often by lifestage and/or season) that indicate a preference or affinity level for selected environmental variables (eg. salinity, depth, temperature, substrate etc). Although interactions commonly occur between environmental variables, the HSI modelling approach assumes their independence from one another. Individual SI's are derived under the assumption that all other variables are held constant at, or near their species-specific optimum (Coyne & Christensen 1997). The suitability indices are typically scaled from 0 (unsuitable) to 1 (optimum habitat) and a combined habitat suitability index can then be calculated by various methods (eg. geometric mean) and represented spatially in a raster GIS.

The application of GIS in marine fishery HSI modelling was pioneered by the National Oceanic and Atmospheric Administration (NOAA) and the Florida Marine Research Institute (FMRI) (see Brown *et al.* 2000, Christensen *et al.* 1997, Clark *et al.* 1999, Rubec *et al.* 1998, Rubec *et al.* 1999). These studies employed a range of qualitative and quantitative techniques to derive SI values for coastal fish species.

In this project we explored a number of approaches to developing Habitat Suitability Index models for commercially significant fish species within Victorian bays and inlets. The main emphasis of the project was to use existing data to develop broad-scale habitat suitability indices using a number of different approaches. This project sought to apply a GIS to address questions about the spatial relationships between fishery species, environmental parameters and habitats in Victorian bays and inlets. The project integrated spatial and non-spatial data on habitats, environmental parameters, species habitat requirements, species life histories and catch and sampling statistics. Spatial modelling techniques were then applied to combine multiple layers of data to produce habitat suitability maps for selected fish species. The system enabled predictions on the likely impact of variations in environmental parameters and habitats to be assessed.

1.2. Need

The shift towards an ecosystem based fishery management approach is, in part, reflected in the reviews of fisheries and seagrass habitat research undertaken for FRDC (Cappo *et al*. 1998; Butler & Jernakoff 1999). Cappo *et al.* (1998) in particular, found that the lack of knowledge of "critical habitats" and habitat links for many fisheries at all scales is a major strategic R&D issue for FRDC.

Cappo *et al.* (1998) stated that "we must know where and what must be conserved for sustainability of fisheries and mariculture, before we determine why and how to do it". At present, key uncertainties exist concerning both the relative values of fisheries habitats and the effects of human disturbance at both regional and local scales. As a result, Cappo *et al.* (1998) found that strategic R&D is needed to overcome the poor ability to predict and manage such disturbances. Specifically, the collection, interrogation and extension of new and existing fisheries and habitat data at scales useful to management are required.

While the habitats which fish are found in association with are generally known, the critical factors that govern fish-habitat usage are poorly understood. In the absence of knowledge about why fish utilise specific habitats, techniques are required to provide managers with information about the relative importance of different habitats and an ability to predict the impact of different pressures on these habitats. A Habitat Suitability Modelling approach enables fisheries managers to identify the spatial component of fish-habitat links and make informed decisions on the management of habitats.

HSI modelling aims to apply a system of determining spatial distributions of important commercial fishery habitats by addressing the habitat requirements of selected fish species. The development of habitat suitability modelling reflects an increasing awareness that management of target populations needs to place greater emphasis on habitat quality and availability. The identification of high/low quality or 'critical' habitat allows areas of high conservation or commercial interest to be identified, impacts of environmental change to be evaluated in an interactive manner; or predictive species distribution maps to be developed for poorly sampled areas.

1.3. Objectives

The project objectives were as follows:

1. Develop a fisheries habitat suitability model for Victorian bays and inlets utilising a Geographic Information System (GIS).
2. Integrate a wide range of existing spatial and non-spatial data for habitat types, environmental parameters, species distribution, species life histories and habitat requirements in the GIS through a relational database.
3. Develop a customised *ArcView* GIS user interface for querying the fisheries habitat suitability model and producing habitat suitability maps.

1.4. Methods

The methods employed in the different stages of the project are described in detail in the relevant sections of the report. The following provides an overview of these methods.

1.4.1. Selection of Fish Species

The aim of this study was to address commercially significant species. These species were initially identified by examining Fishery Victoria annual commercial catch bulletins and a review of habitat influences on commercial catches in bays and inlets (Gunthorpe *et al*. 1998). As this study relied upon existing data, the initial list of species was subsequently modified to reflect the available data. The fish species addressed by this study are discussed in Section 4.

1.4.2. Spatial Distribution of Habitats & Environmental Parameters

Most of the habitat and environmental variables addressed by this study were available as spatial layers in a GIS format from previous mapping studies undertaken by PIRVic (eg. seagrass, depth etc). Where data were not available in a GIS format, new layers were created at a suitable scale (1:25,000) by converting data to a digital format or extrapolating existing digital data. Most of the existing GIS data were in a vector format and needed to be converted to a raster (gridded) GIS format using the ArcView Spatial Analyst extension. Spatial data used during this study are presented in Section 3.

1.4.3. Species Habitat Suitability Indices

Previous reviews of Victorian bays and inlets have summarised most of the available life history information for commercial fish species in these areas (see Gunthorpe *et al.* 1998), but significant gaps in our knowledge remain for many species. This is consistent with Cappo *et al*. (1998), who found that there is a lack of basic life-history information for most of the major fishery species in Australia, and as a result there is a paucity of information on "critical" habitat requirements and processes such as recruitment, post-recruitment mortality and competition, spawning and species interactions.

Undertaking quantitative analyses to define species habitat affinities is dependent on having field-based sampling that provides species catch rates and simultaneous measurements of environmental variables (Monaco & Christensen 1997). In most cases however, habitat affinities are assessed from disparate data collected with multiple sampling strategies over different temporal and spatial scales (Monaco *et al*. 1998). In Victoria, extensive sampling has been undertaken over the past two decades, but the application of much of this data for determining quantitative species habitat suitability indices is limited because of varying temporal and spatial scales of data collection, different sampling strategies and a lack of simultaneous measurements of environmental variables.

Tables of habitat suitability index values were required for the target species that characterised the habitat requirements of different life cycle stages. Fishery independent data was consolidated from a variety of sources and analysed to quantify habitat affinities and thereby generate habitat suitability indices for selected species. Where data were unavailable, habitat suitability indices were determined from a literature review and expert opinion. The literature review of fish habitat requirements is presented in Section 4 and the analysis of fishery independent data is presented in Section 5.

A lack of fishery independent data for many species hindered the development of habitat suitability indices. Commercial catch and effort records constitute the largest fisheries dataset in Victorian bays and inlets. Commercial fishery catch and effort log book data has been found to be useful for augmenting research studies and improving estimates of the distribution and abundance of selected species (Starr and Fox 1997). As a result, an analysis of the commercial fishery catch data was undertaken to determine whether habitat affinities could be derived by characterising log book catch cells according to both the recorded catch and dominant environmental variables. The analysis of commercial catch data to derive habitat suitability models is presented in Section 6.

The resulting habitat suitability indices from the fishery independent and dependent (commercial) data were imported to the ArcView GIS. The table structure for the habitat suitability indices was relatively simple, so a relational database was not required to link these tables to the corresponding spatial data in the GIS.

1.4.4. Habitat Suitability Modelling

Once the spatial habitat data and habitat suitability index tables had been linked in a GIS, it was possible to integrate these datasets to produce combined habitat suitability models for each species. In this study, a model refers to a spatial representation of the predicted distribution of suitable habitat in the form of a map which represents a "model" of reality. This "cartographic model" or mapping approach, differs from the typical concept of statistical modelling applied by fisheries biologists to model species-habitat interactions.

Traditional simulation modelling of biological processes is dependent on a high degree of knowledge of the processes involved and the reasons for the behaviour of the species being modelled. Much of this information is either not available, or poorly understood for marine ecosystems, and an alternative approach is required if fish-habitat links are to be identified.

The HSI approach accounts for the lack of knowledge and empirical links by taking a presence/absence approach for different environmental parameters and habitats to build up a picture of the overall suitability of an area for a particular species. The HSI approach allows users to apply a simple model to combine the suitability ratings assigned to each environmental data layer to calculate an overall habitat suitability for the selected species and area of interest.

1.4.5. GIS Softwater User Interface

A customised user interface for the HSI modelling application was developed with the GIS Software ArcView 3.3. The user interface automates the task of producing habitat suitability maps and is designed for users with limited GIS experience. The ArcView HSI interface is presented in Section 7.

1.5. Study Areas

The study primarily addressed Port Phillip Bay and Western Port and to a lesser extent Corner Inlet/Nooramunga.

1.5.1. Port Phillip Bay

Port Phillip Bay is a large marine embayment with an area of about 1,950 km^2, and a 260 km coastline (Figure 1.1). The State capital of Melbourne sits at the northern end of the Bay with its suburbs extending around the northwestern and eastern shores. With the exception of the area known as The Rip at the entrance to Bass Strait, which reaches depths in excess of 90 m, the majority of Port Phillip Bay is relatively shallow, with depths mostly less than 25 m.

Tides in Port Phillip Bay are semi-diurnal, generally with a range of less than 1 m. There is a time-lag of more than 3 hours between high tide at The Rip and the northern regions of the Bay due to the Bay's narrow entrance and the barrier presented by a shallow region inside the entrance called the Great Sands. The average tidal exchange through The Rip is approximately 1 km^3, or 4% of the Bay's total volume (Vic. Govt. 1992 **in**: Winstanley 1995).

Salinity in the Bay is primarily marine and water temperatures in the Bay range from around 10°C in winter to >20°C in summer (Harris *et al*. 1996).

Sediments ranging from clays to gravels account for approximately 90% of the substrate in Port Phillip Bay. The remaining 10% consists of reefs (Hope Black 1971). These broad substrate types are an important influence on the distribution of flora and fauna within the Bay. Macroalgal communities are generally associated with areas of reef, and form the dominant floral community around The Rip region.

The primary species of seagrass found in the Bay are *Zostera muelleri, Heterozostera tasmanica, Halophila australis* and *Amphibolis antarctica*. A 2001 study mapped a total vegetated area of 169 km^2 (seagrass, macroalgae and *Pyura*) in Port Phillip Bay, of which 68 km^2 or 40% was seagrass or a mixture of seagrass and algae (Blake & Ball 2001a). The dominant category of all vegetation by area was "Undefined Macroalgae" which accounted for an area of 85 km^2 or 50% of the total vegetation mapped. The remaining 10% consisted of different categories of algal species and beds of *Pyura.*

Zostera/Heterozostera was the dominant category of seagrass recorded in the Bay, accounting for 59 km^2 or 95% of the total seagrass mapped. The majority of *Zostera/Heterozostera* was recorded in Swan Bay and along the southern shores of the Geelong Arm and Corio Bay. Significant areas were also present along the north shores of Corio Bay and the Geelong Arm, south of St. Leonards (West Sand Bank), around Mud Islands (Great Sands), off Sorrento (South Sand) and along the Rosebud to Blairgowrie foreshore. Smaller isolated patches were scattered around the Bay, with very little along the eastern shores (Blake & Ball 2001a).

Amphibolis antarctica accounted for 3% of the total seagrass mapped and *Halophila australis* accounted for the remaining 2%. *A. antarctica* is restricted in its distribution to Point Lonsdale Bay, north to Queenscliff Pier and along the shore from Point Nepean to Sorrento. *H. australis* was restricted to the deeper, soft sediments of Swan Bay, the Geelong Arm and Corio Bay (Blake & Ball 2001a).

1.5.2. Western Port

Western Port is a tidal embayment which encloses two large islands. French Island in the northern half of the bay, has an area of approximately 170 km^2 and Phillip Island, to the south, has an area of approximately 100 km^2 (Figure 1.1). The waters of Western Port cover an area of about 680 km^2, of which about 270 km^2 consists of intertidal mud flats that are exposed at low tide (WPRPCC 1992).

Western Port has two entrances located either side of Phillip Island (Figure 1.1). The Western Entrance is the larger of the two, reaching depths of up to 30 m, while the smaller eastern entrance between San Remo and Newhaven has a depth of about 5-10 m. Western Port is characterised by an extensive tidal channel system, with major arms on the western (North Arm) and eastern (Eastern Arm) sides of French Island. Depths in these channels exceed 15 m. An intricate network of smaller tidal creeks feeding into the main channels become swift flowing streams towards low tide as water drains from the exposed intertidal flats (LCC 1993).

Extensive low-relief mud flats are also a characteristic feature of Western Port and some flats may be as much as 1 m above the water level in the channel system at low tide. In the north-eastern sector of the bay, an extensive area known as the Tidal Divide or Watershed separates the headwaters of the tidal channel systems of the north and east arms. Extensive intertidal flats have also developed where sheltered conditions prevail at Hastings Bight, Blind Bight and Rhyll Inlet (LCC 1993).

Western Port features a semi diurnal tidal cycle. The tidal range within Western Port increases towards the north end of the embayment with Flinders having a range of 1.6 m while Tooradin has a range of 2.2 m, with low tide at Tooradin occurring about two hours after Flinders (Shapiro 1975).

Approximately 108 km of the Western Port shoreline is fringed by mangroves. Only one species of mangrove occurs in Western Port, the white mangrove *Avicennia marina*. The white mangrove grows in a 1-4 m high open scrub formation on muddy substrates. Most of the ground beneath the mangroves is bare although a sparse coverage of common salt marsh species may occur in well drained areas and thick mats of brown algae may also occur (Yugovic *et al*. 1993). The Western Port coastal region also features large areas of saltmarsh that extend inland from the behind the mangroves for distances of up to one kilometre in places.

Four species of seagrass were recorded in Western Port during a 2000 survey; *Zostera muelleri, Heterozostera tasmanica, Halophila australis* and *Amphibolis antarctica* (Blake & Ball 2001b). A total area of 154 km^2 of seagrass and macroalgae was recorded in Western Port during the study. Of this, 130 km^2 (84%) was recorded as being either seagrass or a mixture of seagrass and algae. The dominant category of vegetation by area was the mixed category Dense *Zostera/Heterozostera* with algae, accounting for an area of 43 km^2 or 28% of the total vegetation mapped. *Amphibolis* with Macroalgae was recorded as having the second highest area, covering 20 km^2 or 13% of the total vegetation mapped. Undefined algae covered an area of 25 km^2 representing 16% of the total vegetation mapped in Western Port (Blake & Ball 2001b).

1.5.3. Corner Inlet/Nooramunga

Corner Inlet and Nooramunga together form a marine embayment of over 600 km^2, but represent two distinct physical environments. The Corner Inlet/Nooramunga system is generally shallow, with large areas of intertidal mud-sand flats dissected by an extensive network of deep channels and small gutters that drain the embayment through its five permanent entrances to Bass Strait (DCNR 1995).

Corner Inlet or Basin constitutes the western half of the system (Figure 1.1) and is a shallow embayment with extensive intertidal mud-sand flats. Corner Inlet is characterised by a network of tidal drainage systems which converge just inside the main entrance between Wilsons Promontory and Snake Island. Most of the channels are 3-10 m deep, becoming shallower in the northern and western areas of the inlet. Channels near the centre and entrance of the inlet are deeper, reaching depths of about 40 m (DNRE 1996).

The eastern half of the system is known as Nooramunga or Shoal Inlet and consists of numerous small sandy islands separated by shallow channels and intertidal sand-mud flats. Nooramunga is generally shallower than Corner Inlet with most channels being up to about 3 m deep. The deepest channels are near the ocean entrances and south of Sunday Island reaching depths of up to 30 m (DNRE 1996).

Tidal ranges in Corner Inlet/Nooramunga are amongst the highest in Victoria. The tides are semi-diurnal with pronounced fortnightly changes of spring to neap tides. Neap tidal range at Port Welshpool is generally 1-2 m but may be smaller. The spring tidal range, on the other hand, is commonly over 2 m and frequently over 2.5 m. Because of the large intertidal area and associated drainage within the inlets, tidal current velocities in the channels can be very high (DCLS 1980).

The Corner Inlet/Nooramunga region features the most extensive and pristine salt marsh and mangrove communities in Victoria (DNRE 1996).

Seagrass mapping in 1998 found that approximately 25% of Corner Inlet/Nooramunga was seagrass, with the majority occurring in Corner Inlet (Roob *et al*. 1998). Within Corner Inlet, Roob et al. (1998) found extensive beds of *Zostera/Heterozostera* dominated the nearshore intertidal flats and the flats north of Franklin Channel and south of Bennison Channel. The central area of the inlet between Franklin Channel and Bennison Channel was dominated by beds of *Posidonia australis,* while large mixed beds of *Zostera/Heterozostera* and *Halophila australis* were located east of Doughboy Island and between Stockyard Channel and Franklin River Channel. The seagrass in Nooramunga was dominated by *Zostera/Heterozostera* with only limited areas of *Posidonia* (Roob *et al*. 1998).

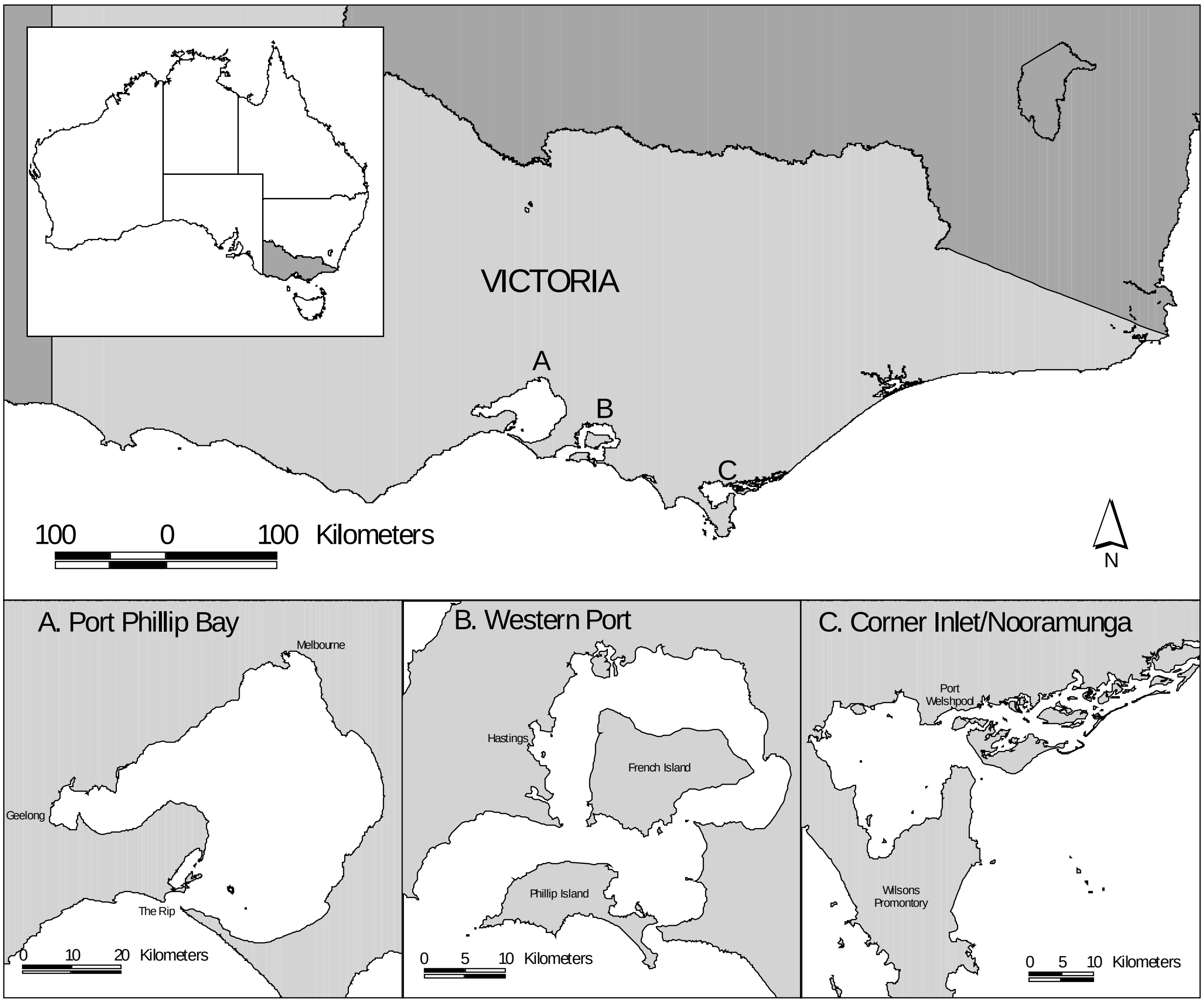


Figure 1.1. Location map

2. Geographic Information Systems and Fishery Habitat Suitability Modelling

2.1. GIS Applications in Fisheries Management

GIS technology has been widely used in the management of terrestrial environments but has until recently remained under-utilised within fisheries science (Isaak & Huber 1997). The slow rate of adoption of GIS in fisheries management can in part be attributed to socio-economic factors which have restricted available funding. Other factors, including the diversity of fisheries activities and the dynamic nature of marine environments, which cannot be readily accounted for in conventional GIS applications and mapping techniques (see below), have also contributed to the low levels of GIS applications by fisheries managers (Meaden 1999).

Meaden (1999) found that most of the early fisheries related GIS applications undertaken during the 1990's were in the field of marine habitat mapping and analysis. A range of other GIS applications began to emerge during the second half of the 1990's, including the mapping of fisheries catch and effort distributions (eg. Ball and Coots 2001) and matching this information to environmental or habitat parameters.

A further extension of GIS to fisheries research and management has been the recent development of approaches to model or simulate the relationships between species abundance and environmental parameters as a means of predicting species distributions in marine areas (Meaden, 1999). The Florida Marine Research Institute and National Oceanic and Atmospheric Administration Center for Coastal Monitoring and Assessment have been at the forefront of the development of this approach through its application of HSI modelling with a GIS to bays and estuaries along Florida's Gulf of Mexico coast (Christensen *et al*. 1997; Rubec *et al*. 1998; Rubec *et al*. 1999).

The development of web enabled GIS software and mapping applications has provided new opportunities for the management and distribution of spatial fisheries information. Australia is well advanced in the application of this new technology. Prominent local examples of web based GIS applications for accessing marine and fisheries data include the Australian Coastal Atlas (see: www.environment.gov.au/marine) managed by Environment Australia and the Coastal Habitat Resources Information System (CHRIS) for Queensland developed by Queensland's Department of Primary Industry with funding from FRDC (see: http://chrisweb.dpi.qld.gov.au/chris/welcome.htm).

2.2. Mapping Fisheries and Marine Environments

Traditional mapping has involved producing spatial representations of the distribution and location of static objects or areas in relation to other entities. This approach has been used extensively in terrestrial environments and is now being applied more widely to non-static objects including those found in the marine environment (see Roob 1999). For fisheries managers though, there is the added complexity in mapping caused by high time/space variability in fish distributions as well as the environment, which is also frequently the causal factor for species distributional change (Meaden & Do Chi 1996).

Mapping of fishery and marine habitats should be among the priority tasks when planning for fisheries management and it has been argued that this should not be postponed until "complete" information is available since redundancies or blanks in the information base will more readily appear during the process of elaboration (Caddy and Garcia 1986, cited in Meaden and Do Chi 1996).

Many coastal or marine features have fuzzy or transitory boundaries and this presents considerable problems in defining linear boundaries for features in order to represent them in a GIS. These difficulties can be partially resolved through mapping extreme, average or seasonal distributions and, where necessary, diurnal distributions (Meaden & Do Chi 1996).

Identification of species distributions and densities will typically be based on sampling methods that can potentially have a large margin of error as surveys can only gather samples from a fraction of the total environment. The margin for error will depend on the accuracy and frequency of the surveys, survey methods and also the behaviour and mobility of the target species (Meaden & Do Chi 1996).

2.3. History of Habitat Suitability Modelling

Habitat Suitability Index (HSI) modelling evolved from habitat evaluation procedures developed by the U.S. Fish and Wildlife Service (USFWS) during the early 1980's. The HSI concept centres around the assumption that the value or importance of a geographic area for a selected species can be defined by estimating a species' habitat requirements and quantifying habitat availability (Monaco and Christensen 1997). Development and use of the HSI models requires a clear understanding of the habitat requirements of the species being evaluated, the characteristics of the different types of HSI models, and the objectives of the study (Terrell *et al*. 1982).

The USFWS procedures required determining a numerical index of habitat suitability ranging from 0 (unsuitable habitat) to 1 (optimally suitable habitat) for each habitat variable selected for inclusion in the model (Terrell *et al.* 1982). A composite HSI model was then developed based on the assumption that a positive relationship exists between an index of habitat suitability and the carrying capacity of a habitat for a given species (Christensen *et al*. 1997). This approach allowed the value of a geographic area as a habitat for a particular species to be quantified, which could then be used in environmental impact assessment and mitigation of resource use conflicts (US Fish and Wildlife Service 1980). The use of Habitat Suitability Models as a management tool is now common in North America in terrestrial (Odum *et al.* 2001), freshwater (Thomas and Bovee 1993), estuarine (Christensen et al. 1997, Rubec et al. 1999), and marine environments (Soniat & Brody 1988; Gallaway *et al*. 1999; Norcross *et al*. 1999).

Specific HSI models applied to mainly freshwater fish species by the USFWS included:

1. Regression models that predicted a measurable response such as standing crop from environmental variables.

2. Descriptive models that assigned an HSI based on the presence or absence of specified levels of environmental variables as judged by the model developer.

3. Mechanistic models that described suitability index ratings for individual variables and aggregated those ratings into an HSI based on hypothesised causal relationships between habitat variable values and habitat suitability (Terrell *et al.* 1982).

Regression models were commonly used for resource planning in reservoirs, while the descriptive and mechanistic models were applied to a broader range of habitat types and conditions. Simple descriptive HSI models were seen to have the advantage of:

- providing a rapid means of comparing habitat conditions;

- are easily modified to meet project goals;

- have low demands for information and generally require few or no extensive field measurements; and

- can be utilised as low effort evaluation tools prior to the application of more detailed models (Terrell *et al.* 1982).

Descriptive models also enabled subjective decisions about optimal levels of habitat related variables to be quantified, but the oversimplified format may have disguised assumptions on ecological processes used to determine the index ratings (Terrell *et al.* 1982).

Mechanistic models provide a way to display and integrate a wide variety of assumed cause and effect relationships between variables when determining habitat suitability. Developing mechanistic models is dependent on firstly; the availability of information to identify a relationship between the model variable and a measure of habitat carrying capacity or quality for a species, and secondly; model variables must be quantifiable and have a measurable value under various habitat conditions. The measurable response by individuals or populations to changes in each model variable must then be converted to a suitability index for the variable. While the accuracy of a mechanistic model may be either low or unknown, its reliability may still be sufficient to use as a planning tool (Terrell *et al.* 1982).

2.3.1. Essential Fish Habitat Legislation in the USA

The Essential Fish Habitat (EFH) provisions of the amended Magnuson-Stevens Fishery Conservation and Management Act enacted by the US Congress in 1996 have been an important factor in the rapid expansion of GIS applications for fishery management in the USA. The Magnuson-Stevens Act defines EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (Section 3(10) Magnuson-Stevens Act).

Under the EFH provisions, Congress directed NOAA's National Marine Fisheries Service (NMFS) and the eight regional Fishery Management Councils to:

1. Describe EFH and identify EFH in each Fishery Management Plan,

2. Minimise to the extent practicable the adverse effects of fishing on EFH, and

3. Identify other actions to encourage the conservation and enhancement of EFH

In order to achieve its objectives, the EFH provisions specify that:

> "The general distribution and geographic limits of EFH for each life history stage should be presented in FMPs (Fishery Management Plans) in the form of maps. Ultimately, these data should be incorporated into a geographic information system (GIS) to facilitate analysis and presentation. These maps may be presented as fixed in time and space but they should encompass all appropriate temporal and spatial variability in the distribution of EFH. If the geographic boundaries of EFH change seasonally, annually, or decadally, these changing distributions should be represented in the maps. Different types of EFH should be identified on maps along with areas used by different life history stages of the species. The type of information used to identify EFH should be included in map legends, and more detailed and informative maps should be produced as more complete information about population responses (e.g., growth, survival, or reproductive rates) to habitat characteristics becomes available. Where the present distribution or stock size of a species or life history stage is different from the historical distribution or stock size, then maps of historical habitat boundaries should be included in the FMP, if known. The EFH maps are a means to visually present the EFH described in the FMP. If the maps and information in the description of EFH varies, the description is ultimately determinative of the limits of EFH."
> (Section 600.810.iii Magnuson-Stevens Act).

The introduction of mandatory requirements for regulatory authorities in the USA to identify the spatial extent of fishery habitats and record this information in a GIS has provided organisations such as NOAA and the Florida Marine Research Institute with the impetus to develop increasingly sophisticated methods of utilising GIS technology in fishery management. As a result, the development of HSI models has been adopted by organisations such as FMRI to provide essential fish habitat information (Rubec *et al*. 1998).

In Australia, the situation is very different. While the habitat suitability model approach has been explored in terrestrial environments (Akcakaya *et al*. 1995; Pearce & Ferrier 2001), there has been less emphasis on identifying and mapping essential habitat in a fisheries management framework to date. The Victorian Fisheries Act 1995, for example, provides a framework for the management of fishery resources through a system of fishing licences, seasonal closures, quotas, size and catch limits and fishery reserves. There are though, a number of international examples of applications of HSI models to aquatic systems (see Brown *et al.* 2000 for examples), and the potential to develop predictive habitat-based fisheries maps in areas with little fisheries information seems particularly relevant in Australia, where fisheries habitat data tends to be limited.

2.4. Fisheries Habitat Suitability Modelling

The National Ocean Service of the National Oceanic and Atmospheric Administration (NOAA) in collaboration with the Florida Marine Research Institute (FMRI) developed a simple spatial model using GIS technology that extends the basic HSI modelling approach developed by the USFWS (Rubec *et al.* 1999). The application of GIS technology enabled a spatial component to be incorporated in the HSI modelling process to produce views or maps of the relative suitability of locations for selected species in geographic space through time (Christensen *et al.* 1997).

The process of developing estuarine fish or "seascape" HSI models applied by NOAA and FMRI involved the following key steps:

1. Conduct a comprehensive data and literature search for the relevant species.

2. Convene an expert review process to select an appropriate set of environmental and biological variables to model (eg. salinity, temperature, depth).

3. Generate suitability index values to relate abundance of fish to environmental gradients.

4. Evaluate the efficacy of the derived suitability index values through either a qualitative literature review and/or a quantitative analysis of fisheries independent monitoring data (Coyne and Christensen 1997).

2.4.1. Environmental Variables

The HSI models developed by NOAA and FMRI involved selecting environmental variables considered sufficient to model suitability and infer potential distributions of estuarine species. The environmental variables used in the models have typically included:

- Water Temperature.
- Salinity.
- Substrate Type.
- Bathymetry.
- Dissolved Oxygen.
- Submerged Aquatic Vegetation (SAV ie. seagrasses).
- Emergent Wetland Macrophytes (ie. mangroves and salt marsh) (Christensen *et al.* 1997 - Figure 2.1).

2.4.2. Suitability Indices

Suitability index (SI) values are assigned according to a scale ranging from 0 (unsuitable habitat) to 1 (optimum habitat) (Monaco and Christensen 1997). Although interactions commonly occur between environmental variables, this model assumes their independence from one another.

Qualitative and quantitative techniques are available for producing SI values. Descriptive or qualitative models for assigning SI values typically involve an extensive literature search to identify any documented tolerances or affinities to gradients in each environmental variable. Expert knowledge and judgement is then required to determine SI increments across environmental variables based on the number of habitat associations that can be reliably identified. Where sufficient field sampling data is available, a quantitative mechanistic modelling approach can be applied to statistically define species-habitat associations through techniques such as ordination procedures, regressions or multinomial response curves (Christensen *et al.* 1997).

It is impractical and inappropriate to quantify the relationship between environmental gradients and species distributions without a robust dataset documenting relative abundance across the complete range of each environmental variable (Christensen *et al.* 1997). In the case of SI values derived from a qualitative descriptive model, this weighting or scaling variables cannot be done in a conventional manner. In developing a descriptive model for SI values for Pensacola Bay, Florida, Christensen *et al*. (1997) addressed this problem by placing each variable in a category of either "critical" or "non-critical" based on their potential effect on species distribution.

Christensen *et al*. (1997) defined "critical" variables as those exhibiting the potential to exclude a population if physiological tolerances are exceeded while "non-critical" variables were defined as those that have an effect on a species distribution, but alone will never exclude a population from utilising a particular habitat. "Critical" variables were assigned a SI value ranging from 0.0 to 1.0 and if any of these variables were scored as 0.0 for a species, the resulting HSI model would predict complete species exclusion ie. the HSI model will indicate unsuitable habitat regardless of the SI value for all other variables. "Non-critical" variables were assigned SI values of 0.2 to 1.0. By scaling the SI values in this way Christensen *et al*. (1997) were able to "weight" the environmental variables without using statistical techniques to quantify the relationships.

Where sufficient species sampling data and associated environmental variables are available quantitative techniques may be used to derive SI's. However, a combination of qualitative and quantitative techniques will usually be required to evaluate SI values as many fisheries independent monitoring datasets exhibit a disproportionate seasonal and geographic sampling effort. The results of NOAA's preliminary analyses indicated that while empirical data will always be necessary to monitor trends in distribution and analyses, qualitative data from scientific literature can provide a reasonable estimate of these measures for estuarine species across broad spatial and temporal scales (Coyne and Christensen 1997).

2.5. Review of Approaches to Suitability Index Modelling

A literature search identified 14 papers that addressed habitat suitability modelling for fisheries management in marine or estuarine environments (Table 2.1 and Table 2.2). Only habitat suitability models that have been tested with independent data tend to get published and this, coupled with the fact that many managers and biologists involved in the "front-line" of natural resource management may have little time to publish the results of their work, means that many of these models do not always appear in the primary literature (Brooks 1997). As a consequence, there are probably more habitat suitability modelling studies presented in the 'grey' literature, but due to difficulties in identifying these studies they are not included in the following review.

Apart from one Australian study, all of the identified studies were from North America. Thirteen of the studies deal with a single species or life history stage of a species and a total of 39 species are included. One of the studies (Williams & Bax 2001) considers habitat suitability at the assemblage level.

Table 2.1 and Table 2.2 summarise the use of habitat suitability models in estuarine and marine environments where habitat or environmental parameters can include a combination of physical, chemical and biological characteristics. There were a variety of approaches used to derive HSI's, with more than one approach typically investigated in each study (Coyne & Christensen 1997; Rubec *et al*. 1999). The methods presented depend to some extent on the amount of data available for the development of the habitat suitability model, but range from simple calculations of scaled indices to more complex classification or GLM/GAM statistical techniques. The methods of deriving habitat suitability models were either 'univariate' (Table 2.1), where the suitability of each habitat parameter was assessed separately and combined into a single index at the final stage of the procedure, or 'multivariate' (Table 2.2) where habitat parameters were assessed simultaneously in a statistical framework. We briefly review these approaches separately and have included a general discussion of issues relevant to both approaches.

2.5.1. Univariate 'Index-Based' Methods of Deriving Habitat Suitability Models

Eight studies present habitat suitability models derived with a univariate or index -based approach (Table 2.1). Three of these studies (Soniat & Brody 1988; Coyne & Christensen 1997; Williams & Bax 2001) also consider a multivariate approach and so appear in both Table 2.1 and Table 2.2.

Consulting a panel of experts coupled with a literature search and reaching some consensus on the important habitat parameters that drive species distributions can be used to develop a habitat suitability model (Christensen *et al*. 1997; Brown *et al*. 2000). The initial stage of the model development process is to use the literature to determine the range of a parameter within which a species occurs. A scale is then assigned to that habitat parameter (either qualitatively or by standardising quantitative data) that ranges from 0, which is considered unsuitable habitat, to 1, which is considered optimal habitat. Where there is a lack of detailed data for a habitat and species, broad categories are assigned to the habitat (e.g. presence/absence of vegetation) and the suitability index calculated accordingly (Christensen *et al.* 1997).

In the case of a finer resolution of data points, the construction of 'biologically relevant categories' of habitat has been recommended (Terrell *et al*. 1982) using cumulative frequencies and regression methods (Rubec *et al*. 1999). An alternative method, where there are relative abundance observations at fine-scale intervals along a gradient of an environmental covariate, is to fit a polynomial regression curve to the data. Predicted density values along the curve can be used to calculate relative suitability index values which can then be scaled accordingly (see Rubec *et al.* 1999 for an example).

It should be noted that the method chosen to derive the component suitability indices could have a considerable effect on the resulting predictions of habitat suitability (Rubec *et al.* 1999). The polynomial regression method was considered the best of these options by Rubec *et al.* (1999), although no mention is made of the increasing spread in the data at higher temperatures in the example they present. In other words, potentially important variability is ignored in these simple model approaches.

Another approach to defining categorical suitability indices is often termed the 'habitat affinity' of a species (Coyne & Christensen 1997; Monaco *et al.* 1998; Williams & Bax 2001). In this case, the presence or density of the species in each habitat class is compared to either the amount of that habitat available or to the amount of that habitat sampled. To quantify habitat affinities, a habitat affinity index (HAI) was developed by Coyne and Christensen (1997) whereby:

$$HAI = (p - r)/r \quad \text{if } p \leq r$$

or

$$HAI = (p - r)/(1 - r) \quad \text{if } p \geq r$$

Where p is the proportion of a species collected in a specific habitat and r is the proportion of area that the habitat comprises in the study area. The HAI has a centre point of 0 and is scaled so that an HAI of -1 corresponds to non-collection or complete avoidance of an area. An HAI of 0 indicates that fish displayed no habitat affinity and an HAI of +1 indicates an apparent exclusive affinity for a specific habitat zone or area. Negative values (other than -1) are used to define avoidance and are not equivalent to complete absence; a negative HAI value in the electivity context reflects a lesser concentration of a species in a particular habitat (Monaco *et al.* 1998). These HAI values can then be scaled from 0 to 1 for use as SI values in the HSI modelling (Coyne and Christensen 1997).

The habitat affinity approach is conceptually more appealing as it allows for any biases in the data caused by higher coverage of a certain habitat type or sampling strategy.

2.5.2. Scaling of Suitability Indices

There are several issues that need to be considered in relation to the scaling of the suitability indices to range between 0 and 1. While it is possible to get habitat that is totally unsuitable as described above, it is harder to be confident that habitat described as optimal in any one study area really is optimal. Genuinely optimal habitat values may not exist in the model development area (Thomas & Bovee 1993), or a history of local human impacts may serve to reduce habitat suitability. This may result in a model underestimating suitable habitat when transferred to other areas.

Where a model is developed for use in a specified area, Brooks (1997) suggested that unless the HSI scores ordinate across the entire range of 0 to 1 they will be of little use in differentiating between sites. He suggested that a calibration exercise should result in a re-scaling of the HSI scores to ensure that the complete range from 0–1 is represented in the area under study. Conversely, Vadas and Orth (2001) did not scale the suitability indices to range between 0 and 1, as such scaling will give each habitat parameter equal importance which is inappropriate as habitat parameters will often vary in their importance to fish (Brown *et al*. 2000). None of the studies reviewed here discuss the issue of calibration or the potential problems associated with the idea of optimal habitat. It seems that any decisions relating to the scaling and calibration of the indices should be dependent on the proposed applications of the model.

2.5.3. Combining Suitability Indices into a Final Habitat Suitability Model

The process of deriving indices for each habitat parameter is repeated for all the variables considered relevant to the model so that there are usually a number of 'suitability indices' that together make up the final habitat suitability model. Terrell *et al.* (1982) point out that including increasing numbers of model parameters does not necessarily improve the predictive power of the model and may even reduce it. They discuss an approach whereby it may be more appropriate to choose the most important habitat parameter and use only that parameter as a predictor of habitat suitability. All of the studies presented here used multiple predictors in the final model outputs (Table 2.1). The choice of predictors or habitat parameters included in the models does not always appear to be considered in much detail and may reflect available data (e.g. Gallaway *et al.* 1999) rather than variables identified as important from the literature review process. Dropping correlated or less important habitat parameters receives little attention in the simpler model building strategies, but see Brown *et al.* (2000).

There are a number of different methods used to calculate the final habitat suitability model or map for a given species (Table 2.1). The habitat suitability model effectively represents a conceptual model whereby environmental parameters are linked to the suitability of a site for the species by an additive, multiplicative or logical function (Burgman *et al*. 2001). The choice of aggregation method will have important assumptions associated with it and will affect the resulting assessment of suitable habitat distributions (Vadas & Orth 2001).

Three of the seven studies in Table 2.1 used a geometric mean that assumes that each environmental variable is equally important and acting independently (Rubec *et al.* 1999, but see Vadas and Orth 2001). The modelling approach adopted by Rubec *et al.* (1999) is a simple deterministic expression that calculates a dimensionless index of habitat quality for a given species. This model requires that a function relating a suitability index SI_i to an environmental or habitat variable X_i is derived for each ith factor as expressed in equation 1.

$$SI_i = f(X_i) \tag{1}$$

The process of determining SI values through qualitative or quantitative means is outlined above. A composite score termed the habitat suitability index (HSI) is then calculated as the geometric mean of SI_i scores for n environmental variables as shown in equation 2 and Figure 2.1.

$$HSI = [\prod_{i=1}^{n} (SI_i)]^{(1/n)} \tag{2}$$

The resulting HSI values can then be grouped into classes from unsuitable to optimum habitat to predict zones of habitat suitability across a bay or estuary (Rubec *et al.* 1999). The geometric mean HSI method assumes that each environmental variable is equally important, that SI functions across environmental gradients are independent, that environmental associations of a species life stage are constant during the time period modelled, and that species distributions are independent across seasons (Rubec *et al.* 1999). In applying this technique at two sites in Florida, Rubec *et al.* (1999) noted that HSI models which treat factors as being independent may be adequate to predict spatial distributions but probably cannot be used to predict actual abundance.

Importantly, the geometric mean method will give a final value of zero when any of the component indices are zero (Brown *et al.* 2000). In other words, there is no compensation for an unsuitable habitat parameter by other more suitable habitat parameters. This method is particularly relevant in environments where 'critical habitat' has been identified (e.g. Christensen *et al*. 1997). For example, a species may not be able to survive at salinities below 20 ppt so salinities below this value are scaled to zero, that is, unsuitable habitat. When aggregated with the other habitat parameters, the habitat is still considered unsuitable if the salinity falls below 20 ppt regardless of how suitable the other habitat parameters may be. In environments where critical habitat has not been identified, an arithmetic mean may be more appropriate.

Three out of the seven studies presented in Table 2.1 used more complex aggregative equations (Brown & Hartwick 1988; Soniat & Brody 1988; Gallaway *et al*. 1999) where parameters considered more important are given more weight in the calculation of the final habitat suitability model. Of these, only the study by Brown and Hartwick (1988) really describes the rationale behind the method used to create the composite index. Without such a description it is impossible to critically evaluate the usefulness of the HSI and an additional component of structural uncertainty is added to the model (Burgman *et al*. 2001).

2.5.4. Multivariate Approaches to Habitat Suitability Modelling

A multivariate approach to habitat modelling has the distinct advantage that several parameters can be modelled simultaneously and interactions between environmental variables can be incorporated into the model. For the regression techniques, polynomials can also be included allowing quadratic, cubic etc response curves to be modelled.

Nine studies used some form of multivariate statistical analysis to derive habitat suitability models (Table 2.2). The majority of these involved regression techniques, with four studies using multiple linear regression and two models derived using presence/absence data in a logistic regression approach (Norcross *et al*. 1999; Stoner *et al*. 2001). The other main approach was a classification approach such as discriminant analysis (Norcross *et al*. 1995; Norcross *et al*. 1997) or tree based regression models (Norcross *et al*. 1997; Norcross *et al*. 1999). Williams and Bax (2001) used non-metric multi-dimensional scaling, cluster analysis and an analysis of similarities to investigate assemblage level data in their studies. All of these methods provide more rigorous, scientifically defensible models providing adequate data is available for their construction and the assumptions of the statistical tests are met.

The use of multiple linear regression has the advantage that it uses abundance or density data to construct the model rather than just presence/absence data, but it also has more restrictive assumptions of normality and homogeneity of variance than logistic regressions. While the use of presence/absence data in a logistic regression uses less detailed information than the linear regression, it is considered a robust, readily available statistical technique that gives easily interpretable results (Norcross *et al*. 1999). The results from these are also easily translated to habitat probability maps (Gregr & Trites 2001) which illustrate the probability of encountering the species in question for each combination of habitat (Odum *et al*. 2001). The probability outputs are usually considered as surrogate measures for abundance, so that a high probability of occurrence in a habitat type is expected to correspond to a high abundance of the species in question (Stoner *et al*. 2001). In a terrestrial environment, Pearce and Ferrier (2001) found that logistic regression models performed as consistently as those based on a relative index of abundance, as did models derived from actual abundance data.

Norcross *et al.* (1995, 1997 & 1999) considered a number of different approaches in their series of publications on critical flatfish nursery habitats including discriminant and regression tree analysis. The discriminant analysis is mathematically equivalent to a multivariate analysis of variance (Tabachnick & Fidell 1989) and as a consequence has the same limiting assumptions which are hard to test (Quinn and Keough 2002).

The regression tree approach, like a logistic regression approach, allows the use of both continuous and categorical predictor variables and progressively splits stations on the basis of their values for one of the predictor variables until a leaf or terminal node is reached (Norcross *et al*. 1997). These types of models use precise values for the continuous parameters of the model, creating branches based on a threshold (e.g. temperature ≤8.9°C – Norcross *et al.* 1999) which may not be appropriate to the quality of the input data (Debeljak *et al*. 2001). The development of a number of models using different methodologies allows a degree of confidence in the model outputs if they all produce similar predictors (Norcross *et al*. 1999).

The study by Williams and Bax (2001) differed from others in that it was an assemblage, as opposed to a population, level approach. The aims of this study were broader in scale than most of the others reviewed here and the outcomes included defining management areas on a scale of inner and outer shelf.

However, the community approach combined with the univariate 'habitat associations' (or affinities) also allowed them to make predictions regarding the types of fish communities that will be present in broad habitat classes. While this type of multivariate approach to assemblage level data is extremely useful as an exploratory and pattern-finding technique, it seems less amenable to the creation of output data that can be spatially modelled in a GIS application.

2.5.5. Measures of Uncertainty

Testing and validation of models is an issue that relates to both the univariate and multivariate methods of model derivation and is discussed in the following section.

Because these simpler models can be derived from a variety of data types and sources, an important aspect of model usage should be the addition of a measure of uncertainty to the final index scores (Bender *et al*. 1996; Burgman *et al*. 2001). Williams and Bax (2001) provide a subjective measure of confidence in their fish association scores based on abundance of the fish species, consistency across gear types and information derived from the literature. None of the other studies presented in Table 2.1 addressed this issue formally.

Variability can, however, be represented by confidence intervals around final index values by using Monte Carlo or bootstrap techniques (Bender et al. 1996). Alternatively, Burgman *et al.* (2001) describe the use of fuzzy numbers to estimate uncertainty in the calculation of habitat suitability indices. The incorporation of uncertainty may affect the management decisions that relate to the model (Burgman *et al*. 2001) and to the conclusions resulting from model testing and validation (Bender *et al.* 1996). While it may not be possible to represent a measure of variability or uncertainty in the data within the mapping techniques, a more detailed examination of a particular area or management strategy could, and should, examine sources of variability and confidence in the model.

2.5.6. Testing and Validation of Models

Testing and validation of habitat suitability models is an important issue that has received attention recently (Thomas & Bovee 1993; Bender *et al*. 1996; Brooks 1997; Roloff & Kernohan 1999; Burgman *et al*. 2001). Most published work on habitat suitability models presents validated or tested models (Roloff and Kernohan 1999) so most of the models reviewed here have been validated or tested in some way (Table 2.1 & Table 2.2).

The validation procedures ranged from consulting an expert review panel (Christensen *et al*. 1997) to utilising commercial fisheries data (Rubec *et al*. 1999) and independent datasets from the same or different localities (Coyne & Christensen 1997; Norcross *et al*. 1997; Clark *et al*. 1999; Brown *et al*. 2000; Stoner *et al*. 2001). Results of the validation process in these studies were variable. Some studies found the habitat suitability models performed well (e.g. Coyne and Christensen 1997, Rubec et al. 1999), while others considered the fit of the models to be dependent on the species or localities (Clark *et al*. 1999), the prediction of presence versus absence (Norcross et al. 1997), or the life-history stage of the species (Stoner *et al*. 2001).

Stoner *et al.* (2001) point out that if there are no strong patterns in the data, models can be misleading in that they can give an impression of more definitive and quantifiable patterns that may not really exist. It follows, that a habitat suitability model is only as good as the data from which it is derived, and Clark *et al.* (1999) emphasise that modelling efforts should be based on large and comprehensive density databases. It is also possible that the models do not measure an important component of a habitat that may exist at a different scale (Norcross *et al.* 1997). Biotic interactions, particularly predator and prey abundances, are also usually ignored, mainly because the extensive survey information required to incorporate this information would make the application of the model to other areas unfeasible (Norcross *et al.* 1997).

Brown *et al.* (2000) discuss some possible limitations of datasets used in model validation. Test datasets tend to consist of point data (that is, a number of sites sampled at a particular time) whereas habitat suitability models are typically intended to be broader characterisations of long-term average conditions (Brown *et al.* 2000). The movement of species in and out of a specific habitat may also mean that a species is not encountered in a limited sampling program. In addition, if the validation dataset included atypical environmental conditions or happened to coincide with a pollution event, then the data may not be representative of more typical or average conditions. Brown *et al.* (2000) also found that validation datasets may not cover the complete range of available habitats. This will be particularly important when commercial fisheries data is used for model testing because this data will tend to focus on areas where abundance is known to be high. As a consequence, the full range of the habitat model cannot be tested because the areas omitted from the data will presumably have both low abundance and low HSI values.

The resultant index of habitat suitability for a particular species (however it is calculated) is assumed to have a positive linear relationship with the carrying capacity of that habitat for that particular species (Terrell 1984). Some measure of density is used as a surrogate measure for carrying capacity (eg Rubec *et al.* 1999) although as Pulliam (2000) points out, species can often be absent from suitable habitat and present in unsuitable habitat due to behavioural, competitive or dispersal dynamics. Where a species is present in numbers below the carrying capacity, it may not occupy all available suitable habitats and similarly in periods of high abundance, a species may occupy low quality habitats that would not normally be utilised (Brown *et al.* 2000). This will obviously affect the predictive power of a habitat suitability model.

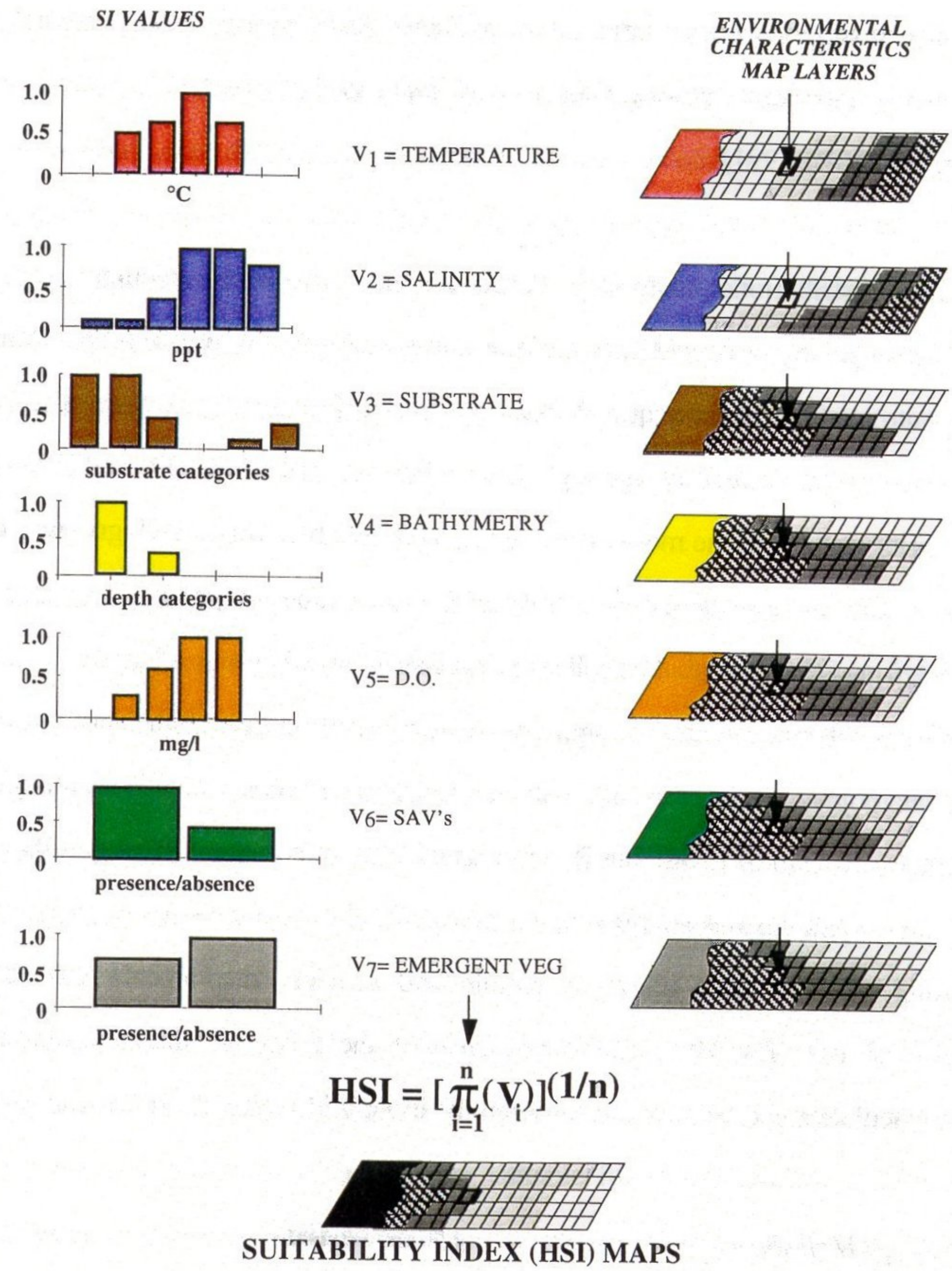


Figure 2.1. Grid-based habitat suitability modelling with a raster GIS (after Christensen *et al.* 1997). In this study, the above categories of substrate, SAV (submerged aquatic vegetation) and emergent vegetation are defined within a single substrate type/biota category.

Table 2.1. Summary of studies that use a 'univariate' analyses in Habitat Suitability modelling (* included in both Table 2.1 & Table 2.2).

Study	Objectives	Species investigated	Habitat Variables investigated	Data Sources	Approach	Outcomes	Validation
Soniat and Brody 1988 *	To test previously developed HSI for oysters in the Gulf of Mexico and see if correlated with oyster densities in new area and modify HSI accordingly	American oyster *Crassostrea virginica*	Temp, salinity, frequency of killing floods, substrate firmness, abundance of oyster drills, disease intensity.	Sampled reef and non-reef sites in Galveston Bay.	No information on how calculated S.I values. Different method of calculating HSI composite index but no explanation given as to why method chosen. Multiple linear regression used to create modified HSI	Modified Gulf of Mexico HSI model that expected to perform better in Galveston Bay.	No independent validation of modified model.
Brown and Hartwick 1988	To develop and evaluate HSI model for year 0 and year 1 Pacific oysters	Pacific oyster *Crassostrea gigas*. Year 0 and year 1	Temp. food, salinity, O_2, Ph, suspended solids, water movement, disease, fouling organisms	Literature search, Relevant literature used for model development referenced.	Suitability curves derived from literature review. Combined into a HSI with components receiving different weighting.	HSI model that predicted suitability of coastal areas for oyster culture	Model evaluated using independent data at 10 sites. Indicated that model worked well.
Christen-sen *et al.* 1997	To develop HSI model with emphasis on freshwater flows	Eastern oyster *C. virginica*, white shrimp, *Penaeus setiferus*, spotted seatrout, *Cynoscion nebulosus*	Temp, salinity, substrate, depth, dissolved oxygen, SAV, emergent vegetation	Literature search Expert opinion	Presence/absence data used for range finding S.I. values assigned to parameters S.I. combined to HSI using geometric mean	Map of HSI to look at effects of changes in freshwater flow.	Validation mainly by experts. Only independent data for oyster distributions. Models considered to be reasonable representations.
Coyne and Christen-sen 1997 *	To generate discussion about Suitability Indices for HSI modelling. Compared qualitative and quantitative models.	Pinfish *Lagodon rhomboides*	Salinity, temperature dissolved oxygen, depth	Literature search Independent data	1. Cumulative frequency graphs used to delimit biologically relevant ranges. Slopes from graphs used for S.I. values. 2. Habitat Affinity Indices – % spp. caught in habitat standardised by availability of habitat.	A discussion of univariate methods to determine suitability indices	Models not far enough along to validate
Gallaway *et al.* 1999	To map habitat utilisation patterns of juvenile red snapper	Red snapper *Lutjanus campechanus* (juveniles)	Depth, temp, salinity, dissolved oxygen, density of trawl 'hang sites', density of petroleum platforms	SEAMAP survey data. Standard methods used in stratified design. Very good coverage of area	Seasonal CPUE data used to define 'high use' habitat etc using quartiles. Scaled SI derived for each habitat by comparing catch with available habitat. Suitability indices composited into HSI – no rationale given.	Seasonal maps produced of HSI values. Summer hypoxia events change map pattern considerably	No independent testing or validation of the model

Table 2.1 continued

Study	Objectives	Species investigated	Habitat Variables investigated	Data Sources	Approach	Outcomes	Validation
Rubec *et al.* 1999	To develop HSI index for a single species in one estuary and test its applicability in nearby estuary.	Spotted seatrout *Cynoscion nebulosus* (juveniles)	Salinity, temp, depth, bottom type (sediment type +/- SAV)	Data included catch using 6 gear types with same mesh size (CPUE standardised by gear efficiency)	3 methods used to create Suitability Indices: cumulative frequency method (presence/absence data); range mean method (above standardised by CPUE data); smooth mean method (CPUE data at fine-scale intervals along gradient)	SIs combined using geometric mean and mapped on GIS. Choice of method for S.I. calculation appears important- polynomial regression considered best.	Validation using independent fisheries data. HSI maps seemed to transfer well to nearby estuary.
Brown *et al.* 2000	To generate habitat distribution maps for selected taxa and identify important habitat in the bays	Alewives, American sand lances, Atlantic salmon, Atlantic tomcods, mummichogs, winter flounder, American lobsters, softshell clams	Temperature, salinity, depth and substrate	Literature review Expert opinion	Suitability indices composited using unweighted geometric means. Models run and evaluated by panel of experts and changed where recommended	Maps of predicted habitat distributions based on models and associated management recommendations	Independent CPUE data used to validate models. Models for well known species with strong affinities for the bottom appeared to perform better than pelagic species.
Williams and Bax 2001 *	To identify associations of fish communities with substratum type.	Species associations for 61 taxa	Depth, substrate type, season.	Multiple gears used to sample 6 areas of mixed substrate type.	A degree of association was defined by comparing the numbers of individuals caught in soft substrate or reef substrates and standardising by the number of samples taken	Associations with sediment type classified as strong, distinct or both and a measure of confidence supplied	No independent testing or validation of predictions.

Table 2.2: Summary of studies that use a 'multivariate' analysis in Habitat Suitability modelling (* included in both Table 2.1 & Table 2.2).

Study	Objectives	Species investigated	Variables investigated	Data Sources	Approach	Outcomes	Validation
Soniat and Brody 1988 *	To test previously developed HSI for oysters in the Gulf of Mexico and see if correlated with oyster densities in new area and modify HSI accordingly.	American oyster *Crassostrea virginica*	Temp. salinity, frequency of killing floods, substrate firmness, abundance of oyster drills, disease intensity.	Sampled reef and non-reef sites in Galveston Bay.	Multiple linear regression used to create modified HSI	Modified Gulf of Mexico HSI model that expected to perform better in Galveston Bay.	No independent validation of modified model.
Coyne and Christen-sen 1997 *	To generate discussion about Suitability Indices for HSI modelling. Compared qualitative and quantitative models.	American oyster *Crassostrea virginica.*	Salinity, temperature dissolved oxygen, depth.	Literature search Independent data.	Multiple linear and polynomial regression techniques.	Quantitative model (multivariate) compared to 'univariate' model approach developed in Soniat and Brody (1988).	Used Texas data to develop models and tested them in Pensacola Bay. Considered both model approaches to perform well but regression models performed particularly well.
Norcross *et al.* 1995	To characterise nursery areas for the 4 most abundant species of juvenile flatfish collected in 6 bays and straits.	Flathead sole *Hippoglossoides elassdon* (age 0), Pacific halibut *Hippoglossus stenolepis* (age 0), yellowfin sole *Pleuronectes asper* (age 1) & rock sole *P. bilineatus* (age 0).	Substrate type, depth, bottom temp, bottom salinity and distance from mouth of nearest bay.	104 stations stratified by depth then randomly chosen.	Descriptive models based on non-parametric correlation coefficients and presence absence data used in discriminate analysis to get 2 best predictor variables.	General descriptions of habitat characteristics for nursery areas considered hypotheses for further testing.	See following studies.
Norcross *et al.* 1997	Repeat linear discriminate analysis with extra data to include wider geographical area. Refined previous habitat models using tree-based regression models on CPUE data	Flathead sole *Hippoglossoides elassdon* (age 0), Pacific halibut *H. stenolepis* (age 0), yellowfin sole *Pleuronectes asper* (age 1) & rock sole *P. bilineatus* (age 0).	Substrate type, depth, bottom temp, bottom salinity, and distance from mouth of nearest bay.	52 extra samples to previous survey. Shallow waters sampled from skiff, deeper waters from trawler.	Spearmans rank correlations, CPUE data (log +1) used in regression trees and discriminate analysis using presence/absence data and extra data.	Depth, temp, substrate type and distance from bay mouth all important predictors. Regression trees of CPUE data generally agree with results of discriminate analysis.	Independent data used to validate model. Absence was less well predicted than presence which they attributed to microhabitat features not measured – predator and prey abundance may also be important.

Table 2.2 continued

Study	Objectives	Species investigated	Variables investigated	Data Sources	Approach	Outcomes	Validation
Norcross *et al.* 1999	To facilitate the identification of critical flatfish nursery habitats in Alaskan waters for 2 age classes of different species of flatfish.	Flathead sole *Hippoglossoides elassdon* (age 0), Pacific halibut *Hippoglossus stenolepis* (age 0), yellowfin sole *Pleuronectes asper* (age 1) & rock sole *P. bilineatus* (age 0).	Depth, sed grain size, bottom temp, bottom salinity, distance to mouth of bay, distance from shore, bay type.	Sampled 5 bays and 85 stations using standard method.	Presence/absence data used in logistic regressions to validate models from discriminant analysis and CART models.	Where data available on sed type, depth, temp, within bay distance or bay aspect ratio then logistic equations can be applied to determine probable locations and areal extent of nurseries for target fish.	3 approaches gave similar results. Discusses advantages & disadvantages of each. Conclude advantage of logistic models is they are readily available stats techniques that give clearly interpretable results.
Clark *et al.* 1999	To develop models and combine with a GIS to provide spatial mosaic of potential Essential Fisheries Habitat	Brown shrimp *Farfantepenaeus aztecus*, white shrimp *Litopenaeus setiferus*, and pinfish *Lagodon rhomboides.*	Habitats (marsh edge, SAV, shallow non-vegetated bottom), season, salinity.	Data from a variety of studies in region that all used same drop-trap sampling method.	Stepwise multiple linear regression used to identify predictors of density. Used temporal subsets where densities were greatest.	Prediction formulae applied to digital habitat geographies in GIS – density estimates classified into 5 equal quantiles and mapped as predicted density classes.	Validation using data from other bays collected in same way as original data plus otter trawl data. Fit of models was variable.
Stoner *et al.* 2001	To test whether there are significant size-related changes in habitat use for winter flounder.	Winter flounder *Pseudopleuronectes americanus* (age 0).	Depth, temp, organic content, salinity, prey abundance, distance from western most station , distance from across bay.	2 years of seasonal surveys at 84 stations. Emphasis on high replication of habitat types and good spatial coverage.	Presence/absence data in Generalised Additive Model (non-parametric generalisation of logistic regression) in multiple regression.	Predicted probabilities of flounder occurrence mapped. Discussion of limitations and significance of spatially explicit habitat models.	Validated on independent dataset where assumed that areas with high probability of capture should yield highest annual capture rates.
Gregr and Trites 2001	To identify coastal regions that may be regarded as critical habitat for several whale species.	Sperm whale *Physeter macrocephalus*, Sei whale *Balaenoptera borealis*, fin whale *B. physalus*, blue whale *B. musculus* & humpback whale *Megaptera novaeangliae.*	Depth, slope, depth class, salinity , temp, month.	Historic whale catch records 1948 – 1967.	Count data used in multiple generalised linear regressions model (Poisson distribution) – then simple transform used to change predicted number of whales into probabilities.	Habitat probability maps depicting the combination of predictors that best describe where the whales found in region – can be considered maps of 'suitable' or possibly 'critical' habitat.	Divided data into two and used half for fitting models and half for testing model predictions. Concern over autocorrelation between cells affecting model testing method.

Table 2.2 continued

Study	Objectives	Species investigated	Variables investigated	Data Sources	Approach	Outcomes	Validation
Williams and Bax 2001* Austral-ian Study	To define broadscale community structure of demersal fish and to identify associations of fish communities with substratum type.	Assemblage level information.	Depth, substrate type (soft, hard bottom), season.	1.Demersal trawl samples at 5 depths on 7 across shelf transects 2. multiple gears used to sample 6 areas of mixed substrate type.	Multivariate analyses including cluster, nMDS, ANOSIM and SIMPER to investigate community structure. Plus (using gill net data) considered number of individuals caught in particular substratum (hard or soft) with number of samples taken to get degree of association for 61 most abundant taxa.	Multivariate analyses and association data indicated distinct fish communities distributed in specific depths, latitude and longitude and substrate type. Recommend broadscale management directed at 3 zones – inner shelf, outer shelf and shelf break.	No independent testing or validation of habitat predictions.

3. Marine Spatial Data

The process of HSI modelling with a GIS requires information to enable identification of species suitability indices and also spatial data for each environmental variable in a GIS format. A review of marine environmental data for Victorian bays and inlets identified GIS data for most of the environmental variables required for the HSI modelling process. This GIS data was in a vector format (points, lines and polygons) and needed to be converted to a raster format with consistent grid cells across the layers for each bay/inlet.

3.1. Raster GIS

Geographic features can be represented in a GIS in either a vector or raster format (Figure 3.1). Vector data represents features as x,y coordinates in the form of single points or a series of points representing lines. Area feature, such as a rocky reef are represented by polygons which consist of a line connected to itself defining an enclosed area (Figure 3.1). A raster is a two dimensional matrix of cells (pixels) (Figure 3.1). The height and width of each cell in a raster are fixed and have the same size, with each cell representing a defined area of the earth such as a square metre or kilometre. Each raster cell has a value that represents the characteristics of the theme at that position (eg. depth, salinity) (Zeiler 1999). The application of a raster GIS rather than a vector GIS to marine environments has the advantage of moving from a system of linear boundaries and points to a cell based system in which the presence/absence or density of fish species and habitats can be identified spatially (Meaden & Do Chi 1996).

A raster GIS also enables mathematical operators (arithmetic, Boolean, relational etc.) to be applied to overlaying digital rasters to produce an output raster (Zeiler 1999). The application of a raster GIS is therefore well suited to HSI modelling which requires the creation of digital rasters for each environmental variable and species suitability index as well as the calculation of combined output habitat suitability rasters through an application of the HSI model to the individual input rasters (see Figure 2.1).

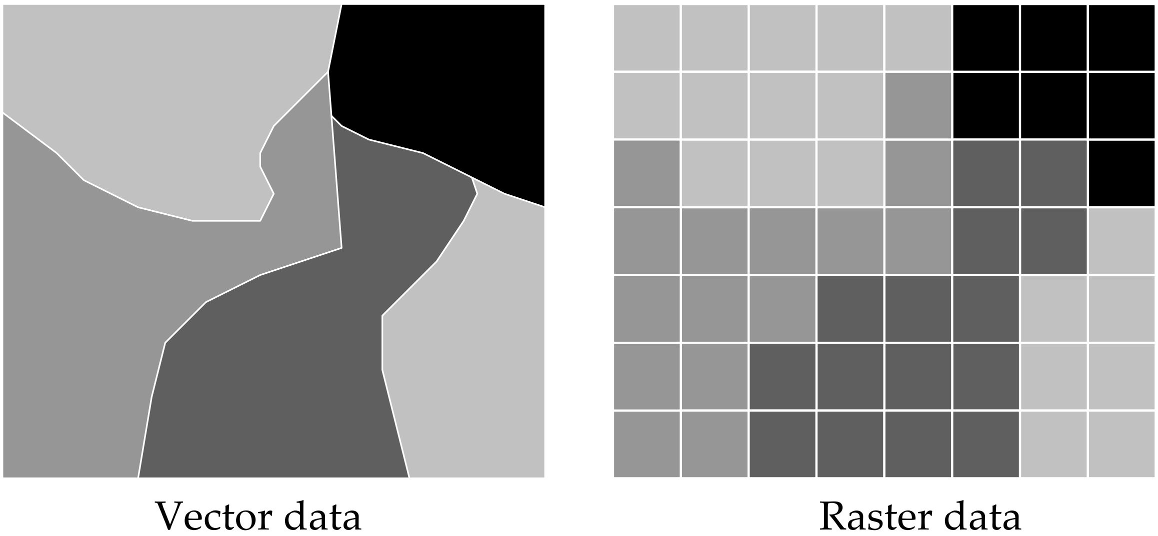


Figure 3.1. Representation of geographic features in a vector polygon versus raster GIS format.

3.2. Habitat and Environmental Parameter Grids

The key spatial data for the three sites was depth and substrate type/biota. Other layers only available for Port Phillip Bay were seabed sediments and salinity variability. Each grid was generated at a standard cell size of 10 m. The spatial grids or rasters used in the HSI modelling are summarised in Table 3.1 and outlined below.

Table 3.1. Spatial grids (rasters) incorporated in HSI modelling

	Bay/Inlet		
Spatial Grid	**Port Phillip Bay**	**Western Port**	**Corner Inlet/ Nooramunga**
Depth	PP_DEPTH	WP_DEPTH	CI_DEPTH
Substrate type/biota	PPHABGD3	WP_HAB2	CI_HAB2
Sediment	PPINSED1	NA	NA
Salinity variability	PPEPAGD	NA	NA

3.2.1. Depth

Depth grids or Digital Elevation Models (DEMs) for Port Phillip and Western Port were created by interpolating depth points sourced from the Port of Melbourne Corporation (formerly Victorian Channels Authority) with a kriging method in Surfer™. The resulting depth grids were then imported to ArcView Spatial Analyst. An existing depth polygon layer for Corner Inlet/Nooramunga was directly converted to a depth grid within Spatial Analyst. The depths in all layers were re-classified to depth zones of intertidal, 0-2 m, 2-5 m, 5-10 m, 10-15 m, 15-20 m, 20-30 m, >30 m.

The depth grids for Port Phillip, Western Port and Corner Inlet/Nooramunga are shown in Figure 3.2, Figure 3.5 and Figure 3.7 respectively.

3.2.2. Substrate Type/Biota

The dominant broad substrate types and biota at the three sites examined in this study were bare sediment, seagrass and macroalgae (attached to both soft sediment and reef), with rocky reef also being present. Spatial data for these substrate types/biota were available from a seagrass mapping program undertaken by PIRVic in the late 1990's (Roob *et al*. 1998; Blake & Ball 2001b; Blake & Ball 2001a). The substrate types and biota categories in the original mapping were grouped or simplified for the HSI modelling (Table 3.2).

Seagrass species and densities vary across the three bays with the four principal species being *Zostera muelleri, Heterozostera tasmanica, Halophila australis* and *Amphibolis antarctica* (LCC 1993). A further species, *Posidonia australis* is only found in Corner Inlet where it occurs as extensive beds on submerged banks (Roob *et al*. 1998). The two species of Zosteraceae; *Heterozostera tasmanica* and *Zostera muelleri*, could not be differentiated by the remote sensing techniques employed in the mapping studies and were grouped into a single category of "*Zostera/Heterozostera*". All species were typically found in sheltered environments with the exception of *A. antarctica,* which is adapted to higher energy environments and was found in the exposed entrances to Western Port and Port Phillip.

Seagrass is a valuable habitat for many fish species including the juvenile stages of many commercially significant species (Edgar *et al*. 1993; Jenkins *et al*. 1993; Jenkins & Sutherland 1997). Jenkins *et al.* (1997) compared fish assemblages of seagrass habitats with fish assemblages of unvegetated habitats in different areas of Victoria and found that diversity was higher in seagrass meadows than in unvegetated areas. While some studies have addressed variations in usage of seagrass species and densities by different fish species (Hyndes *et al*. 2003), we had insufficient data to make this differentiation for the HSI modelling.

As a result, we grouped all the seagrass species and densities into a single seagrass category (see Table 3.2), although *A. antarctica* was retained as a separate category reflecting its preference for higher wave energy environments.

Previous studies have noted the importance of bare sand at the edge of seagrass beds (Ferrell & Bell 1991; Jenkins *et al*. 1997b). Ferrell and Bell (1991) investigated diversity and abundance of fish in seagrass (*Zostera capricorni*) and bare sand (within 10 m and >100 m of seagrass). Their study found that the seagrass supported more species, individuals and different species than bare sand >100 m from seagrass at most of their sites. However, fish assemblages from bare sand adjacent to the seagrass (within 10 m) were more similar to the seagrass than those at the distant bare sand. Interestingly 4 of the 7 common species associated with sand adjacent the seagrass were of commercial or recreational significance.

In order to account for the potential importance of bare areas adjacent to seagrass beds, we created an additional category of seagrass bare edge for Port Phillip Bay, which represented bare sand within 15 m of the edge of seagrass beds. Similarly, a mangrove edge category was also created to enable possible differences in habitat usage of those areas immediately adjacent to mangroves to be differentiated in Western Port and Corner Inlet/Nooramunga.

3.2.3. Seabed Sediments

Data on seabed sediment distribution that was suitable for habitat suitability modelling was only available for Port Phillip Bay. The seabed sediment layer used in this study was derived from sediment sampling and analysis undertaken by the Port of Melbourne Authority between 1969 and 1987 (PMA 1987). The PMA used grain size analysis of samples taken throughout the bay to produce a summary distribution map of seabed sediments in Port Phillip Bay (PMA 1987).

We had limited data to define habitat affinities of fish species to sediment types, so for the purposes of this study the sediment types were simplified to six classes (Table 3.3). The distribution of these classes in Port Phillip is shown in Figure 3.4.

Sediment studies in Western Port during the 1970's (Marsden & Mallett 1974; Donaldson & Marsden 1977; Harris *et al*. 1979) define sediment distribution maps but the scale of the hard copy maps presented in these reports was too coarse to digitise for habitat suitability modelling. Sediment samples collected by Donaldson and Marsden (1977) could have potentially been used to interpolate a sediment layer, but we could not locate either the digital database or hard copy datasheets from 1977.

No seabed sediment distribution maps or data were identified for Corner Inlet/Nooramunga.

Table 3.2. Marine Substrate Type/Biota Classification for Victorian Bays and Inlets.

Substrate type/biota categories used in HSI modelling	Detailed categories in source substrate type/biota mapping
Seagrass	Sparse, Medium and Dense *Zostera/Heterozostera*
	Sparse, Medium and Dense *Zostera/Posidonia* mix
	Sparse, Medium and Dense *Zostera/Heterozostera* & *Halophila* Mix
	Sparse, Medium and Dense *Halophila*
	Zostera & *Heterozostera/Posidonia/Halophila* mix
Seagrass - *Posidonia*	Sparse, Medium and Dense *Posidonia*
	Sparse, Medium and Dense *Posidonia/Halophila* mix
Seagrass & Macroalgae	Sparse, Medium and Dense *Zostera/Heterozostera* & Filamentous Algae Mix
	Sparse, Medium and Dense *Zostera/Heterozostera* & Undefined Macroalgae Mix
	Sparse, Medium and Dense *Zostera/Heterozostera* & *Caulerpa* Mix
	Sparse, Medium and Dense *Halophila* & Macroalgae Mix
Seagrass Bare Edge	Seagrass-Bare (all bare areas in 15 m buffer around seagrass) Port Phillip Bay only
Amphibolis	Sparse, Medium and Dense *Amphibolis*
	Amphibolis & *Zostera/Heterozostera* mix
Amphibolis & Macroalgae	Sparse, Medium and Dense *Amphibolis* & Undefined Macroalgae Mix
	Amphibolis & *Zostera/Heterozostera* mix with algae
Macroalgae on sediment or Macroalgae on reef	Undefined Macroalgae
	Sparse, Medium and Dense *Caulerpa* Dominant Macroalgae
	Sparse, Medium and Dense *Ulva* & *Caulerpa* Dominant Macroalgae
	Sparse, Medium and Dense *Codium* Dominant Macroalgae
	Phyllospora/Ecklonia Dominant Macroalgae
Drift Algae	Sparse, Medium and Dense Drift Algae
Pyura (Cunjevoi)	Sparse, Medium and Dense *Pyura* (generally *Pyura stolonifera*)
Pyura (Cunjevoi) & Macroalgae	Sparse, Medium and Dense *Pyura* (generally *Pyura stolonifera*) & Undefined Macroalgae Mix
Rocky Reef	Intertidal Rocky Reef - Shore Platform
	Subtidal Rocky Reef – high and low profile reef
Bare Sediment	No Visible Bottom
	Bare Sediment/Bare Subtidal Sediment
	Bare Intertidal Sediment (Sand, Mud-Sand and Sand-Silt Clay Flat)
Mangroves	Mangroves *Avicennia marina*
Mangrove Edge – Bare Sediment	Mangrove Edge – Bare Intertidal Sand-Silt-Clay Flat
	Mangrove Edge – Bare Subtidal Sediment
Mangrove Edge - Seagrass	Mangrove Edge – *Zostera/Heterozostera* Dominant Seagrass
	Mangrove Edge – *Zostera/Heterozostera* Dominant Seagrass & Macroalgae
Mangrove Edge - *Amphibolis*	Mangrove Edge – *Amphibolis* Dominant Seagrass
	Mangrove Edge – *Amphibolis* Dominant Seagrass & Macroalgae
Mangrove Edge - Macroalgae	Macroalgae Edge - Macroalgae
Channel – major	Major channels through intertidal flats (Western Port & Corner Inlet only)
Channel – minor	Minor channels through intertidal flats (Western Port & Corner Inlet only)

Table 3.3. Seabed sediment categories in Port Phillip Bay (see Figure 3.4).

Source data sediment classes (PMA 1987)	Grain size mm	Grain size Phi	Simplified sediment classes for HSI modelling
Clay	< 0.063	> 4	Clay
Very fine sand	0.063 - 0.125	4 – 3	Sand-silt-clay
Fine sand	0.125 – 0.25	3 to 2	Fine sand
Medium sand	0.25 – 0.5	2 to 1	Medium sand
Coarse sand	0.5 – 1.0	1 to 0	Coarse sand
Very coarse sand	1.0 – 2.0	0 to –1	Coarse sand
Sand-silt-clay etc.	NA	NA	Sand-silt-clay
Clayey fine sand	NA	NA	Sand-clay
Clayey medium sand	NA	NA	Sand-clay
Clayey coarse sand	NA	NA	Sand-clay

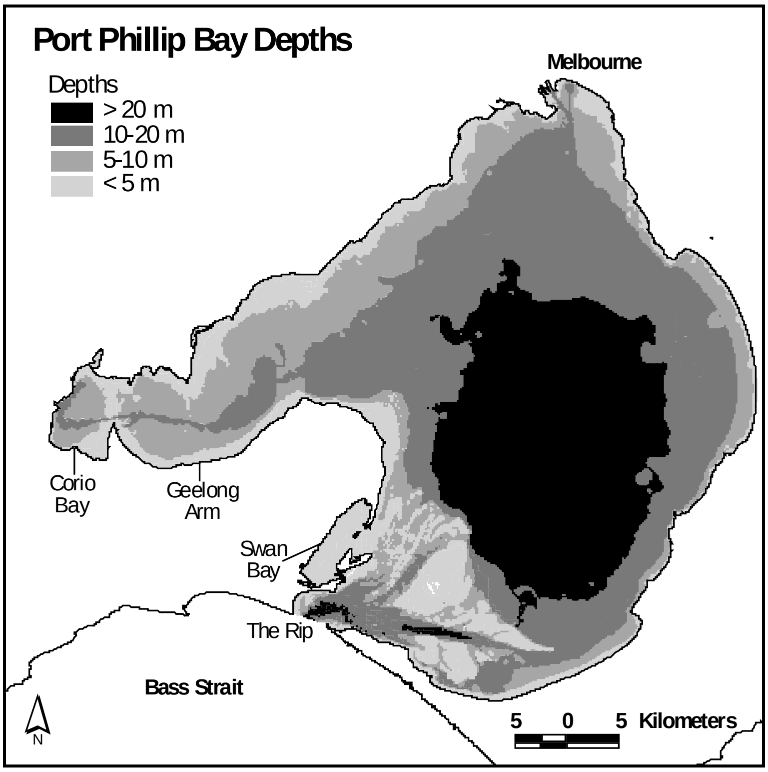


Figure 3.2. Port Phillip Bay depth zones.

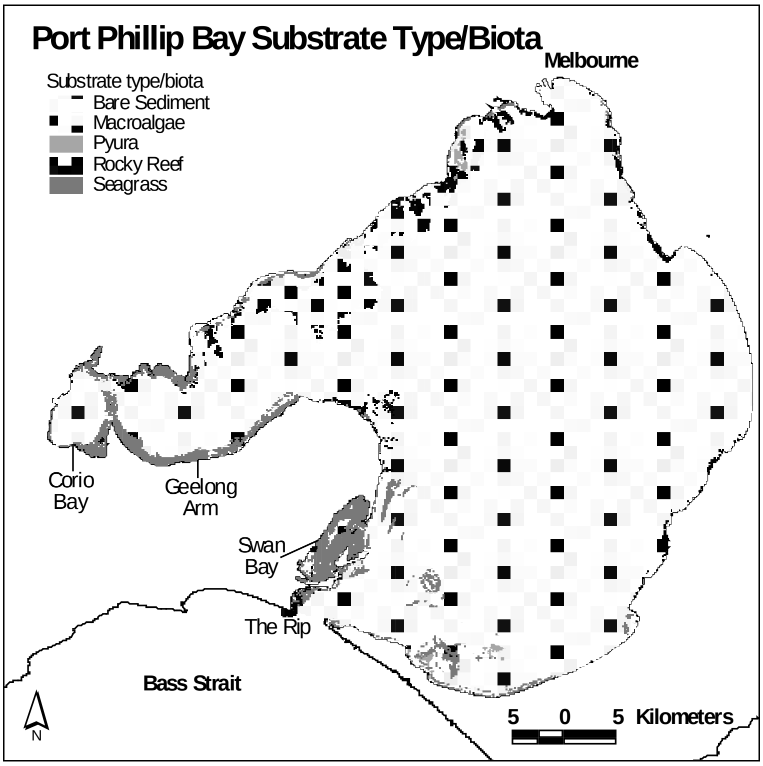


Figure 3.3. Port Phillip Bay substrate type/biota.

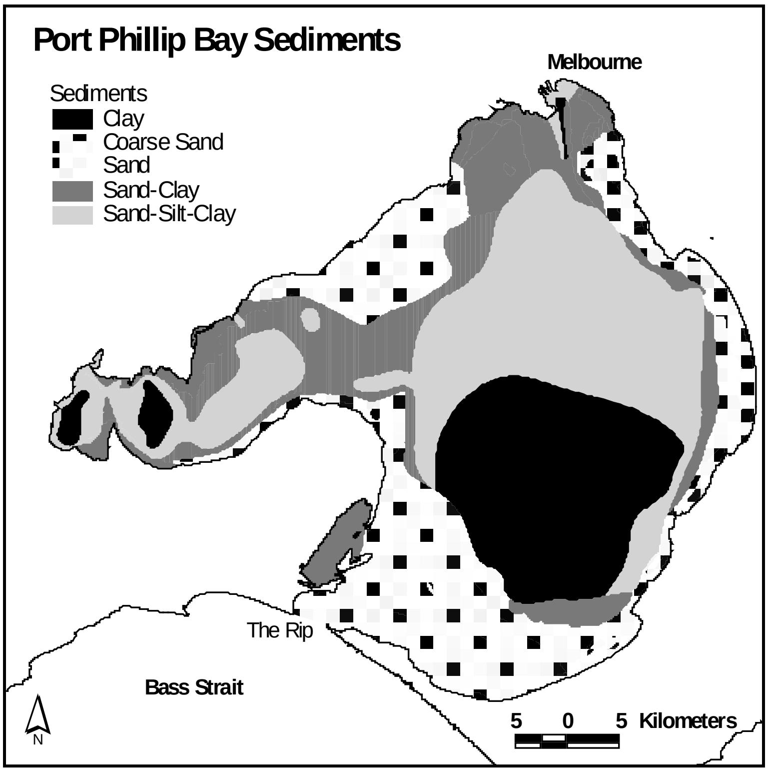


Figure 3.4. Port Phillip Bay seabed sediments (all sand categories have been grouped into a single sand class in this map).

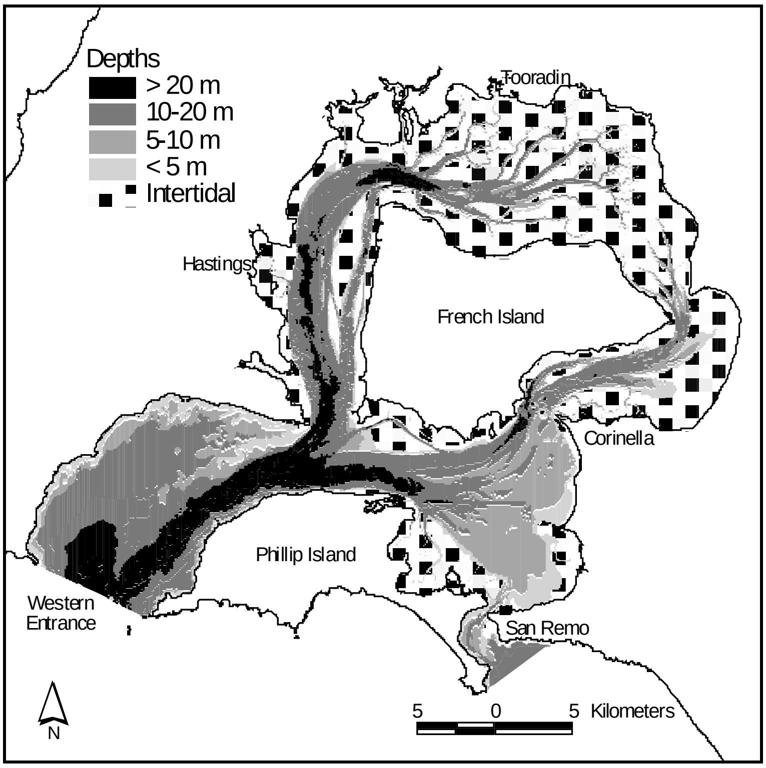


Figure 3.5. Western Port depth zones.

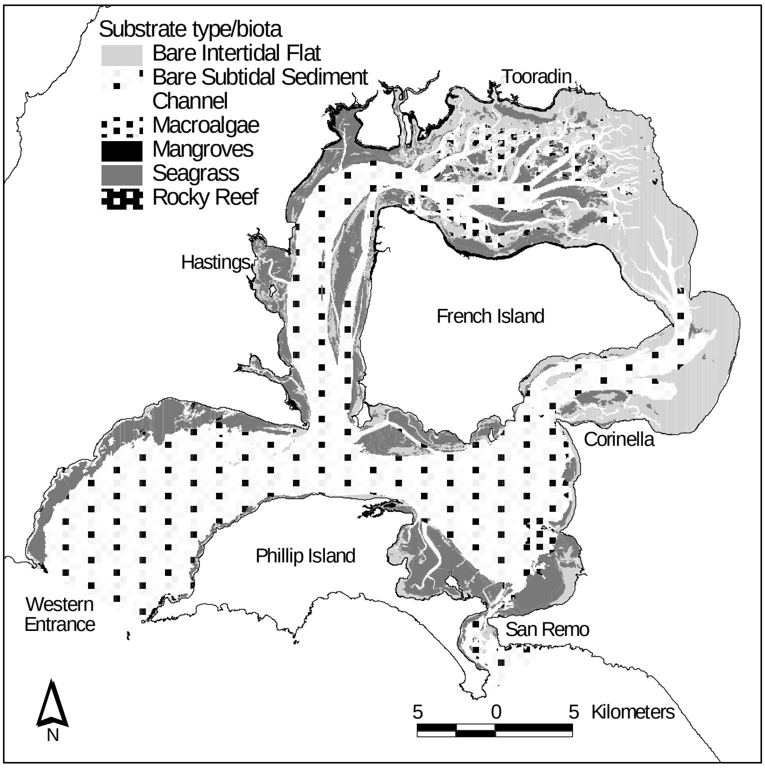


Figure 3.6. Western Port substrate type/biota.

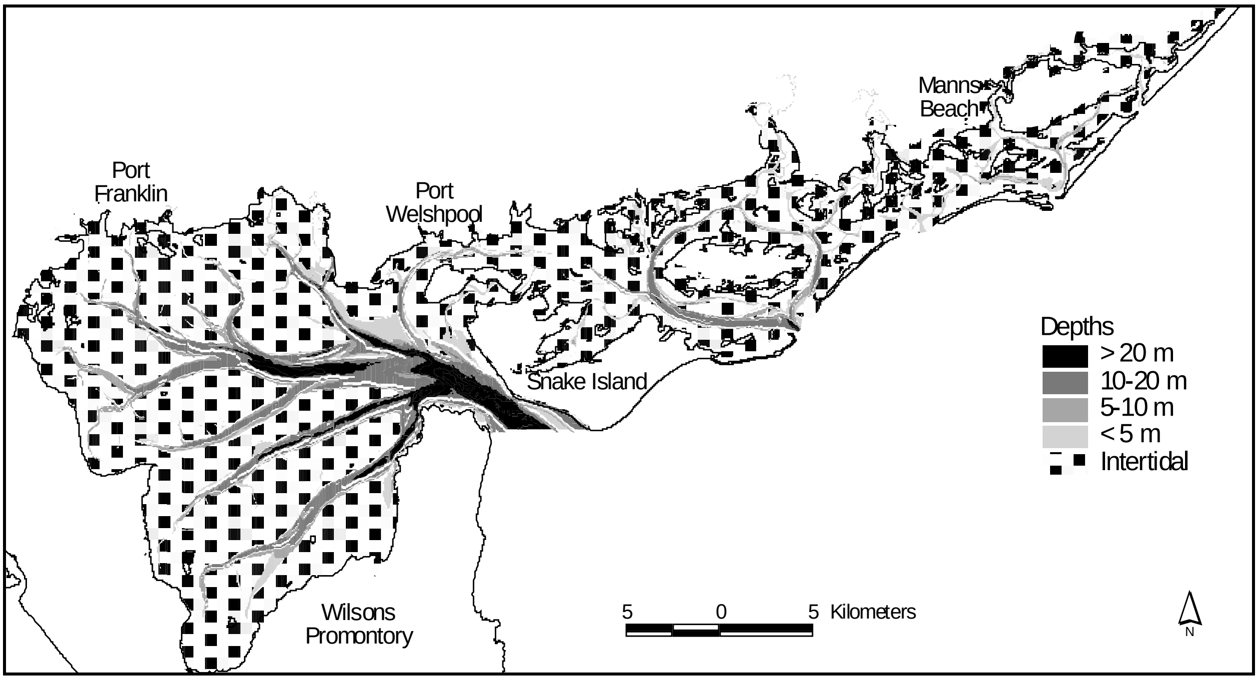


Figure 3.7. Corner Inlet/Nooramunga depth zones.

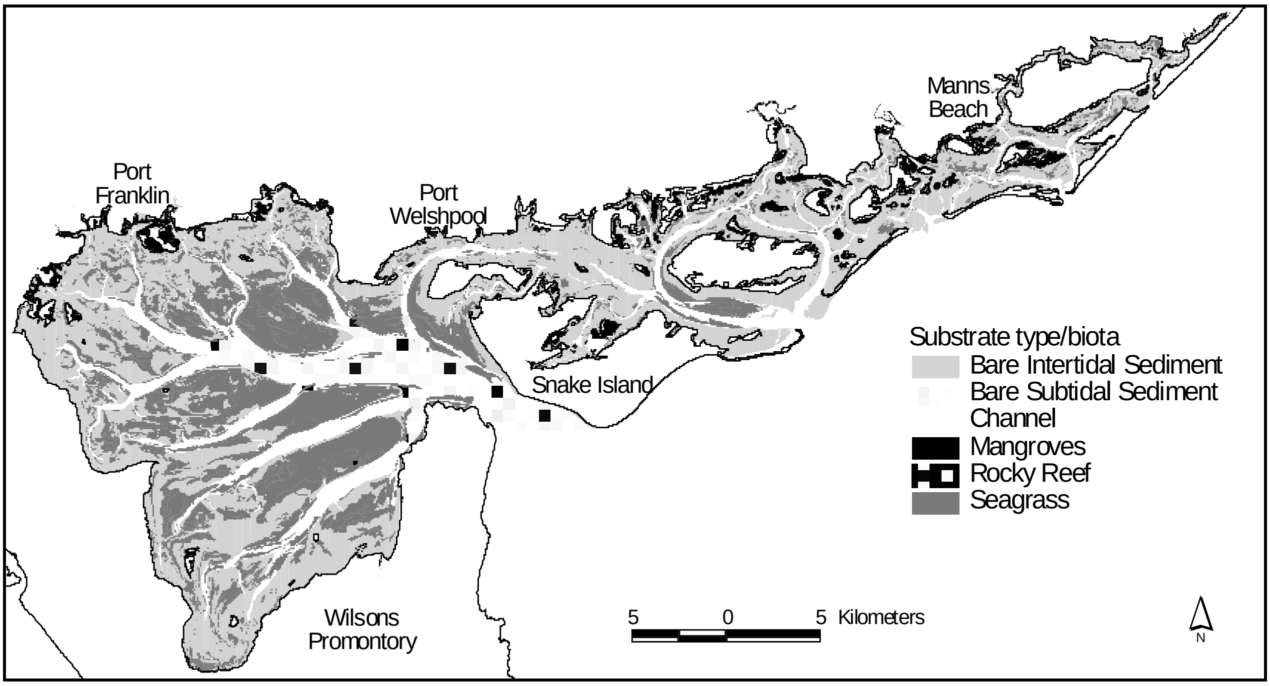


Figure 3.8. Corner Inlet/Nooramunga substrate type/biota.

3.3. Temperature and Salinity

Temperature and salinity are frequently used as environmental parameters in fish habitat suitability modelling, particularly in estuarine environments where large variations can affect spatial distributions of fish (Christensen *et al*. 1997; Rubec *et al*. 1999).

Port Phillip Bay and Western Port are both essentially marine embayments and temperature and salinity fluctuations within the bays are relatively small (Shapiro 1975; Harris *et al*. 1996) and generally within the range of tolerance for most species investigated in this study (Section 4). However, both bays have freshwater inputs and it is possible that regional differences in temperature and salinity variability within the bays could influence local fish distributions.

We examined Port Phillip Bay and Western Port datasets to investigate whether the range and variability in temperature and salinity across the bays could influence the spatial distribution of the target species and whether these parameters should be included in the habitat suitability modelling. The analysis is presented in Appendix 3.

There was insufficient data to analyse salinity and temperature variability in Corner Inlet/Nooramunga. Western Port and Corner Inlet/Nooramunga are similar environments though, and it is expected that the results of analysis of data for Western Port would be applicable to Corner Inlet/Nooramunga.

Available SST satellite imagery was reviewed during this study, but the spatial scale of this data was found to be unsuitable for the bays and inlets under investigation.

3.3.1. Port Phillip Bay

In order to analyse salinity and temperature variations in Port Phillip Bay it was necessary to divide the Bay into regions or zones. We elected to use segments defined by the State Environment Protection Policy (SEPP) - Waters of Port Phillip Bay 1975 (Figure 13.1) as these represent environmental zones within the Bay based on hydrodynamic and water quality influences and also correspond to the distribution of fixed site water quality monitoring undertaken by the EPA.

The analysis of the datasets for salinity indicated that there were differences between segments of the Bay in the mean salinity experienced in each segment, but the differences in mean values tended to be small and well within the tolerance ranges identified for each species (Section 4). Of more interest was the difference in the ranges of salinity experienced between segments and sites, as areas with more variable salinity may provide a more stressful environment for fish.

Early life history stages may be particularly affected by changes in salinity due to their reduced mobility compared to adults and the fact that they are often found in very shallow areas. Pulses of freshwater are less likely to affect the larger life stages of bottom dwelling fish as they can move away from an area and also the changes in salinity are unlikely to affect the deeper habitats. As a consequence we coded the SEPP segments (Figure 13.1) as being 'low', 'medium' or 'high' salinity variability for use in the habitat suitability modelling for juvenile stages of species (Figure 3.9). The only segment coded 'high' was Hobsons Bay, due to the influence of the Yarra River, and the only segment that was coded medium was the Northeastern segment which includes several large permanent watercourses. The remaining segments were coded as 'low' salinity variability. Surprisingly, there was no evidence to suggest that the Werribee segment should be coded as high or medium despite the freshwater discharges, which include two rivers as well as outfall drains from the Western Treatment Plant.

There was some evidence that temperature also differed across the Bay, but while seasonal differences were relatively large they appeared to happen consistently across the whole Bay. Similarly, there was some evidence of differences in ranges of temperatures between sites, but the magnitude of differences was very small and unlikely to be of biological importance. For this reason we did not include temperature as a GIS layer or a predictor in the habitat suitability modelling process for Port Phillip Bay.

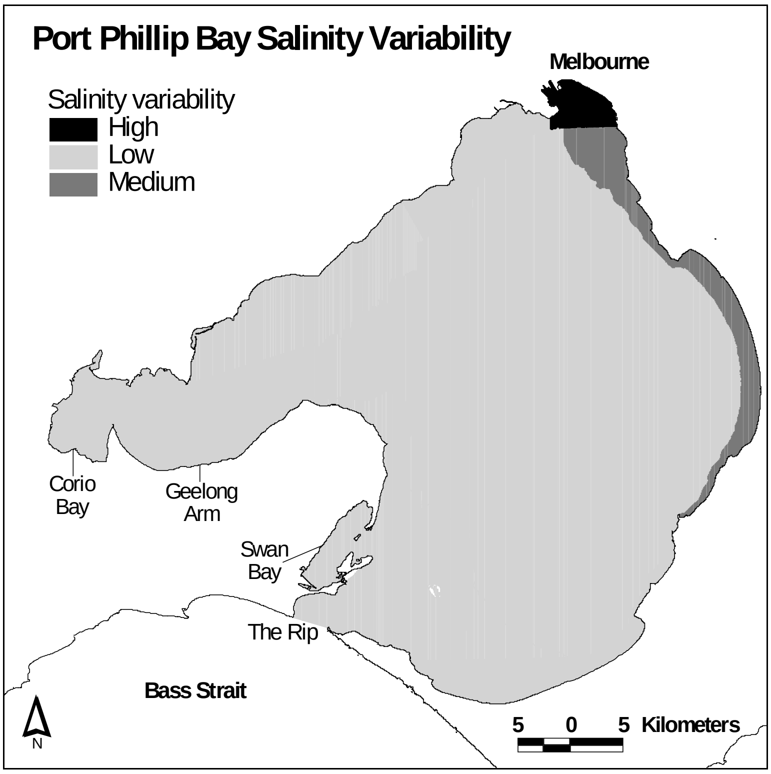


Figure 3.9. Port Phillip Bay zones of salinity variability.

3.3.2. Western Port

The analysis of Western Port salinity and temperature data is presented in Appendix 3. Recent data for Western Port was only available from three EPA fixed sites (Hastings, Barralliar Island and Corinella) sampled during the years 1990 – 2000. There was no difference in the mean temperature recorded at each site and no difference in the range of temperatures experienced at each site. There was also no difference in the mean salinity recorded at each site, but there was a difference in the range of salinities recorded at each site.

The results presented in Appendix 3 are consistent with those recorded by Longmore (1997) and reflect the fact that the majority of freshwater input occurs in the north and east of the bay. The net circulation around French Island is in a clockwise direction and as a consequence freshwater discharging from the northern rivers will primarily move down the eastern arm (Harris and Robinson 1979).

The analysis presented in Appendix 3 suggested that Western Port could be divided into three segments; an eastern segment with a high salinity variability, a northern segment with a medium salinity variability and a western/bay mouth segment with low salinity variability. However, we had insufficient fisheries independent data to effectively derive habitat suitability indices based on salinity variability as a model parameter (Section 5.2.2). As a consequence we did not include salinity variability as an environmental variable in the habitat suitability modelling for Western Port.

3.4. Exploration of other Spatial Data Variables

We recognised that the use of only depth, sediment and substrate type/biota layers may be unable to account for all of the different influences on species distributions and habitat suitability, so we also examined other possible spatial environmental variables for Port Phillip Bay including:

- Distance from Bay entrance (The Rip).
- Bay sectors.
- Orientation/aspect of shoreline.
- Seabed gradient.

In selecting other environmental variables for the habitat suitability modelling, they needed to meet the dual criteria of being able to be represented spatially in the GIS and existing fishery independent data had to be able to be accurately classified by each variable.

A spatial layer representing distance from the Port Phillip Bay entrance (The Rip) was generated based on zones of distance measured from the centre of The Rip in 10 km increments. This layer was investigated as a possible surrogate for tidal current speed and flushing or residence times within the Bay; parameters that are likely to be important in determining fish distributions and also settling of larvae. We calculated habitat suitability index values based on these distance zones from the entrance, but there was a widely uneven spread of sampling data across each 10 km zone. The 10 km distance was chosen somewhat arbitrarily and the boundaries between each zone do not exist as discernible features in the real world. When we included the distance zones as an input parameter to the habitat suitability models they tended to produce a distorted spatial pattern by creating artificial boundaries between suitability categories at the boundary between each 10 km zone. As a consequence, we elected not to include the distance from entrance parameter in the production of the composite habitat suitability models (Section 5). Some alternative to this approach, such as extracting results from hydrodynamic modelling of 3D currents to the GIS may present a more realistic picture of the influence of hydrodynamics on fishery habitat suitability.

Port Phillip Bay was also divided into an 'east' and 'west' sector as it was considered to be a potential predictor for recruits and juveniles found in shallow waters. The sectors were chosen in an attempt to account for the likely presence of juveniles in certain habitats on different sides of the Bay. The differences across the Bay are likely to depend on a number of factors including the origin of the larvae and the prevailing currents delivering them to different areas at different times of the year. After an initial assessment we did not pursue this variable further in our investigation.

Orientation and seabed gradient layers were generated from the depth grid to determine whether the seabed morphology could be considered in determining habitat suitability. While a broad-scale pattern of lower seabed gradients exists on the western side of the Bay, the spread of locations sampled during the fishery independent monitoring was insufficient to determine how variations in seabed morphology across the Bay may influence habitat suitability and these parameters were not investigated further.

4. Habitat Requirements of Commercially Valuable Fish Species in Victorian Bays and Inlets

4.1. Introduction

The first step in generating species suitability index values involves conducting a literature search to identify any documented tolerances or affinities to gradients in each environmental variable. In some cases, where quantitative data is sparse, the literature can be used to create qualitative suitability indices and this has been a common approach in both terrestrial and marine environments. The literature review also provides a broad background against which the final models can be compared.

In this section we review the following commercially important species in Victoria's bays and inlets:

Pelagic Predators

Australian salmon	*Arripis trutta* & *A. truttaceus*
Southern calamari	*Sepioteuthis australis*

Semi-Pelagic

Yellow-eye mullet	*Aldrichetta forsteri*

Demersal

Greenback flounder	*Rhombosolea tapirina*
King George whiting	*Sillaginodes punctata*
Rock flathead	*Platycephalus laevigatus*
Sand flathead	*Platycephalus bassensis*
Snapper	*Pagrus auratus*

The annual commercial catch in tonnes and the dollar value for the species investigated in this study from Port Phillip Bay, Western Port and Corner Inlet/Nooramunga is presented in Table 4.1.

We attempted to review species distributions in relation to the following environmental and habitat variables:

- Salinity.
- Water temperature.
- Depth.
- Substrate type.
- Sediment type.
- Substrate biota eg. seagrass, macroalgae.
- Mangroves and saltmarsh.

The above variables were considered most likely to exert an influence on the distribution of the selected species. In many cases, however, detailed information was not available. A summary of species habitat usage for the key variables is given in Table 4.2.

4.2. Critical and Non-critical Environmental Variables

Christensen *et al*. (1997) describe the principle of critical and non-critical environmental variables in habitat suitability modelling. "Critical" variables are defined as those exhibiting the potential to exclude a population if physiological tolerances are exceeded, while "non-critical" variables were defined as those that have an effect on a species distribution, but alone will never exclude a population from utilising a particular habitat.

Port Phillip Bay, Western Port, and Corner Inlet/Nooramunga are essentially marine systems with respect to their physical and chemical properties, although each system features some estuarine elements. As a result, the environmental variables addressed in this report are unlikely to vary widely enough or to reach levels that are critical to a species. As a result, we consider the environmental conditions within the bays investigated in this study to be non-critical.

4.3. Knowledge Gaps

The following presents a summary of the available information but there remain large gaps in our understanding. These gaps are primarily the result of a lack of information describing how juvenile and adult fish use different habitats, and how habitat use may vary over a variety of spatial and temporal scales, as well as with various environmental conditions.

Where studies have examined relationships between particular species and habitats, the results often suggest that species-habitat relationships are highly variable through time and between different locations within the same bay or inlet. At the same time, some habitats (eg. mangroves and saltmarsh) are widely accepted as important to fish, but only limited quantitative information exists for the areas being examined in this project. Further research using standardised techniques is needed that evaluates the relative importance of available habitats to different life stages of a variety of commercially important species of fish.

Measuring the relative abundances of fish in a particular habitat is just one aspect of describing fish-habitat affinities. Additional information about the nature of the sediments underlying the habitat, local availability of food, relative position of habitat, spatial continuity with regard to other types of habitat, as well as local physical and chemical properties of the local environment, is required to further address why patterns might exist.

Table 4.1: Victorian Bay and Inlet Commercial Fishery Catches 1995-2000 (HSI study species only – Victorian Commercial Fish Production – Information Bulletin 2002).

		1995/96		1996/97		1997/98		1998/99		1999/2000	
Species	**Bay/Inlet**	**Tonnes**	**$000**	**Tonnes**	**$000**	**Tonnes**	**$000**	**Tonnes**	**$000**	**Tonnes**	**$000**
Australian salmon	**Port Phillip**	47	76	36	45	38	51	22	28	80	109
	Western Port	6	10	4	5	3	4	3	3	4	5
	Corner Inlet	11	13	18	25	32	44	15	16	6	8
Southern calamari	**Port Phillip**	18	138	26	191	18	102	25	129	37	234
	Western Port	7	50	5	37	6	35	6	33	6	37
	Corner Inlet	8	54	6	45	6	33	18	93	24	150
Yellow-eye mullet	**Port Phillip**	31	29	29	30	36	40	31	33	19	21
	Western Port	14	13	13	14	12	14	9	9	5	5
	Corner Inlet	22	19	24	15	20	23	21	22	24	25
Greenback flounder	**Port Phillip**	11	65	18	91	12	53	7	40	7	42
	Western Port	3	18	1	6	1	4	1	4	0	1
	Corner Inlet	8	49	19	93	28	129	16	87	9	48
King George whiting	**Port Phillip**	62	677	90	659	131	823	101	774	126	1,260
	Western Port	9	96	13	104	16	97	17	133	12	121
	Corner Inlet	52	537	118	745	98	602	93	687	73	746
Rock flathead	**Port Phillip**	8	33	8	29	12	46	10	36	15	55
	Western Port	15	52	17	64	19	64	17	57	9	34
	Corner Inlet	32	111	23	85	31	100	26	80	36	124
Sand flathead	**Port Phillip**	11	17	13	20	8	12	6	9	5	7
	Western Port	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
	Corner Inlet	8	11	11	17	16	25	7	10	7	11
Snapper	**Port Phillip**	42	358	34	250	40	298	59	403	36	248
	Western Port	0	3	0	1	0	3	1	9	1	4
	Corner Inlet	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS

Table 4.2: Summary of habitat usage of primary commercial fish species in Victoria's aquatic zones (after Gunthorpe *et al.* 1998)

		Habitats									Aquatic Zones				
Species		Pelagic habitat	Seagrass	Shallow bare sediments	Deep bare sediments	Benthic macro-algae	Reef	Snags	Artificial	other	Fresh-water	Estuaries	Bays	Coastal	Oceanic
Australian salmon	Spawning														
	Eggs														
	Larvae														
	Juveniles														
	Adult														
Yellow-eye mullet	Spawning														
	Eggs		?	?	?	?	?	?	?	?	?	?	?	?	?
	Larvae		?	?	?	?	?	?	?	?	?	?	?	?	?
	Juveniles														
	adult														
Greenback flounder	spawning														
	eggs														
	larvae														
	juveniles														
	adult														
King George whiting	spawning	?	?	?	?	?	?	?	?	?					
	eggs														
	larvae														
	juveniles														
	adult														

Table 4.2 continued

Rock flathead	spawning	?	?	?	?	?	?	?	?	?					
	eggs		?	?	?	?	?	?	?	?					
	larvae		?	?	?	?	?	?	?	?					
	juveniles														
	adult														
Snapper	spawning	?	?	?	?	?	?	?	?	?					
	eggs														
	larvae														
	juveniles														
	adult														

LEGEND					?
	critical	important	unimportant	uncertain	not known

4.4. King George Whiting (*Sillaginodes punctata*)

4.4.1. General Information

King George whiting *Sillaginodes punctata* are a demersal species found from northern New South Wales to the south-west coast of Western Australia, including the north coast of Tasmania (Paxton *et al*. 1989). Juvenile fish are thought to be restricted to bays and inlets, while adults are found in open coastal waters (Kailola *et al*. 1993). King George whiting have a life expectancy of 15 years and are thought to reach sexual maturity at three years when fish are 35 cm in length (Jones *et al*. 1990).

Significant commercial and recreational fisheries for King George whiting in Victorian waters occur in Port Phillip Bay, Western Port, Corner Inlet and Shallow Inlet. The commercial fishery for this species in Victorian bays and inlets is based primarily on sub-adult fish (Smith & MacDonald 1997), although adults are occasionally taken in coastal waters.

4.4.2. Early Life History:

King George whiting from Victorian waters are spawned between May to July (Jenkins & Black 1994). While it is known that this species does not use bays or inlets for spawning (Jenkins 1986), the spawning location(s) or habitats of King George whiting are not known. However, recent research suggests that spawning may take place in coastal waters to the west of Victoria's major bays and inlets, and that a significant proportion of Victoria's King George whiting population may be spawned in South Australian waters (Jenkins *et al*. 2000). In South Australia, spawning King George whiting have been found in waters near Kangaroo Island and at the tip of the Eyre Peninsula (Fowler 1997). Eggs and larvae develop in the pelagic environment until larvae are ready to settle (usually around 15-18 mm and 80-150 days old).

Larvae settle into sheltered bays and inlets along Victoria's coast at up to 150 days of age (Jenkins & Black 1994; Jenkins & May 1994). Juvenile King George whiting are known to settle into relatively shallow water that contains either seagrass or rocky reef/algae (Jenkins *et al*. 1997b; Jenkins & Sutherland 1997; Jenkins *et al*. 1998a; Jenkins & Wheatley 1998). Along the Victorian coast, larvae probably settle at western most sites first, a reflection of the overall eastward movement of larvae (Jenkins *et al*. 2000). In Western Port, it is thought that the initial settlement of King George Whiting occurs on the western side of the bay which is consistent with the idea that larvae are arriving from the east.

It is apparent from recent studies in Port Phillip Bay that the importance of particular nursery areas varies depending on their location. Sheltered seagrass/algal nursery areas situated in areas where currents deliver larvae are the most important (Jenkins *et al*. 1997a). In Swan Bay, however, newly-settled individuals have been found in bare mud/sand patches within seagrass beds (Jenkins *et al*. 1997b); the importance of a particular habitat may depend on the amount of food available (Jenkins *et al*. 1996a; Jenkins & Hamer 2001).

Seagrass beds are generally considered to be most important in the provision of habitat for juvenile King George whiting (Jenkins *et al*. 1997b; Jenkins & Sutherland 1997; Jenkins *et al*. 1998a), but habitat preferences are highly site specific. For example, in Jenkins and Wheatley (1998) three sites around Bellarine Peninsula were sampled: Clifton Springs, Grassy Point and St Leonards. Most fish occurred on bare sand at Clifton Springs (although they were slightly larger), while most fish occurred on seagrass and rocky reef at the other two sites. Similarly, in another study (Jenkins *et al*. 1997b), the numbers of fish in Swan Bay were greatest on bare sand than seagrass; at St Leonards, there was no difference between sand and seagrass. In the same study, King George whiting were more abundant on bare sand than seagrass at Corner Inlet.

Little habitat preference is shown by King George whiting juveniles in Western Port (Edgar & Shaw 1995). Depending on the location in which sampling was carried out, juvenile King George whiting were caught in greatest numbers in bare sand, seagrass or in channels. The only constant in all of these studies was that juvenile whiting were caught in less than 2 m of water. However, whether this is a reflection of depth preferences or sampling regimes (only water less than 2 m is sampled) is debatable. Juveniles remain closely associated with shallow seagrass and algal habitats for four to five months after settlement before moving to bare sand patches amongst vegetated habitats (Jenkins & Wheatley 1998).

No work has been done on feeding of early larval stages in coastal waters. In Port Phillip Bay, the diet of late-stage planktonic larvae of King George whiting is composed primarily of planktonic calanoid copepods, however, a range of other zooplankton are also important (Jenkins *et al*. 1998b). The diet of young juveniles is dominated by benthic and epifaunal organisms such as harpacticoid copepods and amphipods, and a range of other small crustaceans that live near the bottom (Robertson 1977; Edgar *et al*. 1993; Jenkins *et al*. 1996b).

4.4.3. Sub-adult/Adult stages

Limited research addresses this stage of the life history of King George whiting in the locations addressed in this study. Sub-adult King George whiting are highly transient, and are capable of using a variety of habitat types. Older juveniles venture into deeper water, where they are more common over sandy, muddy areas with patchy seagrass and algae (G. Jenkins, pers. comm.). King George whiting migrate out of bays and inlets just prior to reaching maturity (at approximately three years of age) (Jones *et al*. 1990). Most relevant information about habitat preferences comes from fishing anecdotes which suggest that King George whiting move deeper with age (1+ years of age). In bays and inlets, King George whiting are more common along edges of reefs and in patches of sand within beds of seagrass in deeper water (2-20 m).

In Western Port, sub-adult fish are often seen on sand patches between fine leaf seagrass meadows, and on the edges of the reefs around Merricks, Balnarring, Somers, Long Island Point (French Island), Bass and San Remo (Gunthorpe & Hamer 1998b). In Corner Inlet, whiting (27-35 cm) feed over sand during the night and move onto seagrass during the day. Larger fish (40 cm) are associated with deeper channels in Corner Inlet, but are also found over seagrass. King George whiting are also found in macroalgal beds, and at certain times of year are thought to actively prefer such areas (most likely related to availability of food).

Recreational fishing guides recommend clean patches of seabed between reef and seagrass at depths less than 5 metres in Port Phillip Bay for King George whiting (AFN 2002). Specific areas noted as being good fishing areas in Port Phillip Bay are found at Swan Bay, West Sand, Beaumaris Bay, Altona Bay, Campbells Cove, Wedge Point, Grassy Point to St Leonards and Curlewis Bank (Classon & Wilson 2002). In Western Port, shallow sand flats (at 1-5 m depths in particular) throughout the bay are considered good fishing areas for King George whiting (Crowley & Worsteling 2000; Classon & Wilson 2002). Similarly, in Corner Inlet/Nooramunga recreational fishers are advised to fish the smaller channels throughout the inlet for King George whiting (AFN 1996; Classon & Wilson 2002).

The diets of larger King George whiting include larger benthic organisms such as polychaete worms, ghost shrimp (*Callianassa*), molluscs and peanut worms (*Sipuncula*) (Robertson 1977). Polychaetes may dominate the diet of the larger King George whiting in Victorian bays (Parry *et al*. 1995).

4.5. Greenback Flounder (*Rhombosolea tapirina*)

4.5.1. General Information

Greenback flounder *Rhombosolea tapirina* are commonly found in estuaries, bays and inshore coastal waters from southern New South Wales to the east coast of West Australia, including Tasmania (Gomon *et al*. 1994). Greenback flounder are demersal and are most common on bare sediments (Hutchins & Swainston 1986; Kuiter 1993). Greenback flounder can tolerate a wide range of salinities and water temperatures (Kailola *et al*. 1993). Greenback flounder may live for 3 to 4 years, reaching a maximum size of 40 cm in length. Maturity occurs at lengths of between 19 and 30 cm (Kailola *et al*. 1993).

In Victoria, small commercial fisheries for flounder occur in Port Phillip Bay, Western Port and Corner Inlet. Flounder are targeted by recreational fishers using hand spears in shallow water and are not often taken with a rod and line. In Nooramunga, the recommended spots for flounder are in the eastern part of the inlet in protected shallow areas (AFN 1996).

4.5.2. Early Life History

Spawning is thought to occur in offshore waters from March through to October (Kurth 1957; Crawford 1984), although the presence of eggs and recently hatched larvae in Port Phillip Bay suggest that spawning also occurs in coastal bays during winter (Jenkins 1986). Eggs are planktonic and hatch about 4 days after fertilisation (Crawford 1984). Fertilisation is greatest between salinities of 35 and 45 ppt (Hart & Purser 1995).

Larvae may spend a month or more drifting in the water column (Jenkins 1986) before metamorphosing and settling onto shallow inshore sand and mud flats during late winter/spring at a length of about 10 mm (May & Jenkins 1992). Larval stages may feed on dinoflagellates and zooplankton such as bivalve veligers, copepods and nauplii (Jenkins 1986). Both the growth and survival of larvae are influenced by day-length, temperature and salinity (Hart *et al*. 1996). Total darkness resulted in 100% mortality, and the optimal light periods were 18-24 hrs (Hart *et al*. 1996). Optimal temperatures were not clearly identified, but temperatures between 19 and 20° C gave better growth than lower temperatures (Hart *et al*. 1996). Lower temperatures (below 16° C) also resulted in greater mortality. A salinity of 15 ppt resulted in lower survival compared with either 25 or 35 ppt, but over this range salinity did not influence growth (Hart *et al*. 1996).

Shallow (< 2 m) bare habitats are important nursery areas for juvenile greenback flounder (Jenkins *et al*. 1997b; Jenkins & Wheatley 1998). In Western Port, large numbers of juvenile flounder (3-4 cm) were caught around Somers in very shallow water (Hyslop & Johnson, pers com. in: Gunthorpe and Hamer 1998b). Areas with patchy macrophyte cover are also likely to be important nursery areas due to the increased food availability resulting from organic enrichment of the sediments by macrophyte detritus (Shaw & Jenkins 1992). Higher numbers of juvenile flounder were often caught in bare sand than seagrass at all locations in both Port Phillip and Corner Inlet (Jenkins *et al*. 1993; Jenkins *et al*. 1997b; Jenkins & Wheatley 1998). In Western Port, flounder were found almost exclusively over bare sand (or in channels) (Edgar & Shaw 1995). Juvenile flounder are probably able to tolerate temperatures up to 28°C and temperatures often reach this level in shallow water during summer (J. Hindell pers. obs.). The diets of juveniles are primarily composed of sediment-associated organisms such as diatoms, harpacticoid copepods and amphipods (Rigby 1984; Shaw & Jenkins 1992).

4.5.3. Adult Life History

Greenback flounder move into deeper bare habitats as they grow, but may still occur in shallow areas as older juveniles. The diets of adults are composed of plant material, polychaete worms, nematodes and small bivalve molluscs. Adult fish are thought to prefer soft sediments, although they are commonly caught in areas where there are large patches of sand interspersed with rocky reef/algae and seagrass. Adult flounder can be caught throughout Corner Inlet, in both shallow and deep water usually on bare substrates near seagrass beds (particularly *Posidonia*). Adult flounder are generally not found in macroalgal habitats. Interestingly, commercial mesh netters often catch flounder when they set their nets over shallow (< 3 m) seagrass. Edgar (1995) caught significantly more flounder in sites with seagrass than corresponding sites in channels or over bare sand. Perhaps fish forage over these habitats at night, or swim over them to get to shallow (inshore) areas.

4.6. Australian Salmon (*Arripis spp*.)

4.6.1. General Information

There are two species of Australian salmon in Victorian waters; the western species *Arripis truttacea* and the eastern species *Arripis trutta*. The eastern species is generally only found east of Cape Otway in Victoria, but range along the eastern Australian coast from Brisbane in Queensland to Tasmania. West Australian salmon is found along the entire Victorian coast; their distribution extends west to the central coast of West Australia (Kailola *et al*. 1993). Both species inhabit continental shelf waters including estuaries and bays (Kailola *et al*. 1993) and make annual spawning migrations.

The main commercial fisheries for Australian salmon in Victoria occur in the coastal waters of Bass Strait and in Port Phillip Bay, with much smaller catches taken from the other bays and inlets (Neira 1997). Most of the catches from inlets are juveniles.

Both species of salmon are taken by recreational fishers along the entire Victorian coastline. The relative proportions of each species in the total catch are unknown because of the difficulty associated with identification which requires counting the number of gill rakers on the first gill arch. The proportion of East Australian salmon in the total catch increases the further east the fishery is located. In Mallacoota Inlet, for example, the Australian salmon catch is predominantly composed of East Australian salmon as opposed to Port Phillip Bay where West Australian salmon dominate the catches (MacDonald *et al*. 1995).

In both Corner Inlet and Western Port the entrances to the bays provide good recreational fishing areas and in Western Port the western side of the bay is also considered good for salmon fishing (Classon and Wilson 2002). These fishing patterns are consistent with the information that in Western Port adult salmon move throughout the bay, though they will congregate near the entrances (Hyslop & Johnson, pers com. in: Gunthorpe and Hamer 1998b). In Port Phillip Bay the entrance is considered a good area for recreational salmon fishing and nearshore areas on the eastern side of the bay at Mount Martha, Mt Eliza and Beaumaris are also recommended (AFN 2002).

***Arripis truttacea* (western species)**

In Victorian waters, western Australian salmon mature at around four to six years of age and may be up to 54 cm long (Cappo 1987). Maturing West Australian salmon migrate from southeastern Australia to waters off the south coast of Western Australia in mid to late summer (Nicholls 1973). Spawning aggregations form and spawning takes place near headlands between February and June, peaking from March to early May (Stanley 1980). Post-spawning adults return eastwards, but not as far as South Australia (Kailola *et al*. 1993).

West Australian salmon is the most common species in Port Phillip Bay and dominates the commercial catch (Coutin 2000a).

***Arripis trutta* (eastern species)**

Eastern Australian salmon mature in their fourth year, at about 39 cm in length (Stanley & Malcolm 1977). Mature East Australian salmon move north from Tasmania and east from central Victoria to waters between Lakes Entrance (in Victoria) and Bermagui (in New South Wales), where spawning takes place between November and February. Spawning usually peaks between December and January off Lakes Entrance (Stanley & Malcolm 1977). Following spawning, fish disperse into Bass Strait and north to New South Wales (Stanley 1978).

4.6.2. Early Life History

***Arripis truttacea* (western Australian salmon)**

Western Australian salmon larvae and juveniles drift east across the Great Australian Bight to southeast Australian waters during winter. This movement is apparently influenced by the eastward flow of the Leeuwin Current (Cappo 1987). Juvenile West Australian salmon (5-8 cm in length) appear in Victoria's bays and inlets between July and September (Cappo 1987).

***Arripis trutta* (eastern Australian salmon)**

Eastern Australian salmon eggs, larvae and juveniles drift/migrate, under the influence of the south-flowing Eastern Australian current, from the spawning grounds to Victorian waters during autumn and winter (Nicholls 1973).

Juveniles of both species are common in shallow waters of bays and estuaries and may spend several years in these nursery areas. Jenkins *et al.* (1996a) reported that juveniles of both species were most common over bare sand/mud and seagrass as opposed to reef. They can tolerate a wide range of salinities and temperatures (Ramm 1986; Kailola *et al*. 1993).

Video footage has shown that juvenile salmon forage much more commonly in bare habitats – in fact they were observed in seagrass only once (J. Hindell unpublished data). Jenkins and Wheatley (1998) also found that juvenile salmon, particularly the eastern species, were more common over shallow bare sand than seagrass. Research by Jenkins *et al*. (1997b) showed that the salmon/habitat associations varied widely depending on the location. For example, juvenile salmon were more abundant in seagrass at St Leonards (which contrasts with previous work) but bare sand at Queenscliff; no salmon were captured at Corner Inlet during this research.

The diet of larval salmon, regardless of species, has not been studied. Juveniles of both species are opportunistic feeders, but there appears to be subtle differences in diets between the two. Juvenile western Australian salmon feed mostly on bottom dwelling crustaceans and fish, while juvenile eastern Australian salmon feed on zooplankton and epibenthic species of fish, squid, crustaceans and polychaete worms (Robertson 1982).

4.6.3. Adult Life History

Large juveniles and adults of both species are generally found in exposed coastal waters around rocky headlands, reefs and sandy beaches (Kailola *et al*. 1993). Adult western Australian salmon feed on squid and a range of fish species including pilchards, anchovies, southern sea garfish and tommy ruff (Kailola *et al*. 1993). In contrast, adult eastern Australian salmon feed mainly on zooplankton such as krill (Stanley 1978). Edgar and Shaw (1995) captured both species in Western Port using gill nets, but fish were more abundant in bare sand than seagrass, although at some sites salmon were more abundant over seagrass than sand.

4.7. Yellow-eye Mullet (*Aldrichetta forsteri*)

4.7.1. General Information

Yellow-eye mullet *Aldrichetta forsteri* are found around the southern half of Australia, from Newcastle in New South Wales to Shark Bay in West Australia, including Tasmania (Kailola *et al*. 1993). Yellow-eye mullet are a schooling species that inhabit waters of estuaries, bays and shallow inshore coastal areas. This species can tolerate a wide range of salinities and temperatures (Kailola *et al*. 1993). Yellow-eye mullet can reach up to 40 cm in length (Hall 1984) and mature at about two to three years of age (Harris 1968). Mature fish form large aggregations in coastal waters and marine embayments prior to spawning (Lenanton 1977).

The major commercial fisheries for yellow-eye mullet in Victoria occur in Gippsland Lakes, Port Phillip Bay, Western Port and Corner Inlet (P. Coutin, pers. comm.). Small commercial fisheries occur in most of the other inlets. They are captured by recreational anglers in most Victorian bays and inlets and are an important component of the shoreline angler's catches in Port Phillip Bay (P. Coutin, pers comm.).

4.7.2. Early Life History

In Victorian waters, spawning may occur from late spring until autumn (Rigby 1984; Ramm 1986). The spawning locations and details of the larval life of yellow-eye mullet in Victorian waters are not well understood, but spawning is suggested to occur predominantly in coastal waters outside bays and inlets (Chubb *et al*. 1981; Jenkins *et al*. 1996a).

Juvenile yellow-eye mullet move into Victorian bays and estuaries from late summer through to early spring when they attain a size of 30 to 40 mm in length (Jenkins *et al*. 1993; Jenkins *et al*. 1996a; Ramm 1986; Robertson 1978). Juveniles are abundant in shallow water over seagrass and bare sand habitats, but are less common over shallow reefs (Jenkins *et al*. 1993; Jenkins *et al*. 1996a). Jenkins and Wheatley (1998) found that mullet did not occur over bare sand at three sites in Port Phillip Bay, but were common over seagrass, and at one site were also caught over rocky reef. Edgar *et al.* (1993) reported that yellow-eye mullet are abundant over both seagrass and bare habitats in Western Port, although there did not appear to be a particular habitat that was consistent between sites (Edgar & Shaw 1995). Ramm (1986) however, suggested that juvenile yellow-eye mullet in Gippsland Lakes prefer habitats with low seagrass cover. In surf zone areas, juvenile mullet have been found associated with drift macrophytes (Lenanton 1982), which is thought to be related to the abundance of food associated with the drift algae (Lenanton 1982). Young juveniles feed mainly on zooplankton.

4.7.3. Adult Life History

Adult mullet occur in a range of water depths over both bare sand and seagrass. They are particularly conspicuous in shallow water over bare sand (J. Hindell pers. obs.). Edgar and Shaw (1995) found that mullet were approximately twice as common in shallow seagrass as over bare sand in Western Port; and their abundances were relatively low in channels. Adult yellow-eye mullet are omnivores, but algae may dominate the diet of larger fish (Edgar *et al*. 1993). Older juveniles and adults also feed on detritus, plankton, filamentous algae, polychaetes and other small invertebrates and epiphytes (Thomson 1957; Rigby 1984).

4.8. Rock Flathead (*Platycephalus laevigatus*)

4.8.1. General Information

Rock flathead *Platycephalus laevigatus* are found from Nowra, New South Wales to Geographe Bay, Western Australia and around Tasmania (Edgar 1997). Rock flathead are generally found in shallow waters (< 20 m) of bays and inlets, most commonly in areas of seagrass and low relief reef (Last *et al*. 1983; Edgar 1997). Little information exists on the life history of rock flathead. Rock flathead can grow up to 50 cm in length, but neither the life expectancy, nor the age and size at maturity is known.

The main commercial fishery for rock flathead in Victoria occurs in Corner Inlet, with smaller quantities taken in Western Port and Port Phillip Bay (P. Coutin, pers comm.). This species is not targeted by recreational fishers but may be caught occasionally by fishers targeting species such as King George whiting in seagrass or reef areas (P. Coutin, pers comm.).

4.8.2. Early Life History

Knowledge of the spawning habits of this species is limited. Spawning is thought to occur in seagrass habitats in Corner Inlet during November and December (Klumpp & Nichols 1983). No information is available concerning the larval biology of this species. Recruitment of juvenile rock flathead in Corner Inlet has been observed during December (Jenkins *et al*. 1993). New recruits have also been captured during summer in Port Phillip Bay (Jenkins *et al.* 1996a, Hamer *et al.*, unpublished data). Jenkins (1993) found that newly-settled rock flathead show a preference for bare habitats while older juveniles and adults prefer seagrass. A similar pattern has been observed in Western Port (Edgar *et al*. 1993). Reef habitats are also used by adults, but the importance of these habitats to juveniles is unknown (Jenkins *et al*. 1996a). Edgar (1995) failed to catch juvenile rock flathead while seine-netting in shallow water, despite the high numbers of this species in the general area. Larval rock flathead are thought to feed on zooplankton, however, little is known about the diet of the larval stages (G. Jenkins, pers comm.). The diet of small juveniles is also unknown (G. Jenkins, pers comm.).

4.8.3. Adult Life History

Large rock flathead are commonly caught in areas containing seagrass, and are particularly common during summer in seagrass < 3 m deep (J.Hindell pers. obs.). In fact, Edgar (1995) found that rock flathead were more common in seagrass than bare sand at several locations in Western Port; although they caught an intermediate number of fish in channels. Conversely, the trawl surveys by Parry *et al* (1995) have caught rock flathead over bare sand/mud habitats in depths between 7 and 22 m (Parry *et al*. 1995). The diets of larger rock flathead consist mostly of organisms associated with seagrass beds (Klumpp & Nichols 1983). Larger juveniles (< 33 cm) feed mainly on shrimp, squid and small fish (Klumpp & Nichols 1983; Hall & MacDonald 1986). The diet of adults in Corner Inlet consists of fish, squid, shrimps and crabs (Klumpp & Nichols 1983). Crabs (in particular *Nectocarcinus integriforms)* and fish are also important components in the diets of larger rock flathead in Port Phillip Bay (Parry *et al*. 1995) and Western Port (Edgar *et al*. 1993).

4.9. Sand Flathead (*Platycephalus bassensis*)

4.9.1. General Information

Sand flathead *Platycephalus bassensis* are endemic to Australia. They are present in southern Australian waters from Red Rock, NSW, west to Lancelin, WA, and around the Tasmanian coast (Kailola *et al*. 1993). Sand flathead inhabit coastal waters, from shallow bays and inlets to depths around 100 m. They occur primarily over sand, shell grit and mud substrates (Kailola *et al*. 1993).

Sand flathead are caught along open coasts by Danish seine and otter trawls, and in bays and inlets using gillnets, haul seines and longlines (Kailola *et al*. 1993). Sand flathead are important to the recreational fishery but are no longer targeted by commercial fishers. Large numbers of sand flathead are caught by both boat and shore-based recreational fishers (Coutin 2000a).

4.9.2. Early Life History

Sand flathead are thought to spawn in bays, inlets and shallow coastal waters. The timing of spawning is influenced by the day length and water temperature (Kailola *et al*. 1993). There is only a single spawning event each year in Port Phillip Bay, between August and October (Kailola *et al*. 1993).

Juvenile sand flathead are found on bare sediment up to depths of about 20-25 m (P. Hamer pers. comm.). Newly recruited juveniles of 3 to 10 cm were predominantly taken in the central and southern regions of Port Phillip Bay during sampling with a small beam-trawl (Hamer *et al*. 1997). Juvenile fish probably feed on small fish and invertebrates such as polychaetes, crabs and shrimps.

4.9.3. Adult Life History

Sand flathead are reported to reach a total length of 46 cm and a weight of over 3 kg. The growth rate of sand flathead is highly variable, however, in Port Phillip Bay, they attain a length of 10-12 cm after 1 year, 22-30 cm after 4 years and up to 43 cm after 9 years. All sand flathead are considered to be mature at a length of 22 cm in Port Phillip Bay.

Adult fish are active foragers and ambush predators (Kailola *et al*. 1993). They feed on a variety of prey, including various species of fish and crustacean, but the composition of the diet varies according to the location and the availability of prey. The diet of sand flathead varies strongly between seasons; during summer, crustaceans are the main dietary component, but in winter fish are more commonly eaten.

In Port Phillip Bay sand flathead appear to have a high affinity with bare muddy-sand or shell habitats in depths of 15-25 m (Coutin 2000a). Sampling with a beam-trawl caught adult sand flathead (mostly 20-30 cm) in all regions of the Bay (Hamer *et al*. 1997). The numbers of adult sand flathead in Western Port appears to be increasing. The increase is thought to be a consequence of the dieback of seagrass that occurred in the late 1970's to early 1980's (Gunthorpe & Hamer 1998b).

Fishing guide books do not always distinguish between sand flathead, yank flathead and rock flathead and so the following recommendations may apply to all these species. In Port Phillip Bay recreational fishing guide books recommend shallower areas on the western side and southern end of the bay (Australian Fishing Network 2002, Classon and Wilson 2002). In Western Port the expanses of shallow mud adjacent to the channels is considered a good habitat for flathead (Australian Fishing Network 2002).

Flathead are found throughout Western Port in depth ranges of 0.5 m to 30 m and deep water areas off Cowes and Tortoise Head, along Tyabb Bank in the North Arm plus the upper part of the bay all return good catches of flathead (Crowley & Worsteling 2000). Flathead are also widely distributed in Corner Inlet/Nooramunga and recreational fishers recommend most of the channels throughout the inlet as good spots to target this species (Classon & Wilson 2002).

4.10. Snapper (*Pagrus auratus*)

4.10.1. General Information

Snapper *Pagrus auratus* are found from Hinchinbrook Island in Queensland to Barrow Island in West Australia, including northern Tasmania (Edgar 1997). Snapper are also widely distributed throughout the Indo-Pacific, including New Zealand and Japan (Paulin 1990). Snapper is a demersal species that is found from the shallow waters of bays and inlets to the edge of the continental shelf (Kailola *et al*. 1993). Snapper may live for up to 35 years and grow to 1.3 m in length (Kailola *et al*. 1993). Snapper reach sexual maturity at about three to four years of age when approximately 27-35 cm in total length (MacDonald 1982). There is little published information on the spawning behaviour of snapper in Victoria, however, it has been established that snapper move into Victorian bays during spring and summer to spawn (MacDonald 1982; Coutin 1996; Neira & Tait 1996).

The extent of coastal spawning is unknown. Spawning is suggested to occur when water temperatures exceed 18° C (MacDonald 1982). Although the spawning areas of snapper in Victoria are not well documented, adult fish aggregate in northern Port Phillip Bay during the spawning season. Eggs and planktonic larvae have also been collected from this region of the Bay during the summer months (Jenkins 1986; Neira & Tait 1996), suggesting that the northern area of Port Phillip Bay is an important spawning ground for snapper.

The main commercial and recreational fisheries for snapper in Victorian bays and inlets are in Port Phillip Bay, Western Port and Corner Inlet (Coutin 1996). The snapper fishery is divided into a long-line fishery that targets larger adult fish and a haul seine and mesh net fishery that targets sub-adults known as 'pinkies'.

4.10.2. Early Life History

The early life history of snapper in Victorian waters is poorly understood. Research from Japan has shown that snapper eggs are buoyant and hatch about two days after fertilisation (Fukuhara 1985). Snapper larvae metamorphose to the juvenile form at about 10 mm length, and begin to move from the pelagic to the demersal environment at about 15 mm length (Azeta *et al*. 1980; Tanaka 1985). In New Zealand, the transition from the pelagic to demersal environment begins shortly after metamorphosis (about one month old) and is suggested to be completed at about two months old when fish reach 30 to 40 mm in length (Francis *et al*. 1992). On the basis of this data, it is likely that larvae in Victorian waters spend one to two months in the water column, before moving to the demersal environment.

Very little information has been gathered on the early demersal stages of snapper in Australia. The available information suggests that the importance of depth, and the type of habitat in determining abundances of juvenile snapper depends on the location. For example, juvenile snapper (35 mm long) have been captured over shallow seagrass in the Gippsland Lakes (Rigby 1984), but a study in Botany Bay (New South Wales) showed that young juveniles (>32 mm long) were only abundant in bare soft sediment habitats deeper than five metres (Anon. 1981).

Juvenile snapper are rarely caught in Western Port, which suggests that this bay is not used as a spawning site by adult fish, despite the fact that recreational fishers catch large fish in this region. But, small snapper (four to nine cm long) are commonly caught at depths > 6 m in the northern half of Port Phillip Bay (Parry and Curry, unpublished data, Hamer *et al.* unpublished data). Recruitment of juvenile snapper into Port Phillip Bay shows strong variability from year to year, with large numbers of new recruits present in some years and absent in other years (Parry and Curry, unpublished data). Extensive sampling by Hamer (unpublished data) has shown that early post-settlement fish can be caught over a variety of habitats, including mud, rocky reef, algal beds and cunjevoi. However, there is some indication that habitat links may be density dependent. In Port Phillip Bay, during years when few snapper are caught, they are caught mostly over cunjevoi, but in years of high recruitment, juvenile snapper can be caught over a greater diversity of habitats. In New Zealand, juvenile recruitment appears to be strongly influenced by water temperatures during the spring spawning season, with warmer temperatures resulting in higher juvenile recruitment during the following year (Francis *et al*. 1992). The importance of environmental factors in influencing recruitment variability of Victorian snapper populations is currently unknown, however, there appears to be some positive relationship between abundances of recruited fish and temperature.

Larval snapper are likely to feed on phytoplankton such as diatoms and dinoflagellates and a range of zooplankton (Tanaka 1985). In Japan, recently-settled juveniles fed primarily on copepods, gammaridean amphipods, caprellids, mysids, fish eggs and polychaete worms (Tanaka 1985).

4.10.3. Adult Life History

Older juveniles and adults appear to utilise a range of habitats from bare soft-sediments to seagrass, algae and reef (MacDonald 1982, Hamer *et al.*, unpublished). They are highly transient and gregarious, and habitat associations are likely to be ephemeral.

Older juvenile and adult snapper do not appear to be selective feeders; they feed on a wide range of benthic organisms. In Victoria, juveniles have been observed to feed on grapsid crabs, isopods, amphipods, polychaete worms, small fish and molluscs (Winstanley 1983).

The diet of larger juveniles and adults may vary depending on the habitat they occupy. Winstanley (1983) suggests that snapper feed on molluscs, crustaceans, polychaetes and fish over reefs. The gastropod mollusc, *Philine angassi*, appears to be an important food item of snapper in soft sediment habitats (Parry *et al*. 1995).

Recreational fishing guide books suggest that snapper in Port Phillip Bay are widely distributed throughout the bay in both the deeper central parts of the Bay and in the shallower nearshore areas (Classon & Wilson 2002). Snapper are associated with reef habitat when they occur in shallower nearshore areas and reefs on the eastern side of Port Phillip Bay provide good land based fishing for snapper (Classon & Wilson 2002). In Western Port, snapper can be caught in all channels and particularly the deeper channels and they are widely distributed throughout the bay (Classon & Wilson 2002). November is considered to be the most productive month, although by March large fish apparently aggregate in the channels prior to leaving the bay for the winter and so this time of year is also considered a good time for recreational fishers to target snapper in Western Port (Crowley and Worsteling 2000, Classon and Wilson 2002). In Corner Inlet/Nooramunga snapper are also fairly widely distributed throughout the bay but found mainly in the main channels or deeper holes (AFN 1996, Classon and Wilson 2002).

4.11. Southern Calamari (*Sepioteuthis australis*)

4.11.1. General Information

Southern calamari *Sepioteuthis australis* inhabit coastal waters, including bays and inlets, around southern Australia (Kailola *et al*. 1993). This species is endemic to southern Australian and northern NZ waters and usually occurs in depths less than 100 m. Southern calamari are quick growing and generally do not live longer than about 18 months. They become reproductively mature at 12 months of age, at which time they have a mantle length of approximately 16 cm. They can reach a maximum length of 38 cm and a weight of 2.1 kg (Kailola *et al*. 1993).

There is little data about the range of salinities and temperatures tolerated by this species, although they are more commonly caught in marine waters (34-37 ppt) and they are likely to tolerate a range of temperatures from less than 10 to more than 20° C. Overall, there is little quantitative information which describes habitat affinities for this species.

4.11.2. Early Life History

Adult squid spawn throughout the year, although spawning activity peaks between August and January in Victorian waters (Kailola *et al*. 1993). Spawning takes place in shallow waters, usually less than 15 m deep, and eggs are laid in 4-5 finger-like capsules. The capsules are attached to rocky substrates, algae or seagrasses, often in masses of 50 to several hundred capsules (Kailola *et al*. 1993). Females sit on nests while squid hatch. Larval squid have a planktonic stage, the length of which is not known. Juvenile squid settle into the habitats in which they hatch from, and remain in these areas until they are about 7 cm in length.

4.11.3. Adult Life History

Squid abundances are generally higher during the breeding season, during which they spend more time in shallow (< 15 m) water. Adult squid are relatively transient, however, they spend most of their time over rocky reef and seagrass (either *Heterozostera* or *Amphibolis*). Adults are found throughout the year in Western Port near Flinders, Hastings and in Cat Bay over seagrass (*Zostera & Heterozostera*), rocky reef and in deeper channels (Hyslop & Johnson, pers com. in: Gunthorpe and Hamer 1998b). Larger juveniles can also be found around piers. By contrast, in Corner Inlet/Nooramunga adults are found throughout the bay and there is no information to suggest that they have any particular habitat affinities (Gunthorpe & Hamer 1998b).

Recreational fishing guide books suggest that good catches of calamari can be taken near the entrance to Port Phillip Bay and particularly between Portsea and Capel Sound with other good fishing areas at Balcombe Bay (Classon & Wilson 2002). The entrances to Western Port, particularly at Flinders and the nearshore areas on the western shores of Western Port, are also identified as sources of good recreational catches (Classon & Wilson 2002). There is less emphasis on recreational fishing for calamari in Corner Inlet/Nooramunga.

It is thought that adult calamari require clean water for spawning and in Western Port spawning occurs at the bottom end of the bay, where water clarity is excellent and extends north to Hastings (Hyslop & Johnson, pers com. in: (Gunthorpe & Hamer 1998b). The turbidity of waters in areas to the north of Hastings within Western Port are thought to restrict spawning (Hyslop & Johnson, pers com. in: Gunthorpe and Hamer 1998b). Fertilisation is internal and males may have 'harems' – although little is known about the ecology of this situation. Adult squid feed mainly on fish.

Overall there is little known about the use of different habitats by calamari.

5. Fisheries Independent Data Analysis

Ideally, habitat suitability modelling would include an extensive data collection stage in which a stratified sampling program ensured that all habitats are sampled. In this project we used existing datasets to create fisheries-independent models of habitat suitability. We analysed data for a number of species and life history stages dependent on the data available for each species and bay.

5.1. Environmental Parameters

We investigated a number of environmental parameters for each bay and the choice of these parameters depended on available information as well as information derived from Section 4 for the different species and life history stages. We present here the information investigated, however not all of these parameters were incorporated in the final models as they did not necessarily add to the model outputs.

5.1.1. Port Phillip Bay

Depth, sediment type and substrate type/biota were identified as being amongst the parameters that would have the most influence on the habitat affinities of the demersal species investigated in this study (Section 4). The spatial distribution of these parameters could also be accurately represented in the GIS with existing data (Section 3).

Salinity and temperature fluctuations in the bay are relatively small over the whole bay area (see Section 3 and Harris 1996) and within the range of tolerance for most species investigated (Section 4). Analysis of long-term data of the shallow areas, however indicated that certain areas within the bay were subject to greater salinity fluctuations than others. As a result of this analysis, the nearshore areas of the bay (within the 10 m depth contour) were classified as subject to 'low', 'medium' or 'high' variability for salinity (Figure 3.9). We only used salinity variability as a predictor variable for juveniles of species that apparently use these shallow areas and assumed that the larger mobile adults would be able to avoid sudden changes in salinity.

5.1.2. Western Port

The available GIS environmental layers for Western Port consisted of depth and substrate type/biota (Section 3). An analysis of salinity data (Section 3.3) also identified zones of salinity variability, but we do not present the salinity data here as without a more extensive spatial coverage of sampling sites we can have very little confidence in these results.

5.1.3. Corner Inlet

Existing GIS layers for environmental data in Corner Inlet consisted of depth and substrate type/biota (Section 3). There was no salinity data available for Corner Inlet.

5.2. Fisheries Independent Data

Suitability indices were derived from existing fisheries independent data using a mix of qualitative and quantitative approaches. The primary datasets used in this analysis are summarised below. Where replicate shots were taken in each area they were treated as independent samples because they often occurred at different depths and so increased the coverage of habitats sampled.

The available data varied for each species examined and not all species had satisfactory datasets even following a process where data was combined. Some species were caught in very low numbers and the majority of the shots recorded zeroes for those species (Table 5.1, Table 5.2 & Table 5.3). The datasets that were available to us had been collected for a range of purposes and so the ideal of a stratified sampling procedure across all the habitats of interest could not be met in this study. As a result, some habitats that are potentially suitable for a species may be under-represented or not sampled at all.

The data sources that were used for each bay are detailed below, however, it is important to note that due to gear limitations not all studies were used for each species or life history stage.

5.2.1. Port Phillip Bay Datasets

While a considerable amount of fishery independent data was available for Port Phillip Bay, most of the sampling either targeted older life stages in depths > 5 m over bare sediment or juveniles in shallow depths (<2 m). As a result, large gaps exist in the data for most substrate type/biota classes except bare sediment in depths >5 m.

5.2.1.1. Port Phillip Bay trawl program

This data provided the main information for all adult or sub-adult life history stages in depths from 5 to 25 m. The data consists of 22 depth stratified stations around the bay sampled annually in summer-autumn since 1990, although in some years seasonal data was also collected (Parry *et al*. 1995; Officer & Parry 1997). Fish were sampled using a wing trawl net (47 m long, 13 m wing spread, 5 m opening height and 45 m between trawl doors) with a mesh size of 44 mm. The sampling targeted four depths (7, 12, 17 & 22 m) on six different transects. For our analysis, we used actual depths recorded for each shot as we were interested in obtaining samples from across the entire depth range in Port Phillip Bay, although the sampled depths ranged around the four set depths. The substrate type/biota at all of the sites sampled in this program was bare sediment. For the purposes of our analysis the sampling sites were also categorised by bottom sediment type using the data outlined in Section 3.2.3.

5.2.1.2. Prawn otter trawl sampling data

A number of sites were sampled at 3 monthly intervals in 1990 and 1991 using a prawn otter trawl (12.8 m headline, 1.5 m opening height) with a mesh size of 44 mm (Hobday *et al*. 1999). No measurements were available for the distance between trawl doors but this was estimated to be approximately 15 m by the vessel master and calculations of area sampled were carried out using this figure. All sites sampled were between 10 and 20 m depth over bare sediment. Due to the low opening height of the net, only flathead were likely to be sampled with the same kind of efficiency as the wing trawl net (G. Parry pers comm), so only the flathead data was extracted from this dataset.

5.2.1.3. Pilot study of newly-settled snapper using modified beam trawl

Between December 1995 to April 1996, 110 sites in Port Phillip Bay were sampled as part of a pilot study aimed at evaluating the distribution of newly-settled snapper (Hamer *et al*. 1997). Sites sampled were all > 5 m in depth and the gear used consisted of a purpose-built beam trawl (5 m in length, 1.2 – 1.5 m opening height and 2.5 – 3 m fishing width) with a mesh size of 3 mm. No newly-settled snapper were caught in this survey, however, juvenile (< 15 cm) and adult flathead were adequately sampled with this gear and these data were extracted for our analysis of sand flathead.

5.2.1.4. Targeted sampling of newly-settled snapper using modified beam trawl

Using the beam trawl described above, 8 sites in Port Phillip Bay were sampled in February and March from 2000 to 2003 (Hamer & Jenkins 2004). The aim of the project was to investigate the spatial and temporal variation in recruitment of newly-settled snapper and, as above, records were also available for sand flathead < 15 cm and > 15 cm.

5.2.1.5. Shallow-water survey in seagrass and bare habitats

Three sites in Swan Bay and one site at St Leonards on the adjacent coast of Port Phillip Bay were sampled using a modified beach seine which was 10 m in length, had a 3 m drop and a mesh size of approximately 1 mm^2 (Jenkins *et al*. 1993). Hauls were 15 m in length and depths sampled were all less than 2 m. At each site both bare and seagrass habitat were sampled. Most sites were sampled monthly between October 1989 and the end of 1990, while the seagrass habitats only continued to be sampled monthly until the end of 1991.

5.2.1.6. Shallow-water surveys in reef, seagrass and bare habitats.

A study by Jenkins (1996a) to examine the importance of shallow water reef-algal habitats as nursery areas for commercial fish from southeastern Australia applied several different sampling approaches in Port Phillip Bay.

Three locations on the Bellarine Peninsula were sampled in reef, seagrass and bare habitat at a depth of approximately 0.5 m in 1993 and 1994 (Jenkins *et al*. 1996a). Fish were sampled with a seine net of 20 m length, a 2 m drop and a mesh size of approximately 1 mm.

Five sites around the bay were sampled over bare sandy habitat in two zones; an inshore zone which was within 10 m of the beach and characteristically of depths less than 1 m, and an offshore zone which was approximately 50 m from the beach and incorporated depths from approximately 0.5 to 1.6 m (Jenkins *et al*. 1996a). Sites were sampled monthly from June 1993 to May 1994 using a modified beach seine that was 10 m in length, had a 3 m drop and a mesh size of approximately 1 mm^2.

Six reef-algal sites in depths of 1-6 m were sampled around the bay by visual transects, gill nets and fish traps. Visual transects were undertaken during four surveys between 1992 to 1994 by SCUBA divers swimming along 50 m transects. Gill nets (with a net size of 30 m long, 3 m deep and a 10 mm stretch mesh size) and fish traps were deployed in March and April 1993 (Jenkins *et al*. 1996a).

Table 5.1. Summary of combined Port Phillip Bay datasets used in analysis for independent habitat suitability indices. Different combinations of datasets were used for different species depending on the ability of the gear types to sample each species – see text for further details and Appendix 4: Table 14.1.

Species	Total number of fish caught	Total area sampled (m^2)	Total number of shots (replicates)	Total number of presence: absence records
Sand flathead > 15 cm	104,599	14,066,662	993	677 : 316
Sand flathead < 15 cm	126	221,857	240	69 : 171
Sand flathead juveniles	19	75,012	1,124	14 : 1,110
King George whiting	1,124	12,659,052	599	62 : 537
King George whiting juveniles	8,204	98,912	1,602	617 : 985
Greenback flounder	441	12,659,052	599	137 : 462
Greenback flounder juveniles	1,213	98,912	1,602	276 : 1,326
Snapper	4,319	12,659,052	599	190 : 409
Snapper juveniles	572	122,799	129	81 : 48
Southern calamari	3,736	12,659,052	599	415 : 184
Rock flathead	17	12,659,052	599	62 : 537

5.2.2. Western Port Datasets

There was less data available for habitat suitability modelling for Western Port than Port Phillip Bay. The following datasets were used for quantitative analysis. In many instances, low numbers of fish were caught and in most cases the majority of shots recorded zero fish for many of the species examined (Table 5.2).

5.2.2.1. Recruitment monitoring of newly-settled snapper using modified beam trawl

Recruitment monitoring was undertaken in Western Port using a purpose-built beam trawl, 5 m in length, 1.2 – 1.5 m opening height and 2.5 – 3 m fishing width, with a mesh size of 3 mm (Hamer & Jenkins 2004). Sampling was undertaken over four summer/autumn recruitment seasons between 2000 – 2003. Snapper recruits occurred in very low densities, so this data was not used to model newly-settled snapper in Western Port. The gear also sampled sand flathead effectively and we used this data for the sand flathead habitat suitability indices.

5.2.2.2. Bottom trawling data from Western Port

Bottom trawls were carried out around Phillip Island using a demersal balloon trawl net (with a 13 m wide by 5 m high opening) during Spring/Summer of 1986 and 1987 (Hobday 1992). Trawling was restricted to the channel areas due to strong tides and risk of gear fouling in shallow waters. The focus of the study was on prey species of the little penguin and so sampling stations were within a 20 km radius of the Phillip Island Penguin Reserve. This dataset was used for the modelling of sand flathead

5.2.2.3. Shallow water sampling in seagrass and bare

Gill and seine nets were used to sample shallow-water seagrass, bare and channel habitats at 5 sites in Western Port seasonally between 1989 and 1990 (Edgar *et al*. 1993). Small fish were sampled with a seine net of 15 m length, a 3 m drop and a 1 mm mesh, while larger fish were sampled with 50 m monofilament gill nets with a 3 m drop and one panel of 64 mm mesh and another panel of 108 mm mesh (Edgar *et al*. 1993). Data was extracted from the published manuscript for this study, so pooled replicate data rather than raw data was used.

5.2.2.4. Shallow water sampling in mangrove and bare habitat

Three sites in Western Port were sampled in all seasons of 2002 in mangrove and bare habitats (Hindell & Jenkins 2004). Gear types used were beach seines, gill nets and fyke nets. The gill nets were used to target larger (>15 cm) mobile fishes and were 1.5 m deep, 35 m long and composed of five panels of different mesh sizes (2.5, 3.8, 5.0, 6.3 and 7.6 cm stretch mesh). Fyke nets were used to target smaller fish and the main 'bag' of each fyke was made with four square rings (70 cm x 70 cm) and a wing (10 m long x 70 cm deep) was attached to each side of the 'bag' opening. A honey comb mesh (6 mm diameter holes) was used. The beach seine net was 10 m long with a 2 m drop and a 10 m rope attached to each end with a mesh of 1 mm.

Table 5.2. Summary of combined Western Port datasets used in analysis for independent habitat suitability indices. Different combinations of datasets were used for different species depending on the ability of the gear types to sample each species – see text for further details and Appendix 4: Table 14.2.

Species	Total number of fish caught	Total area sampled (m^2)	Total number of shots (replicates)	Total number of presence: absence records
Sand flathead > 15 cm trawl	98	1,663,904	56	19 : 37
Sand flathead gill nets	7,722	3,020	ND[1]	ND[1]
Sand flathead juveniles	45	26,380	403	31 : 372
King George whiting gill nets	0.73*	3,020	ND[1]	ND[1]
King George whiting juveniles	66	26,380	403	38 : 365
Greenback flounder juveniles	23	26,380	403	17 : 386
Snapper trawl	37	1,663,904	56	12 : 44
Yellow eye mullet gill nets	146*	3,020		ND[1]
Yellow eye mullet juveniles	21	26,380	403	7 : 396
Southern calamari trawl	440	1,663,904	56	26 : 30
Rock flathead gill nets	9.8*	3,020	ND[1]	ND[1]
Rock flathead juveniles	34	26,380	403	24 : 379

* Pooled gill net data was used to calculate indices for these species – some of this data was extracted from a published manuscript and no raw data was available so these values represent the total of the averages per replicate gill net sample.

[1] As above, no raw data was available and so it was not possible to calculate presence / absence records.

5.2.3. Corner Inlet Datasets

There was considerably less data available for fish habitat suitability modelling for Corner Inlet compared to Port Phillip Bay and even Western Port. The following datasets were used for quantitative analysis and numerous other sources were used for a qualitative analysis and these are referred to where relevant in the text. The majority of the data available was aimed at sampling juvenile fish in shallow water habitats, although total numbers of fish caught were low and the majority of shots did not catch any of the species of interest (Table 5.3).

5.2.3.1. Recruit monitoring of newly-settled snapper using modified beam trawl

Recruitment monitoring was undertaken in Corner Inlet using a purpose-built beam trawl, 5 m in length, 1.2 – 1.5 m opening height and 2.5 – 3 m fishing width, with a mesh size of 3 mm (Hamer & Jenkins 2004). Sampling was undertaken over four summer/autumn recruitment seasons between 2000 to 2003. Snapper recruits occurred in very low densities and so this data was not used to model newly-settled snapper, but the gear also sampled sand flathead effectively and we used this data for the sand flathead suitability indices.

5.2.3.2. Shallow water survey in seagrass and bare habitats

Six sites were sampled with both seagrass and bare habitats at depths ranging from intertidal to approximately 1 m depth (Jenkins *et al*. 1993). Fish were sampled with a modified beach seine, 10 m in length with a drop of 3 m and a mesh size of 1 mm. Hauls were 15 m in length. Sampling was undertaken bimonthly at most sites between October 1989 and August 1991.

5.2.3.3. Shallow water surveys in mangrove and bare habitats

Three sites in Corner Inlet were sampled in each of the four seasons in mangrove and bare habitats (Hindell & Jenkins 2004). Gear types used were beach seines, gill nets and fyke nets. The gill nets were used to target larger (>15 cm) mobile fishes and were 1.5 m deep, 35 m long and composed of five panels of different mesh sizes (2.5, 3.8, 5.0, 6.3 and 7.6 cm stretch mesh). Fyke nets were used to target smaller fish and the main 'bag' of each fyke was made with four square rings (70 cm x 70 cm) and a wing (10 m long x 70 cm deep) was attached to each side of the 'bag' opening. A honey comb mesh (6 mm diameter holes) was used. The beach seine net was 10 m long with a 2 m drop and a 10 m rope attached to each end, the mesh size was 1 mm.

Table 5.3. Summary of combined Corner Inlet datasets used in analysis for independent habitat suitability indices. Different combinations of datasets were used for different species depending on the ability of the gear types to sample each species – see text for further details and Appendix 4: Table 14.3.

Species	Total number of fish caught	Total area sampled (m^2)	Total number of shots (replicates)	Total number of presence: absence records
Sand flathead	202	65,966	65	32 : 33
King George whiting juveniles	76	26,818	388	25 : 363
Greenback flounder juveniles	170	26,818	388	60 : 328
Yellow eye mullet juveniles	85	26,818	388	14 : 374
Rock flathead juveniles	61	26,818	388	45 : 343
Yank flathead juveniles	47	26,818	388	38 : 350

5.3. Generating Habitat Suitability Indices

We adopted a rationale whereby a number of approaches to model generation were trialed. The model outputs were compared and where there was good agreement between the different approaches we could have increased confidence in the model predictions (Norcross *et al*. 1999). The simplest approach produced 'suitability indices' for each level of an environmental variable independently of other environmental parameters to be investigated. The two main methods used to determine these independent suitability indices were a habitat affinity approach (percentage of a fish species caught in relation to the percentage of each habitat sampled) and bivariate logistic regression (Port Phillip Bay only - see below). For Corner Inlet and Western Port, where data was limited, we also reviewed available literature to produce qualitative suitability indices for different habitat types. These indices were expressed in broad terms and combined information from both bays.

A limitation on the current project was that the analysis was dependent on data available from existing surveys. Data was not available across the entire range of depths or habitat types for each species and life stage due to the different aims of the surveys and limitations of the gear types used during these surveys. As a result, we had to use qualitative methods to determine indices for areas not covered by the data. In some cases we extrapolated the logistic regressions beyond the available data range and have indicated where we have done this. In other instances, we used expert opinion and the literature review to provide qualitative estimates of suitability indices where the data did not allow us to calculate them by the methods detailed below. We have indicated where suitability indices are derived from qualitative methods.

Where possible, datasets were pooled to improve the spatial coverage of the input data. Shallow-water datasets were analysed separately to the trawl datasets as the sampling methodologies were different and these surveys were, for the most part, targeting juveniles rather than adults. We did not consider the shallow-water datasets large enough to split and have only presented tables of suitability indices for this data. Where possible we looked at seasons separately in an effort to identify ontogenetic shifts in habitat use in the early life-history stages of the different species.

5.3.1. Qualitative Approaches to Suitability Indices

For Western Port and Corner Inlet, published and non-published reports that included fish sampling data for these bays were examined and statements recorded that linked fish distributions to habitat types. These statements were then combined to provide broad categories of 'high', 'medium' or 'low' habitat suitability for each habitat type. High suitability represented statements that suggested a species was abundant or 'found only' in a certain habitat. Medium suitability tended to be derived from statements that reported that a species was 'less common' in a certain habitat and 'low suitability' statements tended to be that a species 'actively avoided' a habitat or no mention was made of a species in that habitat. There was not enough information available on a bay by bay basis, so the indices that resulted from this process are the same for both Western Port and Corner Inlet.

5.3.2. Habitat Affinities

Habitat affinities for each environmental variable were calculated as the ratio of the proportion of fish caught in a particular habitat category to the proportion of that habitat that was sampled in each bay. In Port Phillip Bay, for example, habitat categories for depth consisted of 6 groups while sediment types fell into 5 classes (Table 5.4). Substrate type/biota categories consisted of reef, seagrass, seagrass-bare edge and bare sediment with the additional categories of mangrove and mangrove edge in Western Port and Corner Inlet.

The resulting ratios for each environmental parameter and species were then scaled to the maximum for each bay, so that all values fell between 0 and 1. These values were considered to be the 'suitability indices' for each particular environmental variable.

5.3.3. Bivariate (single parameter) Logistic Regression

A logistic regression approach allows the use of presence/absence data and is more suited to datasets where there are a large number of zero records. It can also incorporate both continuous and categorical predictor variables. Logistic regression on presence/absence data was carried out for species on each continuous environmental parameter separately. The resulting regression equations allow predictions of the probability of encountering a species to be calculated at each level of the independent variable.

The probabilities of encounter values were treated as 'suitability indices'. Bivariate regression analyses was undertaken using SAS® Version 8e.

Due to a lack of data, logistic regression was only applied to the Port Phillip Bay data. Depth was divided into 6 depth intervals and the regression equations calculated on the depth interval number. Despite re-scaling of the depth it was still treated as a continuous variable in the analyses.

The inclusion of a quadratic function was investigated and where the inclusion of this function improved the fit of the model, predicted probabilities of encounter were calculated with a polynomial term. Sediment type was treated as a categorical variable in the logistic regressions and so the predicted probability of encounter is actually the observed proportion of samples in which a species was present in each sediment-type category. Residuals were examined in order to identify 'outliers' and observations that did not 'fit' the model.

For the logistic regression, the data were randomly split into two halves, wherever practical, and the model developed on one half of the dataset and the other half of the data used for model validation. This is a procedure commonly recommended in statistical texts (e.g. Hosmer and Lemeshow 1989, Quinn and Keough 2002). In some instances, the dataset was considered too small to split and this has been indicated in the following results section. Where possible, plots are presented of 'predicted' versus 'observed' where the 'predicted' represents the regression equation from the model development half of the data and the 'observed' is the data from the 'validation' half of the dataset.

5.3.4. Multiple Logistic Regression

Additive logistic regression models were developed for some of the species examined in Port Phillip Bay following a check for collinearity in the predictor variables. As the available data for Western Port and Corner Inlet was limited, this approach was not pursued for these bays. Quadratic functions were investigated to see if they improved the fit of the model and, as recommended by Hosmer and Lemeshow (1996), where exploratory data analysis indicated that re-scaling of predictor variables would be useful, re-scaled variables were included. Goodness of fit statistics were examined, but because goodness of fit tests for logistic regression models with continuous predictors are difficult to interpret (Quinn & Keough 2002), we also calculated the 'Hosmer-Lemeshow' $\hat{C}$ statistic. A P-value greater than 0.05 for this statistic indicates that the observed data came from a population in which the fitted logistic regression model is true (Quinn and Keough 2002). We also calculated the r^2_L statistic, an analogue of the r^2 value used as a measure of explained variance in ordinary least-squares regression and used this statistic to indicate how well the logistic regression model fitted the data.

Model selection was relatively straightforward due to the small numbers of predictor variables under consideration. As well as the goodness of fit statistics, the Akaike Information Criteria statistic was used to aid in selecting the 'best model' with the fewest number of predictors (Tabachnick & Fidell 1989; Quinn & Keough 2002). In this case the output from the multiple regression model was the Habitat Suitability Index, or the probability of encountering a species for a particular combination of habitat parameters. SAS® Version 8e was used for these analyses.

The multiple logistic regression models for species in Port Phillip Bay only explained a small proportion of the uncertainty in the datasets and as a result this method has not been included in the development of suitability indices presented below.

5.3.5. Linear Regression

Linear regression approaches were also examined, but due to the large number of zero records in most of the datasets only the data for transformed sand flathead (> 15 cm) in Port Phillip Bay met the assumptions of normality and homogeneity of variance of the linear regression methods. Habitat suitability in Port Phillip Bay for sand flathead apparently increased with increasing depth and decreased as sediment grain size increased. Comparing model predictions with observations from the 'validation' half of the dataset indicated that there was a great deal of spread around the predicted model that encompassed several suitability classes. This suggests that using single parameter linear regression analysis to model habitat suitability with the available data is likely to include a wide margin of error and may not be particularly useful. As a result, this method has not been included in the development of suitability indices presented below.

5.3.6. Measuring Confidence in Habitat Suitability Index Values

The habitat suitability index tables include a measure of confidence for each suitability index. The confidence measures were derived from a combination of quantitative and qualitative information.

For the logistic regression method, we randomly split all the datasets in two. Half the dataset was used to generate the suitability index, or the probability of encounter (model dataset) and the remaining half of the dataset was used to provide a simple measure of classification success (test dataset). The proportion of times a species was predicted to be present/absent within a habitat (suitability index) was compared to the proportion of times it was actually observed to be present/absent in the test dataset. The classification success was expressed as the percentage of occasions the model correctly predicted the observation. These classification success values were considered as a guide whereby high classification success (>90%) suggested we could have high confidence in the suitability index. However, where expert opinion indicated it appropriate, we adjusted the confidence measure.

Suitability indices that had been created either from extrapolating outside of the range of our existing data, or based solely on expert opinion as no data existed, were automatically assigned a 'low' confidence rating to indicate that these figures were based solely on qualitative information. If there was a strong body of evidence to support the assigned suitability indices the confidence ratings were upgraded from 'low' to 'medium'.

The confidence ratings for the suitability indices generated with the habitat affinity method were simply assigned as 'medium' if they were calculated from existing data and 'low' if they were habitats that had not been sampled. 'Medium' was the highest rating given to the habitat affinity values as the dataset had significant limitations and without an independent test of classification success we felt unable to assign a higher confidence rating.

5.4. Producing Habitat Suitability Models

The general approach to transferring habitat suitability indices into spatial models or maps of habitat suitability in a GIS is described in Section 2.5.3. In this study, we adopted an approach consistent with Rubec *et al.* (1999) and Christensen *et al.* (1997) which involved applying the following steps in a GIS to produce habitat suitability models based on habitat suitability indices derived from analysis of the fishery independent data:

1. Load environmental parameter grids and habitat suitability index tables to the GIS (ArcView & ArcView Spatial Analyst extension).

2. Select environmental parameter grids (ie. depth, substrate type/biota or sediment) to model for a species.

3. Link habitat suitability index tables to corresponding environmental parameter grids.

4. Re-classify cell values in each environmental parameter grid to equal its corresponding suitability index value.

5. Overlay reclassified grids and calculate the arithmetic mean of suitability index values to a new composite HSI model grid.

6. Re-scale composite HSI model grid to the maximum value by dividing each cell by the maximum cell value found in the grid.

7. Classify re-scaled grid with the HSI legend; low (0 to 0.25), medium (0.25 to 0.75) and high (0.75 to 1.00).

8. Display final re-scaled composite grid and export map.

We used a simple arithmetic mean to combine the reclassified grids into a composite HSI model grid (step 5 above). Previous habitat suitability modelling studies used a geometric mean (Section 2.5.3 and Figure 2.1), but since this study did not assign any parameters with a 0 or unsuitable habitat value, using a geometric mean was unnecessary.

Composite HSI model grids were produced using suitability indices derived from both the habitat affinity approach and logistic regressions and with different combinations of environmental variables. The models were then compared to identify any areas of consistent suitability classification across the models.

A customised ArcView Interface and habitat suitability modelling wizard was developed to simplify the data processing steps above (see Section 7).

5.5. Evaluating Habitat Suitability Models

It is preferable for models to be tested against fishery independent data from the same locality. Our fishery independent data was limited for most of the species under investigation due to the number of fish recorded, the spatial distribution of sampling sites and/or the range of habitats sampled. Most of the samples were records of adult fish over unvegetated sediment in depths > 7 m.

All of the available fishery independent data were used to create suitability indices with the habitat affinity method. In the case of the logistic regression analysis, we randomly split the data into a model half to determine the probabilities of encounter and a validation half. This enabled us to test the predicted probabilities of encounter against real-world observations. However, as the range of habitats sampled were limited (mostly unvegetated sediment at depths >7 m) and had a limited spatial distribution, they were unsuitable for us to also use the validation half of the dataset to test the composite spatial habitat suitability models. As a result, we did not have sufficient data to test our composite habitat suitability models based on SI's generated by either the habitat affinity or logistic regression approach, and our evaluation of these models is restricted to a qualitative assessment and comparison with known patterns of species habitat usage.

5.6. Port Phillip Bay Habitat Suitability Indices & Models

5.6.1. Sand Flathead (*Platycephalus bassensis*) > 15 cm

There were extensive records of sand flathead > 15 cm and this life stage was considered to be adequately sampled by all trawl gears used in the datasets collected in depths > 5 m. As a consequence the four trawl datasets (see Section 5.2.1) were pooled for the analysis of sand flathead which provided a good spatial coverage of Port Phillip Bay.

We had few records of sand flathead < 15 cm (126) or newly recruited flathead juveniles (19) (Table 5.1) and this data were considered to be inadequate to calculate suitability indices. Some of the available shallow-water data did not split newly-recruited flathead into species, so we were further restricted by having to look at a pooled juvenile flathead class. Catches of rock flathead in surveys, where flathead were split into species, were so low that it suggested that most of the fish included in the pooled flathead category were likely to be sand flathead although yank flathead may also feature. No flathead were recorded from the 1997 shallow water-survey, so this data was excluded as flathead recruitment may have been very low that year.

5.6.1.1. Habitat Affinities

Habitat affinities indicated that sand flathead had a higher affinity for deeper areas and clayey sediment (Table 5.4). Our dataset only included samples over bare substrate type/biota, so habitat affinity values for other substrate type/biota categories were based on expert opinion only.

5.6.1.2. Bivariate Logistic Regression

Suitability indices calculated using a single-parameter logistic regression approach indicated that there was a reasonably high probability of encounter (> 0.5) across all depths, except intertidal, and sediment types (Table 5.7). The predicted probability of encounter from the regression analysis was compared with the observed proportions encountered in the validation datasets and indicated a reasonable fit of the model for all parameters, with the exception of the lower depth groups (Figure 5.1 & Figure 5.2). There were very few data points on which to base or test the model for these lower depth groups.

5.6.1.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices are presented in Figure 5.3. The habitat suitability model based on logistic regression SI's for depth and sediment (Figure 5.3A) shows a larger total area of high suitable habitat compared to the model based on habitat affinity SI's for the same variables (Figure 5.3B). This difference reflects the higher overall SI values derived from the logistic regression analysis, but the overall pattern of increasing habitat suitability with increasing depth and smaller sediment grain size is consistent between the two methods. Including substrate type/biota in the habitat affinity model (Figure 5.3C) increased the total area of high and medium suitability habitat and produced a model which is more consistent with the logistic regression model. The main difference between the model based on logistic regression SI's for depth and sediment (Figure 5.3A) versus the model based on habitat affinity SI's for depth, sediment and also substrate type/biota (Figure 5.3C) is the areas of low suitable habitat in the habitat affinity model that corresponded to seagrass and reef habitats.

The combined logistic regression and habitat affinity models results in most of the Bay being identified as medium suitability, with the deeper areas of the central Bay and Corio Bay and Geelong Arm identified as high suitability (Figure 5.3D). The other areas of high suitability habitat in Figure 5.3D correspond to sand-clay in medium depths (10-20 m).

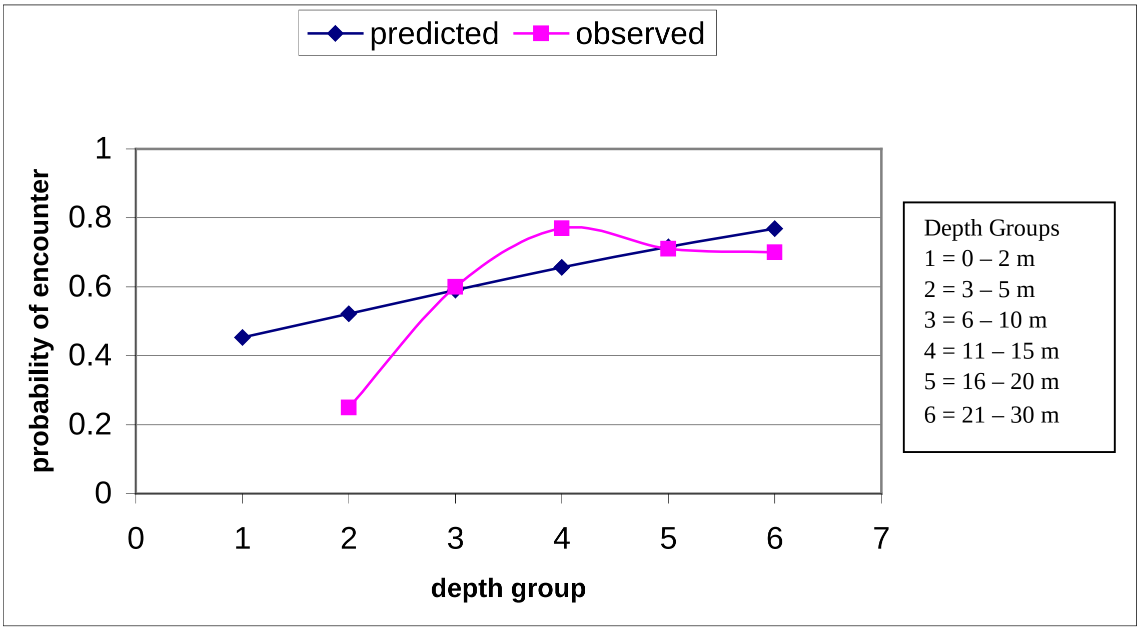


Figure 5.1. Comparison of predicted probabilities of encounter for sand flathead (>15 cm) calculated from the regression analysis on the 'model' half of the dataset and compared with observed proportions encountered in the 'validation' half of the dataset. The regression equation was calculated using the depth group number. Note that the predictions are extrapolated out of the range of the dataset and there were few data points in the 3 to 5 m depth group.

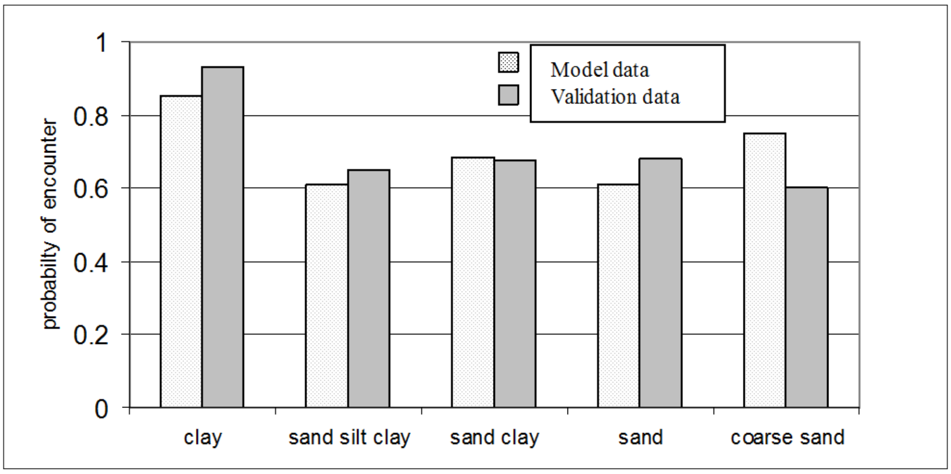


Figure 5.2. Comparison of the observed proportion of sand flathead encountered in each sediment type in the 'model' half of the dataset and the 'validation' half of the dataset.

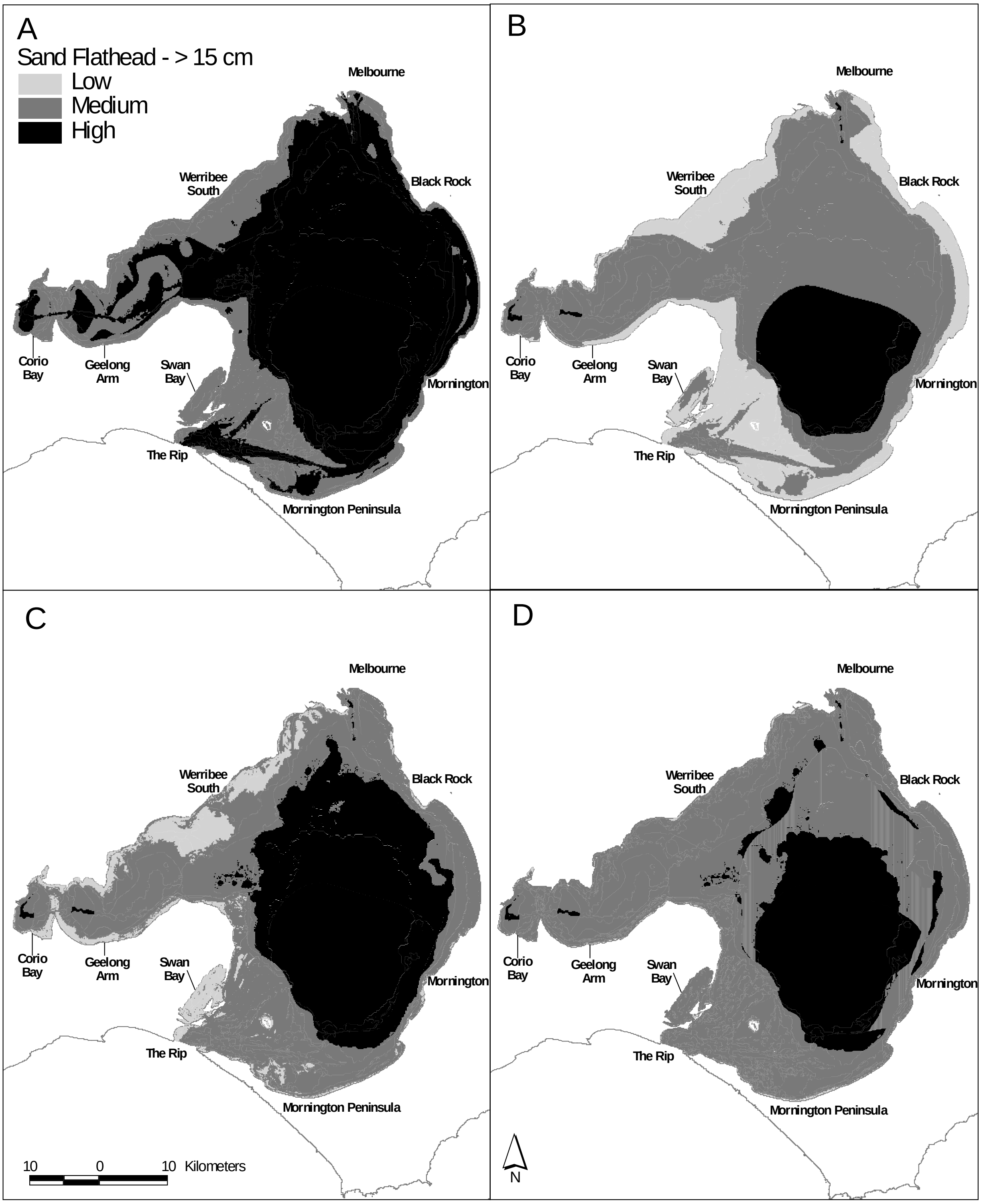


Figure 5.3. Sand flathead (> 15 cm) Port Phillip Bay habitat suitability models. A: Logistic regression SI's – depth & sediment. B: Habitat affinity SI's – depth & sediment. C: Habitat affinity SI's – depth, sediment & substrate type/biota. D: Mean of models A, B & C.

5.6.2. Greenback Flounder (*Rhombosolea tapirina*) Adults

Adult greenback flounder were only considered to be adequately sampled by the larger gear type in depths > 5 m and as a consequence, only the 10 year trawl monitoring dataset (Parry *et al*. 1995; Officer & Parry 1997) was used for the analysis of this species.

5.6.2.1. Habitat Affinities

The habitat affinities (Table 5.4) did not show any clear pattern of species affinities with depth. Lowest values were recorded from depth ranges of 10-20 m with all other depth ranges having a high affinity value for this species. Greenback flounder had a high habitat affinity for clay sediment, with all other sediment types having low values. Habitat affinity values for substrate type/biota were derived from expert opinion only and indicated that bare substrate had the highest habitat affinity.

5.6.2.2. Bivariate Logistic Regression

A single parameter logistic regression was used to calculate suitability indices, and indicated that the 'probability of encounter' was low to medium at all depths (< 0.5) (Figure 5.4 & Table 5.7). The probability associated with clay substrate was considerably greater than the other sediment types (Figure 5.5 & Table 5.7). Including a quadratic function in the depth regressions improved the fit of the model and suitability indices are based on the polynomial regression equation. Comparing the predicted probabilities derived from the regression analysis of the 'model' half of the dataset with observed proportions in each group in the 'validation' half of the model indicates a good fit (Figure 5.4 & Figure 5.5).

5.6.2.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices are presented in Figure 5.6. The models all identify the deep clay area in the central Bay as high suitability and most of the intermediate depths of 5-15 m as medium suitability. The habitat suitability model based on logistic regression SI's for depth and sediment (Figure 5.6A) differs from the other models by highlighting shallow depths of 0-2 m as high suitability and sandy sediment in medium depths of 10-15 m as low suitability. The habitat suitability model based on habitat affinity SI's for depth and sediment (Figure 5.6B) shows a band of low suitability in the medium depths of 15-20 m. Adding the substrate type/biota habitat affinity SI values to the habitat suitability model increased the area of medium suitable habitat, while the high suitable habitat remained unchanged (Figure 5.6C).

The combined logistic regression and habitat affinity model results in most of the Bay being identified as medium suitability, with the deep clayey areas of the central Bay and Corio Bay and Geelong Arm identified as high suitability (Figure 5.6D).

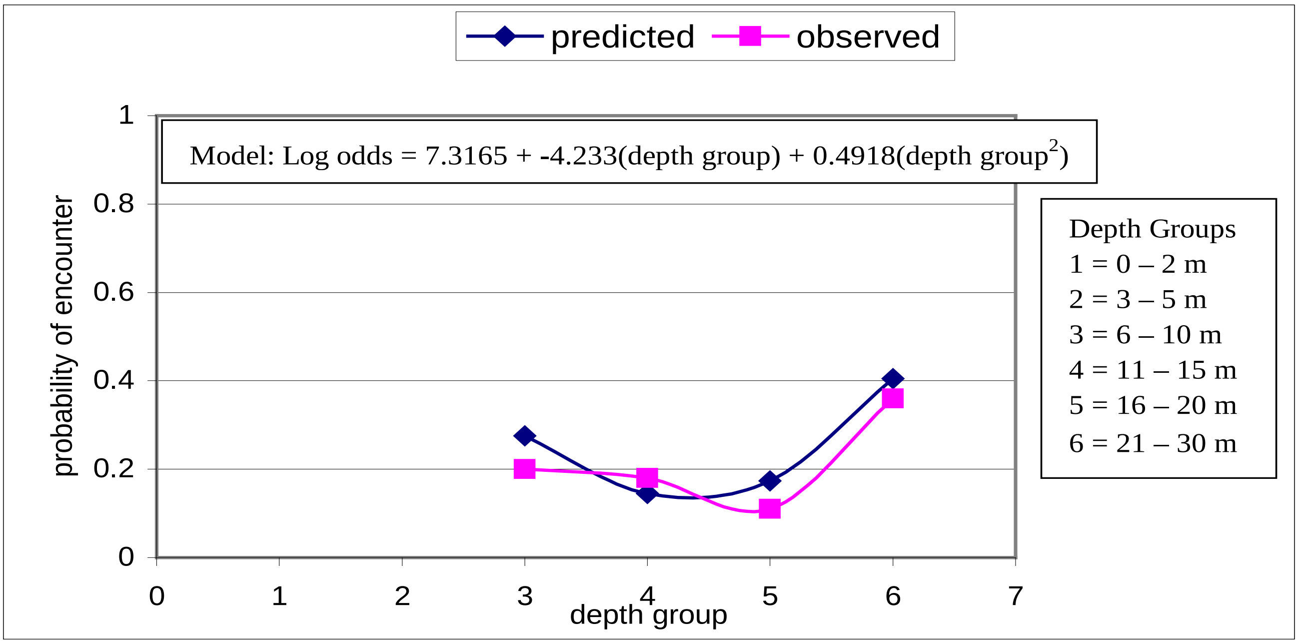


Figure 5.4. Comparison of predicted probabilities of encounter for greenback flounder calculated from the regression analysis on the 'model' half of the dataset and compared with observed proportions encountered in the 'validation' half of the dataset. The regression equation was calculated using the depth group number. The dataset from which the model was derived did not include depths less than 5 m.

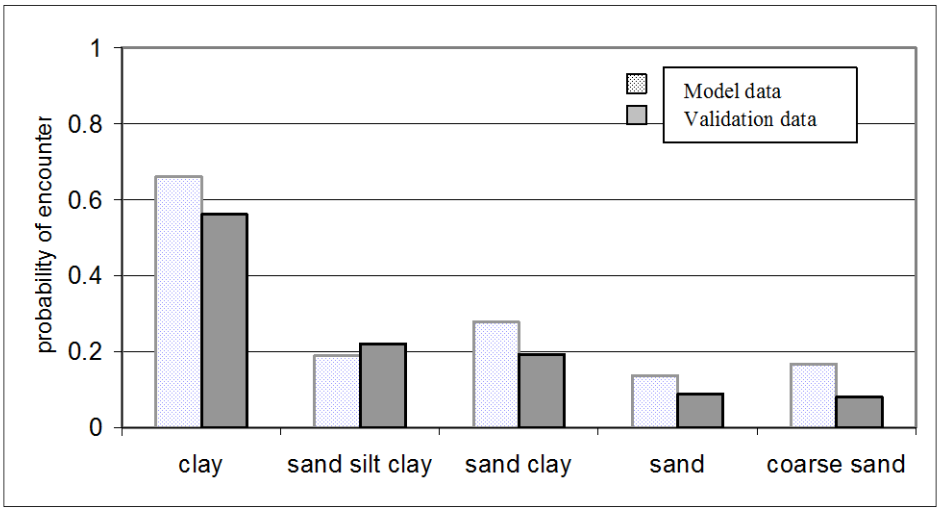


Figure 5.5. A comparison of the observed proportion of greenback flounder encountered in each sediment type in the 'model' half of the dataset and the 'validation' half of the dataset.

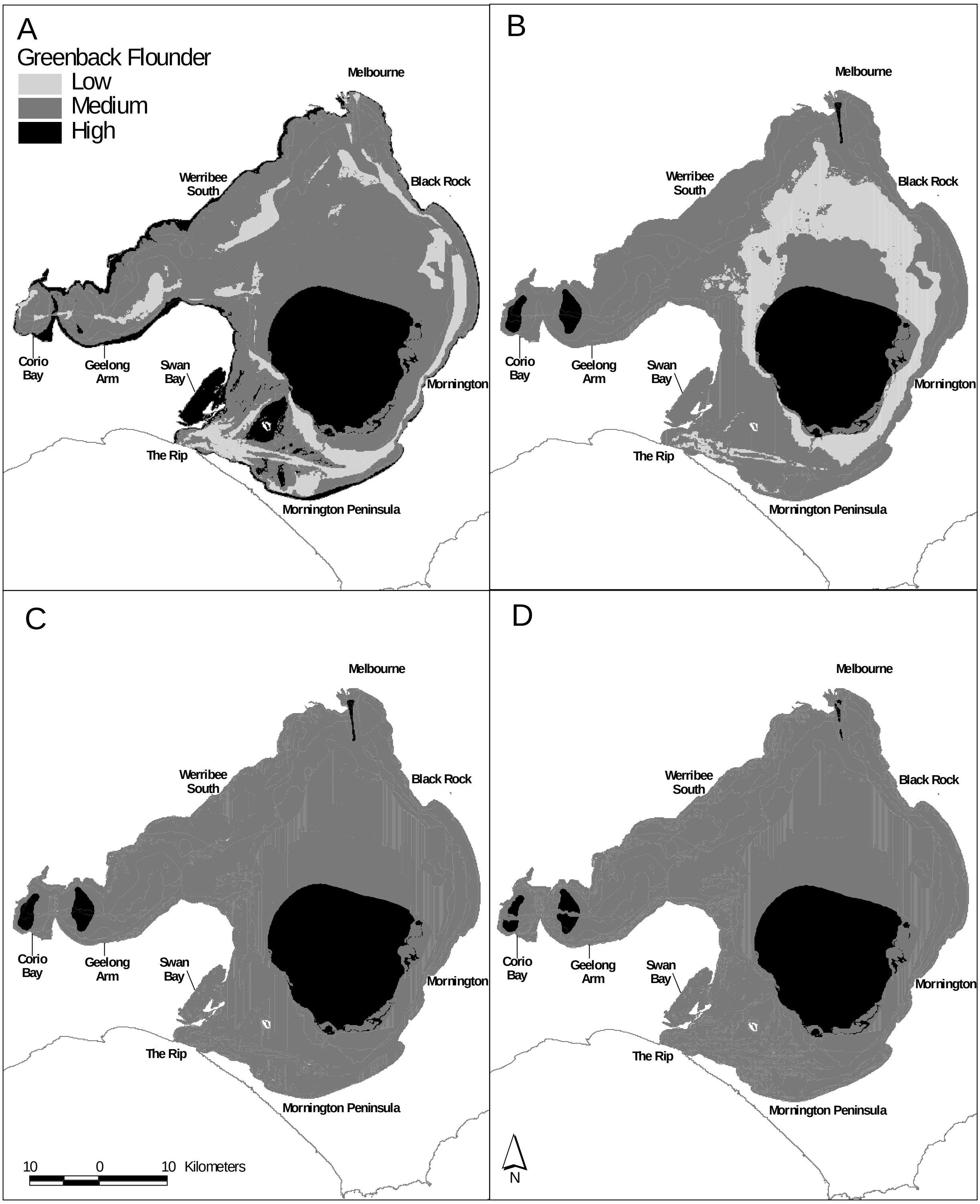


Figure 5.6. Greenback flounder (adults) habitat suitability models. A: Logistic regression SI's – depth & sediment. B: Habitat affinity SI's – depth & sediment. C: Habitat affinity SI's – depth, sediment & substrate type/biota. D: Mean of models A, B & C.

5.6.3. Greenback Flounder Juveniles (shallow-water data)

5.6.3.1. Habitat Affinities

Pooling greenback flounder data across seasons indicated that juvenile fish were more abundant in spring (larvae in Port Phillip Bay settle out in late winter/spring) and had a greater affinity for bare sandy intertidal areas and where salinity fluctuations were liable to be greatest (Table 5.4, Table 5.5 & Table 5.6). Looking at seasons separately, there were a few interesting differences that are worth emphasising. In spring, juvenile flounder showed a high affinity for intertidal areas on bare sediment and on medium/coarse sand. In summer, the juveniles showed a high affinity for bare medium sands and all of the shallow depths sampled. Only 13 fish were caught in autumn, so we did not analyse this data separately.

5.6.3.2. Bivariate Logistic Regression

Probability of encounter was low for juvenile greenback flounder when the data is pooled across seasons. Spring had the highest probability of encounter (Table 5.9). Considering the spring data, probability of encounter for juvenile greenback flounder was greatest on bare fine sand, either intertidal or at 1-2 m (a quadratic relationship with depth). In summer, the highest probabilities were associated with bare substrate with no significant relationship with sediment type for a linear or quadratic relationship ($P > 0.05$). There were so few 'presence' records for autumn and winter that we did not analyse this data separately as probabilities would be very low in all cases.

5.6.3.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices for all seasons are presented in Figure 5.7. The models produced from habitat affinity and logistic regression SI's for depth, sediment and substrate type/biota both showed most of the Bay as medium suitability for juvenile greenback flounder (Figure 5.7A & B). Both models highlight areas of high suitability habitat in shallow bare areas, while the habitat affinity model showed seagrass/macroalgae areas as low suitability. Removing substrate type/biota as a variable from the model resulted in a clear distinction between low suitability in the deeper areas of the Bay (> 5 m) with all shallow areas shown as medium suitability (Figure 5.7C). The combined model showed a general pattern of low suitability in deeper bare areas with most of the shallow-intermediate depths being medium suitability, with the exception of Corio Bay and Geelong Arm which were also mostly low suitability (Figure 5.7D). The shallow macroalgae beds offshore from Werribee on the west coast were also shown as low suitability in the combined model.

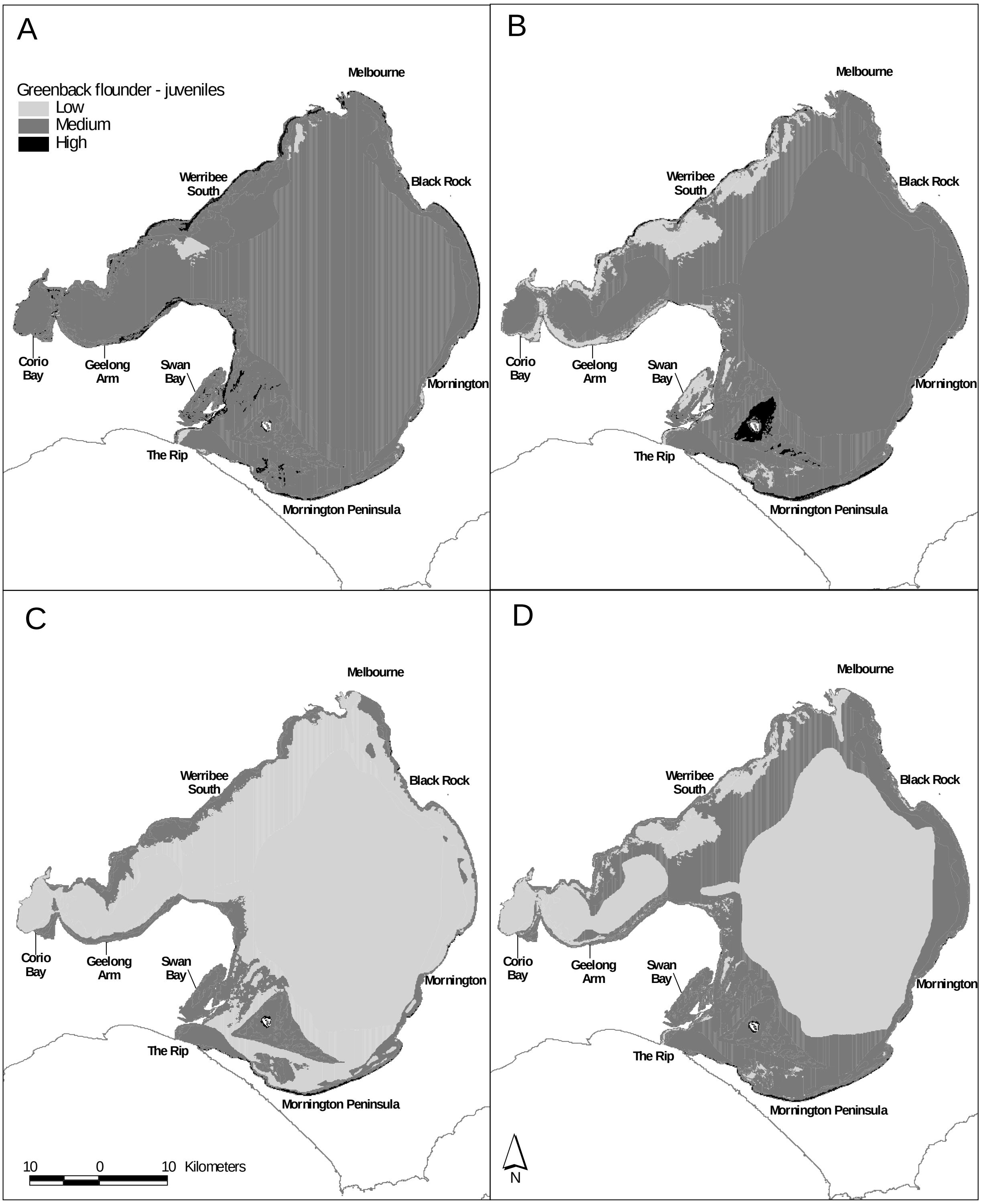


Figure 5.7. Greenback flounder juveniles (all seasons) habitat suitability models. A: Logistic regression SI's – depth, sediment & substrate type/biota. B: Habitat affinity SI's – depth, sediment & substrate type/biota. C: Habitat affinity SI's – depth & sediment. D: Mean of models A, B & C.

5.6.4. King George Whiting (*Sillaginodes punctata*) Sub-adults

Sub-adult King George whiting were only considered to be adequately sampled by the larger gear type in depths > 5 m and as a consequence, only the 10 year trawl monitoring dataset (Parry *et al*. 1995; Officer & Parry 1997) was used for the analysis of this species. Suitability indices for habitat type were estimated from available information and expert opinion and are shown in Table 5.4 & Table 5.7.

5.6.4.1. Habitat Affinities

Habitat affinity values indicate that this species has a very low affinity for depths > 10 m and high affinities for clayey sediment in Port Phillip Bay (Table 5.4). There was insufficient data to calculate habitat affinities for substrate type/biota, but expert opinion indicated that seagrass had the highest affinity.

5.6.4.2. Bivariate Logistic Regression

The single parameter logistic regression approach to creating suitability indices suggested that the entire area sampled by the trawl datasets (depths > 5 m) had a low suitability for King George whiting (suitability index < 0.2, Table 5.7). A comparison of the probabilities associated with the depth groups from the predictive model and the observed 'validation' data suggested the fit allows reasonable confidence in the predictive capabilities of the model (Figure 5.8).

The comparison of observed proportions encountered in each sediment type in the two halves of the dataset further emphasised the low habitat suitability across all substrate types (Figure 5.9).

5.6.4.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices are shown in Figure 5.10. The habitat suitability models all show a clear pattern of decreasing habitat suitability with increasing depth. Adding substrate type/biota to the habitat affinity model (Figure 5.10C) highlights shallow seagrass and reef areas as high suitability.

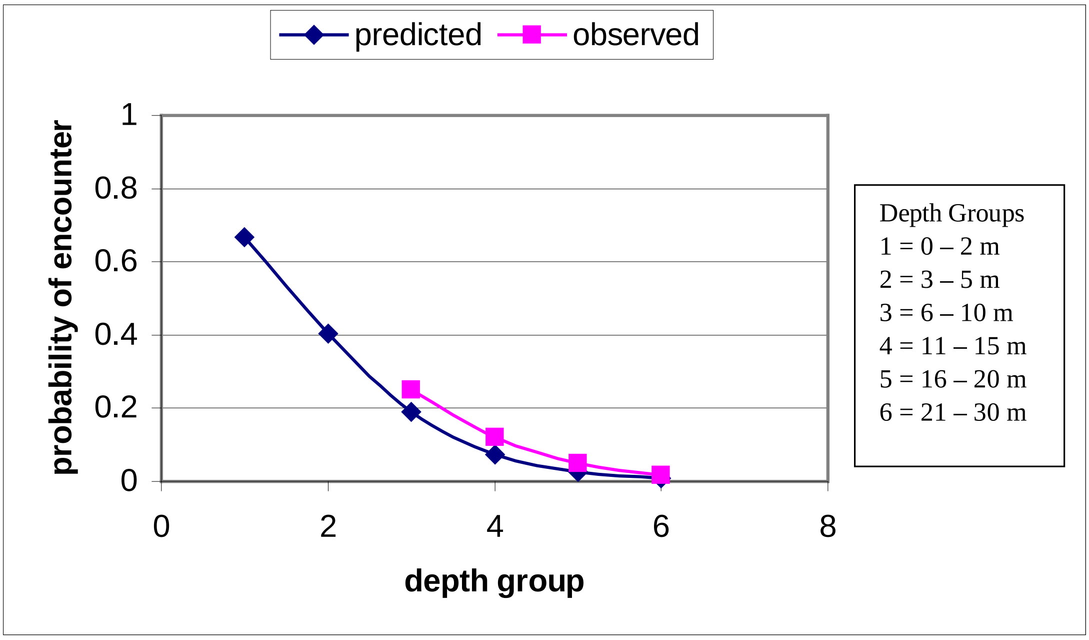


Figure 5.8. **Comparison of predicted probabilities of encounter for King George whiting calculated from regression analysis on the 'model' half of the dataset and compared with observed proportions encountered in the 'validation' half of the dataset. The regression equation was calculated using the depth group number. Note that the predicted probabilities have been extrapolated beyond the available data.**

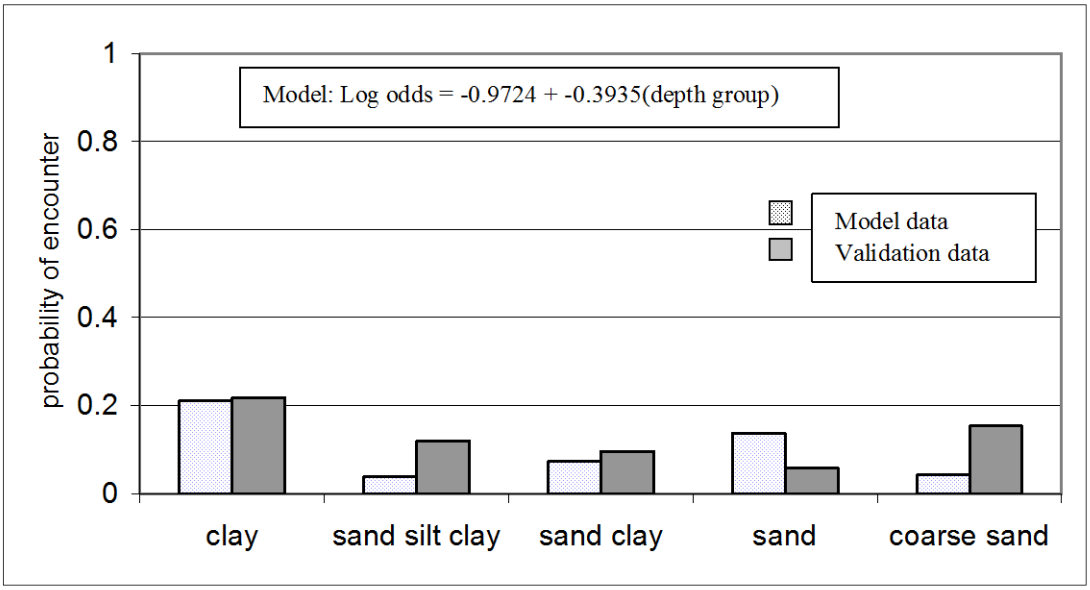


Figure 5.9. **comparison of the observed proportion of King George whiting encountered in each sediment type in the 'model' half of the dataset and the 'validation' half of the dataset.**

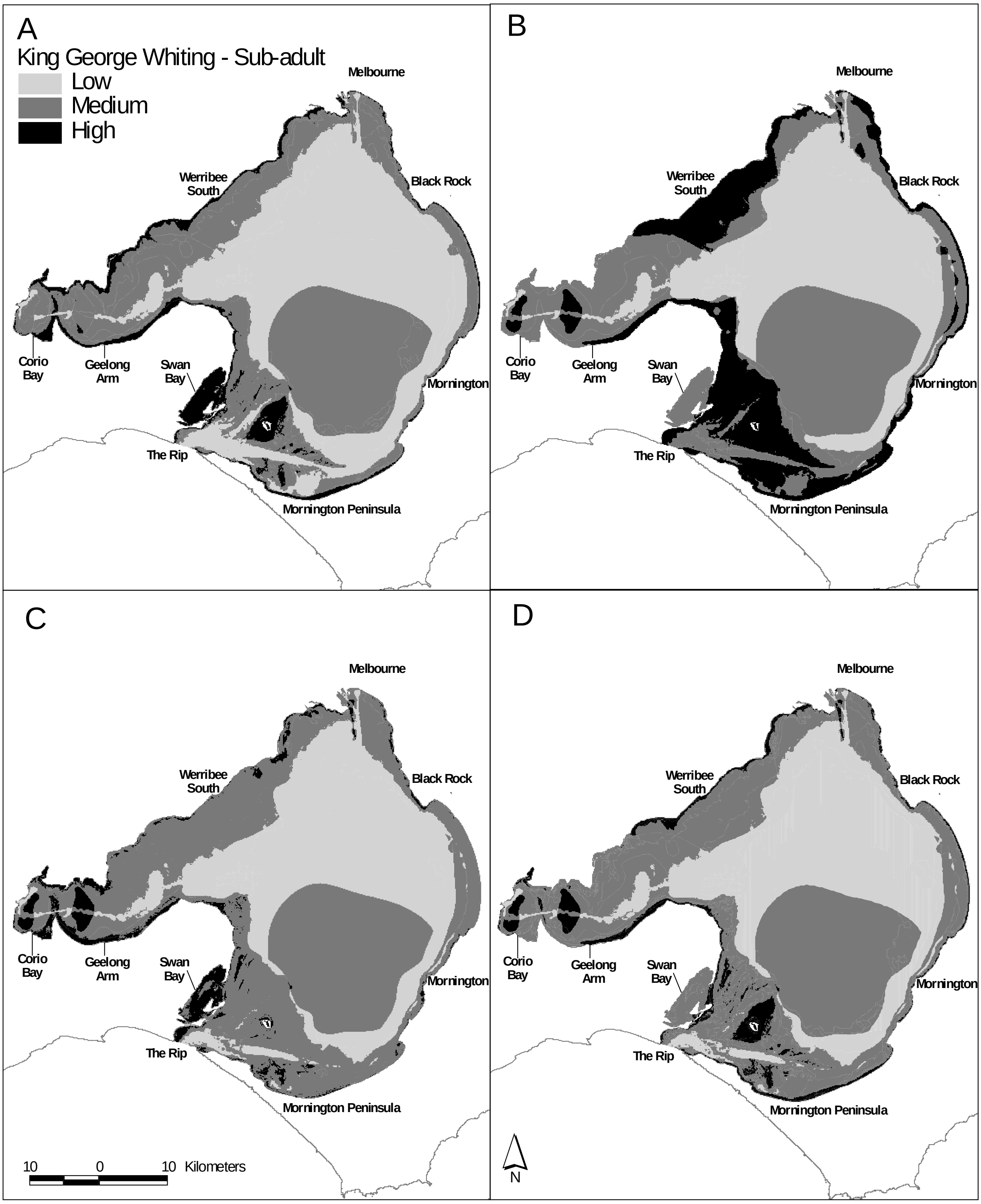


Figure 5.10: King George whiting (sub-adult) habitat suitability models. A: Logistic regression SI's – depth & sediment. B: Habitat affinity SI's – depth & sediment. C: Habitat affinity SI's – depth, sediment & substrate type/biota. D: Mean of models A, B & C.

5.6.5. King George Whiting Juveniles (shallow-water datasets)

5.6.5.1. Habitat Affinities

The suitability indices suggested that juvenile King George whiting had a high habitat affinity for reefs and medium sands at shallow depths (Table 5.4). Seagrass and bare edges of seagrass had a higher habitat affinity than bare habitat. Low habitat affinity values for deeper areas (>3 m) are based on expert opinion.

5.6.5.2. Bivariate Logistic Regression

Suitability indices for King George whiting calculated from bivariate logistic regression are presented in Table 5.7. The probabilities of encounter were greatest in spring and summer (Table 5.9), and for shallower depths. There were no significant relationships with substrate type/biota or sediment type when data was pooled over seasons. In summer, there were no significant relationships with substrate/biota and the highest probabilities were associated with sand-clay in intertidal areas.

5.6.5.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices for all seasons are presented in Figure 5.11. The models for juvenile King George whiting were relatively similar, showing deep areas as low suitability and shallow areas as medium to high suitability. The main areas of difference in the models were the areas of medium sand in the south of the Bay shown as medium suitability in the habitat affinity model with depth, sediment and substrate type/biota Figure 5.11C. The logistic regression model (Figure 5.11A) also gave a higher suitability rating to the shallow areas than the habitat affinity models.

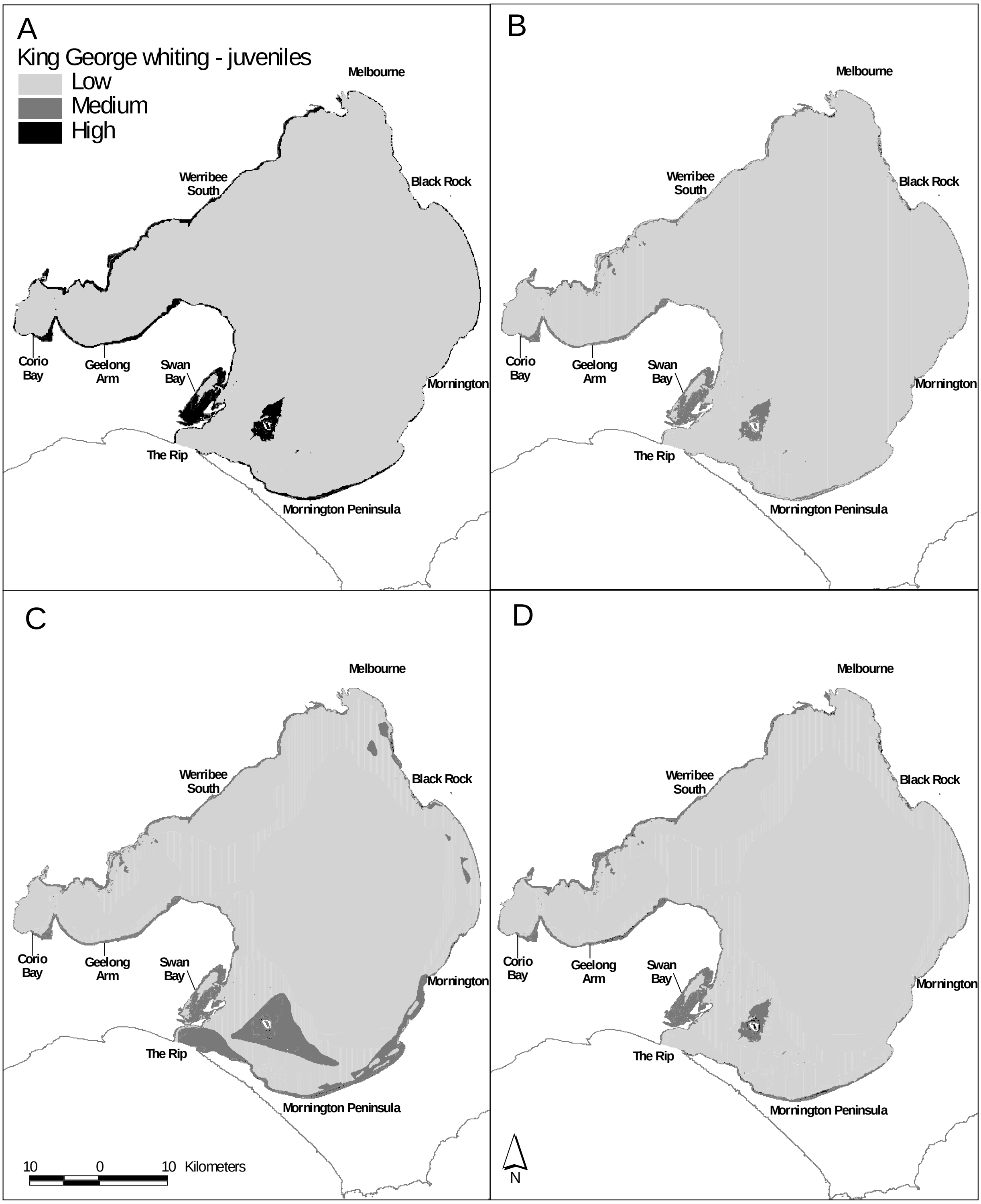


Figure 5.11: King George juveniles (all seasons) habitat suitability models. A: Logistic regression SI's – depth. B: Habitat affinity SI's – depth & substrate type/biota. C: Habitat affinity SI's –depth, sediment & substrate type/biota. D: Mean of models A, B & C.

5.6.6. Snapper (*Pagrus auratus*) Juveniles and Sub-adults

Only the larger trawl gear was considered suitable for sampling snapper, so only the 10-year trawl data (Parry *et al*. 1995; Officer & Parry 1997) was used for the analysis of this species. The snapper sampled by this trawl gear were generally juveniles between 1 and 3 years of age, but also included some sub-adults ('pinkies') that were greater than 27 cm in length which is the legal minimum size to catch this species.

5.6.6.1. Habitat Affinities

Habitat affinity values for snapper were highest in depths from 5-15 m and were high for all sediment types except fine-medium sand (Table 5.4). There was insufficient data to calculate habitat affinities for substrate type/biota, but expert opinion suggested that reef had the highest suitability.

5.6.6.2. Bivariate Logistic Regression

A logistic regression approach on each predictor separately indicated that the probability of encounter was medium to low (<0.5) across all depths and sediment types (Table 5.7). These values gave some indication that deeper, finer sediments provided more suitable habitats for snapper in Port Phillip Bay (Table 5.7). A comparison of the predicted probabilities with those observed in the 'validation' half of the dataset indicated a relatively poor fit to the model for depth (Figure 5.12), while there was a better fit for sediment type (Figure 5.13). This suggests that we can have little confidence in the predictive capabilities of the depth suitability indices for snapper in Port Phillip Bay.

5.6.6.3. Habitat Suitability Models

Habitat suitability models produced from the suitability indices are presented in Figure 5.14. The habitat suitability models based on depth and sediment SI values are very different for the logistic regression and habitat affinities (Figure 5.14A & B). The habitat affinity model showed large areas of low suitability corresponding to fine sand (Figure 5.14B). The logistic regression model showed the central deep clay zone of the Bay as high suitability (Figure 5.14A), while the habitat affinity model showed this area as medium suitability and the intermediate depth (10-15 m) sand-clay areas as high suitability (Figure 5.14B). Adding substrate type/biota as a variable to the habitat affinity model (Figure 5.14C) increased the value of the low suitability areas in Figure 5.14B to medium suitability, but the total high suitability areas remained largely the same. The combined model (Figure 5.14D) is similar to the overall distribution of medium and high suitability in the logistic regression model (Figure 5.14A).

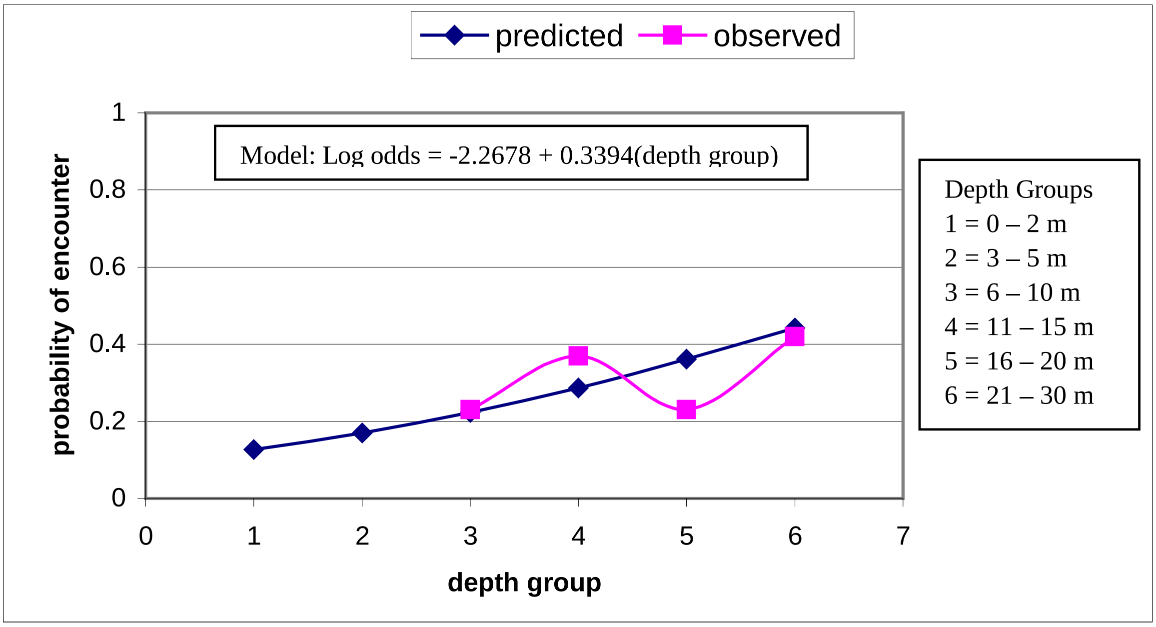


Figure 5.12. Comparison of predicted probabilities of encounter for snapper juveniles and sub-adults calculated from the regression analysis on the 'model' half of the dataset and compared with observed proportions encountered in the 'validation' half of the dataset. The regression equation is calculated using the depth group number. Please note that the predicted probabilities have been extrapolated outside the range of available data.

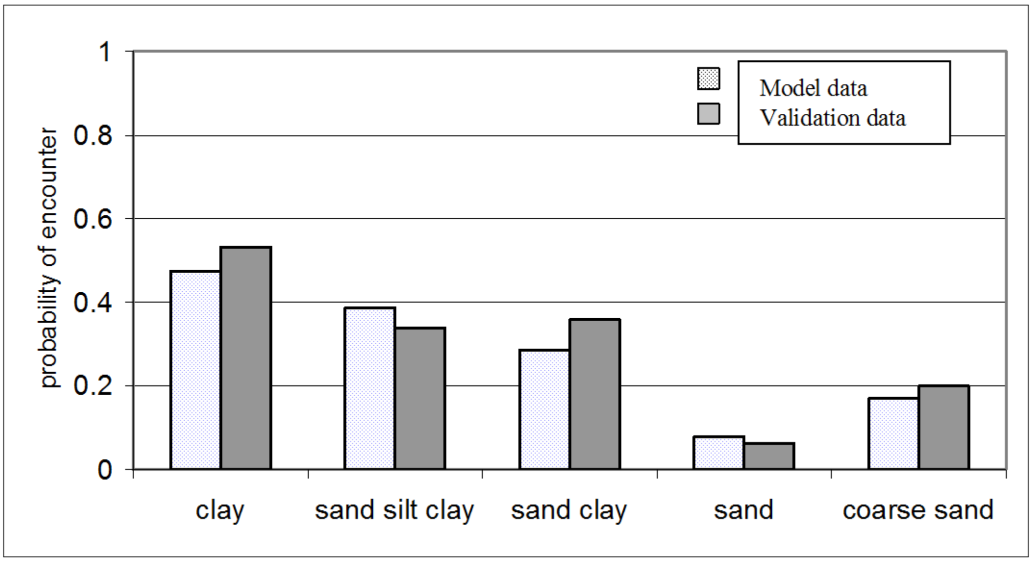


Figure 5.13. Comparison of the observed proportion of juvenile/sub-adult snapper encountered in each sediment type in the 'model' half of the dataset and the 'validation' half of the dataset.

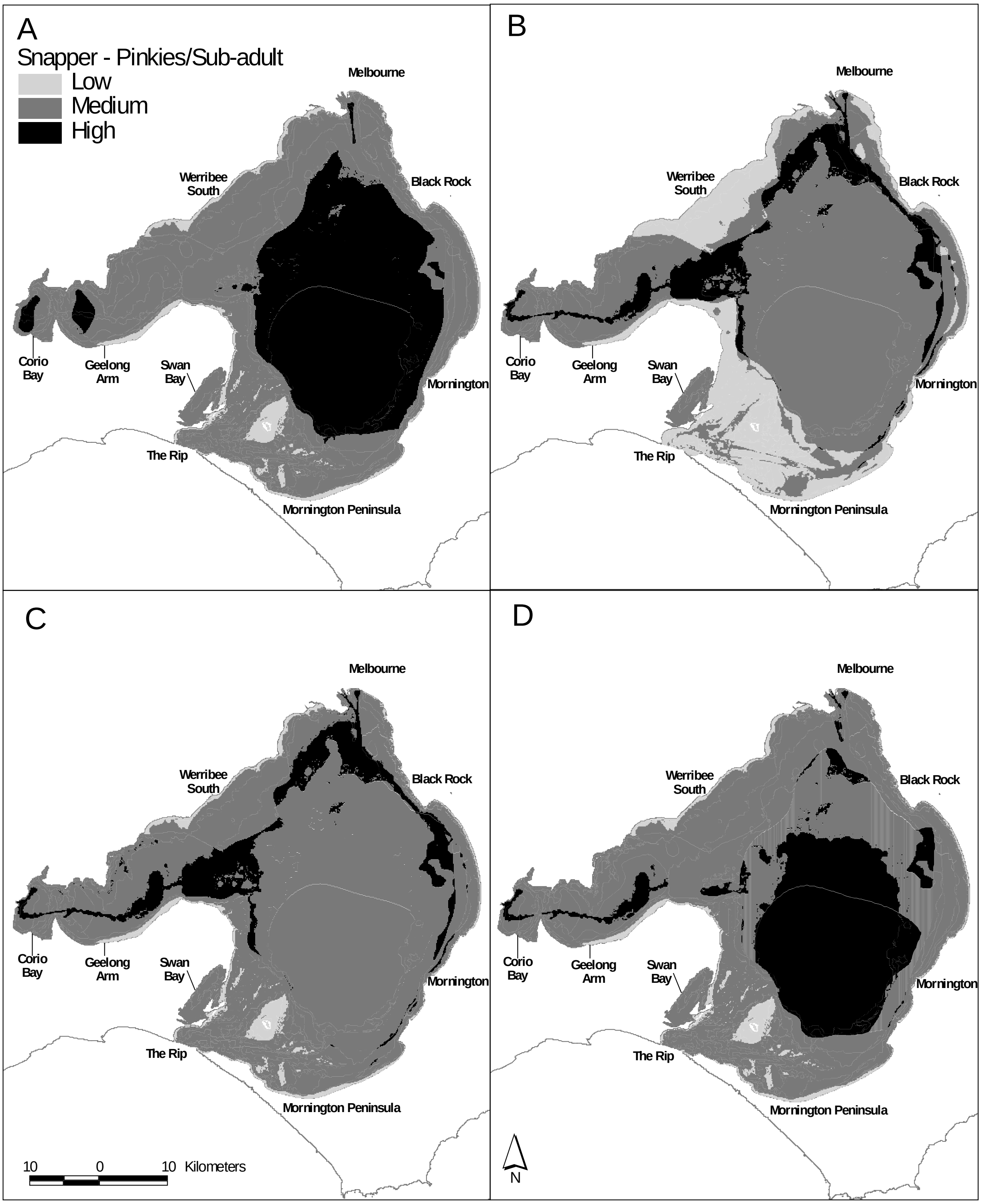


Figure 5.14: Snapper (juveniles/sub-adult) habitat suitability models. A: Logistic regression SI's – depth & sediment. B: Habitat affinity SI's – depth & sediment. C: Habitat affinity SI's – depth, sediment & substrate type/biota. D: Mean of models A, B & C.

5.6.7. Snapper (*Pagrus auratus*) 0+ year class

Data were available for newly-settled snapper (0+) from a targeted survey of this age class (Hamer & Jenkins 2004). The survey only sampled during February/March, so we could not calculate seasonal suitability index values.

5.6.7.1. Habitat Affinities

Habitat affinity values suggested that newly-settled snapper have an affinity for intermediate depths (5-10 m) and intermediate sediment grain sizes (Table 5.4). We did not have sufficient data to calculate habitat affinities for substrate type/biota and expert opinion did not enable us to discriminate between these habitats.

5.6.7.2. Bivariate Logistic Regression

There were insufficient records to justify splitting the dataset into a 'model development' half and a 'validation' half in this case and so Figure 5.15 only shows the observed proportions of newly-settled snapper in each sediment type. There was no significant logistic relationship with depth in the single parameter analysis.

Suitability indices for newly-settled snapper in Port Phillip Bay indicate that clay substrate had a lower suitability for this life history stage than other substrate types (Table 5.7, Figure 5.15).

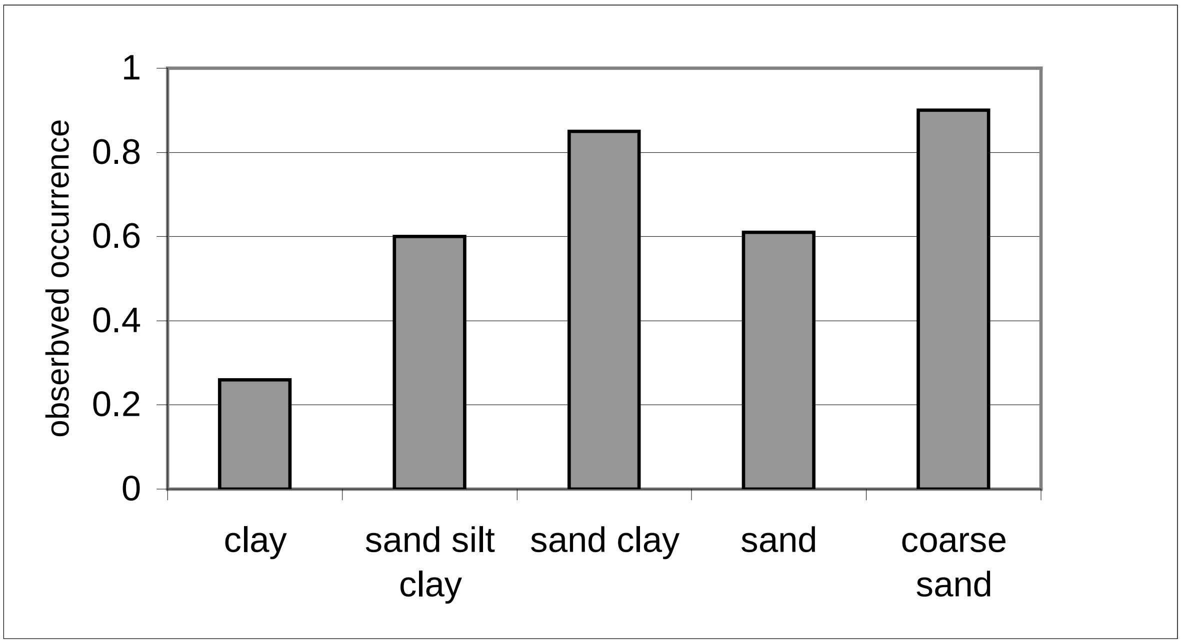


Figure 5.15. Observed proportions of newly settled snapper encountered in each sediment type.

5.6.7.3. Habitat Suitability Models

We did not produce habitat suitability models for snapper recruits as SI's were only available for sediment from the logistic regression. The habitat affinity SI's were similar to snapper sub-adults except for higher SI values for fine/medium sand and lower values for clay and sand-silt-clay.

5.6.8. Southern Calamari (*Sepioteuthis australis*)

The 10-year trawl data (Parry *et al*. 1995; Officer & Parry 1997) was the only dataset used for the analysis of southern calamari.

5.6.8.1. Habitat Affinities

In general, there is little known about the use of different habitats by southern calamari (Section 4). Habitat affinity values were medium-high at most depths and sediment types (Table 5.4). There was no data available for substrate type/biota and habitat affinity values are based on expert opinion only.

5.6.8.2. Bivariate Logistic Regression

There were no significant regression coefficients for any of the logistic regressions undertaken separately. As a consequence, none of the separate regressions could be used to create suitability indices. In this case, examination of the data suggests that the whole area should be given a high habitat suitability index.

5.6.9. Rock Flathead (*Platycephalus laevigatus*)

There were very few records (17) of rock flathead in the 10-year trawl dataset (Parry *et al*. 1995; Officer & Parry 1997). As a consequence, we did not attempt to quantify suitability indices for this species.

Table 5.4. Port Phillip Bay habitat affinities (values are scaled to the maximum so that they range between 0 and 1). See text for details of datasets used to calculate habitat affinities. Values in italics have been extrapolated beyond existing data range or are based on expert opinion and are of low confidence, all other values are of medium confidence. ND = no data.

	Depth groups (metres)								Substrate type/biota				Sediment type					
Species life history stage and season	**0**	**0-1**	**1-2**	**3-5**	**6-10**	**11-15**	**16-20**	**> 21**	**Bare**	**Sea-grass bare edge**	**Sea-grass**	**Reef**	**Clay**	**Sand silt clay**	**Sand clay**	**Sand fine**	**Sand medium**	**Coarse sand**
Sand flathead >15 cm (all seasons)	*0.01*	*0.10*	*0.10*	0.11	0.15	0.54	0.92	1.0	*1.0*	*0.5*	*0.01*	*0.01*	1.0	0.44	0.47	0.22		0.17
King George whiting sub-adults (all seasons)	*1.0*	*1.0*	*1.0*	*1.0*	1.0	0.13	0.01	0.01	*0.10*	*0.5*	*1.0*	*0.5*	1.0	0.17	0.04	0.55		0.03
KGW juveniles (all seasons)	0.44	1.0	0.03	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	0.01	0.4	0.21	1.0	*0.1*	*0.1*	*0.24*	0.34	1.0	0.24
KGW juveniles (spring)	0.18	1.0	0.01	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	0.02	0.20	0.19	1.0	ND	ND	ND	0.61	1.0	0.45
KGW juveniles (summer)	1.0	0.87	0.01	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	0.02	1.0	0.16	0.93	ND	ND	ND	0.27	1.0	0.2
Greenback flounder adults (all seasons)	*0.95*	*0.95*	*0.95*	*0.95*	0.95	0.52	0.18	1.0	*1.0*	*1.0*	*0.01*	*0.01*	1.0	0.22	0.27	0.20		0.06
Greenback flounder juvs (all seasons)	1.0	0.31	0.33	*0.10*	*0.01*	*0.01*	*0.01*	*0.01*	1.0	0.35	0.01	0.0	*0.15*	*0.15*	*0.42*	0.42	1.0	0.43
Greenback flounder juvs (spring)	1.0	0.25	0.57	*0.30*	*0.10*	*0.01*	*0.01*	*0.01*	1.0	0.78	0.03	0.0	ND	ND	ND	0.52	1.0	0.66
Greenback flounder juvs (summer)	0.88	0.78	1.0	*0.75*	*0.50*	*0.25*	*0.01*	*0.01*	1.0	0.9	0.01	0.0	ND	ND	ND	0.20	1.0	0.51

Table 5.4 continued

	Depth groups (metres)								Substrate type/biota				Sediment type				
Species life history stage and season	**0**	**0-1**	**1-2**	**3-5**	**6-10**	**11-15**	**16-20**	**> 21**	**Bare**	**Sea-grass bare edge**	**Sea-grass**	**Reef**	**Clay**	**Sand silt clay**	**Sand clay**	**Sand fine medium**	**Coarse sand**
Snapper sub-adults (all seasons)	*0.0*	*0.10*	*0.20*	*0.30*	0.41	1.0	0.41	0.30	*0.5*	*0.5*	*0.5*	*1.0*	0.89	0.99	1.0	0.07	0.85
Snapper 0+ (all seasons)	*0.0*	*0.10*	*0.20*	*0.30*	0.40	1.0	0.88	0.39	*0.5*	*0.5*	*0.5*	*0.5*	0.06	0.44	1.0	0.64	0.47
Southern calamari adults (all seasons)	*0.50*	*0.50*	*0.50*	0.42	0.75	0.61	0.57	1.0	*1.0*	*0.5*	*0.5*	*0.5*	0.71	0.67	0.28	0.53	1.0

	Low Confidence in SI Value		**Medium Confidence in SI Value**

See Section 5.3.6 for definition of SI confidence values.

Table 5.5. Port Phillip Bay habitat affinities for salinity variability for juveniles sampled in shallow-water datasets

	Salinity variability		
Species life history stage and season	**Low range**	**Medium**	**High range**
King George whiting juveniles (all seasons)	1.0	0.01	0.01
King George whiting juveniles (spring)	1.0	0.01	0.03
King George whiting juveniles (summer)	1.0	0.01	0.01
Greenback flounder juveniles (all seasons)	0.4	0.3	1.0
Greenback flounder juveniles (spring)	0.21	0.38	1.0
Greenback flounder juveniles (summer	0.22	0.10	1.0

Table 5.6. Port Phillip Bay habitat affinity values by season for juveniles of certain species for entire bay.

Species	Spring	Summer	Autumn	Winter
King George Whiting juveniles	0.98	1.0	0.12	0.01
Greenback flounder juveniles	1.0	0.26	0.06	0.32

Table 5.7. Port Phillip Bay habitat suitability indices calculated from univariate logistic regression (probability of encounter). Dependent variable modelled is presence/absence of each species (see text and figures for details of regression equations). Indices in italics have been extrapolated from the regression equation outside the available data range. N/S = non-significant relationship and ND = no data.

	Depth groups (metres)								Substrate type/biota				Sediment type					
Species life history stage and season	**0**	**0-1**	**1-2**	**3-5**	**6-10**	**11-15**	**16-20**	**> 21**	**Bare**	**Sea – grass bare edge**	**Sea-grass**	**Reef**	**Clay**	**Sand silt clay**	**Sand clay**	**Sand fine**	**Sand medium**	**Coarse sand**
Sand flathead >15 cm (all seasons)	*0.01*	*0.45*	*0.45*	*0.52*	0.59	0.66	0.72	0.77	ND	ND	ND	ND	0.85	0.61	0.69	0.61		0.75
King George whiting sub-adults (all seasons)	*0.67*	*0.67*	*0.67*	*0.41*	0.21	0.04	0.03	0.02	ND	ND	ND	ND	0.21	0.04	0.07	0.14		0.04
KGW juveniles (all seasons)	0.33	0.41	0.04	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
KGW juveniles (spring)	0.72	0.57	0.40	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	0.49	*0.56*	0.62	0.82	NS	NS	NS	NS	NS	NS
KGW juveniles (summer)	0.80	0.32	0.05	*0.01*	*0.01*	*0.01*	*0.01*	*0.01*	NS	NS	NS	NS	*0.42*	*0.42*	0.42	0.26	0.35	*0.20*
Greenback flounder adults (all seasons)	*0.97*	*0.97*	*0.97*	*0.69*	0.28	0.15	0.17	0.41	ND	ND	ND	ND	0.66	0.19	0.28	0.14		0.17
Greenback flounder juvs (all seasons)	0.33	0.16	0.21	*0.1*	*0.01*	*0.01*	*0.01*	*0.01*	0.35	0.35	0.04	0.01	*0.15*	*0.15*	0.15	0.23	0.11	*0.05*
Greenback flounder juvs (spring)	0.57	0.21	0.63	*0.20*	*0.01*	*0.01*	*0.01*	*0.01*	0.46	*0.46*	0.05	0.01	*0.21*	*0.21*	0.21	0.29	0.09	*0.05*
Greenback flounder juvs (summer)	0.35	0.12	0.26	*0.1*	*0.01*	*0.01*	*0.01*	*0.01*	0.33	*0.33*	0.02	0.01	NS	NS	NS	NS	NS	NS

Table 5.7 continued

	Depth groups (metres)								Substrate type/biota				Sediment type				
Species life history stage and season	**0**	**0-1**	**1-2**	**3-5**	**6-10**	**11-15**	**16-20**	**> 21**	**Bare**	**Sea – grass bare edge**	**Sea-grass**	**Reef**	**Clay**	**Sand silt clay**	**Sand clay**	**Sand** **fine medium**	**Coarse sand**
Snapper sub-adults (all seasons)	*0.01*	*0.13*	*0.13*	*0.17*	0.22	0.29	0.36	0.44	ND	ND	ND	ND	0.48	0.39	0.29	0.08	0.17
Snapper 0+ (all seasons)	NS	NS	NS	NS	NS	NS	NS	NS	ND	ND	ND	ND	0.26	0.60	0.85	0.61	0.90
Southern calamari Adults (all seasons)	NS	NS	NS	NS	NS	NS	NS	NS	ND	ND	ND	ND	NS	NS	NS	NS	NS

	Low Confidence in SI Value		**Medium Confidence in SI Value**		**High Confidence in SI Value**

See Section 5.3.6 for definition of SI confidence values.

Table 5.8. Port Phillip Bay suitability indices (probability of encounter) for salinity variability for juveniles sampled in shallow-water datasets

	Salinity variability		
Species life history stage and season	**Low range**	**Medium**	**High range**
King George whiting juveniles (all seasons)	0.41	0.01	0.04
King George whiting juveniles (spring)	0.61	0.06	0.01
King George whiting juveniles (summer)	0.35	0.01	0.17
Greenback flounder juveniles (all seasons)	0.16	0.28	0.58
Greenback flounder juveniles (spring)	0.21	0.83	1.0
Greenback flounder juveniles (summer)	0.13	0.06	0.75

	Low Confidence in SI Value		**Medium Confidence in SI Value**		**High Confidence in SI Value**

See Section 5.3.6 for definition of SI confidence values.

Table 5.9. Port Phillip Bay suitability indices (probability of encounter) values by season for juveniles of certain species.

Species	**Spring**	**Summer**	**Autumn**	**Winter**
King George whiting juveniles	0.54	0.36	0.05	0.03
Greenback flounder juveniles	0.24	0.14	0.05	0.08
Sand flathead juveniles	0.06	0.03	0.13	0.06

5.7. Discussion of Port Phillip Bay Habitat Suitability Models

5.7.1. Sand Flathead >15 cm

Sand flathead are most common in Port Phillip Bay on muddy sand or shell habitats at 15-25 m (Coutin 2000a) and this is consistent with the areas shown as highly suitable habitat in each of the models in Figure 5.3. Sand flathead are widespread in the Bay though, and have been identified as the most abundant demersal fish species and dominant benthic predator (Parry *et al*. 1995; Officer & Parry 2000). Commercial catch data indicates that sand flathead are caught throughout the Bay by both nets (mesh and haul seines) and hooks/lines. As a consequence, the representation of most of the Bay as high and medium suitability for sand flathead is a reasonable conclusion.

The model based on logistic regression SI's for depth and sediment identifies most of the Bay at depths >5 m as high suitability for sand flathead, with the remaining areas shown as medium suitability (Figure 5.3A). This general pattern is refined in the model based on habitat affinity SI's for depth, sediment and substrate type/biota which distinguishes seagrass and macroalgae on sediment as low suitability (Figure 5.3C). Sand flathead are more common on bare sediment and this model option probably provides the best representation of the overall distribution of relative habitat suitability for sand flathead in the Bay.

5.7.2. Greenback Flounder Adults

Greenback flounder appear to be a species with a wide distribution in Port Phillip Bay with commercial catches being taken in shallow areas and over seagrass, and are commonly caught in areas where there are large patches of sand between reef, algae and seagrass. Recreational fishers mostly catch flounder with hand spears in shallow water offshore from sand beaches.

The model which appears to present a pattern most generally consistent with existing habitat preference statements is based on logistic regression SI's for depth and sediment (Figure 5.6A). This model highlights shallow areas and the central deeper region as high suitability habitat, while showing most of the rest of the Bay as medium suitability.

5.7.3. Greenback Flounder Juveniles

Shallow (< 2 m) bare habitats have been identified as important nursery areas for juvenile greenback flounder (Jenkins *et al*. 1997b; Jenkins & Wheatley 1998). Higher numbers of juvenile flounder have also often been caught in bare sand than seagrass in Port Phillip Bay (Jenkins *et al*. 1993; Jenkins *et al*. 1997b; Jenkins & Wheatley 1998). The habitat suitability models which incorporate SI's for substrate type/biota identify very shallow bare sediment as high suitability for juvenile greenback flounder (Figure 5.7A & B) and this is consistent with existing knowledge of the species' behaviour. Only the habitat affinity model separates seagrass/macroalgae areas as low suitability (Figure 5.7B).

5.7.4. King George Whiting Sub-Adults

Sub-adult King George whiting are highly transient, and are capable of using a variety of habitat types. Commercial fishers predominantly catch King George whiting with haul seines and mesh nets. The majority of the commercial catch is taken from the Corio Bay/Geelong Arm region and inside the Bay entrance, with good catches also recorded along the western coast between Williamstown and Point Wilson. These broad areas are mostly shown as medium or high suitability habitat in the models presented in Figure 5.10.

Specific areas noted by recreational fishing guide books as being good fishing spots in Port Phillip Bay are found at Swan Bay, West Sand, Beaumaris Bay, Altona Bay, Campbells Cove, Wedge Point, Grassy Point to St Leonards and Curlewis Bank (Classon & Wilson 2002). Most of these areas are identified as high suitability in the models shown in Figure 5.10. However, it is the model based on depth and habitat logistic regression SI's (Figure 5.10A) and the model based on habitat affinity SI's incorporating substrate type/biota (Figure 5.10C) that show distributions of high and medium suitability habitat most consistent with these areas.

5.7.5. King George Whiting Juveniles

Juvenile King George whiting are known to settle into relatively shallow water that contains either seagrass or rocky reef/algae (Jenkins *et al*. 1997a; Jenkins *et al*. 1997b; Jenkins *et al*. 1998a; Jenkins & Wheatley 1998). King George whiting spawn on the open coast from May to July and the larvae settle in bays and inlets up to 150 days later in spring (Jenkins & May 1994; Jenkins *et al*. 2000). In Port Phillip Bay, larvae initially settle in seagrass and reef-algal habitats between September and December and juveniles later migrate to unvegetated sand between February and April (Jenkins and Wheatley 1998, G. Jenkins pers. comm.). This ontogenetic shift was only partly reflected in the higher habitat affinity SI value assigned to seagrass bare edge habitat between spring and summer (Table 5.4).

Reef habitat has been assigned a much higher SI value than seagrass in the habitat affinities (Table 5.4). This analysis was strongly influenced by data from Jenkins and Wheatley (1998) where seagrass, reef-algal and unvegetated habitats were sampled at three locations on the Bellarine Peninsula. This study found that immediately after settlement, King George whiting were associated with both shallow seagrass and reef-algal habitats, but not unvegetated sand. After a couple of months growth, an increasing preference was shown for reef-algal habitat over seagrass at one of the locations (Grassy Point) and it is suggested that this may have been due to higher food levels in reef-algal habitat compared to seagrass habitat at this location (Jenkins & Wheatley 1998).

The models for King George whiting juveniles shown in Figure 5.11 relate to all seasons, so the areas of high suitability need to encompass both the shallow seagrass/reef habitat and bare habitat used by the different larval/juvenile life stages. The models that only use depth and substrate type/biota seem to identify most of these regions as medium-high suitability habitat(Figure 5.11A & B).

Introducing seabed sediment habitat affinity SI values to the composite habitat suitability model highlights areas of medium suitability habitat at the Bay entrance, around the Great Sands and along the Mornington Peninsula shore (Figure 5.11C). These areas are characterised by sandy sediment in depths <5 m and probably overestimate the total area of medium suitability habitat in this region.

All of the models in Figure 5.11 effectively define the deeper central zone of the Bay as low suitability.

5.7.6. Snapper Juveniles/Sub-Adults

Older juveniles and sub-adults appear to utilise a range of habitats from bare soft-sediments to seagrass, algae and reef (MacDonald 1982, Hamer *et al.* unpublished). They are highly transient and gregarious, and habitat associations are likely to be ephemeral. Recreational fishing guide books suggest that snapper in Port Phillip Bay are widely distributed throughout the Bay in both the deeper central areas and in shallower nearshore areas (Classon & Wilson 2002).

The habitat suitability model based on logistic regression SI values (Figure 5.14A) does not appear to be a good representation of likely habitat distribution. This is consistent with the assessment of the probability of encounter values for snapper versus depth which suggested that we could have little confidence in the predictive capabilities of these suitability indices for snapper in Port Phillip Bay (Figure 5.12).

The habitat suitability model based on habitat affinity SI values for depth and sediment presents a more likely distribution of suitable habitat (Figure 5.14B), but still fails to identify reef areas (eg. between Beaumaris and Sandringham) which are considered to be good snapper fishing sites (AFN 2002). Reef habitat was given a high SI value, but we had insufficient data to accurately differentiate the relative significance of other substrate type/biota categories (Table 5.4). As a result, adding the substrate type/biota SI values to the composite habitat suitability model does not improve the predicted distribution of suitable habitat (Figure 5.14C). However, even if the SI values are refined for substrate type/biota, our existing GIS layer for these habitats does not accurately define many of the reef systems in depths >5 m or other important habitat such as *Pyura* in depths of 5-15 m, which are also known to support good snapper fishing areas (AFN 2002).

The wide distribution of snapper in Port Phillip Bay is partly reflected in the large areas of medium and high suitable habitat shown in the combined model (Figure 5.14D).

5.8. Western Port Habitat Suitability Indices & Models

It was not possible to combine all of the different datasets for Western Port due to the sparse data and the range of gear types used. As a result, different combinations of data were used to determine the quantitative suitability indices depending on the life history stage and sampling efficiency of a gear type for a fish species. There were still large gaps in the available data for the different habitats, and suitability indices for habitats that had not been adequately sampled were determined by extrapolation or from qualitative information.

In the following sections, only habitat suitability models that are based on habitat affinity values for adult life-stages are presented where habitat affinity values could be calculated for both depth and sediment type/biota categories.

5.8.1. Sand Flathead (*Platycephalus bassensis*)

Data collected with gill nets and trawls were used to calculate quantitative suitability indices (see Section 5.2.2). Only the gill net data included samples in different substrate type/biota classes and so habitat affinity suitability indices were calculated using this data. Suitability indices for depth were calculated using the trawl data as this data had the biggest spread of different depths sampled. Where no data were available for a substrate type/biota class or a depth group, extrapolations or expert opinion have been used to fill these gaps.

5.8.1.1. Habitat Affinities

Habitat affinity data indicated that bare and channel habitats at depths >5 m were the most suitable for sand flathead (Table 5.10). The habitat suitability model for habitat affinities is shown in Figure 5.16.

5.8.1.2. Qualitative Indices

Compilation of habitat statements for sand flathead suggested that bare or channel habitats were the most suitable for sand flathead, with all types of substrate type/biota having a low suitability for this species. No depths were considered to have a low suitability for sand flathead, but depths of 10 m and greater were considered more suitable than the shallower depths (Table 5.12).

5.8.1.3. Habitat Suitability Model

A habitat suitability model based on the habitat affinity SI values is shown in Figure 5.16. Deep (>10 m) unvegetated areas at the Western Entrance, the North Arm and between French and Phillip Islands are highlighted as highly suitable habitat. The areas classified as low suitability habitat correspond to shallow (<5 m) seagrass and macroalgae.

5.8.2. Sand Flathead Juveniles

Quantitative suitability indices were calculated using data from seine net sampling that was pooled across different surveys which focused on shallow water habitats (Edgar *et al.* 1993; Hindell & Jenkins 2004).

5.8.2.1. Habitat Affinities

As with the adult sand flathead, bare and channel habitats were found to be the most suitable for juvenile sand flathead with structural habitat having a low suitability (Table 5.10). While our data was limited, the 2-5 m depth range had the highest habitat affinity for juvenile sand flathead (Table 5.10). Interestingly, there was very little difference in habitat affinity values for the different seasons, implying that sand flathead are recruiting throughout the year in Western Port, but this analysis is based on only limited data (Table 5.11).

5.8.2.2. Qualitative Indices

An examination of habitat statements relating to juvenile sand flathead suggests that bare and channel habitats deeper than 3-5 m are the most suitable habitats for this life stage of this species (Table 5.12). There was little information regarding the depth distributions of juvenile sand flathead, but juveniles and adults were considered to use the same habitat in Corner Inlet (Gunthorpe & Hamer 1998a) and juveniles were considered likely to be found at depths > 5 m (Section 4).

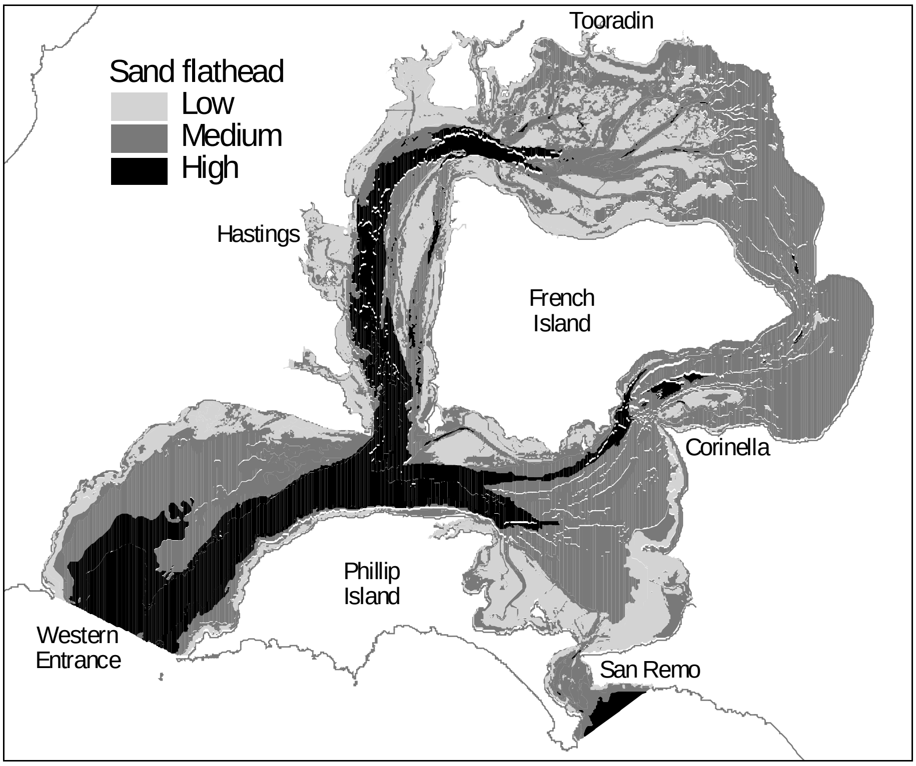


Figure 5.16. Habitat suitability model for sand flathead in Western Port, habitat affinity SI's – substrate type/biota and depth.

5.8.3. King George Whiting (*Sillaginodes punctata*)

The most useful information for the quantitative analysis for King George whiting came from the pooled gill net data (Edgar *et al.* 1993; Hindell & Jenkins 2004), but it was only possible to use this data for habitat affinities because raw data was not available for one of the studies (i.e. Edgar *et al.* 1993).

5.8.3.1. Habitat Affinities

Shallow depths (up to 10 m) had the greatest habitat affinity for King George whiting and all habitats except for bare had reasonably high habitat affinities for this species (Table 5.10). The habitat suitability model is shown in Figure 5.17.

5.8.3.2. Qualitative Indices

Habitat statements suggested that only the deepest areas in Western Port would be of low suitability for this species and the highest suitability depths were considered to be between 3 and 15 m (Table 5.12). All categories of substrate type/biota were of high or medium suitability, with the exception of mangroves that were coded low suitability although there has been very little sampling in Victorian mangroves and so the confidence in this coding is low.

5.8.3.3. Habitat Suitability Model

The habitat suitability model for King George whiting (Figure 5.17) showed most of Western Port as medium suitability habitat, with the highly suitable habitat located in the channels cutting through the shallow sand-mud banks and depths of about 2-5 m.

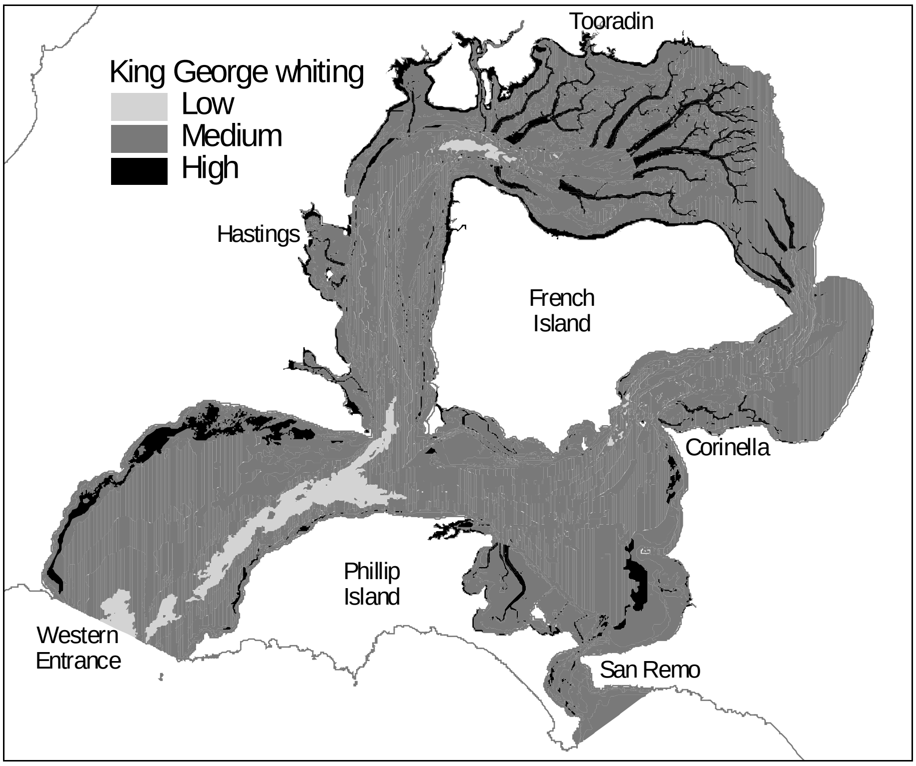


Figure 5.17. Habitat suitability model for King George whiting in Western Port, habitat affinity SI's – substrate type/biota and depth.

5.8.4. King George Whiting Juveniles

The most useful information for the quantitative indices for juvenile King George whiting came from the seine net datasets (Edgar *et al.* 1993; Hindell & Jenkins 2004).

5.8.4.1. Habitat Affinities

Habitat affinity values suggest that shallow water (up to 2 m depth) is the most suitable for juveniles of this species with a steep drop off in affinity with increasing depth (Table 5.10). Habitat affinities were high for all habitats except for channels. There was no direct data to support the habitat affinities that have been assigned to reef and validation of these values is required in Western Port. Habitat affinity values were highest in Spring suggesting that juvenile King George whiting recruit to Western Port mainly in Spring, but late arrivals will still be picked up in summer and autumn (Table 5.11).

5.8.4.2. Qualitative Indices

Examination of habitat statements for juvenile King George whiting suggested that only the 0-2 m depth range would be highly suitable for this species and depths > 10 m would be of low suitability (Table 5.12). No habitat statements were made about the association of juvenile King George whiting with mangroves in Western Port despite the fact that they were sampled in the study undertaken by Hindell & Jenkins (2004). As with the quantitative analyses, all substrate type/biota categories, except for channels, were considered of similar suitability and in this case were assigned a high habitat suitability (Table 5.12).

5.8.5. Rock Flathead (*Platycephalus laevigatus*)

The only information we were able to use for the quantitative indices for rock flathead came from the gill-net datasets (Edgar *et al.* 1993; Hindell & Jenkins 2004).

5.8.5.1. Habitat Affinities

Shallow subtidal and seagrass habitat had the highest affinity for this species (Table 5.10). No rock flathead were caught in the intertidal, but we have adjusted the habitat affinity value up from zero as there was insufficient data to conclude that rock flathead are not found in intertidal habitat. Only bare intertidal was sampled and it may be that rock flathead have a higher affinity for intertidal seagrass. The habitat suitability model for rock flathead is shown in Figure 5.18.

5.8.5.2. Qualitative Indices

The habitat statements indicated that shallow depths and seagrass or reef habitat would be the most suitable habitat for rock flathead (Table 5.12). Bare substrate, channel habitat and possibly mangrove habitat were considered of low suitability for rock flathead, as were depths > 10 m.

5.8.5.3. Habitat Suitability Model

The habitat suitability model for rock flathead is shown in Figure 5.18. Only small areas of seagrass habitat in depths of 0-2 m were identified as highly suitable habitat, while seagrass-macroalgae habitat and depths of 2-5 m were shown as medium suitability habitat.

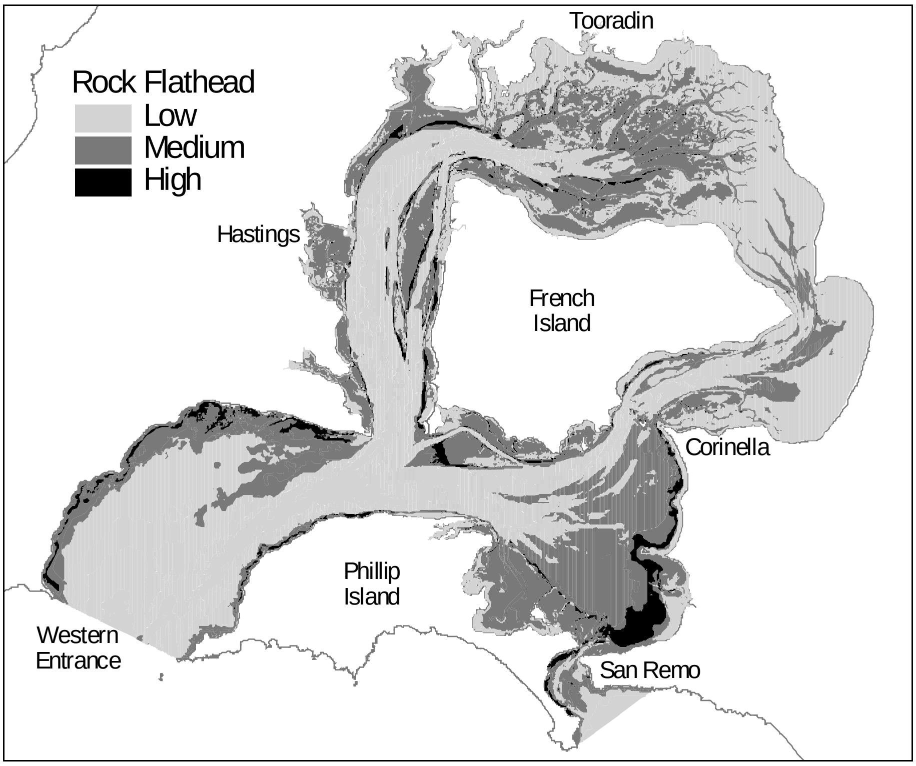


Figure 5.18. Habitat suitability model for rock flathead in Western Port, habitat affinity SI's – substrate type/biota and depth.

5.8.6. Rock Flathead Juveniles

The most useful information with which to analyse the juvenile rock flathead was the seine net data (Edgar *et al.* 1993; Hindell & Jenkins 2004) and this was used to calculate habitat affinities.

5.8.6.1. Habitat Affinities

Patterns of habitat affinities for juvenile rock flathead were similar to those of adults. Shallow subtidal and seagrass habitat had the highest values of habitat affinity for juveniles of this species (Table 5.10). Habitat affinity values were greatest in spring and summer for juvenile rock flathead suggesting that this species mainly recruits during this time period (Table 5.11).

5.8.6.2. Qualitative Indices

The qualitative analysis also gave similar suitability indices for juvenile rock flathead as were found for adults for depth (Table 5.12). The main difference between juveniles and adults was in the suitability values assigned for substrate type/biota. Juveniles were considered to have a high suitability for bare habitat and only seagrass was considered to be of medium suitability in contrast to the high suitability given to adults for seagrass (Table 5.12). There were no statements relating to reef habitat for juvenile rock flathead for Western Port and so reef is undefined.

5.8.7. Yellow-eye Mullet (*Aldrichetta forsteri*)

Only the combined gill net data (Edgar *et al.* 1993; Hindell & Jenkins 2004) was suitable for the quantitative indices for this species.

5.8.7.1. Habitat Affinities

Shallow water and seagrass habitats had the highest habitat affinity values for yellow-eye mullet. Channel habitats had a low habitat affinity and mangrove, reef and bare substrate all had medium habitat affinity values (Table 5.10). The habitat suitability model for yellow-eye mullet is shown in Figure 5.19.

5.8.7.2. Qualitative Indices

Patterns of habitat suitability resulting from the qualitative analysis were similar to habitat affinity values. Shallow water habitat was considered to be of high suitability for this species, as was seagrass habitat. Bare substrate and mangrove habitats were both given a high suitability index and reef was coded medium with a degree of uncertainty (Table 5.12).

5.8.7.3. Habitat Suitability Model

The habitat suitability model for yellow-eye mullet (Figure 5.19) is similar to the model for rock flathead (Figure 5.18). Only small areas of seagrass habitat and depths of 0-2 m were identified as highly suitable habitat, while all habitats at depths of 0-5 m were shown as medium suitability habitat.

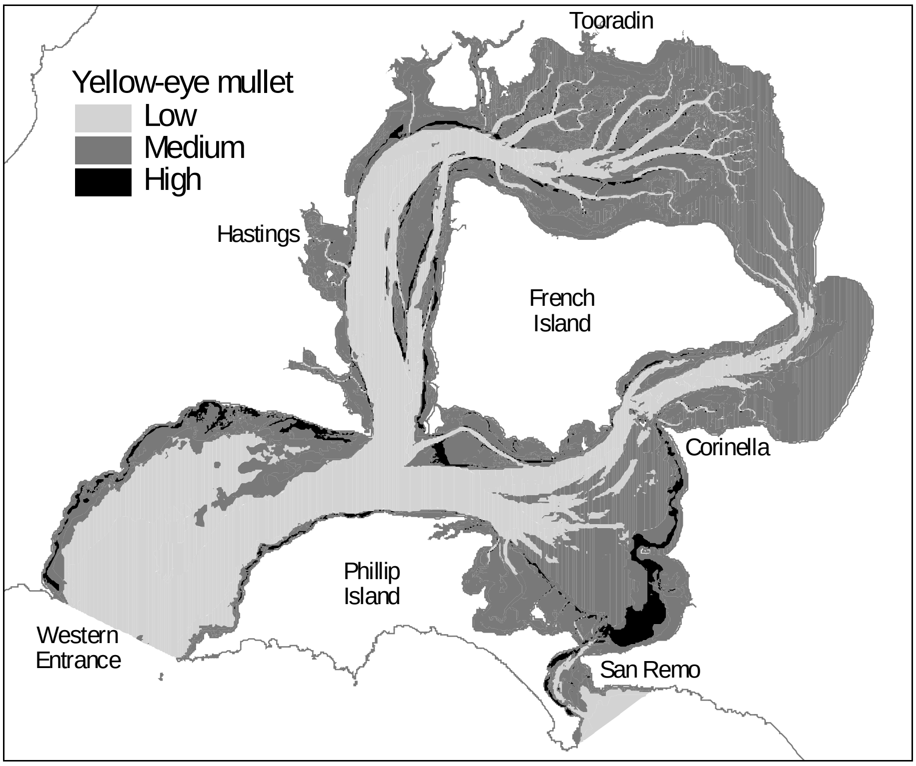


Figure 5.19. Habitat suitability model for yellow-eye mullet in Western Port, habitat affinity SI's – substrate type/biota and depth.

5.8.8. Yellow-eye Mullet Juveniles

The most useful information for the analysis of juvenile yellow-eye mullet was the seine-net data (Edgar *et al.* 1993; Hindell & Jenkins 2004) and this was used to calculate habitat affinities.

5.8.8.1. Habitat Affinities

Juvenile yellow-eye mullet had a greatest affinity for shallow water habitat (Table 5.10). Both mangrove and bare habitats had high values and seagrass also had a reasonably high habitat affinity value for juvenile yellow-eye mullet. Only channel habitat had a low suitability for this life stage of this species. The habitat affinity values for juvenile yellow-eye mullet were highest in summer (Table 5.11) suggesting that the majority of recruitment takes place at this time for this species.

5.8.8.2. Qualitative Indices

Indices derived from the examination of the habitat statements were consistent with the habitat affinity values. Mangrove and bare habitats were both considered of high suitability, as was the shallow water habitat (Table 5.12). Reef and seagrass habitats were considered of medium suitability and the main channels were considered to be of low suitability for juvenile yellow-eye mullet.

5.8.9. Greenback Flounder (*Rhombosolea tapirina*)

Adult greenback flounder did not appear to be adequately sampled by any gear types in the data we had available, so we were unable to produce habitat affinity indices for the adults of this species. Examination of habitat statements indicated that greenback flounder were more likely to be found at depths > 5m in bare habitat. Seagrass and channel habitats were given a medium suitability value and reef and mangrove habitats were considered to be of low suitability (Table 5.12).

5.8.10. Greenback Flounder Juveniles

Data from seine-net sampling (Edgar *et al.* 1993; Hindell & Jenkins 2004) was the most suitable for determining habitat affinities for juvenile greenback flounder.

5.8.10.1. Habitat Affinities

Juvenile greenback flounder had a high affinity for shallow water habitat, and particularly intertidal areas, and for bare or mangrove habitats (Table 5.10). Conversely, habitat affinity values for seagrass and reef were low (Table 5.10). Spring and summer had the highest habitat affinity values suggesting that greenback flounder mainly recruit to Western Port at this time (Table 5.11).

5.8.10.2. Qualitative Indices

Patterns of suitability derived from qualitative statements were similar to the habitat affinity values. Bare habitat in intertidal or shallow depths was considered to be of high suitability (Table 5.12), while seagrass or mangrove habitat were considered to be of medium suitability and reef of low suitability.

5.8.11. Southern Calamari (*Sepioteuthis australis*)

Calamari only appeared to be taken by trawls and not the other gear types and so only trawl data (Hobday 1992; Hamer & Jenkins 2004) was used to calculate habitat affinities. The trawls were over a range of depths but only sampled limited substrate type/biota types and so it was not possible to use quantitative data to generate suitability indices for the different substrate type/biota categories for this species.

5.8.11.1. Habitat Affinities

Habitat affinity values suggest that depths between 0-2 m are the most suitable for this species with depths up to 10 m being of medium suitability (Table 5.10).

5.8.11.2. Qualitative Indices

There were very few habitat statements for southern calamari and so it was not possible to construct qualitative habitat suitability indices for this species.

5.8.12. Snapper (*Pagrus auratus*)

Snapper were only adequately sampled by the trawls and so the combined trawl datasets (Hobday 1992; Hamer & Jenkins 2004) have been used to calculate suitability indices. No data was available for the different substrate type/biota and so we only calculated habitat affinities for depth.

5.8.12.1. Habitat Affinities

Habitat affinity values indicate that depths of 10-15 m have the highest suitability with 15-20 m having a medium suitability for this species (Table 5.10). The shallower depths had low habitat affinity values.

5.8.12.2. Qualitative Indices

The pattern for the qualitative indices differs slightly from the quantitative indices in that depths > 3 m were considered of high suitability, with only the very shallow depths considered to have low suitability. No substrate type/biota were considered to be of low suitability and bare and reef habitats were considered to be of high suitability (Table 5.12).

5.8.13. Snapper Juveniles

There were no suitable datasets from which to construct quantitative indices for snapper juveniles and so only qualitative indices were produced.

5.8.13.1. Qualitative Indices

There was a high degree of uncertainty associated with the qualitative suitability indices for juvenile snapper. The highest suitability habitat was considered to be at depths of 6-15 m and on bare habitat (Table 5.12). Low suitability habitat was considered to be shallow depths and reef, mangrove or channel habitat (Table 5.12). These suitability indices need testing in Western Port and Corner Inlet as habitat statements were conflicting and there was no quantitative data with which to test these statements.

5.8.14. Australian Salmon (*Arripis spp.*)

Australian salmon appeared to be very strongly schooling as there were large numbers of fish from only a few of the total samples and so it was not possible to use the quantitative data to calculate suitability indices for this species.

5.8.14.1. Qualitative Indices

Suitability indices for Australian salmon derived from qualitative data suggested that depths of 3-10 m were of high suitability and bare, reef and main channel habitat were of high suitability for this species. Low suitability habitat was intertidal and depths > 15 m and the minor channel habitat (Table 5.12). There is a degree of uncertainty with some of these suitability indices (Table 5.12).

5.8.15. Australian Salmon Juveniles

There was no suitable quantitative data with which to determine habitat affinities for juvenile Australian salmon and so only qualitative suitability indices were produced.

5.8.15.1. Qualitative Indices

Shallow bare or seagrass habitat was considered to be highly suitable for this species (Table 5.12), while deeper areas including the main channels were considered to be of low suitability. There were no habitat statements about the suitability of mangrove habitat for juvenile salmon and the suitability indices presented for channel and seagrass habitat require verification.

5.9. Discussion of Western Port Habitat Suitability Models

5.9.1. Sand Flathead

Sand flathead are found in Western Port on shallow unvegetated sand banks (Gunthorpe & Hamer 1998b) and these areas are shown as medium suitability in the habitat suitability model (Figure 5.16). Deeper (>10 m) unvegetated habitats were identified as high suitability and while we have limited information to confirm this within Western Port, these areas are consistent with the high suitability habitat within Port Phillip Bay and it is reasonable to assume that a similar situation occurs in Western Port.

There is no significant commercial catch of sand flathead in Western Port to compare with the model and recreational fishing guide books do not distinguish between different species of flathead.

5.9.2. King George Whiting

In Western Port, sub-adult King George whiting are often seen on sand patches between seagrass and on the edges of reef areas (Gunthorpe & Hamer 1998b). Western Port is recognised as a good seasonal recreational fishery for King George whiting between October and May. Recreational fishing guide books identify drop-offs on the edge of shallow or exposed banks as the most likely places to catch sub-adult fish, and while whiting can be caught in depths >10 m, they are more commonly caught in the 1-5 m depth zone (Crowley & Worsteling 2000).

The habitat suitability model for King George whiting in Western Port showed most of the bay as medium suitability habitat, while the highly suitable habitat is located in the channels and seagrass habitats at depths of 2-5 m (Figure 5.17). This model gives an indication that King George whiting are widespread in Western Port and that the channels represent highly suitable habitat.

5.9.3. Rock Flathead

Rock flathead are generally found in shallow waters (<20 m) and most commonly in areas of seagrass and low relief reef (Gunthorpe & Hamer 1998b). The habitat suitability model for rock flathead in Western Port (Figure 5.18) showed the intertidal and shallow (<5 m) seagrass and reef areas as medium suitability habitat. The only areas identified as highly suitable habitat are seagrass habitats in 0-2 m depths. This model is probably conservative in its representation of medium suitability habitat and many of the areas could possibly be considered as highly suitable habitat.

5.9.4. Yellow-eye Mullet

Yellow-eye mullet are broadly distributed across Western Port in shallow waters (<6 m) (Gunthorpe & Hamer 1998b). Edgar and Shaw (1995) found that mullet were approximately twice as common in shallow seagrass as over bare sand in Western Port; and their abundances were relatively low in channels. This broad pattern is reflected in the habitat suitability model shown in Figure 5.19.

Table 5.10: Western Port habitat affinities (values are scaled to the maximum so that they range between 0 and 1). Values in italics have been extrapolated beyond the existing data range or are based on expert opinion and are of low confidence, all other values are of medium confidence.

	Depth group (m)						Substrate type/biota				
Species life history stage and season	**0+**	**0-2**	**3-5**	**6-10**	**11-15**	**16-20**	**Mangrove & Mangrove Edge**	**Bare**	**Seagrass**	**Reef**	**Channel**
Sand flathead (all seasons)	*0.05*	*0.05*	*0.25*	*0.50*	0.79	1.0	0.05	1.00	0.08	*0.08*	0.93
Sand flathead juveniles (all seasons)	0.01	0.25	1.0	*0.75*	*0.50*	*0.25*	0.02	1.0	0.05	*0.05*	0.80
King George whiting sub-adults (all seasons)	0.71	0.83	1.00	*0.75*	*0.50*	*0.25*	0.94	0.29	0.67	*0.50*	*1.00*
KGW juveniles (all seasons)	1.00	0.77	0.07	*0.05*	*0.02*	*0.01*	1.00	0.70	0.96	*0.50*	0.09
Rock flathead (all seasons)	0.05	1.00	0.48	*0.25*	*0.15*	*0.10*	0.05	0.09	1.00	0.50	0.41
Rock flathead juveniles (all seasons)	0.21	1.00	0.70	*0.50*	*0.25*	*0.05*	*0.02*	0.38	1.00	0.50	*0.64*
Yellow-eye mullet (all seasons)	0.45	1.00	*0.25*	*0.05*	*0.01*	*0.01*	0.36	0.32	1.00	*0.50*	0.04
Yellow-eye mullet juveniles (all seasons)	0.54	1.00	0.02	*0.02*	*0.02*	*0.02*	1.00	0.91	0.59	*0.50*	0.02
Greenback flounder juveniles (all seasons)	1.00	0.32	0.11	*0.05*	*0.05*	*0.02*	0.79	1.00	0.08	0.08	0.21
Calamari (all seasons)	*0.20*	*1.00*	*0.43*	*0.43*	*0.28*	*0.02*	ND	ND	ND	ND	ND
Snapper (all seasons)	*0.01*	*0.02*	*0.05*	*0.25*	1.00	0.48	ND	ND	ND	ND	ND

	Low Confidence in SI Value		**Medium Confidence in SI Value**

See Section 5.3.6 for definition of SI confidence values.

Table 5.11: Western Port habitat affinity values by season for juveniles of certain species for entire bay.

Species	Spring	Summer	Autumn	Winter
Sand flathead	0.72	0.74	0.86	1.00
King George whiting	1.00	0.34	0.28	0.07
Rock flathead	1.00	0.69	0.27	0.29
Yellow-eye mullet	0.12	1.00	0.13	0.01
Greenback flounder	1.00	0.39	0.10	0.13

Table 5.12: Qualitative suitability indices for Western Port and Corner Inlet. Habitat suitability has been classified as high, medium or low suitability based on habitat requirements stated in the literature and question marks indicate uncertainty in the classification. Channels have been divided into main channels and minor channels and classified separately.

	Depth group (m)						Substrate type/biota				
Species life history stage and season	**0+**	**0-2**	**3-5**	**6-10**	**11-15**	**16-20**	**Mangrove**	**Bare**	**Seagrass**	**Reef**	**Channel**
Sand flathead	Low	Medium	Medium	Medium	High	High	Low	High	Low	Low	Main=High Minor=Medium
Sand flathead juveniles	Low	Low	Medium	High	High?	High?	Low?	High	Medium	Low	Main=High Minor=Medium
King George whiting	Medium	Medium	High	High	High	Medium	Low?	High	High	High	Main=Medium Minor=High
KGW juveniles	High	High	Medium	Medium	Low	Low	?	High	High	High	Main=Low Minor=Medium
Rock flathead	High	High	High	Medium	Low	Low	Low?	Low	High	High	Main=Low Minor=Medium
Rock flathead juveniles	High	High	High	Medium	Low	Low	?	High	Medium	?	Main=Low Minor=Medium
Yellow-eye mullet	High	High	High	Medium	Low	Low	High?	High	High	Medium?	Main=Low Minor=Medium
Yellow-eye mullet juveniles	High	High	Medium	Low	Low	Low	Medium	High	Medium	Medium	Main=Low Minor=Medium
Greenback flounder	Low	Medium	Medium	High	High	High	Low?	High	Medium	Low	Main=Medium? Minor=Medium?
Greenback flounder juveniles	High	High	Medium	Low	Low	Low	Medium	High	Medium	Low	Main=Low Minor=Medium
Snapper (includes pinkies)	Low	Low	High	High	High	High	Medium	High	Medium	High	Main=High Minor=High
Snapper juveniles	Low?	Low?	Medium ?	High?	High?	Medium?	Low?	High	Medium	Low?	Main=Low Minor=High
Australian salmon	Low	Medium	High	High	Mediu m	Low?	Medium?	High	Medium	High	Main=High Minor=Low
Australian salmon juveniles	High	High	Medium	Low	Low	Low	?	High	High	Medium	Main=Low? Minor=High?

5.10. Corner Inlet Habitat Suitability Indices

Only limited data were available for Corner Inlet and it was difficult to combine data because of the different gear types and habitats sampled. Trawl data (Hamer & Jenkins 2004) were examined for relationships between depth and substrate type/biota for adult sand flathead, but there were no significant relationships between either habitat category and presence/absence of sand flathead.

There was not enough data to allow us to construct suitability indices for adults of any species, but we have used the limited number of shallow water surveys to construct suitability indices for juveniles of certain species. All suitability indices were calculated from the combined seine net data and were restricted to shallow depths (<2 m) and only three substrate type/biota categories (mangrove, seagrass and bare). In some cases, different habitats were sampled in different surveys and so inferences about habitat suitabilities were confounded by possible differences between surveys (ie. year or season of sampling). For this reason, the restricted number of species that we present habitat suitability values for should all be considered to have a low confidence associated with them.

We have not presented composite habitat suitability models for Corner Inlet as the habitat affinity values only relate to a limited number of habitat variables and are based on limited data. Qualitative indices identified for Corner Inlet are identical to those presented for Western Port (Table 5.12) and are not repeated here.

5.10.1. King George Whiting Juveniles (*Sillaginodes punctata*)

Juvenile King George whiting had high affinity values at 1 m depth on bare sediment (Table 5.13). Habitat affinity values suggest that juvenile King George whiting start to appear in Corner Inlet in winter and spring but peaked in summer.

5.10.2. Rock Flathead Juveniles

Juvenile rock flathead had high habitat affinity values at 1 m depth and in seagrass or bare habitat (Table 5.13). Summer and autumn had the highest values suggesting that the majority of rock flathead recruit to Corner Inlet at this time.

5.10.3. Yank Flathead Juveniles

Habitat affinity values were greatest for juvenile yank flathead at 1 m depth and in bare substrate (Table 5.13). Interestingly, there was no real difference in habitat affinity values by season for juvenile yank flathead suggesting that this species recruits year round in Corner Inlet.

5.10.4. Yellow-eye Mullet

Intertidal mangrove habitat had the highest values of habitat affinity for yellow-eye mullet and summer appeared to be the main season for recruitment of this species in Corner Inlet (Table 5.13).

5.10.5. Greenback Flounder Juveniles

Intertidal and 1 m depth in mangrove or bare habitat had the highest affinity values for juvenile greenback flounder. Spring appeared to be the main season for recruitment for this life stage of this species (Table 5.13).

Table 5.13 Habitat affinity values for juveniles of certain species in Corner Inlet. Data was restricted to shallow depths only and three substrate type/biota classes.

Species	Depth			Substrate type/biota		
	Intertidal	1 m	2 m	Mangrove	Seagrass	Bare
King George whiting juveniles	0.01	1.0	0.01	0.01	0.06	1.0
Rock flathead juveniles	0.01	1.0	0.01	0.01	1.0	0.92
Yank flathead juveniles	0.01	1.0	0.01	0.01	0.04	1.0
Yellow-eye mullet juveniles	1.0	0.18	0.01	1.0	0.17	0.25
Greenback flounder juveniles	1.0	0.30	0.01	0.74	0.06	1.0

Table 5.14: Corner Inlet habitat affinity values by season for juveniles of certain species.

Species	Spring	Summer	Autumn	Winter
King George whiting juveniles	0.33	1.0	0.14	0.38
Rock flathead juveniles	0.27	1.0	0.89	0.22
Yank flathead juveniles	0.82	0.91	0.92	1.0
Yellow-eye mullet juveniles	0.04	1.0	0.35	0.001
Greenback flounder juveniles	1.0	0.13	0.07	0.04

6. Fisheries Dependent Data Analysis

Commercial fisheries data recorded by vessel monitoring systems and logbooks are routinely collected for stock assessment and fishery management purposes in Australia. Fisheries dependent data has intrinsic problems when used as a surrogate for independent sampling data which relate to sampling bias, scale of information, reporting issues and confidentiality (Mace 1997; Starr & Fox 1997; Zheng *et al*. 2001). Despite this, fisheries scientists and managers are becoming increasingly interested in accessing the large amount of information that exists within the fishing community (Bowen 1997; Maurstad 2002). In areas where the characteristics of the fishery are well known, one way of sourcing fisher knowledge is to use commercial catch and effort data along with the implicit assumption that the regions in which the fishers are operating have the highest densities of the targeted species. Several recent studies have explored this approach through the use of logbook data or vessel monitoring systems to investigate the spatial distributions of fish and linking this information to environmental data (Denis & Robin 2001; Zheng *et al*. 2001; Denis *et al*. 2002; Kemp & Meaden 2002; Marrs *et al*. 2002). In this Section, we extend this approach to predict the distribution of suitable habitat, and by extension, fish distributions, based on catch and effort data in Port Phillip Bay.

An analysis of the available fishery independent data for Port Phillip Bay (Section 5) identified considerable limitations in the application of this data for habitat suitability modelling. This was due to a number of factors, including; a lack of accurate location details recorded at sampling sites and poor records of environmental parameters that correspond to sampling locations and times; insufficient sampling effort in important habitat and a low spatial coverage of sampling effort. In order to try and fill some of the gaps in the independent data modelling, we investigated the development of habitat suitability models using commercial fisheries catch data.

6.1. Bay and Inlet Fisheries

There are 59 licensed commercial fishers operating in Port Phillip Bay who primarily fish from small vessels with seine nets (haul, purse and beach), mesh nets and long lines (Anon. 2001). The fishery is managed through a licensing system with restrictions on fishing gear types and minimum allowable fish sizes, with some areas within the Bay also being closed to fishing (eg. marine national parks). Catch and effort data is collated from daily logbook returns that all commercial fishers in Victoria are required to submit on a monthly basis. PIRVic Marine and Freshwater Systems maintains the Victorian catch and effort database on behalf of Fisheries Victoria. The returns from the bay and inlet fishery record information including species caught, weight, time expended fishing and the fishing gear employed (eg. long lines, seines, mesh nets etc). To assist in the analyses of this data, PIRVic Marine and Freshwater Systems developed an ArcView GIS application known as *Catch and Effort Info* (Ball & Coots 2001).

Commercial fishers are required to identify the location of their catch in their returns and in the bay and inlet fishery this is defined by a system of coded fishing blocks. Since April 1998, fishing blocks for Port Phillip and Western Port have been defined by a 5′ x 5′ (approximately 9 km x 9 km) grid. In Port Phillip Bay this represents a system of 41 fishing blocks (Figure 6.1). Prior to 1998, catch and effort returns for Port Phillip Bay were based on seven unevenly sized fishing blocks, which did not provide a satisfactory spatial resolution for the analysis in the present study. As a result, the analysis presented here only addresses catch and effort data for the three year period from 1998-2001, with 2001 being the most recent complete year of data available at the time of the analysis.

We also assessed the Western Port fishery, but determined that the smaller number of fishing blocks (23) in this bay (Figure 6.2) was too limited to apply the habitat suitability modelling technique presented below. Similarly, the Corner Inlet fishery only has 5 unevenly sized fishing blocks and was unsuitable for this analysis.

Catch and effort data is subject to strict confidentiality restrictions. Information from the catch and effort database can only be published or distributed in an aggregated form (with information from at least five fishers) so that no individual fisher’s operations can be identified. As a result, the analysis of the catch and effort data undertaken for this study can only present data that meets this confidentiality rule.

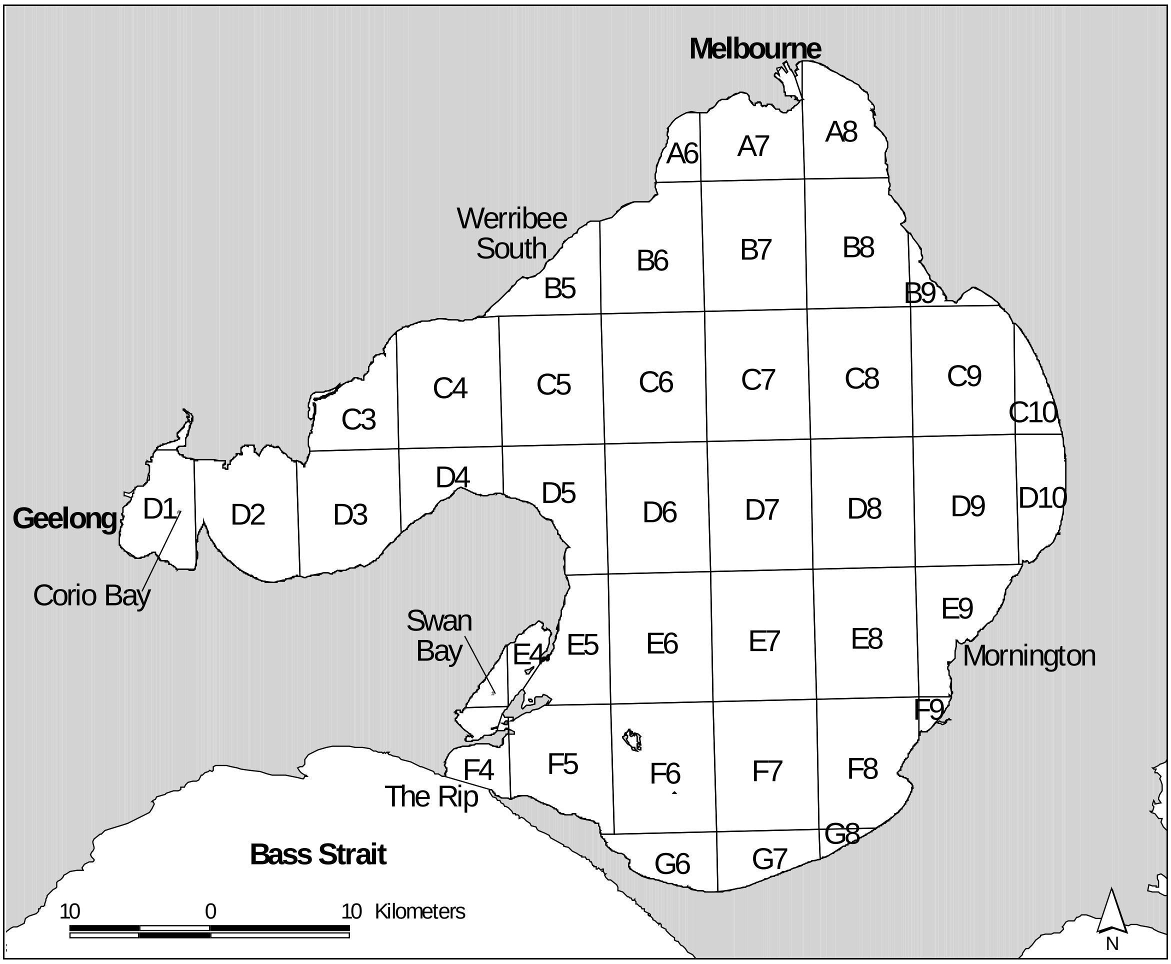


Figure 6.1: Port Phillip Bay commercial fishing catch blocks for fishery returns (blocks in operation since April 1998 and based on a 5′ grid).

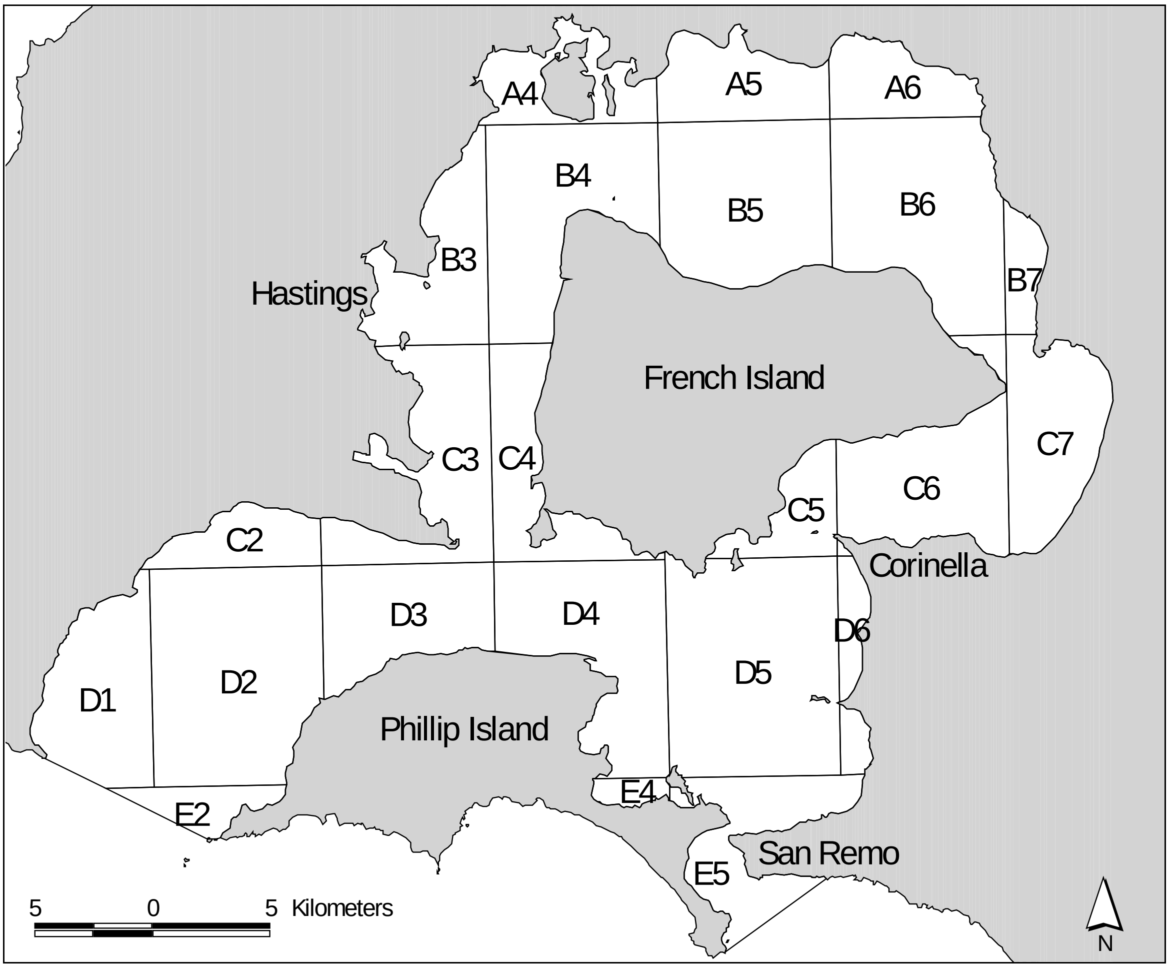


Figure 6.2: Western Port commercial fishing catch blocks for fishery returns (blocks in operation since April 1998 and based on 5′ grid).

6.2. Data Extraction and Analysis

Catch statistics were extracted from the Catch and Effort database for the period April 1998 to June 2001 to provide three complete years of data (Table 6.1). The data was aggregated by seasonal, yearly and three yearly total and average catches and the inclusion of gear type and effort information enabled catch per unit effort (CPUE) values to also be calculated. The data was then integrated with the GIS by linking the fishing block codes to the corresponding codes in the spatial layer representing the boundaries of the fishing blocks.

We analysed catch data for King George whiting *Sillaginodes punctata*, greenback flounder *Rhombosolea tapirina*, yellow eye mullet *Aldrichetta forsteri* and snapper *Pagrus auratus*. Australian salmon catches were also analysed and as the western species *Arripis truttaceus* is the most common in Port Phillip Bay and dominates commercial catches (Coutin 2000a), for the purposes of this study we treated catch records for Australian salmon as being *A. truttaceus*. A summary of catch statistics used in the analysis is given in Table 6.1 and Table 15.1.

The selected species are major components of the Port Phillip Bay fishery and include demersal species (King George whiting, greenback flounder, snapper, sand flathead), a semi-pelagic species (yellow eye mullet) and a pelagic predator (Australian salmon). We assumed that the models for the demersal species would be more reliable because of the closer association of these species with the types of environmental parameters used in this study.

Sand flathead are not specifically targeted by commercial fishers, but is a by-catch to other species and its catch per unit effort is relatively even across the fishing blocks. As such, this species was not suitable for deriving habitat suitability at this scale of data. This highlighted that the analysis presented here is best suited to species that are being specifically targeted by commercial fishers.

The fisheries for King George whiting and Australian salmon are based on juveniles and sub-adults, while the snapper fishery can be divided into juveniles/sub-adults and larger adult fish. The snapper fishery is divided into a long-line fishery that targets larger adult fish and a haul seine and mesh net fishery that targets sub-adults known as 'pinkies'. Haul seining is restricted to the shallower areas of the Bay and while long-lines can be used in shallower areas, they tend to only be used in the deeper waters, due to possible snagging on the seabed in the nearshore areas of the Bay. Mesh nets are used throughout the Bay, but most of the effort for this gear type occurs in the same catch areas as the haul seines.

We used catch per unit effort (CPUE) values where effort was measured by metre lifts for mesh nets, number of shots for haul seines and the number of hook lifts for long lines and catch was weight in tonnes (Figure 6.1). Snapper data were divided into the long line fishery, which tends to target larger adult fish, and the haul seine and mesh net fishery, which primarily targets 'pinkies'. 'Pinkies' have a minimum legal catch size of 27 cm in total length and range in size up to about 35 cm.

One problem with using this type of fisheries dependent data is that CPUE data tends to be on very different scales across the different gear types due to the various units of measurements used and the different efficiencies of the different gears. A recommended approach to this problem is to standardise CPUE data in order to provide a consistent index of species' abundance (Hilborn and Walters 1992). We chose a simple method to standardise CPUE values across the different gear types where we assumed that the average CPUE for each gear type represented a similar density of fish. We divided the CPUE within each block for a specific gear type by the average CPUE for that gear type over the whole bay. Once we had standardised the CPUE values so that data from all gear types was effectively unit-less and at the same scale, we combined these relative values by calculating the mean relative CPUE for each fishing block.

There are some areas within Port Phillip Bay where fishing is excluded (eg. Marine National Parks and Sanctuaries) and other areas where fishing may be restricted to a certain gear type only. We excluded block E4 (Figure 6.1) from the analysis because it is entirely within a Marine National Park (formerly the Swan Bay Fishery Reserve) where commercial fishing was not permitted. The total area in Port Phillip Bay where fishing is prohibited through other regulations is small and an analysis which separated those areas where fishing is prohibited made no difference to the final outcomes. On 16 November, 2002 a new system of Marine National Parks and Sanctuaries was proclaimed in Victoria, including nine sites in Port

Phillip Bay which has increased the area of no-take zones and future analysis should incorporate these areas into the modelling process. Similarly, if the methods for modelling habitat suitability using fisheries dependent data presented here are used in regions with large no-take areas, they will also need to be excluded from the analysis.

Table 6.1. Commercial catch data summary for Autumn 1998 to Summer 2001 (see Appendix 5 - Table 15.1).

Species	Number Of Fishing Blocks	Days	Hours	Shots	Hook-lifts	Hook-hours
Snapper (Mesh Nets/Haul Seines)	233	7,483	48,267	11,279	NA	NA
Snapper (Lines)	174	2326	NA	NA	732,779	2,738,995
King George whiting (Mesh Nets/Haul Seines)	387	9,767	62,085	14,243	NA	NA
King George whiting (Lines)	19	303	NA	NA	31755	62,403
Australian salmon (Mesh Nets/Haul Seines)	256	8,050	51,201	11,621	NA	NA
Australian salmon (Lines)	11	189	NA	NA	79	2,153
Flounder (Mesh Nets/Haul Seines)	265	7,843	51,388	11,774	NA	NA
Flounder (Lines)	0	NA	NA	NA	0	0
Yellow-eye mullet (Mesh Nets/Haul Seines)	282	8715	55,152	12,725	NA	NA
Yellow-eye mullet (Lines)	0	NA	NA	NA	0	0
Total	**1,627**	**44,676**	**268,093**	**61,643**	**764,613**	**2,803,551**

6.3. Development of Habitat Suitability Models

The nature of fisheries dependent data is quite different to the fisheries independent data and so the approach to developing habitat suitability models from this data also had to be different. Unlike the fisheries independent data which consisted of point samples, the fisheries dependent data consists of unspecified fishing locations within a larger area or fishing block.

6.3.1. Habitat Data

GIS polygon layers developed by PIRVic Marine and Freshwater Systems for depth, sediment and substrate type/biota (Section 3) were used to provide the habitat information to characterise each fishing block (Figure 6.1). As Port Phillip Bay is predominantly marine, salinity and temperature were not considered to be significant influences on the distribution of the fish species investigated in this study and were not included in this analysis.

The Identity command in ARCINFO was used to overlay the fishing block, substrate type/biota, depth and sediment polygon layers and calculate the geometric intersection of each layer. Two layers were intersected at a time with the Identity command so that the output layer would then form one of the input layers to intersect with the next habitat layer (ie. a geometric intersection was calculated on the fishing block and substrate type/biota layers first and the output from this was then intersected with the depth layer and so on until all layers had been intersected). The output of this process was a single combined layer which retained the spatial features and attributes values for each of the input layers. Figure 6.3 illustrates this process. A composite habitat code for each feature in the output layer was then calculated by combining the habitat codes from each input layer (Figure 6.3). The attributes of the GIS habitat layers used in this analysis are summarised in Table 6.2.

The attribute table for the combined fishing block/habitat layer from the Identity process (Figure 6.3) included all of the attribute values from the input layers, a composite habitat code and the area in m^2 of each combined spatial feature. This table was exported to Excel and a pivot table created to summarise each fishing block in terms of the total area of every possible combination of habitat parameters (ie. substrate type/biota, depth and sediment). A total of 135 habitat combinations present in Port Phillip Bay were identified in this analysis (Table 6.2).

6.3.2. Statistical Analysis

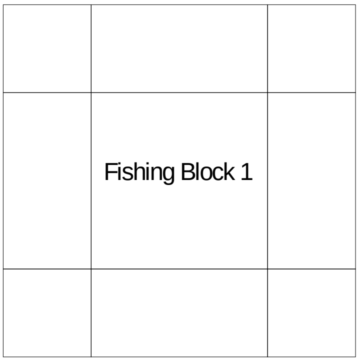
Interaction plots were used to examine catch data for each species through time and across all the fishing blocks in order to determine whether there were seasonal differences in the pattern of catch (CPUE) by fishing block. Examination of the graphs indicated that the pattern of catches across fishing areas was consistent between seasons except for large snapper that had low to non-existent catches in winter. The winter data was excluded for this life history stage of this species. As we were interested in spatial patterns in fish distributions and not in differences between years, data was then pooled across all seasons and years so that each fishing block had one value only for each species.

In order to link fish distributions with habitat parameters we have assumed that habitats with higher CPUEs also have higher habitat suitability for that particular species. Due to the problems associated with working with commercial catch data and in particular the difference in scale between the CPUE data and the habitat data, we decided to use a multivariate approach to linking habitat to fish distributions. The first step was to create a data matrix of habitat that was independent of the fact that fishing blocks had different sizes. To do this we summarised the habitat parameters of each fishing block by the proportion of each habitat combination (Table 6.2) that was present in that block, so that our data matrix consisted of an array of rows (habitat combinations) and columns (fishing blocks). We then created a similarity matrix whereby the similarities between each fishing block were calculated using the Bray Curtis coefficient and which underlies the subsequent multivariate analysis.

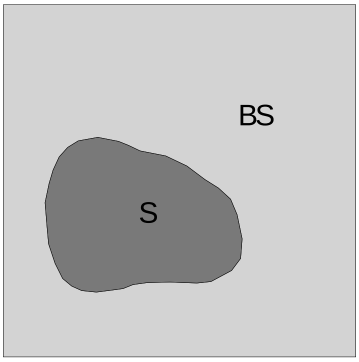
In order to test the *a priori* hypothesis that fishing blocks where fish were caught differed in terms of their habitat parameters from areas where fish of that species were not caught, a multivariate Analysis Of Similarities (ANOSIM) was carried out. Where there was a significant difference ($P<0.05$) between the habitat parameters of fishing blocks where a particular species was caught versus the fishing blocks where that species was not caught, the relationship between the habitat parameters of these fishing blocks was further explored using ordination and cluster analyses. For the ordination we used non metric multidimensional scaling (nMDS) which attempts to place the fishing blocks on a 'map' in such a way that the rank order between the fishing blocks on the 'map' represents the rank order of the similarities in the similarity matrix. The cluster analysis progressively links the samples based on the calculated similarities in to hierarchical groups and the analysis is represented in the form of a dendrogram. Primer Version 5 (© PRIMER-E 2000) was used for all multivariate analysis.

Fishing blocks were arbitrarily grouped according to the cluster analysis at a 40% cut-off level (Figure 6.4) and the average of the standardised CPUE of each species overlayed on the ordination in the form of a bubble plot, such that the bigger the bubble, the greater the relative CPUE of that species. Groups of fishing blocks determined from the cluster analysis were then assigned to high catch or low catch groups according to the size of the relative CPUE values within each group. A series of simple rulings were then used to determine whether environmental combinations were 'high', 'medium' or 'low' suitability for the species in question (Table 6.3). These rulings were based on the assumption that the consistent presence of an environmental parameter in a cluster group will be important in determining the yield measured by mean CPUE. Primer Version 5 (© PRIMER-E 2000) was used to produce the bubble plots and to extract the information regarding the presence of habitat parameters in the cluster groups.

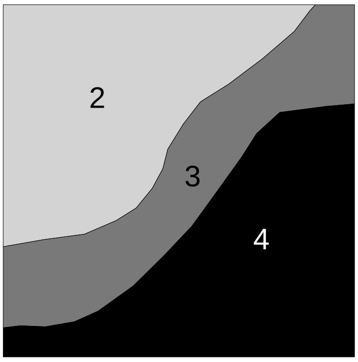
Once the combined habitat parameters were defined as high, medium, low or undefined for each species (Table 6.4) the composite GIS habitat layer was reclassified according to these categories to create a predictive map of overall habitat suitability.



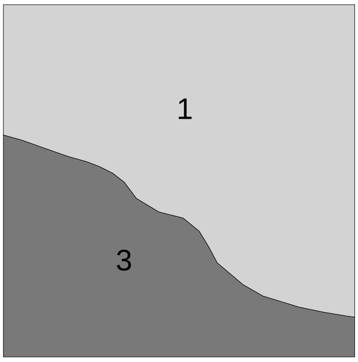
A. Fishing Block



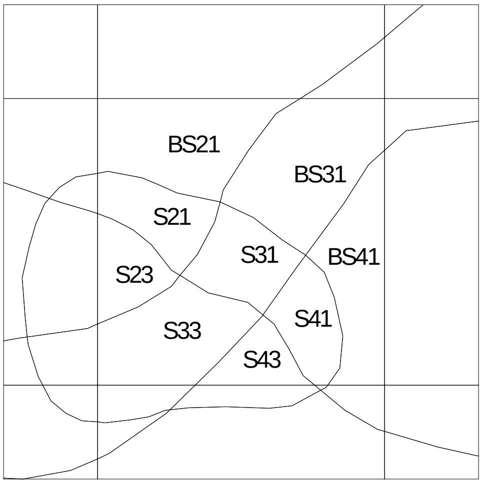
B. Substrate type/biota



C. Depth



D. Sediment



E. Output combined habitat layer

Fishing Block	Sub code	Depth code	Sed code	Composite
Fishing Block 1	BS	2	1	BS21
Fishing Block 1	BS	3	1	BS31
Fishing Block 1	BS	4	1	BS41
Fishing Block 1	S	2	1	S21
Fishing Block 1	S	3	1	S31
Fishing Block 1	S	2	3	S23
Fishing Block 1	S	4	1	S41
Fishing Block 1	S	3	3	S33
Fishing Block 1	S	4	3	S43

F. Output layer attribute table

Figure 6.3: Illustration of process applied to characterise each fishing block by its habitat characteristics. Input GIS layers (A-D above) were overlayed and a geometric intersection calculated in ARCINFO to produce a single output layer which retained the spatial features of each input layer (E above). The output layer also retained the attribute table items of each input layer (see Table 6.2) and these were combined into a single "Composite" code (E & F above). Each row in the output layer attribute table (F above) corresponds to a polygon in the combined habitat layer (E above).

Table 6.2. Habitat parameters used to classify substrate type/biota, depth and sediment types in commercial fishing blocks. Habitat parameter composite code = substrate type/biota code & depth code & sediment code (eg. composite code S43 = seagrass at depth 5-10 m on sand-clay substrate).

Substrate type/biota class	Substrate type/biota code	Depth class	Depth code	Sediment class	Sediment code
Seagrass (predominantly *Heterozostera tasmanica* & *Zostera muelleri* – includes seagrass & macroalgae mix)	S	intertidal	1	Clay	1
Macroalgae (undefined species)	M	0-2 m	2	Sand-silt-clay	2
Amphibolis antarctica (includes *A. antarctica* & macroalgae mix)	A	2-5 m	3	Sand-clay	3
Subtidal Rocky reef (includes macroalgae on reef)	R	5-10 m	4	Fine sand	4
Pyura stolonifera (includes *Pyura* & macroalgae mix)	P	10-15 m	5	Coarse sand	5
Bare intertidal	BI	15-20 m	6	Rocky reef	6
Bare subtidal	BS	20-30 m	7	Medium sand	7
Drift algae	D	30 m +	8		
Intertidal rocky reef	IR				
Seagrass bare edge (bare sediment within a 15 m buffer from edge of seagrass beds)	SE				

Table 6.3. Summary of steps taken in assigning habitat combinations to a 'high', 'medium' or 'low' suitability.

Ruling	Suitability
Habitat combination is present in 100% of fishing blocks in a cluster group assigned 'high' catch (CPUE).	High
Habitat combination is present in 100% of fishing blocks in a cluster group assigned 'high' catch (CPUE) and habitat combination is also present in 100% of fishing blocks in a cluster group assigned 'low' catch (CPUE).	Medium
Habitat combination is present in >50% of fishing blocks in a cluster group assigned 'high' catch (CPUE) but has not already been assigned a suitability value.	Medium
Habitat combination is present in 100% of fishing blocks in a cluster group assigned 'low' catch (CPUE).	Low
Habitat combinations present in <50% of fishing blocks in a cluster group assigned 'high' catch (CPUE).	Undefined
Habitat combinations present in <100% of fishing blocks in a cluster group assigned 'low' catch (CPUE).	Undefined

Table 6.4. Habitat suitability classification by species for habitat parameter combinations (see Table 6.2) using commercial fishery catch statistics. Greenback flounder classifications were identical to King George whiting (see text for further details).

Habitat composite codes (See Table 6.2)	King George whiting (sub-adult)	Australian salmon	Yellow eye mullet	Snapper - pinkies	Snapper - adults
A14	undefined	undefined	undefined	undefined	undefined
A17	undefined	undefined	undefined	undefined	undefined
A24	undefined	undefined	undefined	undefined	undefined
A27	undefined	undefined	undefined	undefined	undefined
A34	undefined	undefined	undefined	undefined	undefined
A37	undefined	undefined	undefined	undefined	undefined
A44	undefined	undefined	undefined	undefined	undefined
A47	undefined	undefined	undefined	undefined	undefined
BI11	undefined	undefined	undefined	undefined	undefined
BI12	medium	medium	medium	medium	medium
BI13	high	high	high	high	low
BI14	medium	high	high	medium	medium
BI15	undefined	undefined	undefined	undefined	undefined
BI17	medium	medium	low	low	low
BS22	high	high	high	high	low
BS23	high	high	high	high	low
BS24	medium	high	medium	medium	medium
BS25	undefined	medium	medium	undefined	undefined
BS27	medium	medium	low	low	medium
BS31	medium	medium	medium	medium	undefined
BS32	high	high	high	high	low
BS33	high	high	high	high	medium
BS34	medium	medium	medium	medium	medium
BS35	low	high	high	low	low
BS37	medium	medium	low	low	medium
BS41	high	high	high	high	low
BS42	high	high	high	high	medium
BS43	high	high	high	high	medium
BS44	medium	medium	medium	medium	medium
BS45	low	medium	medium	low	low
BS47	medium	medium	medium	low	medium
BS51	high	high	high	high	low
BS52	high	high	high	high	medium
BS53	medium	medium	medium	high	medium
BS54	medium	medium	medium	medium	medium
BS55	low	medium	medium	low	low
BS57	low	low	low	low	medium
BS61	low	low	low	low	high
BS62	medium	medium	medium	medium	high
BS63	medium	medium	medium	medium	high
BS64	low	low	low	low	high
BS65	undefined	undefined	undefined	undefined	undefined
BS67	medium	medium	undefined	undefined	medium

Table 6.4 continued

BS71	low	low	low	low	high
BS72	undefined	undefined	undefined	undefined	medium
BS73	undefined	undefined	undefined	undefined	medium
BS74	medium	medium	undefined	undefined	medium
BS77	undefined	undefined	undefined	undefined	undefined
BS81	undefined	undefined	undefined	undefined	undefined
BS83	undefined	undefined	undefined	undefined	undefined
BS84	medium	medium	undefined	undefined	undefined
BS87	undefined	undefined	undefined	undefined	undefined
D14	medium	medium	medium	medium	medium
D23	medium	medium	medium	medium	undefined
D24	medium	medium	medium	medium	undefined
D34	medium	medium	medium	medium	undefined
EI12	medium	medium	medium	medium	undefined
EI13	high	high	high	high	low
EI14	high	high	high	high	medium
EI17	high	high	low	low	low
ES22	high	high	high	high	low
ES23	high	high	high	high	low
ES24	high	high	high	high	medium
ES27	high	high	low	low	low
ES32	high	high	high	high	low
ES33	high	high	high	high	low
ES34	high	high	low	low	low
ES37	high	high	low	low	low
ES42	high	high	high	high	low
ES43	high	high	high	high	low
ES44	high	high	low	low	low
ES47	high	high	low	low	low
ES52	high	high	high	high	low
ES53	medium	medium	medium	medium	undefined
ES54	medium	high	undefined	low	undefined
ES57	undefined	undefined	undefined	undefined	undefined
IR16	medium	medium	medium	medium	medium
M12	medium	medium	medium	medium	undefined
M13	medium	medium	medium	medium	undefined
M14	medium	medium	medium	medium	undefined
M17	undefined	undefined	undefined	undefined	undefined
M22	high	high	high	high	low
M23	medium	medium	medium	medium	medium
M24	medium	medium	medium	medium	medium
M27	medium	medium	undefined	undefined	undefined
M32	high	high	high	high	low
M33	medium	medium	medium	medium	medium
M34	medium	medium	medium	medium	medium
M37	undefined	undefined	undefined	undefined	undefined
M42	medium	medium	medium	medium	undefined
M43	medium	medium	medium	medium	medium
M44	medium	medium	medium	medium	medium

Table 6.4 continued

M47	undefined	undefined	undefined	undefined	undefined
M53	undefined	undefined	undefined	undefined	undefined
M54	medium	medium	medium	undefined	medium
P12	medium	medium	medium	medium	undefined
P17	medium	medium	undefined	undefined	undefined
P22	medium	medium	medium	medium	undefined
P23	high	high	high	high	low
P24	medium	medium	undefined	undefined	undefined
P27	medium	medium	undefined	undefined	undefined
P32	high	high	high	high	low
P33	high	high	high	high	low
P34	high	high	low	low	low
P35	undefined	undefined	undefined	undefined	undefined
P37	medium	medium	undefined	undefined	undefined
P42	medium	medium	medium	medium	undefined
P43	undefined	undefined	undefined	undefined	undefined
P44	high	high	low	low	low
P52	medium	medium	medium	medium	undefined
P54	medium	medium	undefined	undefined	undefined
P64	undefined	undefined	undefined	undefined	undefined
R26	medium	medium	medium	medium	medium
R36	high	high	high	high	medium
R46	high	high	high	high	low
R56	medium	medium	medium	medium	undefined
S12	high	high	high	high	low
S13	high	high	high	high	low
S14	high	high	high	high	medium
S17	high	high	low	low	low
S22	high	high	high	high	low
S23	high	high	high	high	low
S24	high	high	medium	medium	medium
S27	high	high	low	low	low
S32	high	high	high	high	low
S33	high	high	high	high	low
S34	high	high	low	low	low
S37	high	high	low	low	low
S42	high	high	high	high	low
S43	high	high	high	high	low
S44	high	high	low	low	low
S47	high	high	low	low	low
S52	high	high	high	high	low
S53	medium	medium	medium	medium	undefined
S54	medium	medium	undefined	undefined	undefined

6.4. Habitat Suitability Modelling Results

There were significant differences in habitat characteristics between fishing blocks where fish had been caught, compared to fishing blocks where no fish were caught for all species examined except sand flathead (Table 6.5). Ten cluster groups were defined from the 40% similarity level (Figure 6.4) and there was good correspondence with the cluster groupings and the 2-dimensional nMDS ordination (Figure 6.5). Blocks A6 and F4 clustered out singly (Figure 6.4) and so could not be used in the analysis described above, as more than one block was required per cluster group. Fishing block E4 (Swan Bay) was excluded from the cluster analysis as it is entirely within a Marine National Park and no fishing is permitted in this area. Fishing block E9 clustered out in group 1 (Figure 6.4) but we have excluded this area from the subsequent analysis as it appeared to be a large outlier in several cases (eg. Australian salmon, large snapper and yellow-eye mullet) and is discussed further below.

Table 6.5. Results of ANOSIM comparing the proportion of habitats of fishing blocks where a species was taken from with the habitats where that species was not caught.

Species	R-value	P-value
King George whiting	0.385	0.001
Greenback flounder	0.248	0.002
Snapper pinkies	0.225	0.002
Large snapper	0.184	0.014
Sand flathead	-0.001	0.49
Australian salmon	0.331	0.001
Yellow-eye mullet	0.206	0.004

6.4.1. King George whiting

The cluster groups designated as 'high' and 'low' catch (CPUE) groups for King George whiting are shown in Figure 6.7. The relative CPUE values for King George whiting were dominated by fishing block G6 (Figure 6.7), located at the southern end of the Bay (Figure 6.1). The correspondence between the cluster groupings and the relative CPUE was reasonably good across all groups. The only exceptions were the moderate relative CPUE values in areas B7, B9 and D6, all of which were in cluster groups assigned a 'low' catch (CPUE) and the comparatively low CPUE in area C5, which was in a 'high' catch cluster group (Figure 6.7).

Assigning suitability codes to the habitat composites following our simple ruling system resulted in the predictive model of habitat suitability for King George whiting (> 27 cm TL) shown in Figure 6.8. The most notable feature of this model is the 'high' suitability of all habitats that include seagrass or seagrass-edge, which are primarily located in the southern and western areas of the Bay. The shallow bare areas on fine substrate in the northern part of the Bay were also predicted to provide suitable habitat for King George whiting and similarly the reef areas along the north eastern shores of the Bay. The areas classed as low suitability habitats are mainly the deeper bare substrate in the centre of the Bay and the coarse sand habitat on the eastern side of the Bay.

6.4.2. Greenback flounder

The flounder catch in Port Phillip Bay consists of greenback flounder and the less abundant long-snouted flounder *Ammotretis rostratus* (Coutin 2000a). Catch and effort data for the period 1998-2001 includes more records for "flounder – unspecified" than greenback flounder and as a result it appears that many of the catches of greenback flounder may have actually been recorded along with long-snouted flounder in the "flounder – unspecified" category. For the purposes of this modelling we combined the catches from both the greenback flounder and "flounder – unspecified".

The catch for flounder came from exactly the same fishery blocks as King George whiting and is probably in reality a by-catch of the more highly valued King George whiting. As a result, the habitat suitability model was identical to King George whiting (Figure 6.8) and is not included as a separate figure.

6.4.3. Australian salmon

The relative CPUE for Australian salmon was dominated by the large value for fishing block E9 (Figure 6.9) but this block was treated as an outlier and is discussed further below. There was very good agreement with the CPUE values and the classification of the cluster groups to 'high' or 'low' catch (CPUE). The only exceptions were blocks C5 and F5, which had low relative CPUE values and were in cluster groups assigned to a 'high' catch (CPUE). The resultant suitability codes and habitat suitability model (Figure 6.10) were similar to that of King George whiting in that the majority of seagrass associated habitat and shallow finer sediment is classified as 'high' suitability while the low suitability habitat is the deeper central region of the Bay. The main difference in predicted habitat suitability in comparison to King George whiting was the shallow coarse sand strip on the eastern edge of the Bay that had been classed as medium suitability for Australian salmon and the very shallow strip of sandy sediment along the western shore classed as high suitability.

6.4.4. Yellow-eye mullet

The cluster groups assigned to high or low catch (CPUE) groups for yellow-eye mullet is shown in Figure 6.11. In this case there was less agreement between the relative CPUE values and the assigned catch groups. We assigned cluster group 10 (G6, G7, F5, F6) to a 'low' catch (CPUE) group although, arguably, it could have been assigned a 'high' catch (CPUE) rating, in which case models for Australian salmon and yellow-eye mullet would have been identical. Even with the different classification of group 10 the models were very similar; with the majority of the shallow and seagrass associated habitat having a high or a medium suitability and the deeper, bare areas having a low suitability. The main difference in the predictive map for the yellow-eye mullet versus Australian salmon was the greater amount of low suitability habitat in the southern region of the Bay (Figure 6.12).

6.4.5. Snapper 'pinkies'

The majority of the catch of snapper pinkies came from cluster groups 4, 5, 8 and 9 (Figure 6.13 & Figure 6.5) with fishing block A7 dominating the relative CPUE values. There was reasonably good agreement with the assignment of cluster groups to high or low catch (CPUE) groupings and the relative CPUE. The main exceptions were blocks B9 and F5, both of which were in cluster groups assigned 'low' catch ratings although they had moderate relative CPUE values, and C5 which had a low relative CPUE and was in a 'high' catch cluster group. The predictive map of habitat suitability for snapper pinkies was similar to the other species, although there was more high suitability habitat in the northern part of the Bay and at the entrance to the Geelong Arm, and more low suitability habitat in the southern and eastern fringe of the Bay (Figure 6.14).

6.4.6. Snapper (large)

The catch of large snapper was mainly taken from different fishing blocks to the other species (Figure 6.15) and the assignment of cluster groups to 'low' or 'high' catch groups is reasonably consistent with the relative CPUE values. Cluster group 5 was assigned a 'high' catch rating although blocks A8, A7 and C3 had comparatively low relative CPUE values and this group could arguably have been assigned a 'low' catch rating although all blocks within the group did return at least some large snapper. The resulting habitat suitability map (Figure 6.16) for large snapper differs to those for the other species. Suitable habitat for large snapper is predicted to be the deeper areas of the Bay while the very shallow seagrass associated areas and the coarser sediment on the eastern side of the Bay are mainly predicted to be of low habitat suitability for this life history stage (Figure 6.16).

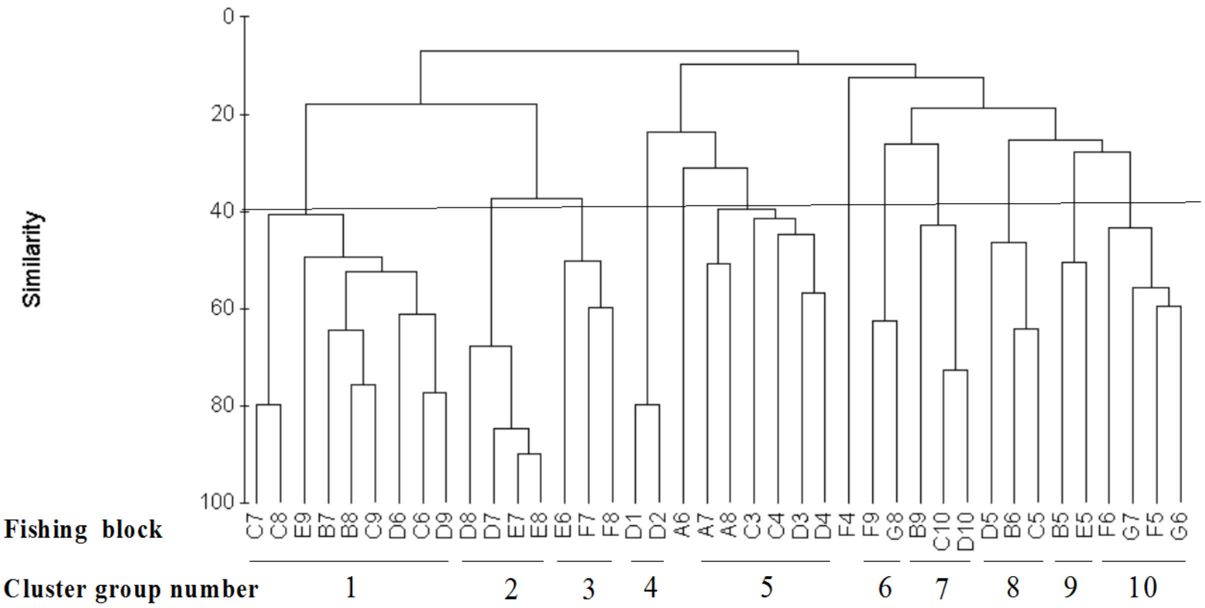


Figure 6.4: Dendrogram resulting from cluster analysis. Solid line indicates 40% cut-off point used for determining cluster groups used in subsequent data exploration. See Figure 6.6 for spatial distribution of cluster groups.

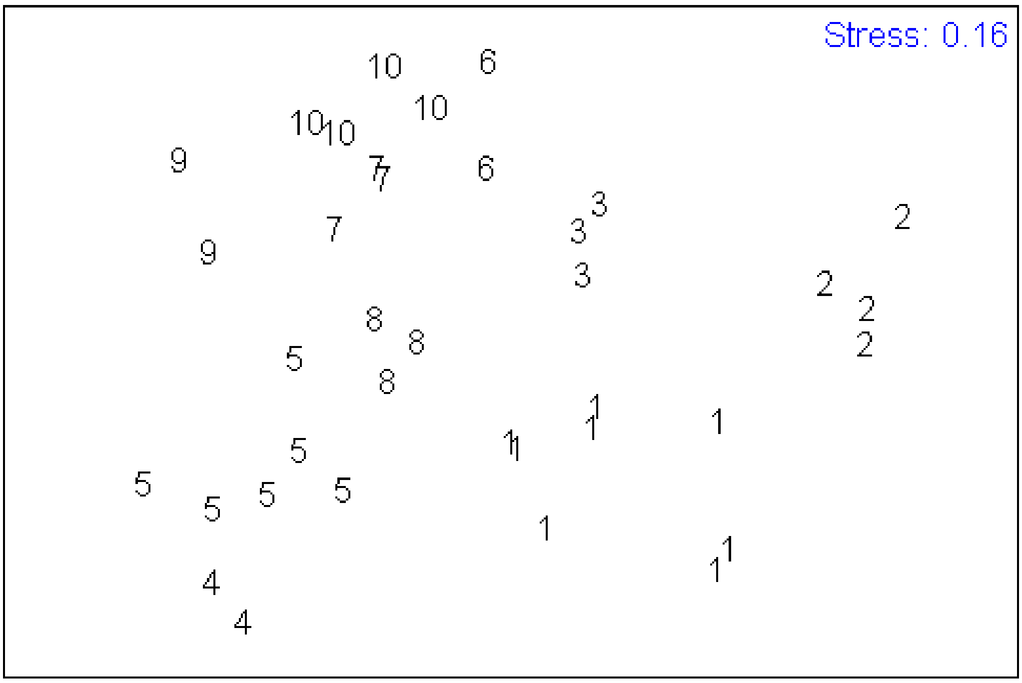


Figure 6.5: Ordination following nMDS of fishing block environmental data. Cluster groups (see Figure 6.4) are indicated on the ordination. Note that fishing blocks not used in subsequent habitat classification (ie. areas A6, E4, E9, F4) are not shown on this ordination (see Section 6.4 for further details).

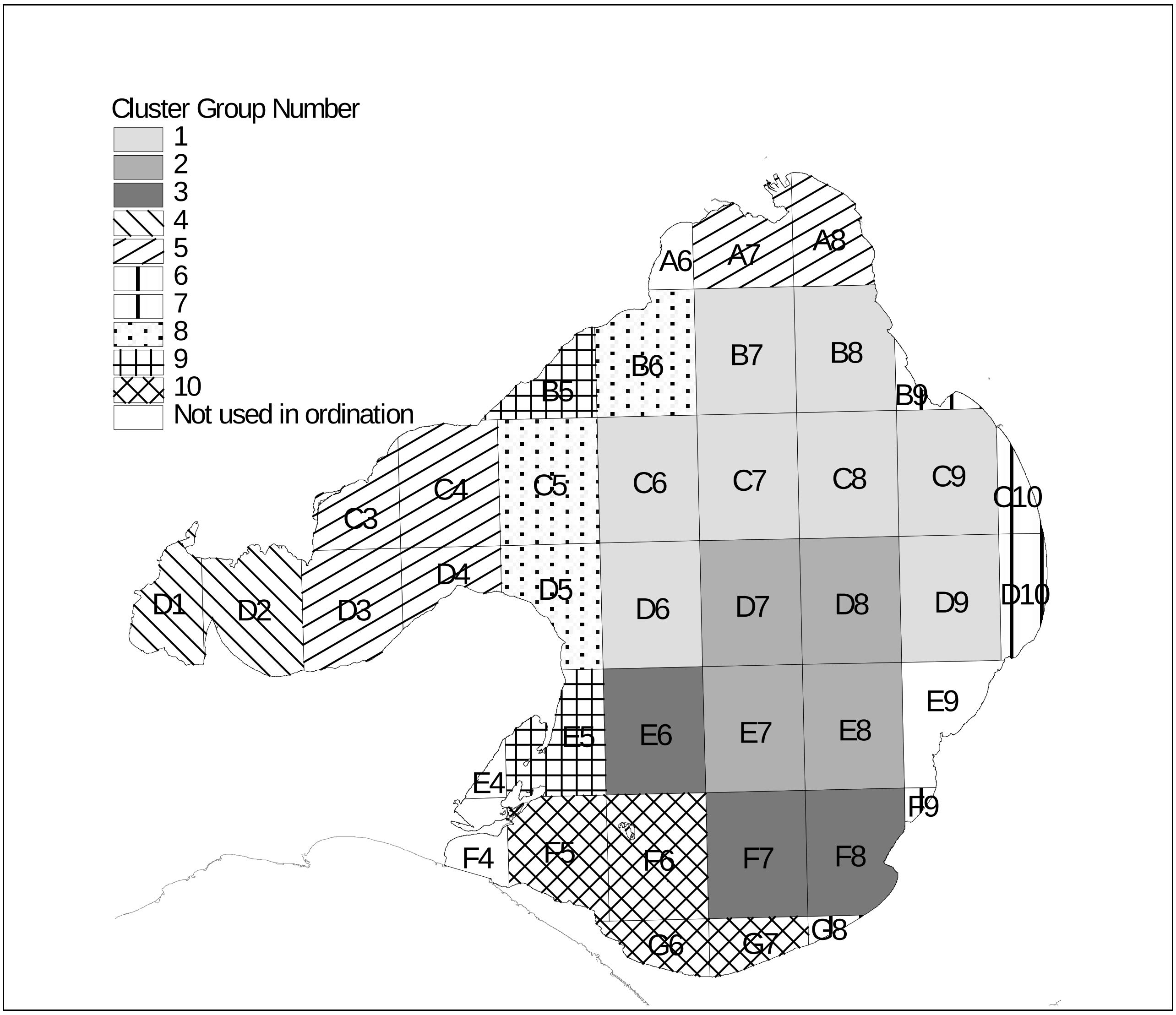


Figure 6.6: Spatial distribution of fishing block cluster groups from Figure 6.4.

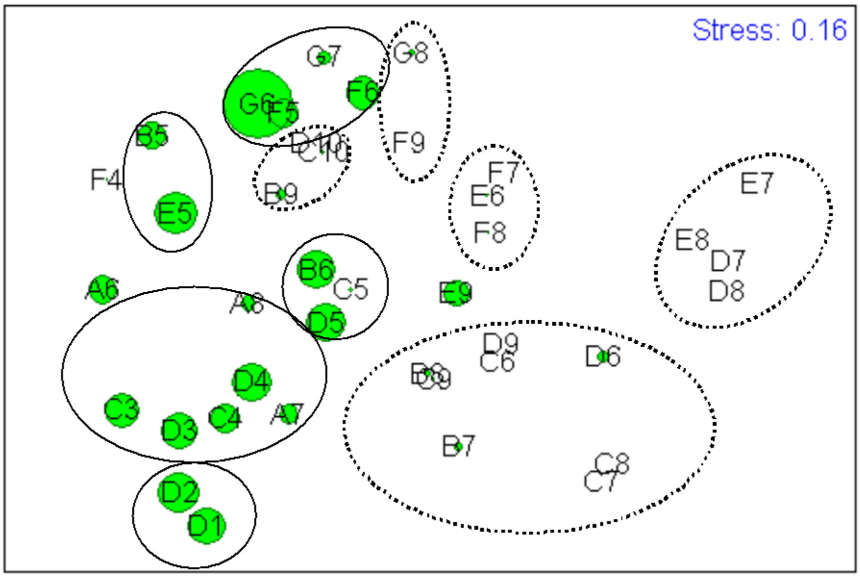


Figure 6.7: MDS based on percentage data of habitat composites with standardised CPUE of King George whiting overlayed in the form of bubbles where the bigger the bubble the larger the relative CPUE. Groups from the cluster analysis are also outlined on the plot and cluster groups assigned to a high catch are enclosed by a solid line and cluster groups assigned to a low catch are enclosed by a dotted line.

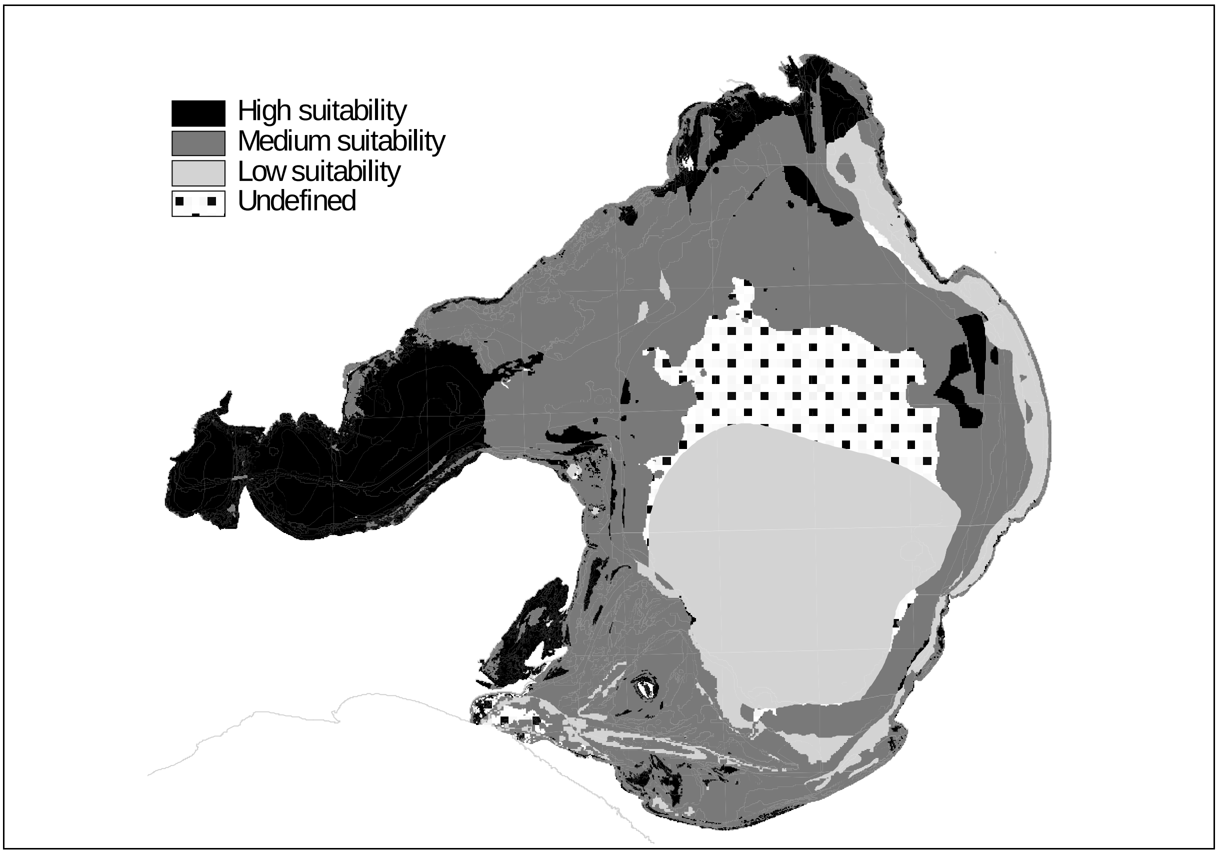


Figure 6.8: Fishery dependent habitat suitability model for King George whiting in Port Phillip Bay (model also applies to greenback flounder)

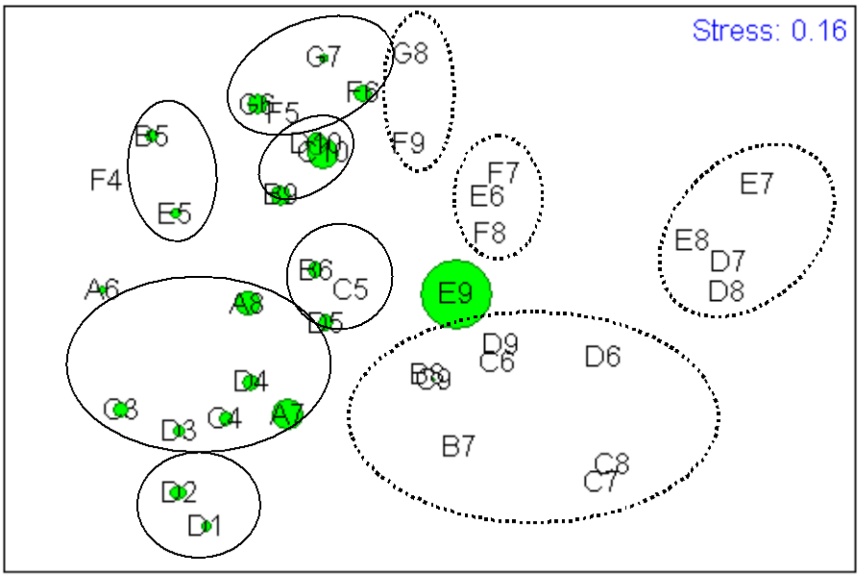


Figure 6.9: MDS based on percentage data of habitat composites with standardised of CPUE of Australian salmon overlayed in the form of bubbles where the bigger the bubble the larger the relative CPUE. Groups from the cluster analysis are also outlined on the plot and cluster groups assigned to a high catch are enclosed by a solid line and cluster groups assigned to a low catch are enclosed by a dotted line.

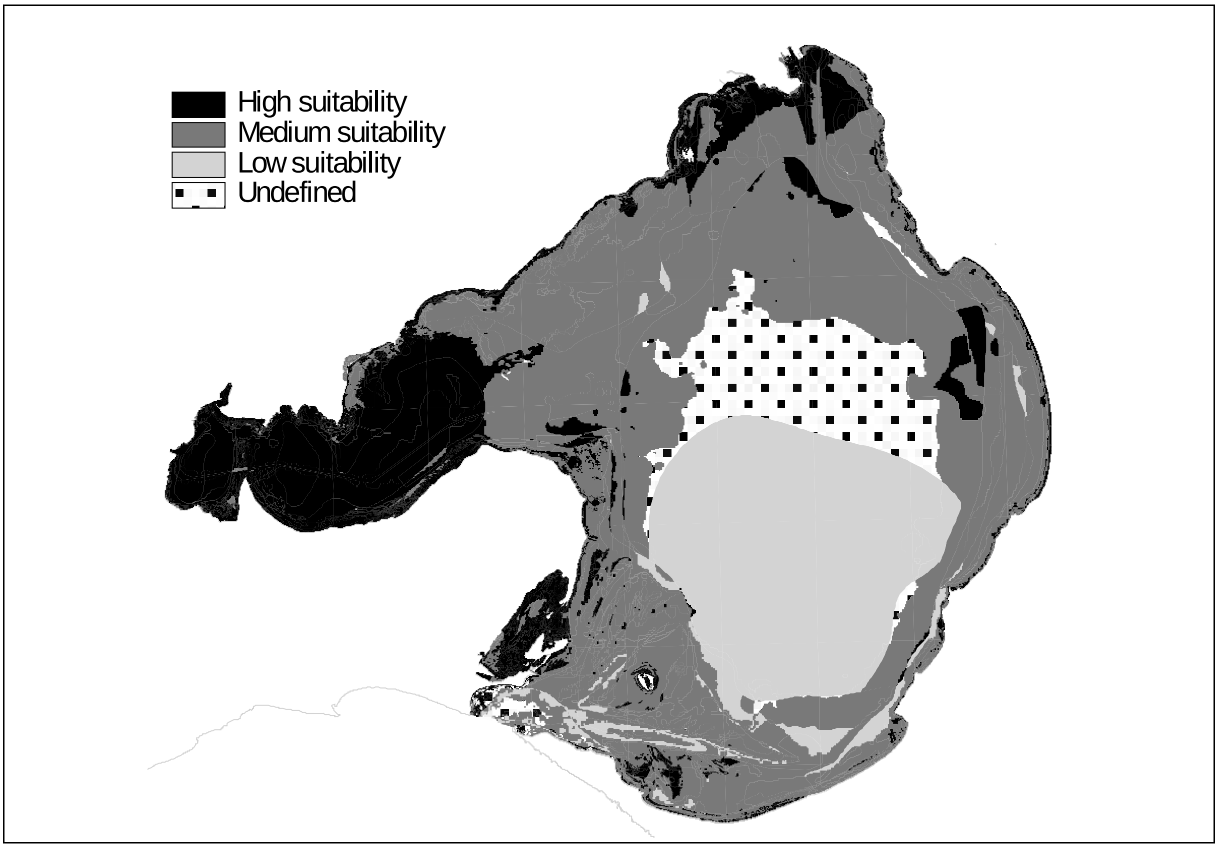


Figure 6.10: Fishery dependent habitat suitability model for Australian salmon in Port Phillip Bay.

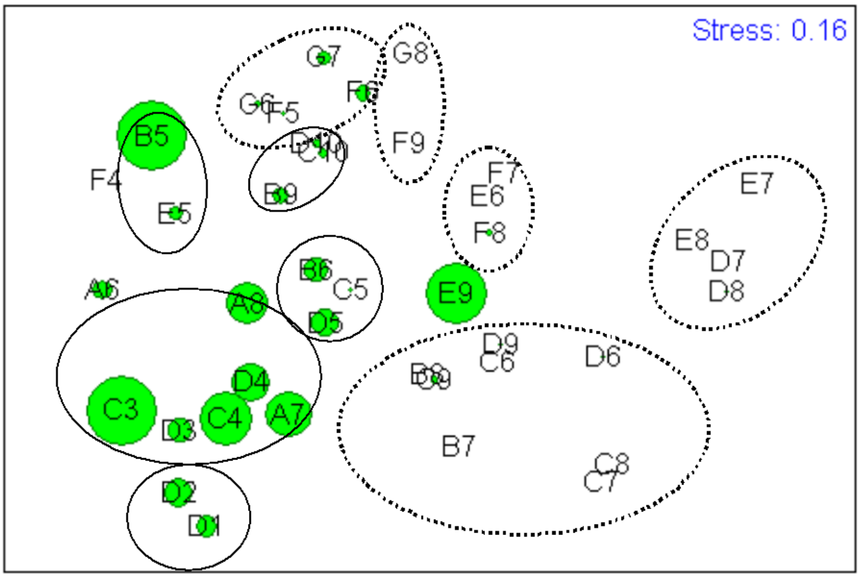


Figure 6.11: MDS based on percentage data of habitat composites with standardised CPUE of yellow-eye mullet overlayed in the form of bubbles where the bigger the bubble the larger the relative CPUE. Groups from the cluster analysis are also outlined on the plot and cluster groups assigned to a high catch are enclosed by a solid line and cluster groups assigned to a low catch are enclosed by a dotted line.

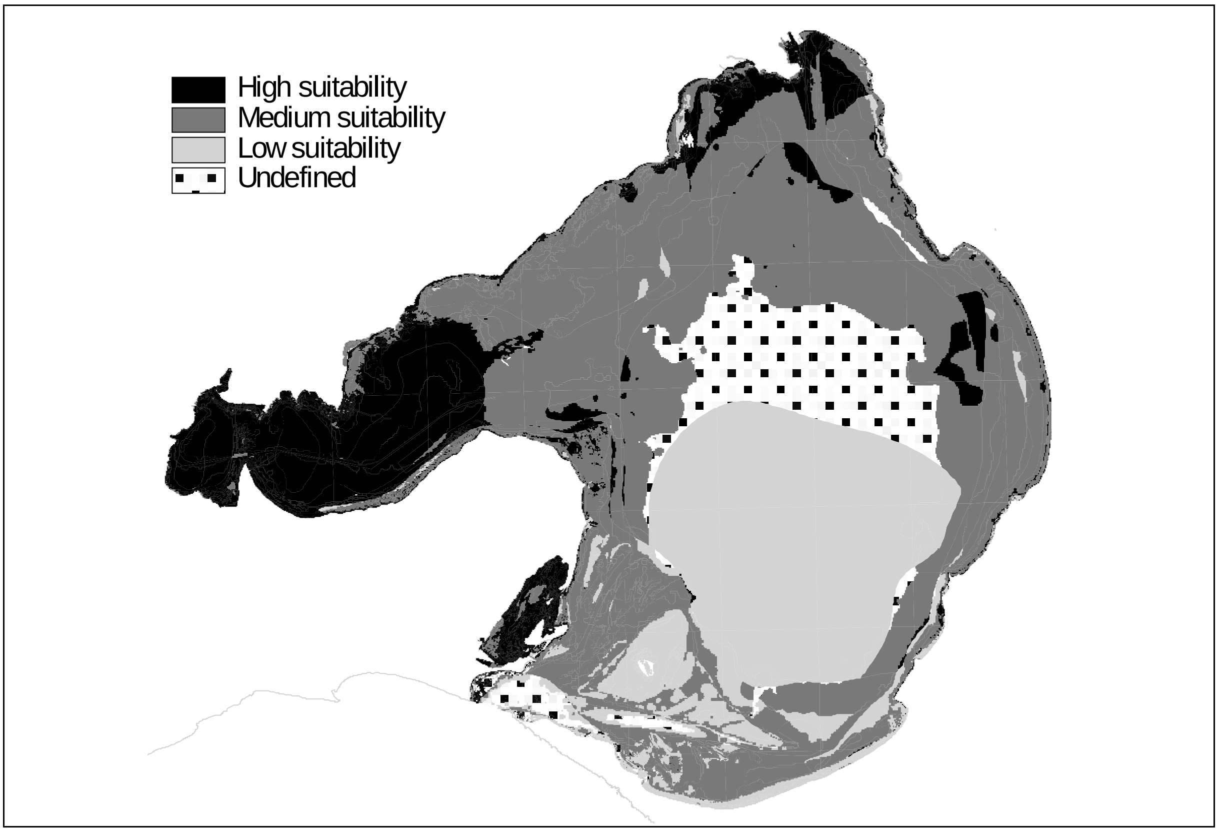


Figure 6.12: Fishery dependent habitat suitability model for Yellow-eye mullet in Port Phillip Bay.

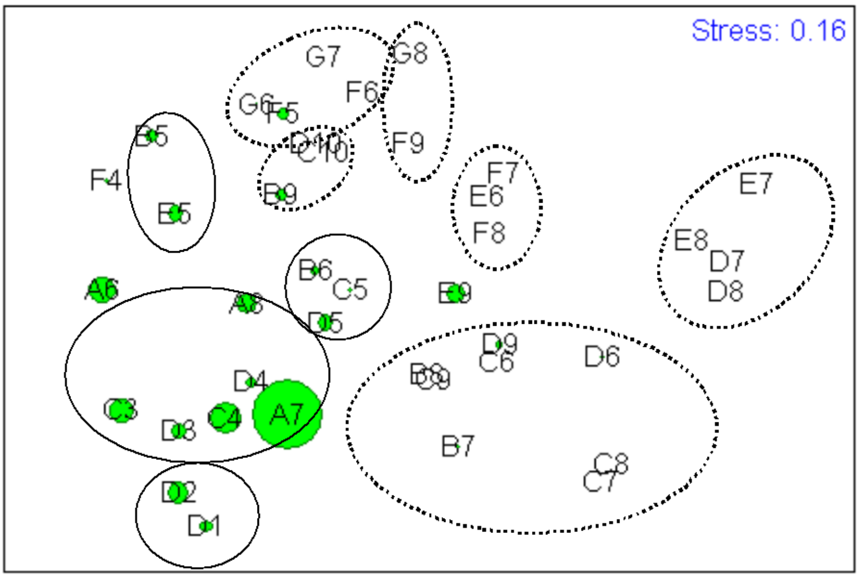


Figure 6.13: MDS based on percentage data of habitat composites with standardised CPUE of snapper pinkies overlayed in the form of bubbles where the bigger the bubble the larger the relative CPUE. Groups from the cluster analysis are also outlined on the plot and cluster groups assigned to a high catch are enclosed by a solid line and cluster groups assigned to a low catch are enclosed by a dotted line.

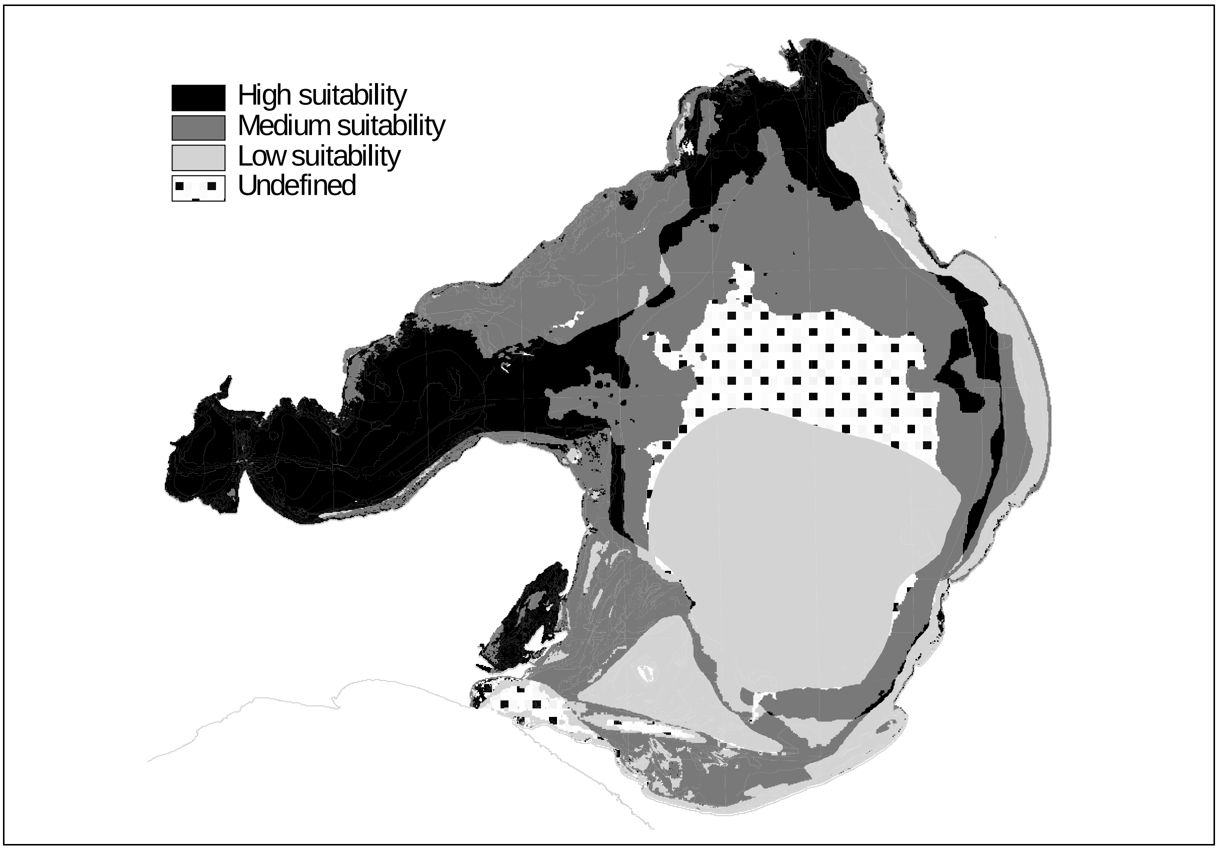


Figure 6.14: Fishery dependent habitat suitability model for snapper "pinkies" in Port Phillip Bay.

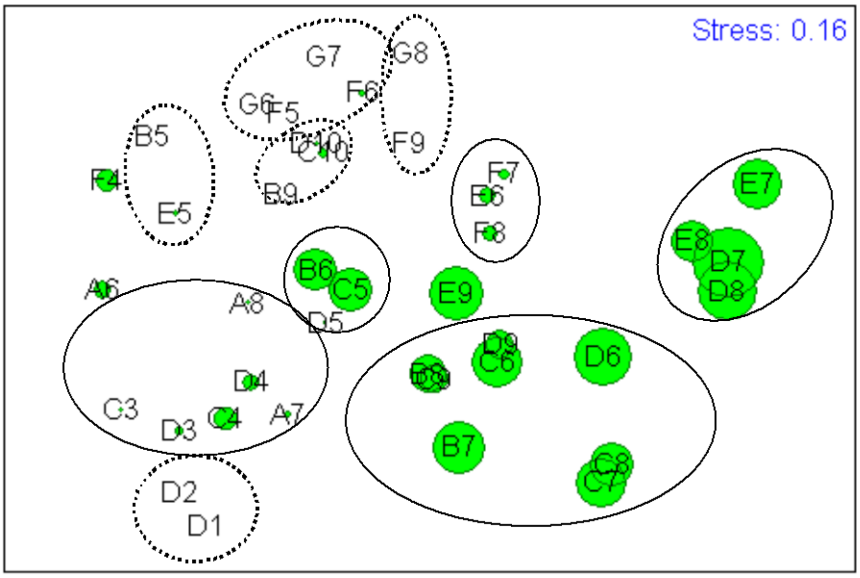


Figure 6.15: MDS based on percentage data of habitat composites with standardised CPUE of large snapper caught with long lines overlayed in the form of bubbles where the bigger the bubble the larger the relative CPUE. Groups from the cluster analysis are also outlined on the plot and cluster groups assigned to a high catch are enclosed by a solid line and cluster groups assigned to a low catch are enclosed by a dotted line.

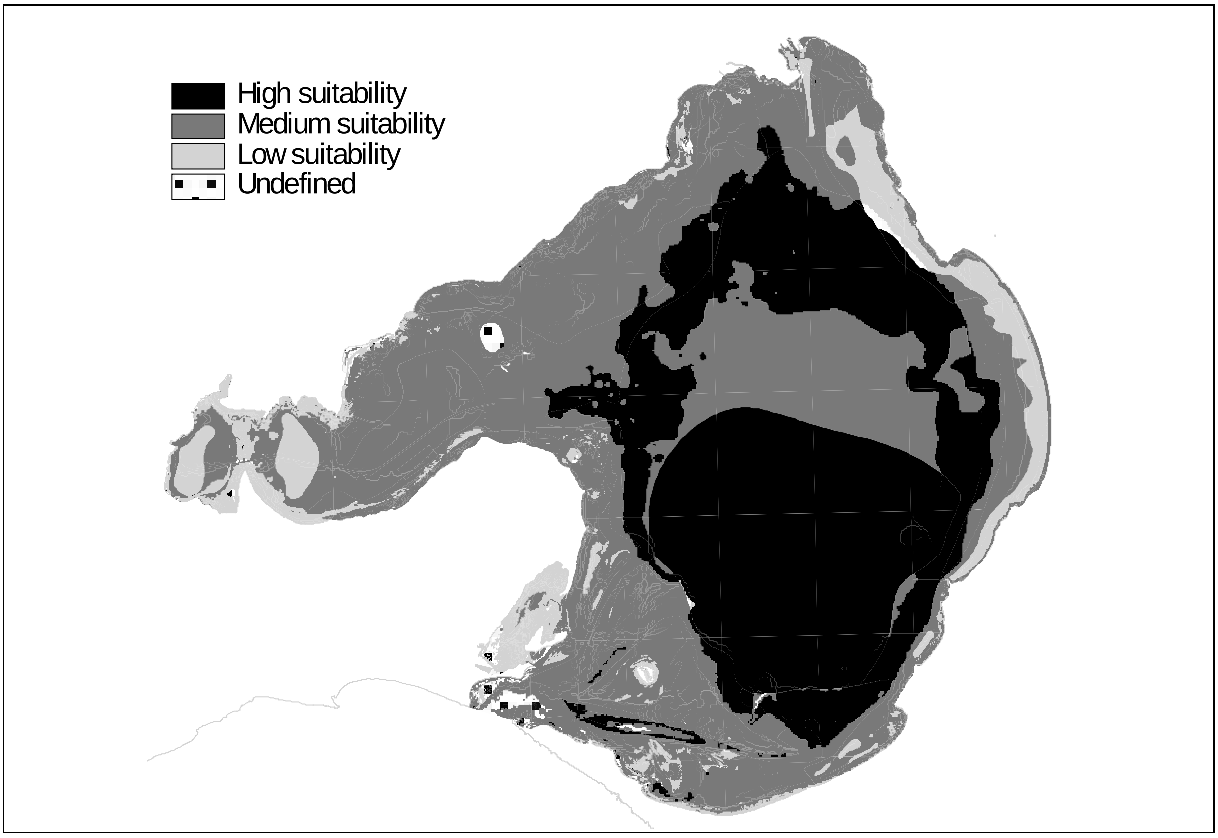


Figure 6.16: Fishery dependent habitat suitability model for large snapper in Port Phillip Bay.

6.5. Fishery Dependent Habitat Suitability Model Evaluation

While the habitat suitability models for King George whiting, flounder, Australian salmon, and snapper pinkies are all broadly similar, in that they emphasised shallow-water habitat, the method we used highlights differences between species.

Validating and testing the habitat suitability models presented here is hindered by a lack of fishery independent data. While data exists for some of the species (see Table 5.1), its unequal spatial distribution and the concentration of sampling over only some of the possible habitat types (mostly bare sediment in depths > 7 m) restricts its usefulness for model validation. As a consequence we have provided a qualitative assessment of the overall patterns of suitable habitat distribution.

6.5.1. King George whiting

The habitat suitability model for sub-adult (2-3 year old) King George whiting is consistent with existing information available about the habitat affinities and distributions for this species. King George whiting are known to recruit into shallow seagrass dominated areas and move into reef and bare shallow areas as they get older (Fowler & McGarvey 1995; Smith & MacDonald 1997; Jenkins & Wheatley 1998). The model predicted that the sheltered, shallow and seagrass associated areas would be of high suitability, including Swan Bay (fishing area E4) and the seagrass areas in fishing area G6, which are both important nursery areas for this species (Jenkins *et al*. 1993, Jenkins and Hamer 2001). The strip of low suitability habitat along the east coast of the Bay corresponds to results from other surveys (Parry *et al*. 1995) and recreational angling returns (Coutin *et al*. 1995; Conron & Coutin 1998) and appears to be driven by the lack of structural habitat and the coarse sandy sediment (Figure 3.4). While independent validation of the model will be necessary, the consistency of the model for King George whiting with other sources of information suggests that we can be reasonably confident in the model predictions.

6.5.2. Greenback flounder

Greenback flounder are probably less restricted to the shallow areas than the model in Figure 6.8 suggests (Kuiter 1993; Gomon *et al*. 1994). Data from trawl surveys in Port Phillip Bay report flounder in the deeper, more central areas of the Bay (Parry *et al*. 1995). The majority of flounder are caught with haul seines and fishing effort using this gear type is concentrated in the shallower depth range (< 10 m). The mesh nets that are used in slightly deeper areas do not target flounder particularly well, creating a bias in the catch data towards the shallower areas for this species. Flounder are also associated with bare organic-rich substrate and have been recorded in large numbers in the bare areas interspersed between patches of seagrass in Swan Bay (Jenkins *et al*. 1993). The high suitability area in Corio Bay is compatible with the large areas of seagrass and organic rich clays and similarly the high suitability classification around the mouth of the Yarra also corresponds with an organic-rich clay sediment.

6.5.3. Australian salmon

Juvenile Australian salmon are known to recruit to a wide range of coastal habitats such as medium energy sandy areas to sheltered mangrove-lined tidal creeks (Jones 1999). As they get older, schooling behaviour becomes more apparent. In Port Phillip Bay they have been described as transient and gregarious and have been recorded from shallow water over mosaics of seagrass and rocky reef interspersed with patches of bare sand (Hindell *et al*. 2000a; Hindell *et al*. 2000b). Dietary studies from Port Phillip Bay have also found that Australian salmon consume juveniles of seagrass associated fish (Hindell *et al*. 2000b). The habitat suitability map for Australian salmon in Port Phillip Bay is fairly consistent with this information; in the shallow areas there is very little habitat classed as low suitability and, in the very shallow areas, a wide range of habitats are classed as highly suitable.

Fishing block E9 had only a small amount of habitat classed as highly suitable for Australian Salmon, despite the very large catch of this species that was recorded from this area. Fishers utilising this area tend to be based locally, very experienced and target transient schools of Australian salmon (S. Morison pers. comm.). The schooling behaviour of salmon combined with a targeted effort may mean that it is possible to get very high catches from a very small area of highly suitable habitat, or alternatively from a reasonably large area of medium suitability habitat. The comparatively small amount of low suitability habitat within the Bay overall is also consistent with the fact that this species is a pelagic predator and so less likely to be strongly tied to a particular habitat. Instead, they are likely to move relatively large distances through the water column in search of suitable prey species (Hoedt & Dimmlich 1994).

6.5.4. Yellow-eye mullet

Yellow-eye mullet are a semi-pelagic species and have a low dollar value at market, which means that they are unlikely to be specifically targeted, but are instead a useful by-catch of the more lucrative species. Both of these factors suggest that the model for this species is liable to be less accurate than for other species. This is supported by the difficulty in assigning group 10 to either a 'high' or 'low' catch rating (Figure 6.110. However, the habitat suitability map (Figure 6.12) does predict that shallow seagrass associated areas in muddy sediment (sand-clay) are highly suitable habitat for this species and this is certainly consistent with information from the nearby Western Port (Edgar & Shaw 1995). The large number of yellow-eye mullet caught by shore-based recreational anglers (Coutin *et al*. 1995) also shows that this species is abundant in shallow water habitats.

Yellow-eye mullet are frequently found in estuarine environments and may even venture into freshwater (Kuiter 1993; Gomon *et al*. 1994). All the major freshwater inputs to Port Phillip Bay are in the northern part of the Bay and while the greater amount of low suitability habitat in the central and southern end of the Bay is in line with this fact, inclusion of a measure of salinity would probably improve the predictive model. The majority of catch for this species comes from fishing areas adjacent to the Yarra river discharge (blocks A7and A8) and the Werribee area (blocks C3, C4, B5); the two major freshwater inputs to the Bay.

6.5.5. Snapper

The habitat suitability maps for the snapper pinkies and the larger, older snapper caught with long-lines differ considerably (Figure 6.14 & Figure 6.16). The snapper pinkies are predicted to occur in the shallower areas in the northern and western parts of the Bay, findings which correspond with studies of the Port Phillip Bay recreational fishery that primarily targets pinkies (Conron & Coutin 1998). While the majority of the shallow areas along the eastern edge of the Bay are predicted to be of low suitability for pinkies, there are small areas of reef along this strip that are predicted to be of high suitability (Figure 6.14). This is consistent with information provided by recreational fishing guide books (Wilson 1986; Classon & Wilson 2002)) and anecdotal evidence about this species.

While the nearshore shallow reefs along the north-eastern shores are identified as high suitability for snapper pinkies, an adjacent band of low suitability habitat is predicted to extend along this and the eastern shores up to depths of about 15 m (Figure 6.14). However, this area is also known to feature reefs, rubble, lace coral and cunji beds (*Pyura stolonifera*) which are recognised as good fishing sites for pinkies by recreational fishers (Classon and Wilson 2002, Wilson 1986). Two studies (GHD 1997; Hamer *et al*. 1997) also report the presence of *P. stolonifera* beds in this area and Hamer et al. (1997) suggest that juvenile snapper may be associated with the presence of sessile organisms such as *P. stolonifera*. These habitats are not presently represented in the GIS habitat layers due to the limitations of mapping these depths in Port Phillip Bay from aerial photography and this area is primarily defined as bare coarse sand (Figure 3.4). As a result, the presence of these habitats was not accounted for in either the analysis of fishing blocks versus environmental variables or in the production of the predictive habitat suitability maps. This discrepancy between the habitat suitability model and anecdotal evidence (recreational fishing guide books) provides a specific hypothesis to be tested.

Interestingly, Swan Bay (block E4) is predicted to be primarily of high suitability, due to the intertidal seagrass coverage on muddy substrate, however anecdotal information and surveys undertaken by PIRVic Marine and Freshwater Systems as part of other studies, suggest that 'pinkies' are not particularly abundant in this area (e.g. Jenkins *et al.* 1993). Good pinkie catches are recorded in deeper areas immediately adjacent to shallow seagrass beds in other parts of Port Phillip, so at the scale of modelling undertaken here this may be influencing the classification of this habitat type as high suitability.

Snapper are known to move into deeper water with age (Gomon *et al*. 1994; Coutin 2000b) and the habitat suitability map for larger fish certainly predicts that the deeper areas of Port Phillip Bay have the most suitable habitat for this life stage of snapper (Figure 6.16).

Most of the Geelong Arm/Corio Bay is classified to be of low suitability for large snapper (Figure 6.16) with the balance of this area being classified as medium suitability. Recreational fishers catch good numbers of adult snapper in the western Geelong Arm and it is recognised as a good winter recreational fishery for this species (G. Jenkins pers. comm.). Commercial fishers do not long-line in this relatively shallow and enclosed area, but instead use haul seines that target sub-adult or 'pinkie' sized snapper. As a result, the presence of any adult snapper in this area would be underestimated by this method of modelling. This is an area of the predictive habitat suitability map that requires further verification.

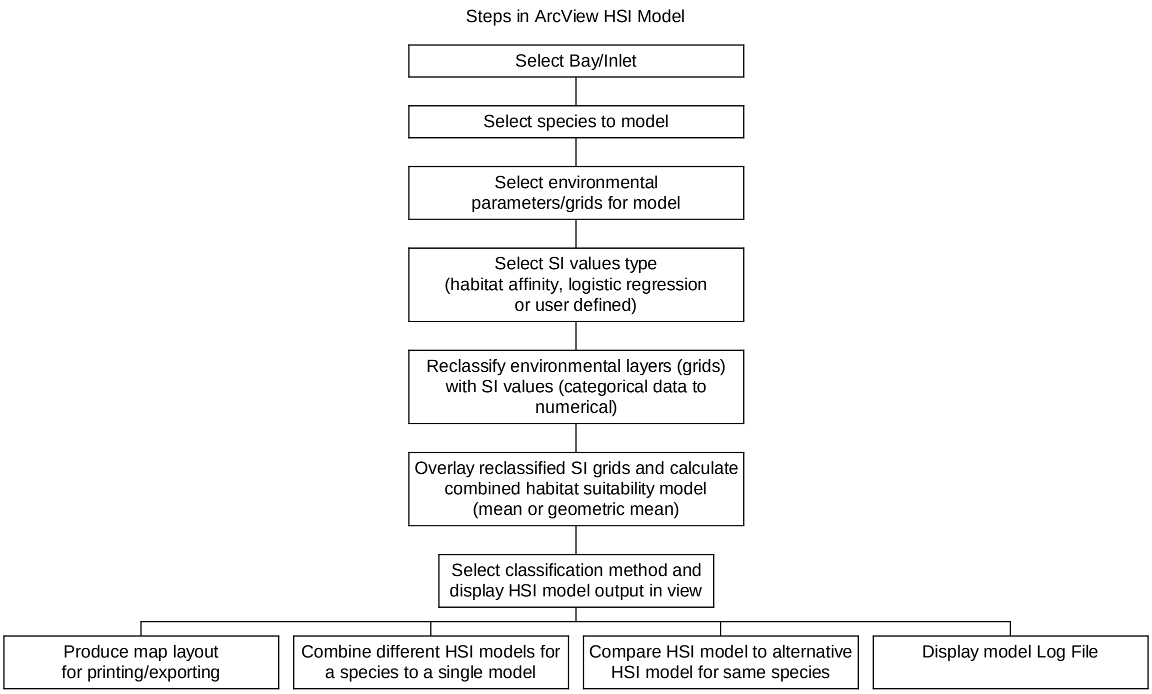
If the two habitat maps for snapper were to be combined, the majority of Port Phillip Bay would be predicted to be of high or medium suitability. This is consistent with available information about this species and, in particular, dietary data. Snapper are demersal predators but appear to eat a wide range of prey species from a variety of habitats (Parry *et al*. 1995; Coutin 2000b). They are also highly aggregative species that can move considerable distances and so are likely to utilise a range of habitats, and the predictive maps of habitat suitability are in agreement with this information.

7. ArcView Habitat Suitability Model Interface

An ArcView Habitat Suitability Model Interface was developed for ArcView 3.3 and ArcView Spatial Analyst with the Avenue programming language. The appearance of the ArcView interface main data view is show in Figure 7.1.

In developing our ArcView Habitat Suitability Model Interface we assessed beta or test versions of ArcView habitat suitability applications being developed by the NOAA Biogeography Program (http://biogeo.nos.noaa.gov/products/apps/hsm/) and the Florida Marine Research Institute. While having similar functionality to the NOAA and FMRI applications, our ArcView Interface was developed to work with the different types of suitability index values produced during this study. In order to overcome some of the limitations of the available data, we also designed the system to provide opportunities for user input during the modelling process to allow users to adjust suitability index values according to their own knowledge of species/habitat interactions.

In order to simplify the process of producing habitat suitability models presented in Section 5 and allow users to operate the model with limited instruction, a habitat suitability modelling wizard was also developed that takes the user through step by step instructions on how to operate the model. The key steps in the modelling process are shown in the following chart.



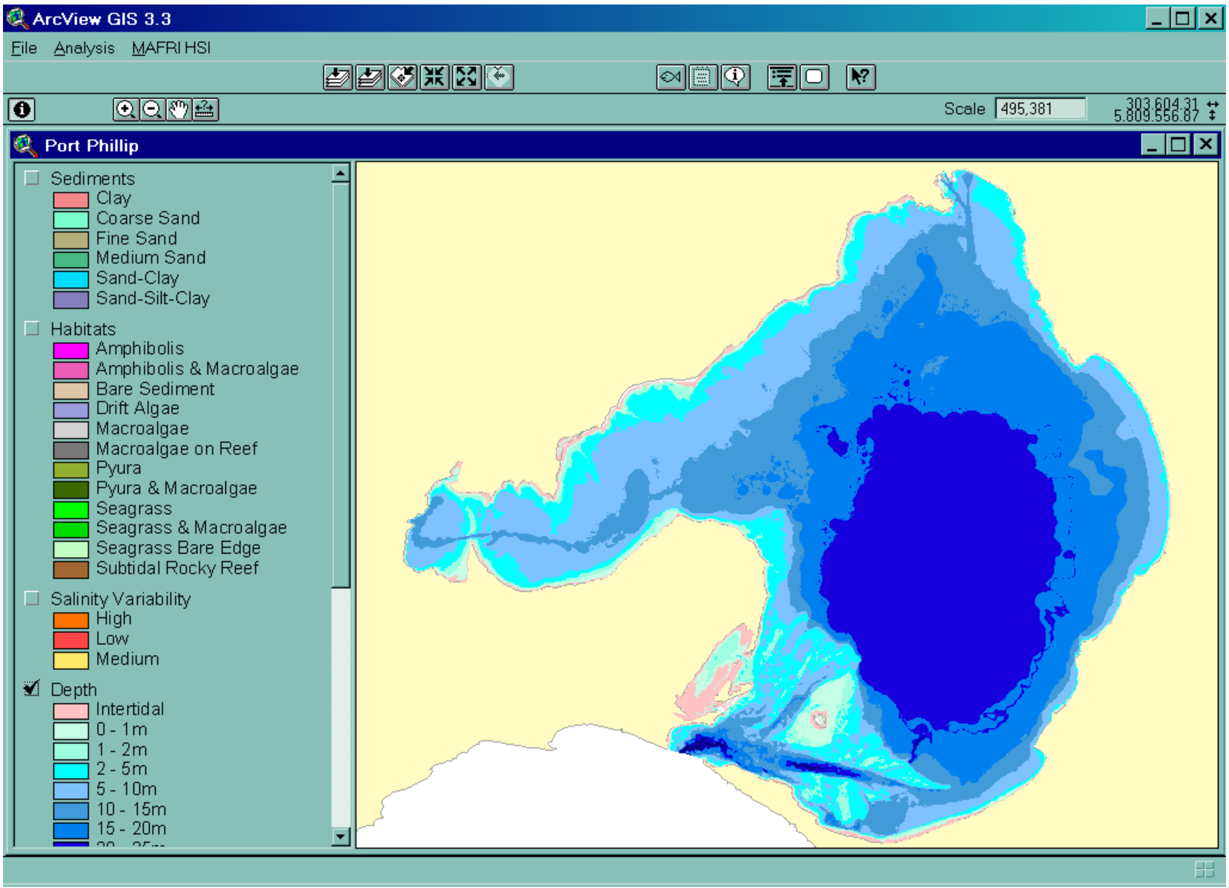


Figure 7.1: ArcView Habitat Suitability Model Interface data view for Port Phillip Bay. Habitat layers are in an ArcView grid format. Note that the standard ArcView interface has been simplified to only display buttons and menus directly required for the HSI modelling (the standard ArcView buttons/menus can be toggled on or off).

7.1. System Requirements

The Habitat Suitability Model Interface requires ArcView 3.3 and the ArcView Spatial Analyst extension to operate. ArcView version 3.3 was chosen as this was the predominant version of the ArcView software in use by relevant Victorian State Government agencies, including Fisheries Victoria, at the time the interface was developed.

It is also possible to save the outputs from the GIS modelling (ie. the habitat suitability distribution layers/maps) to a shapefile format that can be viewed using GIS data viewers (eg. ArcExplorer available at http://www.esri.com/software/arcexplorer/index.html) or imported to other GIS software packages.

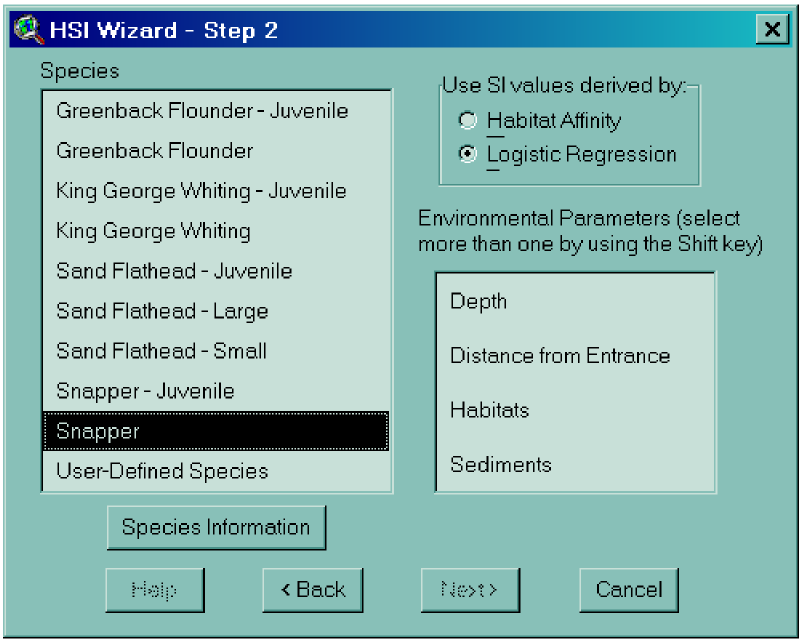
7.2. Habitat Suitability Model Wizard Dialogue Boxes

The Habitat Suitability Model wizard guides the user through the modelling process through a series of dialogue boxes. The HSI wizard is opened by clicking the following button in the ArcView project window or from within a data view.

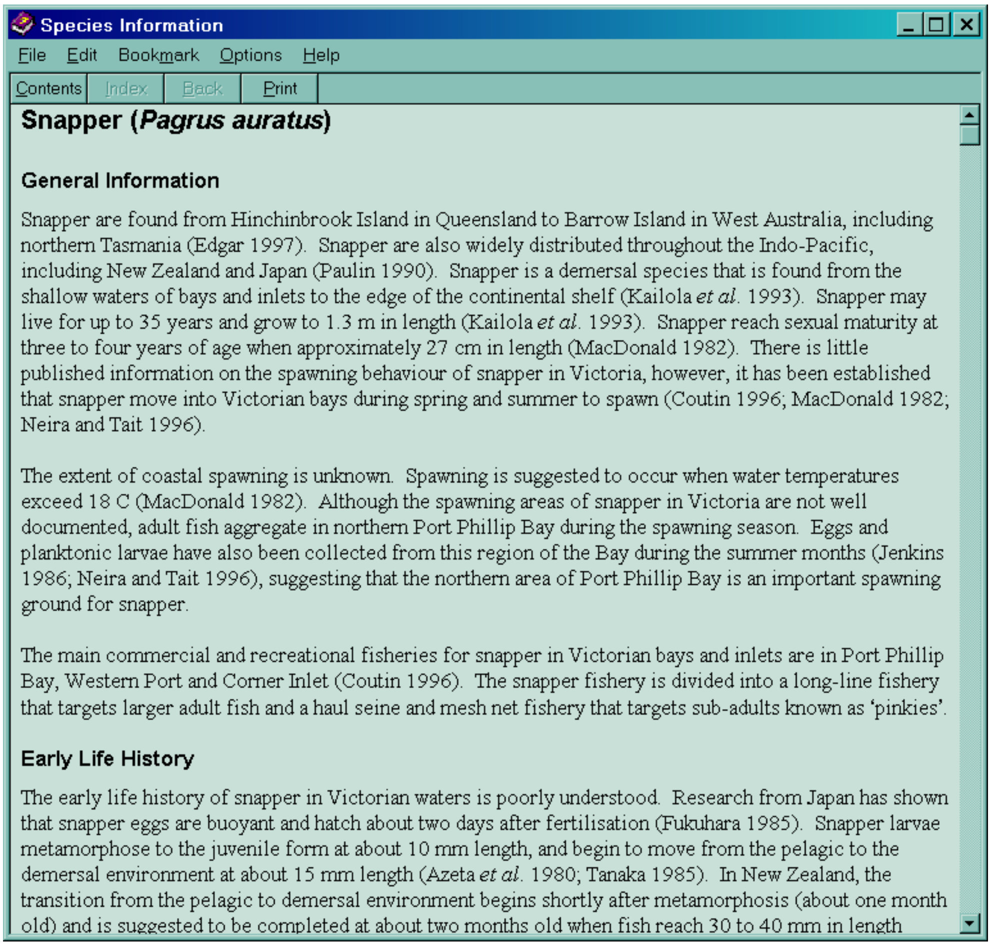
 Start Habitat Suitability Model wizard button opens Step 1 of the HSI wizard.

Step 1: requires the user to select the bay/inlet that they want to model.

Step 2: requires the user to select the species to be modelled, the environmental parameters to include in the model (available parameters are dependent on the species selected) and the method used to derive the Suitability Index (SI) values. The species available to model is dependent on the bay/inlet selected in Step 1.



The Species Information button in Step 2 opens a text box containing species information derived from a literature review (see Section 4). The species information box for Snapper is shown below.



Snapper (*Pagrus auratus*)

General Information

Snapper are found from Hinchinbrook Island in Queensland to Barrow Island in West Australia, including northern Tasmania (Edgar 1997). Snapper are also widely distributed throughout the Indo-Pacific, including New Zealand and Japan (Paulin 1990). Snapper is a demersal species that is found from the shallow waters of bays and inlets to the edge of the continental shelf (Kailola *et al.* 1993). Snapper may live for up to 35 years and grow to 1.3 m in length (Kailola *et al.* 1993). Snapper reach sexual maturity at three to four years of age when approximately 27 cm in length (MacDonald 1982). There is little published information on the spawning behaviour of snapper in Victoria, however, it has been established that snapper move into Victorian bays during spring and summer to spawn (Coutin 1996; MacDonald 1982; Neira and Tait 1996).

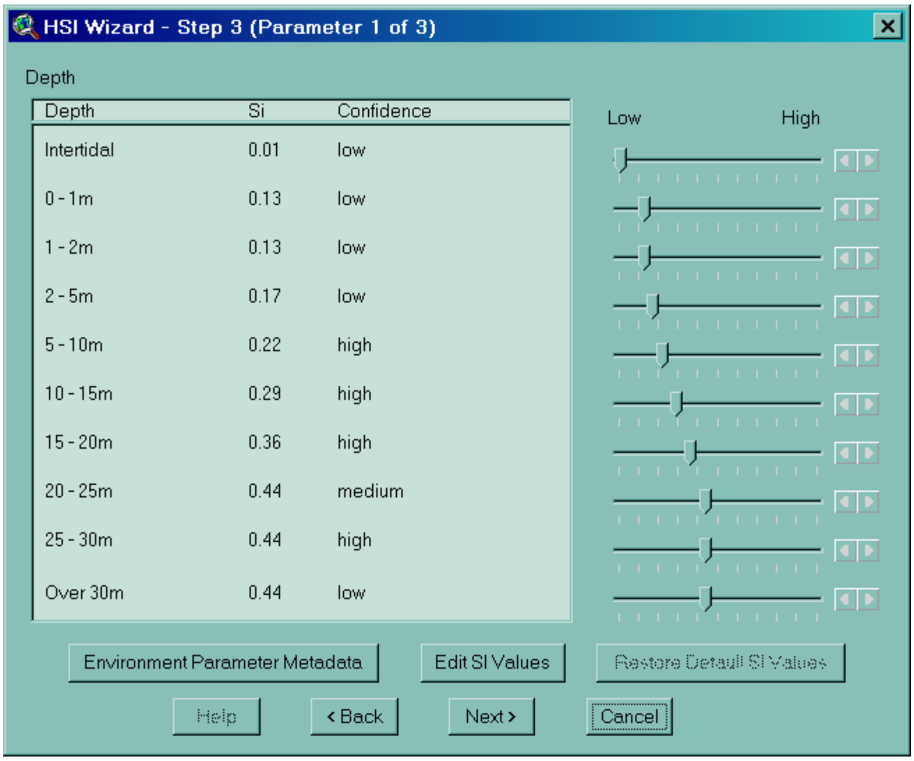
The extent of coastal spawning is unknown. Spawning is suggested to occur when water temperatures exceed 18 C (MacDonald 1982). Although the spawning areas of snapper in Victoria are not well documented, adult fish aggregate in northern Port Phillip Bay during the spawning season. Eggs and planktonic larvae have also been collected from this region of the Bay during the summer months (Jenkins 1986; Neira and Tait 1996), suggesting that the northern area of Port Phillip Bay is an important spawning ground for snapper.

The main commercial and recreational fisheries for snapper in Victorian bays and inlets are in Port Phillip Bay, Western Port and Corner Inlet (Coutin 1996). The snapper fishery is divided into a long-line fishery that targets larger adult fish and a haul seine and mesh net fishery that targets sub-adults known as 'pinkies'.

Early Life History

The early life history of snapper in Victorian waters is poorly understood. Research from Japan has shown that snapper eggs are buoyant and hatch about two days after fertilisation (Fukuhara 1985). Snapper larvae metamorphose to the juvenile form at about 10 mm length, and begin to move from the pelagic to the demersal environment at about 15 mm length (Azeta *et al.* 1980; Tanaka 1985). In New Zealand, the transition from the pelagic to demersal environment begins shortly after metamorphosis (about one month old) and is suggested to be completed at about two months old when fish reach 30 to 40 mm in length

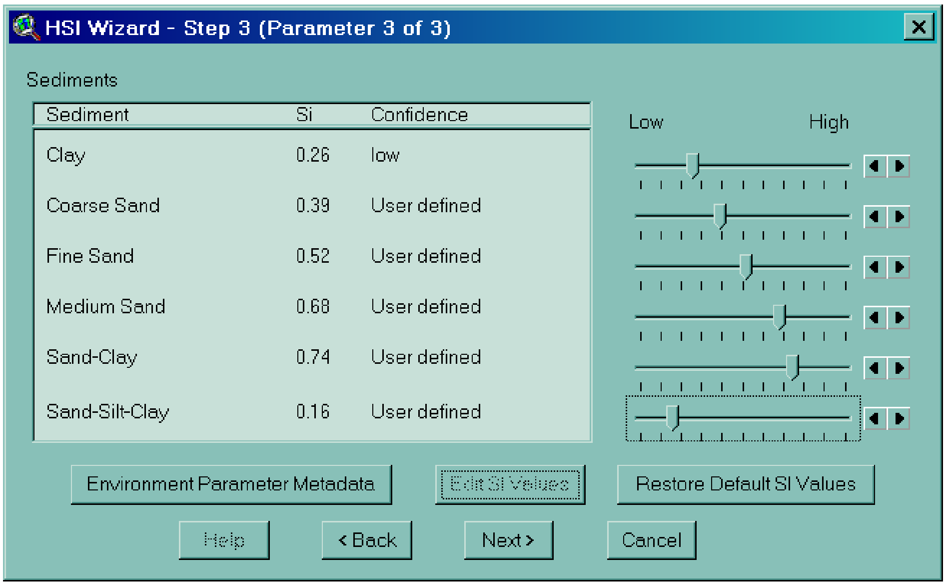
Step 3: presents the user with tables showing the SI values for each of the environmental values selected in Step 2. The relevant GIS grid layer is reclassified from the categorical values to the numerical SI values.



The tables in Step 3 include a measure of confidence for each suitability index. The confidence measure was derived from a combination of quantitative and qualitative information and is intended to give the user an indication of the reliability of the suitability indices relative to the source data. The approach adopted for defining confidence measures is outlined in Section 5.3.6.

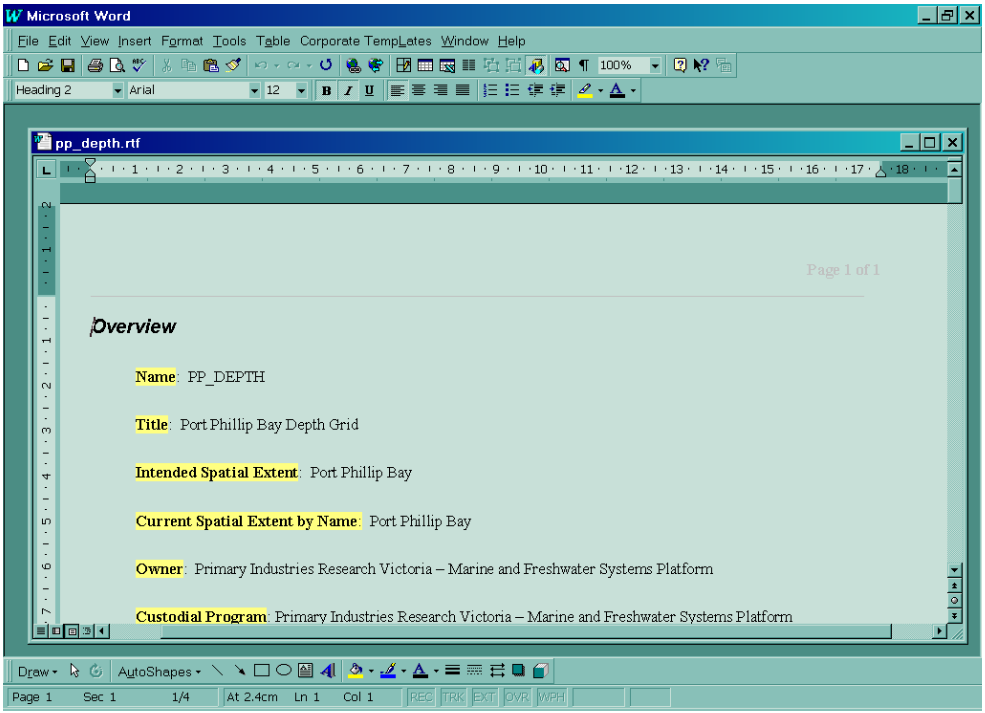
Edit SI Values

The default SI values provided for each species in Step 3 can be edited by clicking on the Edit SI Values button. This activates the sliders to the right of the table that can be moved left or right to increase or decrease the SI value for any of the environmental categories. An example is given below showing the dialogue box after the SI values for Sediment have been changed (note that the Confidence values change to User defined for the categories that have been changed).



Metadata

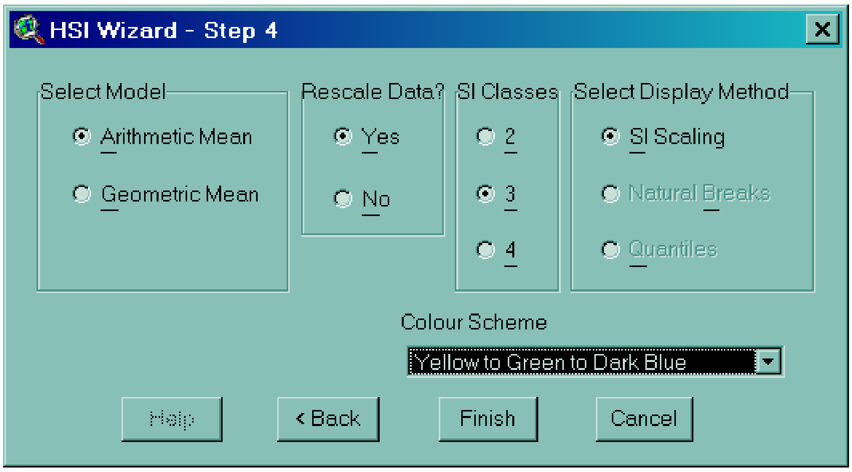
The Environment Parameter Metadata button in Step 3 retrieves the corresponding metadata in a rich text file format for the selected grid layer and opens the file in MS Word as shown below.



Step 4: is the final stage in the wizard and requires the user to select how the reclassified environmental grids will be combined to produce the habitat suitability model. The Re-scale Data option recalculates all the values in the model grid so that they range from 0-1 by dividing each cell value by the maximum cell value found in the raw model (ie. if the SI values in a model range from 0-0.7 then all values are divided by 0.7 so that the re-scaled model ranges in SI values from 0-1).

The SI Classes option allows users to select the default 3 suitability classes (low 0-0.25, medium 0.25-0.75 or high >0.75) or the alternatives; 2 classes (low 0-0.5, high >0.5) or 4 classes (low 0-0.25, medium 0.25-0.5, high 0.5-0.75, very high >0.75).

If the user selects No in the Rescale Data option, they are able to select from either Natural Breaks or Quantiles as options in the Select Display Method as alternatives to the predetermined SI Scaling classes.



After the user selects Finish in Step 4 the software provides the output View shown in Figure 1 The final view includes the original environmental parameter layers, the environmental parameter layers reclassified by SI values and the final model layer calculated from these individual SI layers.

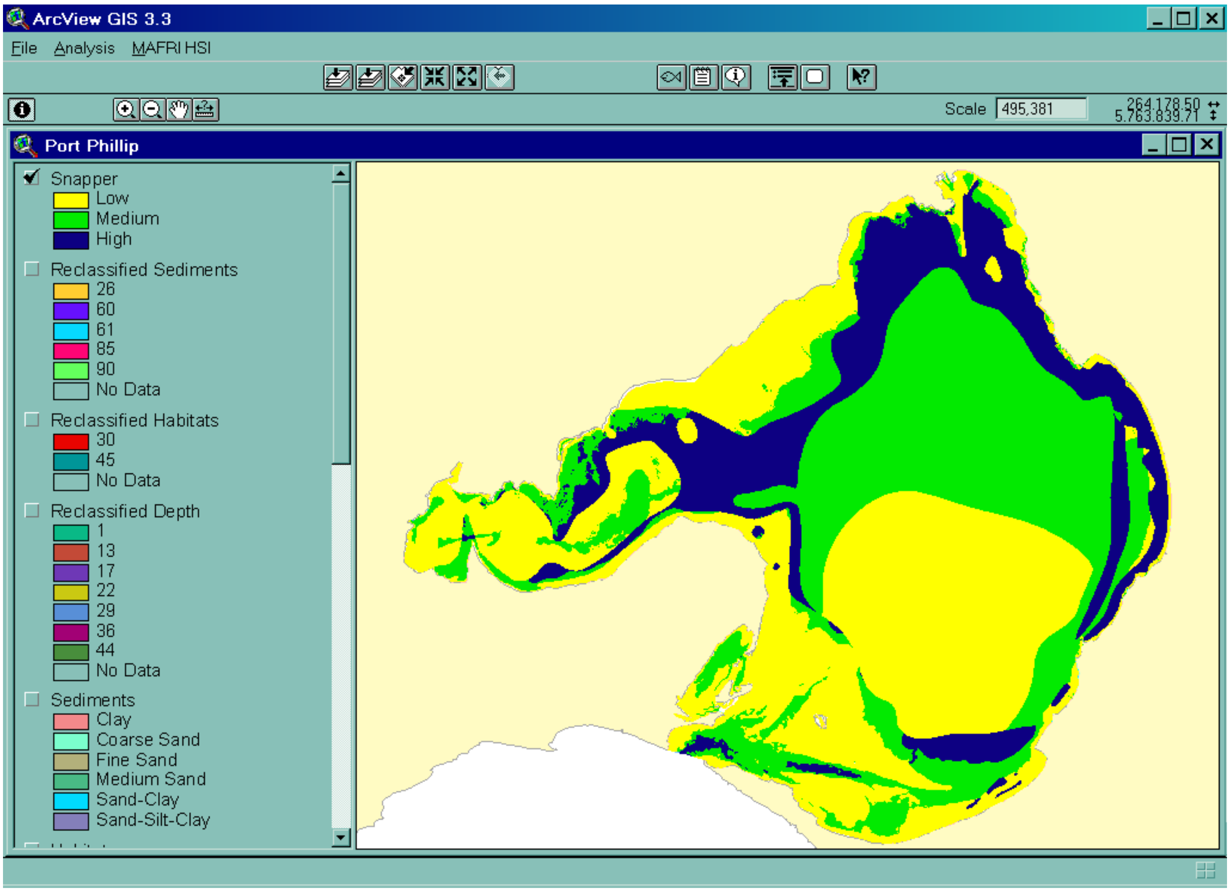
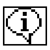


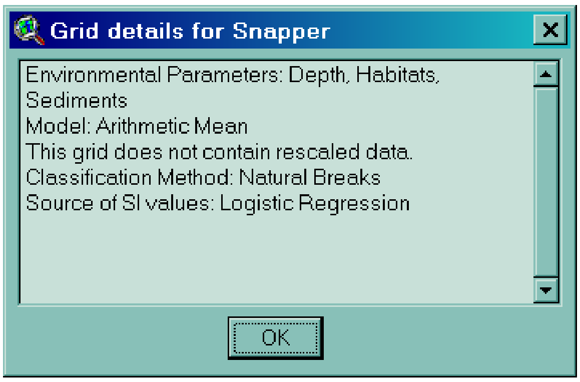
Figure 7.2: Final Data View displaying output from HSI wizard outlined above. Each of the input environmental parameter layers has been reclassified to its SI value based on the values from Step 3 in the wizard (Note that to simplify the display and analysis of the data the SI values have been multiplied by 100 to create integer grids).

7.3. Model Grid Details

Summary details of each HSI model can be viewed by selecting the relevant model in the Data View and clicking the Show Model Details button.

 Show Model Details button

This function returns a dialogue box like the following for the HSI model shown in Figure 7.2.



7.4. Model Log File

A detailed log file is automatically produced for every model generated by the HSI application. The Log file preserves information about the model inputs, SI values and model classification method. The software also maintains a link between the GIS model grid and its Log text file. The Log file can be viewed by making the model grid the active theme in the view and then clicking on the Show Model Log File button in the Data View.

 Show Model Log File button

Sample Log File

Report for result grid pslan1
Creation Date: 09:26 Tue Jan 07 2003
User Selections
Bay/Inlet: Port Phillip Bay HSI Model
Species: Snapper
Habitat Parameters:
 Depth
 Substrate type/biota
 Sediments

Model: Arithmetic Mean
Classification Method: Natural Breaks
Colour scheme used for display: Yellow to Green to Dark Blue
Source of SI values: Logistic Regression

Habitat Parameter Information

Parameter: Depth

Depth,SI,Confidence
Intertidal,0.01,low
0 – 1 m,0.13,low
1 – 2 m,0.13,low
2 – 5 m,0.17,low
5 – 10 m,0.22,high
10 – 15 m,0.29,high
15 – 20 m,0.36,high
20 – 25 m,0.44,medium
25 – 30 m,0.44,high
Over 30 m,0.44,low

Parameter: Sediments

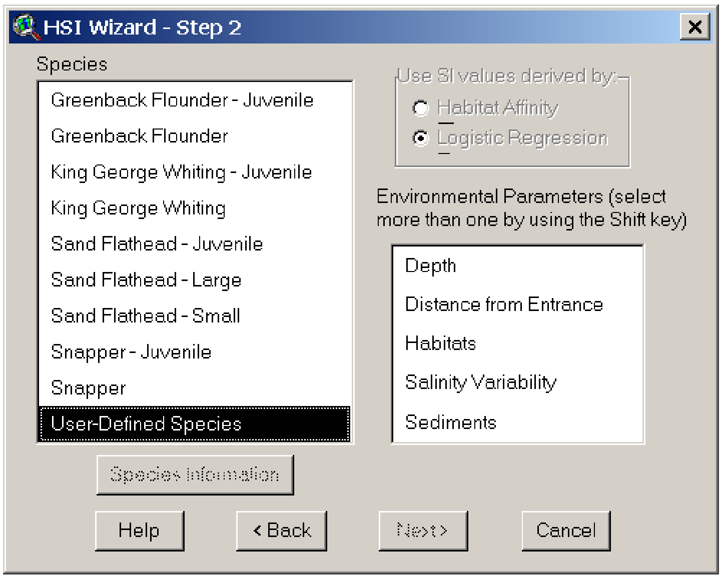
Sediment,SI,Confidence
Clay,0.26,low
Coarse Sand,0.9,medium
Fine Sand,0.61,medium
Medium Sand,0.61,medium
Sand-Clay,0.85,medium
Sand-Silt-Clay,0.6,medium

Parameter: Substrate type/biota

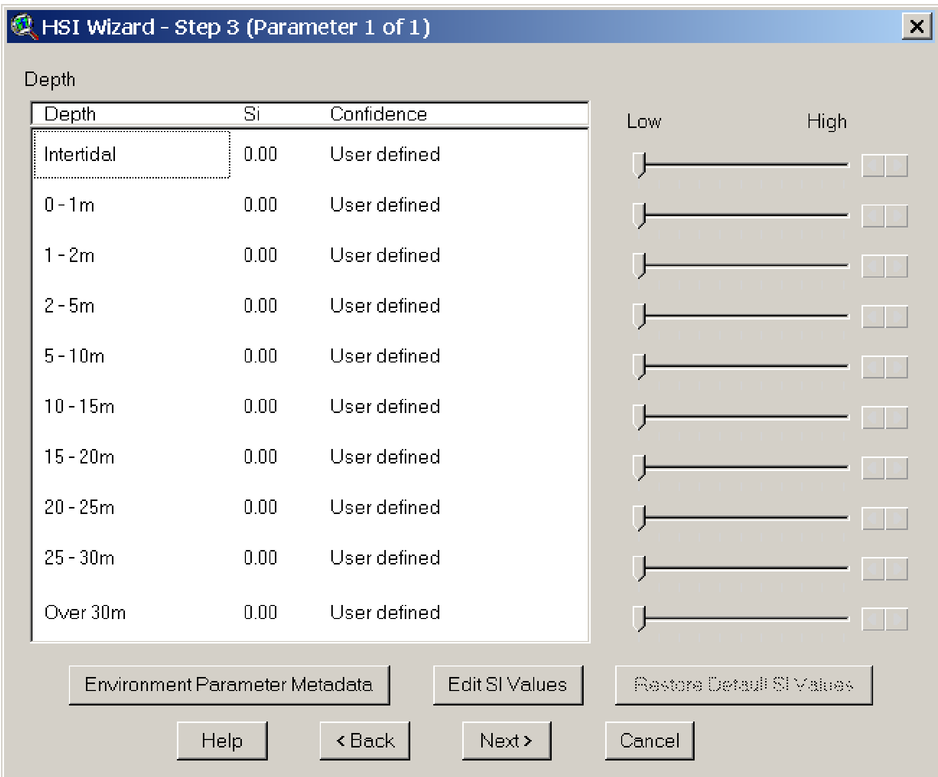
Substrate type/biota,SI,Confidence
Amphibolis,0.3,low
Amphibolis & Macroalgae,0.3,low
Bare Sediment,0.3,low
Drift Algae,0.3,low
Macroalgae,0.3,low
Macroalgae on Reef,0.3,low
Pyura,0.3,low
Pyura & Macroalgae,0.3,low
Seagrass,0.3,low
Seagrass & Macroalgae,0.3,low
Seagrass Bare Edge,0.3,low
Subtidal Rocky Reef,0.45,low

7.5. User Defined Species

Step 2 of the HSI application (see above) provides the user with a list of species for the selected bay/inlet being examined. This list of species is limited to the key commercial species for which suitable fishery independent monitoring data was available to generate SI values for the different habitat parameters. It is also possible for the user to model a new species by selecting the User-Defined Species option and the required Environmental Parameters.

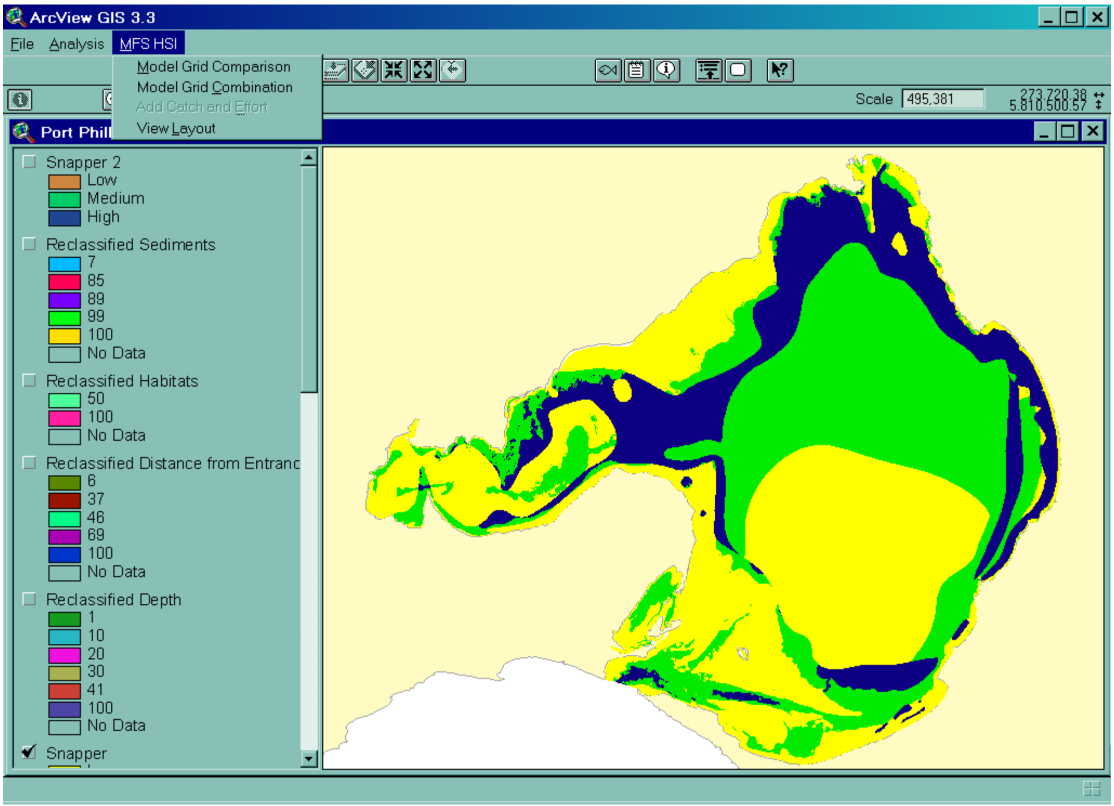


After selecting User-Defined Species, Step 3 creates Habitat Parameter tables with all the SI values set to 0 by default (see below). The user can then edit these values according to their own information or requirements.



7.6. MFS HSI Menu

The Data View includes a drop-down menu shown below. The menu title is an abbreviation of Marine and Freshwater Systems (the Section at PIRVic where the work was completed) Habitat Suitability Index. The MFS HSI menu provides tools for combining or comparing grids/models and for producing map layouts. These functions are described below.



7.7. Map Layout

The View Layout option in the MFS HSI menu in the Data View creates an A4 map layout as shown in Figure 7.3. This layout can either be printed directly from ArcView or exported to an image file for use in other software eg. MS Word.

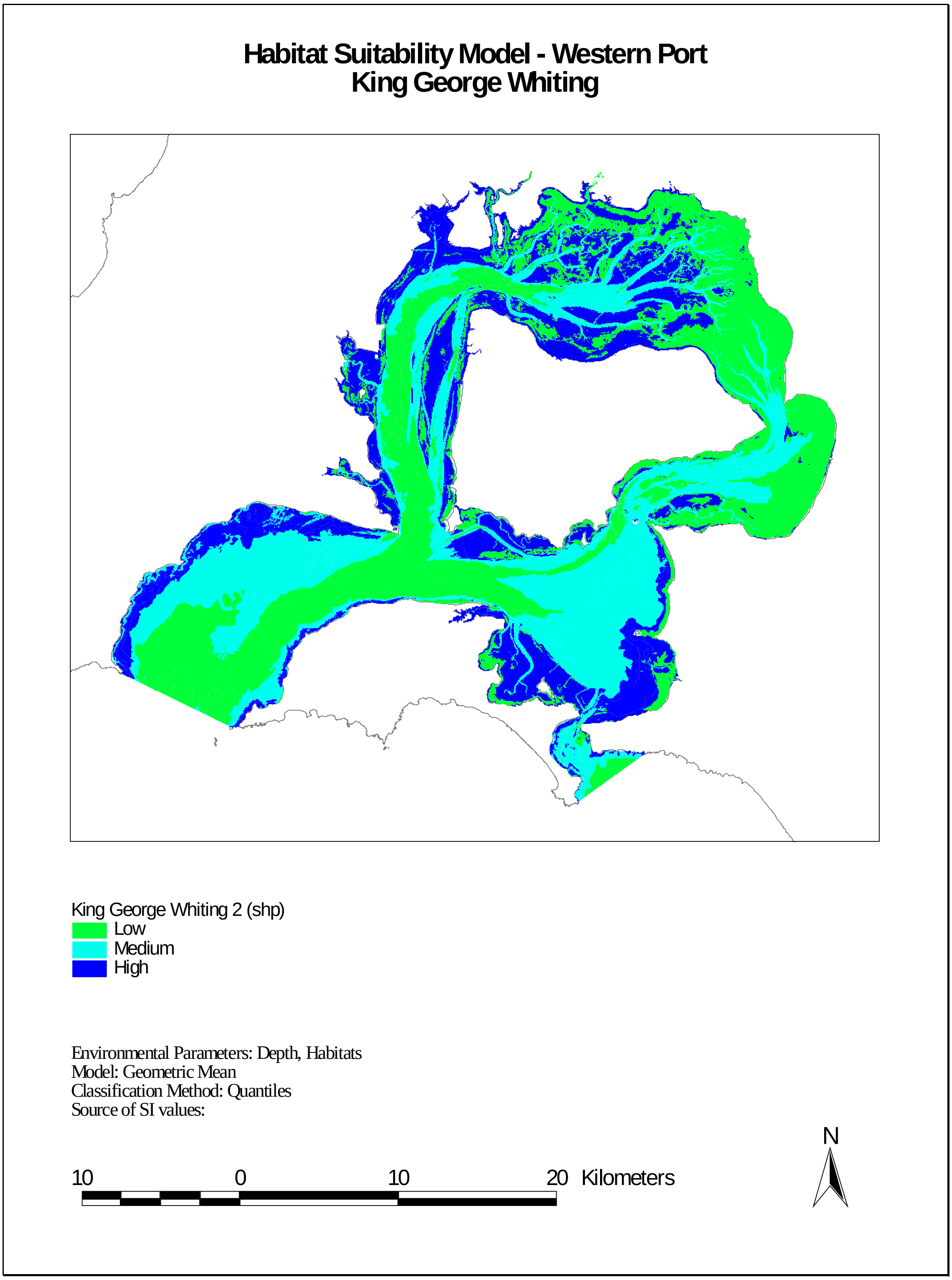
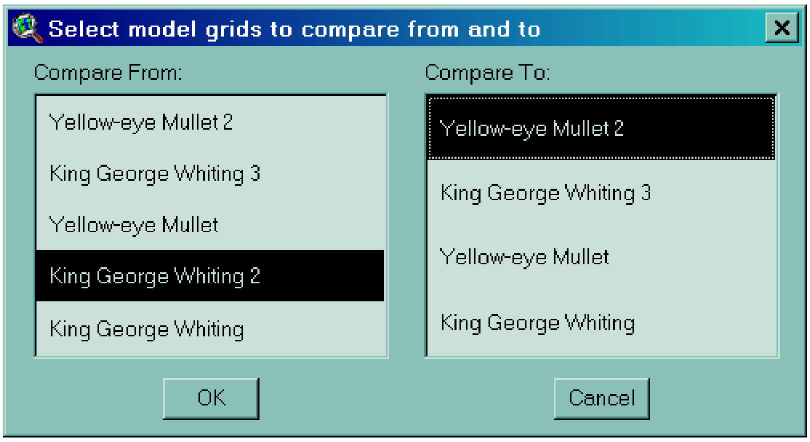


Figure 7.3: Map layout produced automatically with the View Layout option.

7.8. Model Grid Comparison Tool

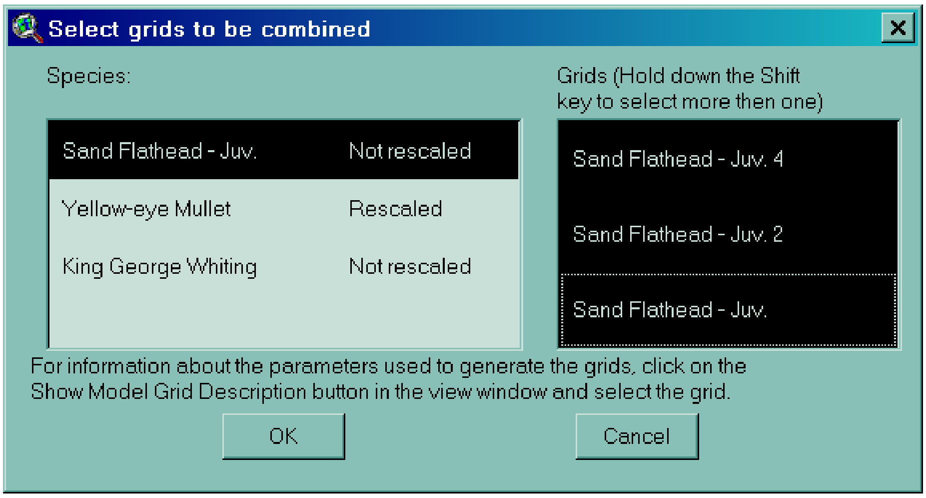
Where more than one HSI model is produced using different methods, a grid comparison tool enables differences and similarities between the models to be highlighted. Selecting Model Grid Comparison from the HSI menu in the Data View opens the following dialogue box which identifies models available for comparison.



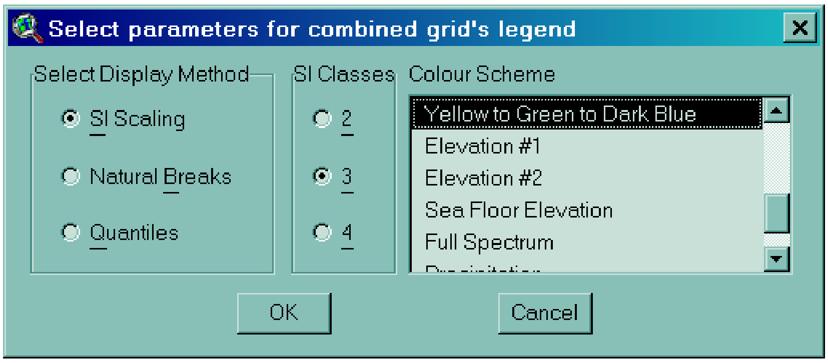
The cells in the output comparison grid are classified as either Models Match, 1 Class Different, 2 Classes Different etc.

7.9. Model Grid Combination Tool

This tools allows HSI models created with different inputs for the same species to be joined to produce a single combined model. Selecting Model Grid Combination from the HSI menu in the Data View opens the following dialogue box which identifies the models available to be combined. The combined model represents an arithmetic mean of the selected input models. Models can only be combined if they were created with the same scaling method at Step 4 in the HSI modelling wizard.

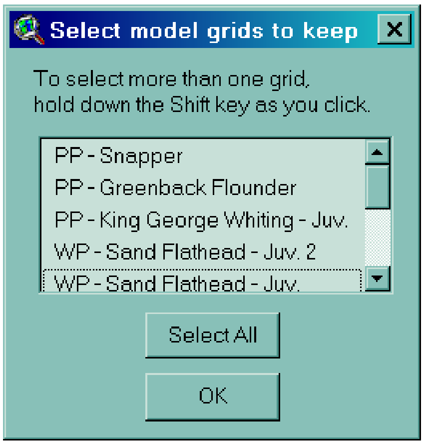


After selecting the grids to combine, the user is prompted by the following screen to select a display method or data scaling option for the combined grid.



7.10. Saving Models

The above steps create temporary grids on the C:\ drive of the user's PC. When the user selects to either shut down the ArcView HSI application or to model a different bay/inlet, the application prompts the user to select whether any models created during the current session are required to be saved through the following dialogue box. Any models not selected are deleted when ArcView is closed.



7.11. Additional Functions

7.11.1. Test Habitat Suitability Model Against Fishery Independent Monitoring Data

It was initially envisaged that a function to allow testing habitat suitability models against fishery independent monitoring data would be included in the ArcView application. However, existing fishery independent monitoring data was inadequate for this purpose for all species in all bays. As a result this function has not been enabled for this version of the ArcView application.

7.11.2. Load Fishery Dependent Habitat Suitability Models

Due to the restricted availability of fishery independent monitoring data for Port Phillip Bay, we also developed habitat suitability models from an analysis of commercial catch and effort statistics (Section 6). These models can also be loaded to the ArcView HSI application as stand-alone layers and for comparison to models produced by the methods described above.

8. Model Discussion

We applied two different approaches to habitat suitability modelling for Victorian bays and inlets. The first approach made use of existing fishery independent data to derive suitability indices and then produced composite habitat suitability models within a GIS (Section 5). This approach was consistent with previous habitat suitability modelling of marine and estuarine systems (see Christensen *et al.* 1997, Rubec *et al.* 1998, Clark *et al.* 1999, Rubec *et al.* 1999, Brown *et al.* 2000). The second approach made use of commercial fishery catch and effort returns and was a new method of habitat suitability modelling that was developed during this study (Section 6). The following presents a discussion of the relative merits of the two approaches.

8.1. Fishery Independent Habitat Suitability Modelling

We were able to successfully apply an existing habitat suitability modelling approach to Port Phillip Bay and Western Port, and also established the model framework for Corner Inlet. The modelling approach enabled us to produce spatial models or maps of the predicted distribution of different classes of habitat suitability for selected species and life-stages across the bays.

Some of the assumptions about species behaviour and interactions underlying this modelling approach are discussed in Section 2.5 and have equal relevance to the models presented for Port Phillip Bay and Western Port. The habitat suitability modelling approach assumes that each environmental variable used in the model is equally important in determining habitat suitability for a species and that the relationships between habitat quality for a species and each environmental variable are independent (Brown *et al.* 2000). In view of the data limitations and gaps in our knowledge about the species under investigation, adopting a simplistic model seems appropriate as a first approach to defining the likely distribution of suitable habitat in the bays.

We attempted to identify seasonal variations in habitat preferences where possible, but we mostly focussed on producing models that represented total suitable habitat across all seasons. Without more comprehensive data, important seasonal variations in species habitat preferences may be overlooked. Ontogenetic shifts may also be disguised, such as the process of King George whiting initially settling as post-larvae in seagrass and reef-algal habitats between September and December and young juveniles later migrating to unvegetated sand between February and April (Jenkins and Wheatley 1998, G. Jenkins pers. comm.). Such variations in species habitat preferences can be highlighted through the production of separate habitat suitability models for each different life stage, provided that data is available to determine differences in habitat preferences through a species lifecycle.

We applied two methods of calculating suitability index values for Port Phillip Bay; a modified habitat affinity method and logistic regression. The composite habitat suitability models based on logistic regression SI values (Section 5.5) appeared to produce the most likely distributions of habitat suitability for Port Phillip Bay, based on a qualitative assessment. Rubec *et al*. (1999) highlighted the importance of the method of determining SI's in producing reliable habitat suitability maps. As the complexity of the method of calculating SI values increases, so does the requirement for more comprehensive data. As a result, we were limited in the methods of data analysis that we could apply by the inherent limitations of the data available to us.

8.1.1. Fishery Independent Data Limitations

We had limited existing fishery independent data and this presented a restriction on both calculating SI values and testing/validating the habitat suitability models. A habitat suitability model is only as good as the data from which it is derived and Clark *et al.* (1999) emphasised that modelling efforts should preferably be based on large and comprehensive fishery databases. We did not have the benefit of comprehensive fishery independent sampling data and many habitats were either under-represented or missing from our data.

The fisheries independent data used in the development of suitability indices in this study were derived from existing datasets. This meant that we were relying on data that had been collected for a number of different purposes and many of the studies did not measure key environmental variables at the sampling locations. In many cases we had to retrospectively determine the environmental characteristics of sampling locations by overlaying these positions on depth, sediment and/or substrate type/biota layers in the GIS.

The largest dataset available to us was the 10 year trawl data for Port Phillip Bay. This data also had limitations as it was restricted to sampling unvegetated sediment at four depths zones > 5 m. While there have been a number of surveys in the shallow Bay habitats (< 2 m), these were primarily targeted at juveniles. This meant that for the adults and sub-adults we had to estimate suitability indices for the shallower parts of the bays based on expert opinion. We had limited sampling data for seagrass and reef in depths of 2-10 m, which are generally considered to represent highly suitable habitat for some species and life stages (eg. sub-adult King George whiting and snapper). As a result some of the habitat suitability models have been based on data where, potentially, the most important habitats may be missing from the dataset.

Similarly, we did not have any data from the deeper sections of the bays for juveniles. This reflects assumptions that these areas provide poor habitat for juveniles of most species, however these assumptions have not really been adequately tested. Reef habitat was also under-represented in the juvenile data, again reflecting a bias towards the seagrass and finer sediment habitats in Port Phillip Bay. Reef habitat is relatively sparse in Western Port and Corner Inlet, and so the lack of sampling in reef habitat in these bays was considered to be less of a problem.

Future habitat suitability modelling would be greatly improved by incorporating a targeted data collection program at the start of the project. Using a random sampling technique that was stratified by habitats, and where sampling effort reflected the amount of each habitat available, would provide an improved basis for modelling habitat suitability over using data collected for other purposes.

8.1.2. Spatial Data Limitations

The existing spatial marine data presented two limitations on the habitat suitability modelling. Firstly, we only had consistent spatial layers for two environmental parameters across all of the bays under investigation (depth and substrate type/biota). Secondly, the spatial layers utilised in this study may not define important habitat categories or define habitat boundaries at a sufficient resolution for habitat suitability modelling.

While we were only able to use depth and sediment type as the input parameters for the composite habitat suitability models based on logistic regression SI values for Port Phillip Bay (Section 5.6), these still allowed us to produce models that depicted broad-scale habitat preferences of the species under investigation. In the case of habitat suitability models based on habitat affinity SI values, adding the substrate type/biota layer did not necessarily improve the model outcome. This is probably more an indication of the lack of sampling data available to us identifying species presence/absence for the different substrate type/biota categories, than an indication that these features are not important in determining the distribution of suitable habitat for these species.

Terrell *et al.* (1982) point out that including increasing numbers of model parameters does not necessarily improve the predictive power of a habitat suitability model and, may even reduce it. They discuss an approach whereby it may be more appropriate to choose the most important habitat parameter and use only that parameter as a predictor of habitat suitability. Our modelling appears to generally support this approach, but ultimately it depends on the characteristics of the individual species.

The spatial data itself had some limitations in the definition of certain habitats. Our spatial layers for substrate type/biota were created from aerial photography interpretation that only allowed mapping seabed features at depths up to about 5-7 m. As a result, features such as reef and *Pyura* beds at depths >5 m, that are known to provide important habitat for sub-adult snapper for example, were not defined in the substrate type/biota layer. Consequently, the composite habitat suitability models could not identify the presence of highly suitable habitat at these locations. Some substrate type/biota categories, such as seagrass and algal beds may also display seasonal and annual variations in their distribution.

Similarly, the seabed sediment layer for Port Phillip Bay defines distinct boundaries between sediment categories which do not always exist in the real-world, where a more gradual transition of one sediment type grading into another is more likely to exist. At the scale of mapping available to us, the presence of distinct sediment features (eg. patches of coarse shelly sediment amongst fine-medium sands) are also likely to be overlooked. While these problems of scale did present some limitations on the habitat suitability modelling, including seabed sediment as an input parameter usually improved the overall accuracy of the models for Port Phillip Bay. We did not have sediment layers for Western Port or Corner Inlet and it appears reasonable to assume that incorporating a sediment layer and SI values would also improve the habitat suitability models for these bays.

Continued improvements and updates to the substrate type/biota and sediment layers for Port Philip Bay through the application of improved mapping techniques (eg. hydro-acoustics) and the definition of sediment layers for Western Port and Corner Inlet would be beneficial to the production of habitat suitability models.

8.2. Fishery Dependent Data Habitat Suitability Modelling Approach

While the available information suggests that, with the exception of greenback flounder, we can have confidence in the broad patterns predicted by the fishery dependent habitat suitability models, particularly for King George whiting, Australian salmon and snapper, there were still certain anomalies. A fishing block that had similar habitat parameters to blocks where catches were high can have a consistently low catch as well as effort for all species (e.g. C5, Figure 6.1). There may be a number of reasons for this type of anomaly and they may affect our original assumption that fishers will target areas with the highest densities of fish. For example, there is a considerable amount of drifting macroalgae in block C5 (Figure 6.1), which may make fishing difficult or less cost effective. Alternatively, fishers may not consider it cost effective to travel large distances from port, or if they do they will have less time available for fishing. As a result fishing effort may not be equally distributed amongst high quality habitats, or certain gear types might be excluded from high quality areas.

There may also be sources of variation in effort that we have not measured and that have the potential to affect fishing efficiency and, in turn, CPUE. These may include certain intangibles such as experience and skill of the particular fishers that work an area, as well as differences in technology used by fishers targeting different areas. Conversely, there may be suitable habitat for a species, but other important environmental factors that have not been measured may also be important in determining a species' distribution, such as the effects of pollution, introduced marine pests, distribution of prey items and hydrodynamics.

The reverse situation is one where a fishing block, (eg. E9, Figure 6.1), had high CPUE for several species but had habitat more similar to blocks with low catches. In this case, fishers might apply their experience and concentrate their efforts within a small area and, as previously discussed, if schooling fish are successfully targeted, CPUE values can be very high.

An advantage of the approach we have presented in this study is that the above anomalies in CPUE did not affect the predictions we made about suitable habitat. Fishing blocks were classified into 'high' or 'low' catch groups depending on the standardised CPUE values for all the blocks in that cluster group and so effectively we were averaging across all blocks that had similar habitat. Alternatively, obvious outliers (such as block E9) were excluded from the analysis. At the same time, the habitats within blocks such as E4 (Figure 6.1), that were excluded from the original analysis as no fishing takes place within the area, were still included in the predictive map of habitat suitability.

There are several important assumptions that underlie our method of determining suitable habitat presented in Section 6. The first of these was that the areas targeted by fishers correspond to high densities of the species under investigation. For species that are not the primary target, this may not be the case. For instance, flounder are mainly caught with haul seines in the same areas that King George whiting (the primary target species) is caught. As haul seines are not used in the deeper, unvegetated soft-sediment areas of Port Phillip Bay that are known to provide suitable habitat for flounder (Parry 1995), the assumption that fishers only target areas of high density is unlikely to be true for this species. We predicted that demersal species would provide habitat suitability models that were more reliable than pelagic species, but it appears that the degree to which a species is actively targeted is at least, if not more, important than the life history characteristics of that species.

Similarly, the operational limitations of a gear type may also exert an influence on where species are caught. Long lines are not typically used in Corio Bay or most of the Geelong Arm due to the relatively shallow and enclosed nature of this area. As a result, species that are targeted by these gear types and may be present in the area, such as large snapper, do not appear in the data.

The second major assumption of our method relates to the fact that the consistent presence of a habitat type in areas that have similar environmental characteristics is considered to be important in determining the yield. In fact, we do not know over what scale habitat is important and this type of scale-dependent habitat information would, without doubt, improve models of habitat suitability.

The modelling of habitat suitability from fishery dependent data was also influenced by the accuracy of the spatial data for substrate type/biota and sediment. The limitations of the spatial data are discussed in Section 8.1.2.

Catch and effort returns are subject to a number of checks or quality control measures which aim to identify outliers and inconsistencies in the data. We assumed that the data we have analysed is accurate, that is, fishers have accurately recorded the catch, gear type and fishing block from where the catch was taken. While the use of on-board data loggers that record the exact location of catches would improve the usefulness of fisheries dependent data for research purposes, a closer examination of the data by gear type is also likely to provide extra information on the general area that a catch came from within a fishing block. For example, haul seines tend to be used in shallow areas, so there is the potential to use the haul seine data to model fish distributions and suitable habitat in depths <10 m. This would be a logical next step to further develop the habitat suitability models based on fisheries dependent data and to provide a way of increasing the resolution of the models while still using existing data collection techniques.

The models we have produced are likely to be conservative, due to the methods we used to classify habitat suitability. We have placed the emphasis mainly on fishing blocks where fish were caught rather than the blocks where they were not caught. This means that the models probably have more high and medium quality habitat than is actually the case. A model that is 'precautionary' though, seems preferable where it is likely to be used in identifying and protecting important habitat.

8.3. Testing and Validating Models

Previous studies have identified the importance of testing and validating habitat suitability models (Thomas & Bovee 1993; Bender *et al*. 1996; Brooks 1997; Roloff & Kernohan 1999; Burgman *et al*. 2001). Validation procedures have ranged from qualitative expert reviews (Christensen *et al*. 1997) to quantitative approaches using commercial fisheries data (Rubec *et al*. 1999) and/or independent datasets from the same or different localities (Coyne & Christensen 1997; Norcross *et al*. 1997; Clark *et al*. 1999; Brown *et al*. 2000; Stoner *et al*. 2001).

We did not have enough data to undertake a quantitative statistical test of the models produced from the fishery independent data. We attempted to 'test' the SI values produced for depth and sediment type for Port Phillip Bay through logistic regression by randomly splitting the data into a model half and a test half. While this provided an indication of the reliability of the predictions of probability of encounter, the spatial distribution of these sampling locations was mostly limited to unvegetated habitats in depths >7 m that restricted the application of this test half of the data to validating the spatial habitat suitability model. As a result, we had to rely on a qualitative assessment of the habitat suitability models based on a comparison with statements of habitat preferences in the literature and expert opinion. This approach to model-testing has limited application where the suitability indices were developed from the same qualitative information.

Our lack of comprehensive data to both calculate suitability index values and also test the habitat suitability models was the major restriction on the outcomes of this project.

8.3.1. Comparing Outputs from Different Model Approaches

Development of a number of habitat suitability models using different methodologies can allow a degree of confidence in the model outputs if they all produce similar predictions (Norcross *et al*. 1999). Under this assumption, we compared selected models for Port Phillip Bay produced by the composite suitability index method (Section 5) and the fishery dependent data analysis (Section 6).

Snapper (sub-adult) and King George whiting (sub-adult) had both fishery independent and dependent models. We chose composite habitat suitability models produced from habitat affinity suitability index values as the models to compare with the fishery dependent models.

Model comparisons were undertaken in the GIS to determine areas of commonality and difference between habitat suitability values for the corresponding grid cells in each model. The GIS processing steps were as follows:

- Habitat suitability values for each model grid were assigned a consistent numeric value (ie. high suitability = 3, medium suitability = 2 and low suitability = 1).
- The fishery independent and fishery dependent model grids for each species were overlayed in the GIS and one grid subtracted from the other.
- The values in the resulting grid corresponded to how many habitat suitability classes different a grid cell in one model was from the other (ie. a value of 0 meant they had the same habitat suitability, a value of 1 or –1 meant they were one habitat suitability class apart and a value of 2 or –2 meant they were two habitat suitability classes apart).

While the models from the two different modelling approaches presented in this study were derived from data at very different spatial resolutions (point samples for the fishery independent data and fishing blocks for the fishery dependent data), they were worth comparing to identify any trends in habitat distribution at a bay-wide scale.

Results of the comparison between the two habitat suitability models for King George whiting (sub-adults) and snapper (sub-adults) are shown in Figure 8.1 and Figure 8.4 respectively. Both comparisons indicate that there were substantial areas of agreement in habitat suitability classifications between the fishery independent and dependent models, and most of the remaining areas were only 1 class apart in their definition of habitat suitability.

The comparison between model grids for sub-adult King George whiting (Figure 8.1) showed that most of the Bay in depths <10 m, and particularly the southern region of the Bay, the western shores and depths <5 m in Corio Bay/Geelong Arm, had the same habitat suitability classification in both the selected fishery independent and dependent models. The comparison between model grids for sub-adult snapper (Figure 8.4) showed that most of the Bay in depths of 10-20 m had the same habitat suitability classification in both the fishery independent and dependent models.

In order to determine whether there were any patterns in the class difference categories across the habitat suitability model classes, we overlayed the individual habitat suitability model grids on the class difference grids. Each habitat suitability model was overlayed on the corresponding class difference grid in the GIS (eg. Figure 5.10C was overlayed on Figure 8.1), and a cross-tabulation undertaken to extract the total area of each habitat suitability class (low, medium and high) that lay within each class difference category (same, 1 class different, 2 classes different).

The distribution of class difference categories by area for habitat suitability classifications in the King George whiting (sub-adult) fishery dependent model is shown in Figure 8.2. Almost all of the high suitability habitat in the habitat affinity model was the same as the fishery dependent model (Figure 8.2). The medium suitability habitat was split between classifications that matched the fishery dependent model or were 1 class different. The low suitability habitat was mostly 1 class different to the fishery dependent model.

The distribution of class difference categories by area for habitat suitability model classifications in the King George whiting (sub-adult) fishery dependent model is shown in Figure 8.3. This distribution is similar to that observed for the fishery independent habitat affinity model (Figure 8.2), but the main difference is that the high suitability habitat had more area classed as 1 or 2 classes different by comparison to the fishery independent model.

The distribution of class difference categories by area for the snapper (sub-adult) habitat affinity model is shown in Figure 8.5. In this case most of the high suitability habitat in the habitat affinity model was the same as the fishery dependent model (Figure 8.5). The medium and low suitability habitat classifications were split between classifications that were the same or 1 class different by comparison to the fishery dependent model.

The distribution of class difference categories by area for habitat suitability model classifications in the snapper (sub-adult) fishery dependent model is shown in Figure 8.6. In this case most of the high and medium suitability habitat was either the same or 1 class different to the fishery independent model (Figure 8.6). Most of the low suitability habitat was 1 class different to the fishery dependent model.

The overall pattern shown in the comparison between the habitat suitability model grids and the class difference grids was that there was a greater agreement between the definition of high and medium suitability habitat than the low suitability habitat. There were also only small areas where the model comparison showed that the selected models were 2 classes different (ie. the models contradicted each other with one classifying an area as low suitability habitat and the other as high suitability habitat).

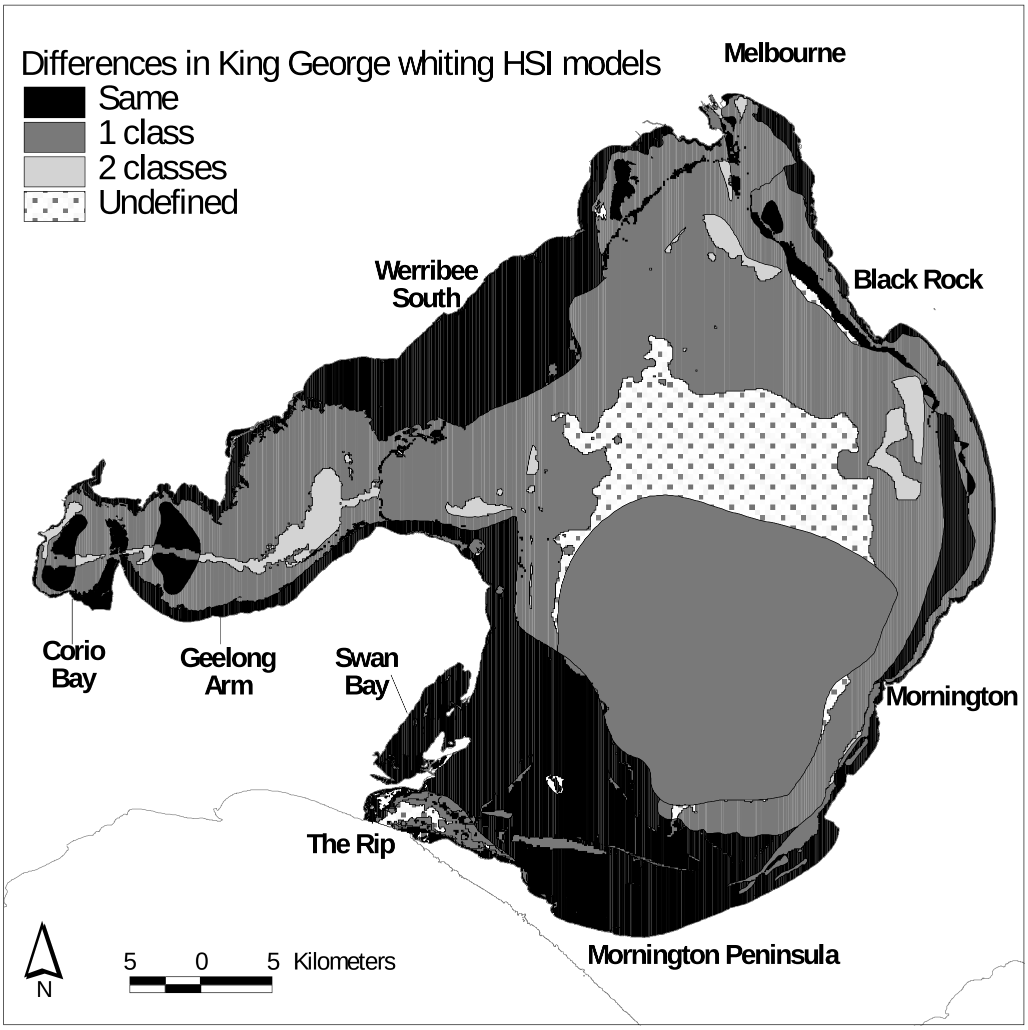


Figure 8.1. Differences in habitat suitability class categories (low, medium, high) between the King George whiting (sub-adults) habitat suitability model (produced with habitat affinity SI's for depth, sediment and substrate type/biota; Figure 5.10C) and the model produced from fishery dependent data (Figure 6.8).

Map legend:
Same = no difference between the habitat suitability class assigned by either model.
1 Class = one class difference between the habitat suitability assigned in either model (eg. model 1 shows a grid cell value as low suitability while model 2 shows the same cell value as medium suitability).
2 classes = two classes of difference between the habitat suitability assigned in either model (eg. model 1 shows a grid cell value as low suitability while model 2 shows the same cell value as high suitability).

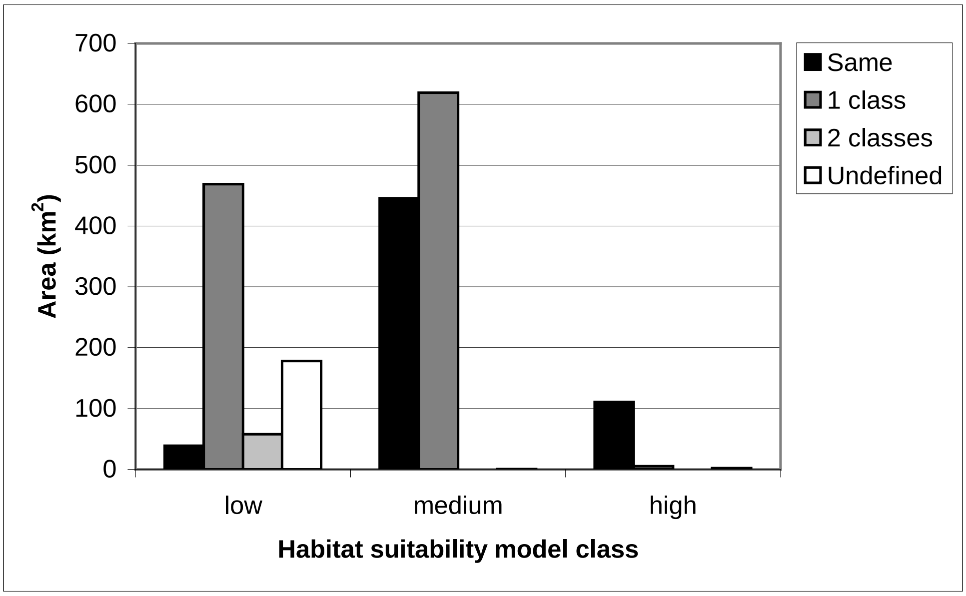


Figure 8.2. Total area of habitat suitability class difference categories from Figure 8.1 by habitat suitability classes within the King George whiting (sub-adult) habitat suitability model Figure 5.10C - produced with habitat affinity SI's for depth, sediment and substrate type/biota

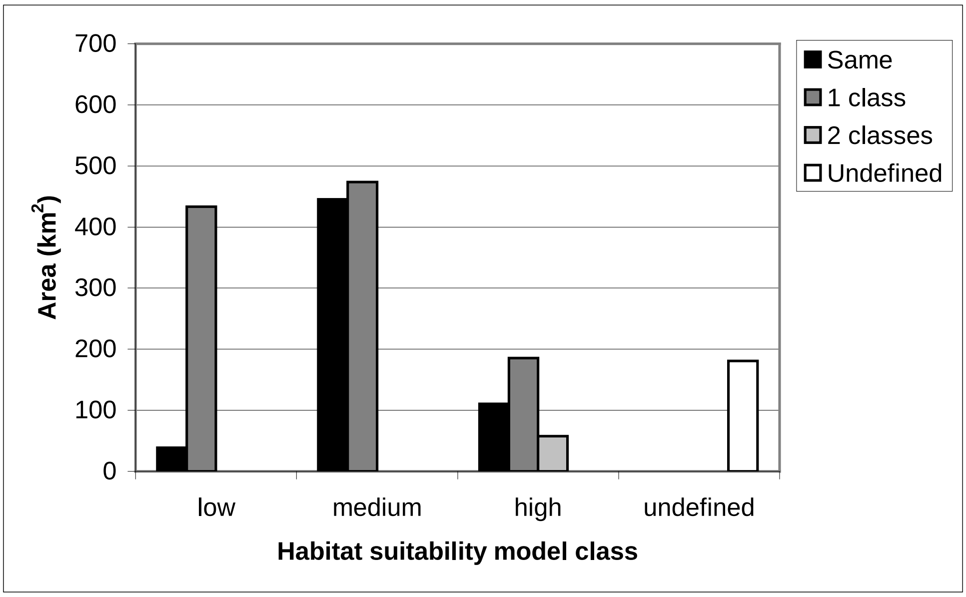


Figure 8.3. Total area of habitat suitability class difference categories from Figure 8.1 by habitat suitability classes within the King George whiting (sub-adult) fishery dependent habitat suitability model (Figure 6.8).

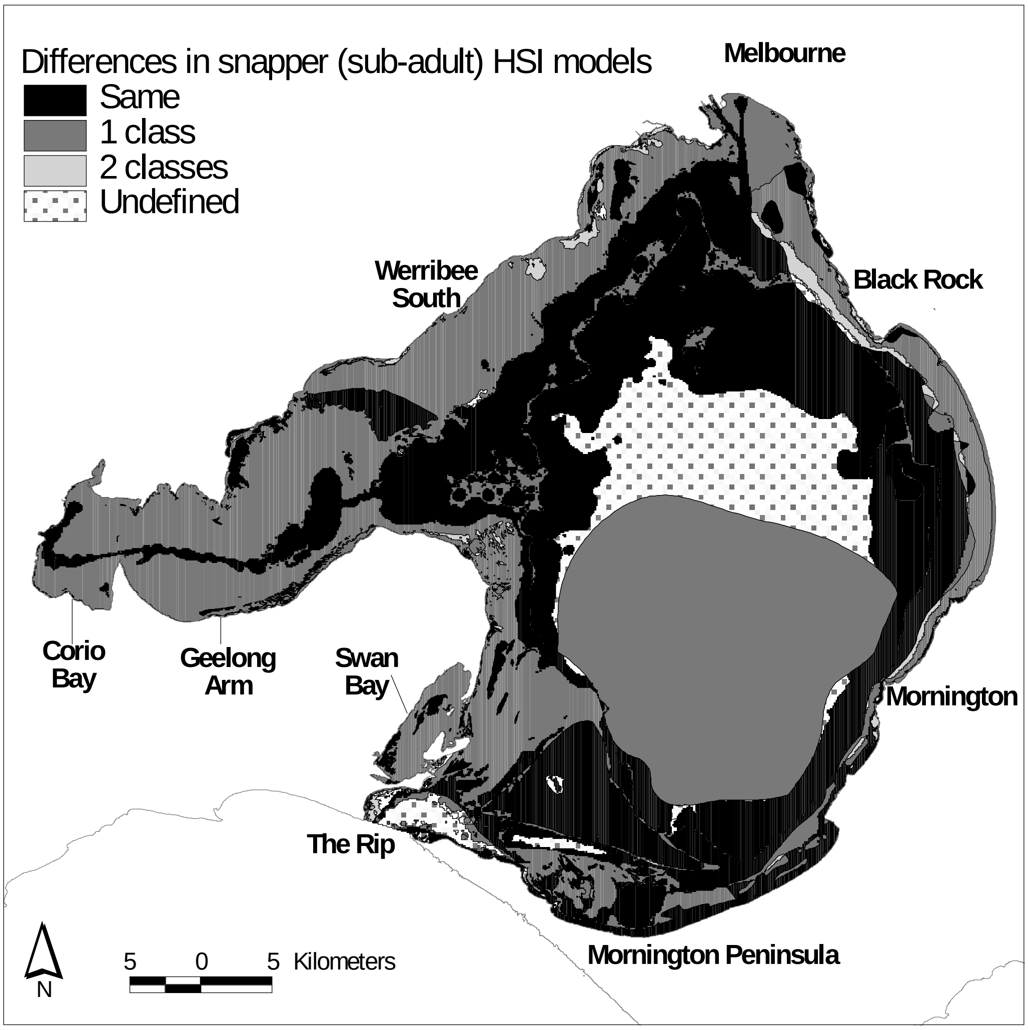


Figure 8.4. Differences in habitat suitability class categories (low, medium, high) between the Snapper (sub-adult) habitat suitability model (produced with habitat affinity SI's for depth and sediment; Figure 5.14B) and the model produced from fishery dependent data (Figure 6.14).

Map legend:
Same = no difference between the habitat suitability class assigned by either model.
1 Class = one class difference between the habitat suitability assigned in either model (eg. model 1 shows a grid cell value as low suitability while model 2 shows the same cell value as medium suitability).
2 classes = two classes of difference between the habitat suitability assigned in either model (eg. model 1 shows a grid cell value as low suitability while model 2 shows the same cell value as high suitability).

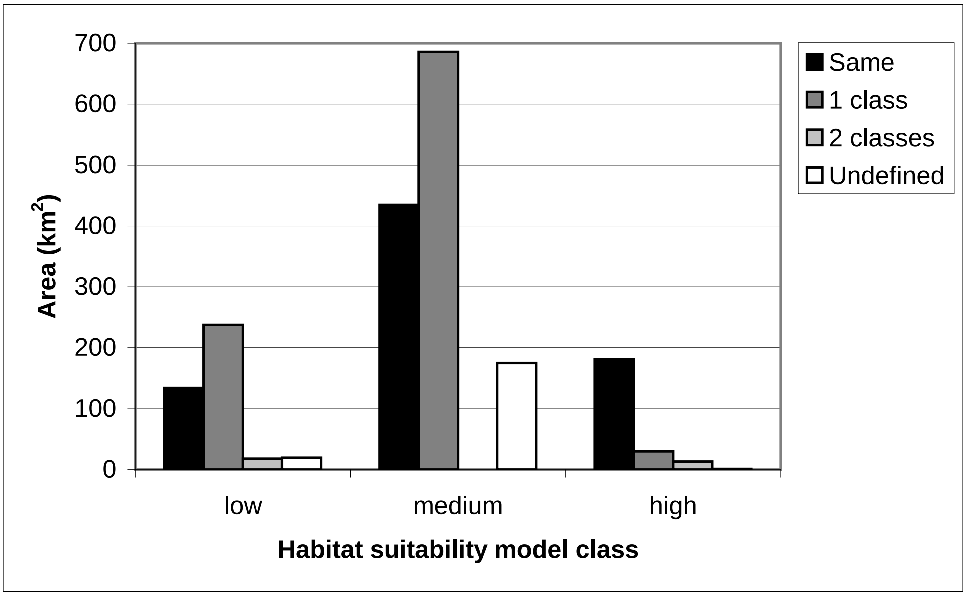


Figure 8.5. Total area of habitat suitability class difference categories from Figure 8.4 by habitat suitability classes within the snapper (sub-adult) habitat suitability Figure 5.14B - model produced with habitat affinity SI's for depth and sediment.

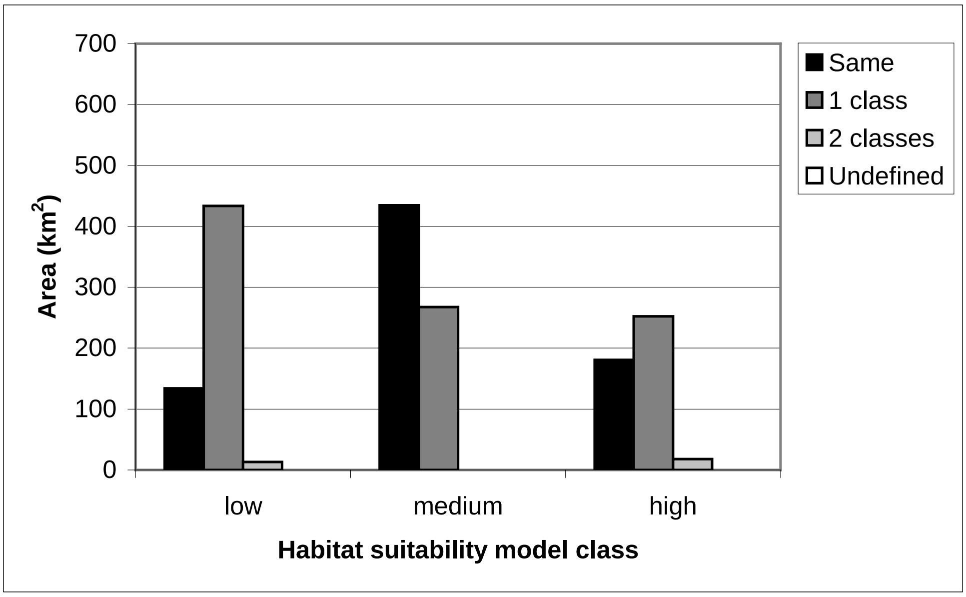


Figure 8.6. Total area of habitat suitability class difference categories from Figure 8.4 by habitat suitability classes within the snapper (sub-adult) fishery dependent habitat suitability model (Figure 6.14).

9. Outcomes and Conclusion

9.1. Benefits

The original project proposal predicted that the flow of benefits from this project would be equally split between the commercial and recreational sectors for the bay and inlet fisheries. While we have focussed on commercial species, the key species addressed by this project (ie. King George whiting and snapper) are equally targeted by recreational fishers. We also produced habitat suitability models for sand flathead which are not targeted by commercial fishers, but represent a large component of the recreational fishery.

The habitat suitability models presented in this study are in themselves useful, but the greater benefit of the project comes from having establishing a framework for fishery habitat suitability modelling within a GIS. This application provides fishery managers with an effective tool to develop and assess possible scenarios when establishing fishery management plans and responding to development proposals. The likely impact that changes in one environmental variable may have on the overall distribution of suitable habitat for a species can also be readily incorporated into the model outputs.

9.2. Further Development

PIRVic Marine and Freshwater Systems will continue to enhance the habitat suitability model GIS application beyond the current project as new fishery and/or spatial data becomes available for Victorian bays and inlets.

In recognition of the lack of fishery independent data for Port Phillip Bay habitats in depths of 2 to 7 m, Fisheries Victoria funded PIRVic Marine and Freshwater Systems to undertake a sampling program during summer 2003-04. The field sampling focussed on three distinct habitats in this depth range; unvegetated sediment, seagrass and reef. The data collected during this program is presently being analysed and will be used to improve the suitability index values presented in Section 5 and to test the habitat suitability models.

The suitability index values for individual environmental parameters used in the fishery independent habitat suitability models should not be seen as static tables and should be updated as other new information and sampling becomes available. It is expected that enhancements to SI values over time will improve the composite habitat suitability models. We therefore recommend that future fishery independent monitoring programs seek to undertake simultaneous measurements of key environmental variables. The advent of GPS technology enables sampling locations to be recorded to a high degree of precision. New generation depth sounders and underwater video also provide the means to make accurate measurements of seabed characteristics.

The ArcView habitat suitability model application interface developed in this project (Section 7) can readily be adapted to model other locations across Australia, provided that at least one environmental variable is available in a raster grid format for that area. The ArcView application will be made available to fishery management or research agencies that want to investigate habitat suitability modelling for their region. The extension of habitat suitability modelling to other locations will have the added benefits of improving the modelling application and also testing localised assumptions/knowledge about fishery habitat.

As questions about likely impacts on fishery habitat from possible greenhouse effects begin to gain prominence, the habitat suitability modelling application could potentially be modified in the future to address how changes in one environmental or habitat variable in response to global warming could in turn affect the distribution of suitable habitat for commercially significant species.

9.3. Planned Outcomes

The spatial models produced in this study present a simplified picture of habitat suitability and do not account for many complex relationships and interactions both between species and environmental variables. However, in the absence of a more complete knowledge of the nature of these relationships and the spatial scales at which they operate, the habitat suitability modelling approach presents a relatively effective method for conducting a 'first-pass' at identifying likely distributions of important fishery habitat. This ability to address the spatial aspects of fishery habitat has to date not been available to fishery and natural resource managers.

The Victorian bay and inlet fisheries are subject to considerable competitive pressures for access to fishery resources between the commercial and recreational sectors. At the same time, major developments, continually raise questions about the locations and status of fishery habitat in Victorian bays and inlets. While the habitat suitability modelling approach presented in this study cannot answer all questions about the spatial distribution of important fishery habitat in these areas, it represents an important step forward in highlighting the importance of these habitats. We have also produced a GIS application that can be readily updated to incorporate new information about interactions between species and habitat and habitat affinities as they become known. Similarly, the model can also incorporate new spatial data representing the spatial distribution of important environmental variables or habitats.

9.4. Conclusion

The primary objective of this project was to develop a fisheries habitat suitability model for Victorian bays and inlets with a Geographic Information System (GIS). An extensive literature search and direct contact with specialists in fishery GIS in the USA resulted in the selection of a habitat suitability index approach for modelling from fishery independent data. We were able to successfully apply the habitat suitability index modelling approach to Port Phillip Bay and Western Port, while establishing a modelling system that could also be applied to Corner Inlet with further refinement of the suitability index values (Section 5).

Our modelling was limited by a lack of suitable fishery independent data to quantify some of the habitat affinities or preferences for the species under investigation. Despite this limitation, we were able to use qualitative approaches to identify suitability index values for these species and parameters. These values were less reliable than the quantitative assessments. Similarly, the lack of data meant that our testing and validating of the habitat suitability models was restricted to qualitative assessments. Within these limitations we were able to produce habitat suitability models that were relatively reliable for the key species under investigation in Port Phillip Bay, and simplified models for selected species in Western Port.

In the absence of suitable fishery independent data, we investigated other data sources and have shown that catch and effort data from the commercial fishery can be used to create habitat suitability models (Section 6). We developed a method that allows predictions of suitable habitat and, by extension, fish distributions, at a finer scale than the catch returns. Within this method, the best models related to species that were highly targeted by commercial fishers due to the implicit assumption that areas targeted by commercial fishers had the highest densities of the species being modelled. While there is evidence that our habitat suitability models provided good predictive information on fish habitat and fish distributions for some species, the models and the hypotheses generated from the modelling process require further testing. In the absence of high quality data from fishery independent monitoring, the approach described here provides a useful adjunct in developing spatial models to define important fishery habitat at a bay-wide scale.

The second project objective was to integrate a wide range of existing spatial and non-spatial data for habitat types, environmental parameters, species distribution, species life histories and habitat requirements in the GIS through a relational database. This objective was achieved through the development of multiple tables of habitat suitability index values for the fishery independent analysis (Section 5) and combined habitat classification values from the fishery dependent analysis (Section 6). The tables were all used in the modelling and have been incorporated in the GIS habitat suitability modelling application (Section 7). These tables had a relatively straight-forward data structure which could be readily linked or joined within the GIS to each other or the spatial data tables as required. As a result, we did not need to establish a separate relational database that was external to the GIS to manage these datasets or integrate them into the modelling.

The final objective of the project was to develop a customised GIS user interface for querying the fisheries habitat suitability model and producing habitat suitability maps. A customised application was developed for ArcView 3.3 and the ArcView Spatial Analyst extension software using ArcView's own programming language Avenue. The application consists of a customised user interface, menus, buttons and a habitat suitability modelling wizard which was designed to guide users with limited knowledge of GIS through the modelling process (Section 7). Some notable features of the habitat suitability modelling wizard include the ability for users to edit the species suitability index values or to model an entirely new species that has suitability index values completely defined by the user.

The ArcView Habitat Suitability Model Interface was primarily set-up to allow users to produce composite models of habitat suitability from the suitability index values calculated from fishery independent data. Habitat suitability models generated from an analysis of fishery dependent data can also be added directly to a view in the interface.

Copies of the ArcView Habitat Suitability Model Interface are available on CD-ROM from PIRVic Marine and Freshwater Systems.

10. References

AFN (1996). Australian Fishing Network Fishing Map 7. Port Welshpool and Port Albert. Australian Fishing Network, Croydon.

AFN (2002). Australian Fishing Network Fishing Map 1. Port Phillip Bay. Australian Fishing Network, Croydon.

Akcakaya, H. R., McCarthy, M. A. & Pearce, J. L. (1995). Linking landscape data with population viability analysis: management options for the helmeted honeyeater *Lichenostomus melanops cassidix*. *Biological Conservation* **73**, 169-176.

Anon. (1981). The ecology of fish in Botany Bay - biology of commercially and recreationally valuable species. pp. 287. State Pollution Control Commission, Sydney.

Anon. (2001). Fisheries Victoria Commercial Fish Production Information Bulletin. pp. 29. Marine and Freshwater Resources Institute, Queenscliff.

Azeta, M., Ikemoto, R. & Azuma, M. (1980). Distribution and growth of demersal 0-age Red Sea bream, *Pagrus major*, in Shijiki Bay. *Seikei Reg. Fish. Res. Lab.* **54**, 259-278.

Ball, D. & Coots, A. G. (2001). Catch and Effort Info: An ArcView GIS application for querying and displaying commercial fishery catch data. In 'AURISA 2001, 29th Annual Conference of AURISA, 19-23 November'. Melbourne.

Bender, L. C., Roloff, G. J. & Haufler, J. B. (1996). Evaluating confidence intervals for habitat suitability models. *Wildlife Society Bulletin* **24**, 347-352.

Blake, S. & Ball, D. (2001a). Seagrass mapping of Port Phillip Bay. Geospatial Systems Section, Marine and Freshwater Resources Institute, Queenscliff.

Blake, S. & Ball, D. (2001b). Victorian Marine Habitat Database: Seagrass Mapping of Western Port. Marine and Freshwater Resources Institute, Queenscliff.

Bowen, B. K. (1997). What are the roles of science, economics, sociology and politics in fisheries management. In 'Developing and sustaining world fisheries resources. The state of science and management. 2nd World Fisheries Congress.' (Eds D. A. Hancock, D. C. Smith, A. Grant & J. P. Beumer). CSIRO Publishing, Melbourne.

Brooks, R. P. (1997). Improving habitat suitability models. *Wildlife Society Bulletin* **25**, 163-167.

Brown, J. R. & Hartwick, E. B. (1988). A habitat suitability index model for suspended tray culture of the Pacific oyster, *Crassostrea gigas thunberg*. *Aquaculture and Fisheries Management* **19**, 109-126.

Brown, S. K., Buja, K. R., Jury, S. H., Monaco, M. E. & Banner, A. (2000). Habitat suitability index models for eight fish and invertebrate species in Casco and Sheepscot Bays, Maine. *North American Journal of Fisheries Management* **20**, 408-435.

Burgman, M. A., Breininger, D. R., Duncan, B. W. & Ferson, S. (2001). Setting reliability bounds on habitat suitability indices. *Ecological Applications* **11**, 70-78.

Butler, A. & Jernakoff, P. (Eds) (1999). Seagrass in Australia: Strategic review and development of an R & D plan. CSIRO, Melbourne. 210 pp.

Cappo, M., Alongi, D. M., Williams, M. & Duke, N. (1998). A review and synthesis of Australian fisheries habitat research: major threats, issues and gaps in knowledge of coastal and marine fisheries habitats. Australian Institute of Marine Science, Townsville, Qld.

Cappo, M. C. (1987). The fate and fisheries biology of sub-adult Australian salmon in South Australian waters. pp. 162. S.A. Department of Fisheries Research Report, Adelaide.

Christensen, J. D., Battista, T. A., Monaco, M. E. & Klein, C. J. (1997). Habitat suitability modeling and GIS technology to support habitat management: Pensacola Bay, Florida case study. pp. 89. NOAA, Strategic Environmental Assessment Division, Silver Spring, MD.

Chubb, C. F., Potter, I. C., Grant, C. J., Lenanton, R. C. J. & Wallace, J. (1981). Age structure, growth rates and movements of sea mullet, *Mugil cephalus* L., and yellow-eye mullet, *Aldrichetta forsteri* (Valenciennes), in the Swan-Avon River System, Western Australia. *Australian Journal of Marine and Freshwater Research* **32**, 605-628.

Clark, R. D., Minello, T. J., Christensen, J. D., Cladwell, P. A., Monaco, M. E. & Mathews, G. A. (1999). Modeling nekton habitat use in Galveston Bay, Texas: an approach to define essential fish habitat (EFH). pp. 70. NOAA/NOS Biogeography Program, Silver Spring, MD and Galveston, Texas.

Classon, B. & Wilson, G. (Eds) (2002). Fishing Victoria's Coastline. Australian Fishing Network, 112 pp.

Conron, S. & Coutin, P. (1998). The recreational snapper catch from Port Phillip Bay: a pilot survey of the boat based fishery. Marine and Freshwater Resources Institute, Queenscliff.

Coutin, P. (1996). Snapper 1996, Victorian Fisheries Stock Assessment Report. Marine and Freshwater Resources Institute, Queenscliff.

Coutin, P. (Ed.) (2000a). Port Phillip Bay Fin Fisheries - 1998. Compiled by the Bay & Inlet Fisheries and Stock Assessment Group. pp. 79. Marine and Freshwater Resources Institute, Queenscliff.

Coutin, P. (2000b). Snapper - 1998. pp. 43. Marine and Freshwater Resources Institute, Queenscliff.

Coutin, P., Conron, S. & MacDonald, M. (1995). The daytime recreational fishery in Port Phillip Bay. pp. 49. Department of Conservation and Natural Resources, Queenscliff.

Coyne, M. S. & Christensen, J. D. (1997). Habitat suitability index modeling: species habitat suitability index values. pp. 19. NOAA, Biogeography Program, Silver Spring, MD.

Crawford, C. M. (1984). An ecological study of Tasmanian flounder. Thesis, University of Tasmania.

Crowley, J. & Worsteling, P. (Eds) (2000). Fishing Western Port. Sportsfish Australia Publications Pty. Ltd., Noorat.

DCLS (1980). Corner Inlet/Seaspray Coastal Study. Prepared for the Coastal Management and Co-ordination Committee by the Division of Crown Land Management, Department of Crown Lands and Survey in collaboration with the Division of the Town and Country Planning Board of the Department of Planning, Soil Conservation Authority and the Ports and Harbours Division of the Public Works Department. pp. 238. Department of Crown Lands and Survey, Melbourne.

DCNR (1995). Corner Inlet Fisheries Management Plan (draft). pp. 59. Flora, Fauna & Fisheries Division, Department of Conservation & Natural Resources, East Melbourne.

Debeljak, M., Dzeroski, S., Jerina, K., Kobler, A. & Adamic, M. (2001). Habitat suitability modelling for red deer (*Cervus elaphus L.*) in South-central Slovenia with classification trees. *Ecological Modelling* **138**, 321-330.

Denis, V., Lejeune, J. & Robin, J. P. (2002). Spatio-temporal analysis of commercial trawler data using general additive models: patterns of Loliginid squid abundance in the north-east Atlantic. *ICES Journal of Marine Science* **59**, 633-648.

Denis, V. & Robin, J.-P. (2001). Present status of the French Atlantic fishery for cuttlefish (*Sepia officinalis*). *Fisheries Research* **52**, 11-22.

DNRE (1996). Corner Inlet & Nooramunga Marine & Coastal Parks Draft Management Plan. pp. 41. National Parks Service, Department of Natural Resources & Environment, East Melbourne.

Donaldson, A. K. & Marsden, M. A. H. (1977). Facies and bottom sediment characteristics of Western Port. Report to Western Port Bay Regional Study, Environmental Studies Section, Ministry for Conservation, Victoria. pp. 79. Geology Department, University of Melbourne, Melbourne.

Edgar, G. J. (Ed.) (1997). Australian Marine life: the plants and animals of temperate waters. Reed Books, Melbourne. 544 pp.

Edgar, G. J., Hammond, L. S. & Watson, G. F. (1993). Consequences for commercial fisheries of loss of seagrass beds in southern Australia. pp. 130. Victorian Institute of Marine Sciences, East Melbourne.

Edgar, G. J. & Shaw, C. (1995). The production and trophic ecology of shallow-water fish assemblages in southern Australia. I. Species richness, size structure and production of fishes in Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* **194**, 53-81.

Ferns, L. W. & Hough, D. (Eds) (2000). Environmental inventory of Victorias marine ecosystems stage 3 - understanding biodiversity representativeness of Victorias rocky reefs. Parks, Flora and Fauna Division, Department of Natural Resources and Environment, East Melbourne.

Ferrell, D. J. & Bell, J. D. (1991). Differences among assemblages of fish associated with *Zostera capricorni* and bare sand over a large spatial scale. *Marine Ecology Progress Series* **72**, 15-24.

Fowler, A. J. (1997). Aspects of the population biology of the King George whiting (*Sillaginodes punctata*) in South Australia - consequences for evolutionary processes. In 'ASFB/FARMAA'. Darwin.

Fowler, A. J. & McGarvey, R. (1995). Development of an integrated fisheries management model for King George whiting (*Sillaginodes punctata*) in south Australia. pp. 231. SARDI, Henley Beach.

Francis, M. P., Williams, M. W., Pryce, A. C., Pollard, S. & Scott, S. G. (1992). Daily increments in otoliths of juvenile snapper, *Pagrus auratus* (Sparidae). *Australian Journal of Marine and Freshwater Research* **43**, 1015-1032.

Fukuhara, O. (1985). Functional morphology and behaviour of early life stages of red sea bream. *Bulletin of the Japanese Society of Scientific Fisheries.* **51**, 731-743.

Gallaway, B. J., Cole, J. G., Meyer, R. & Roscigno, P. (1999). Delineation of essential fish habitat for juvenile red snapper in the northwestern Gulf of Mexico. *Transactions of the American Fisheries Society* **128**, 713-726.

GHD (1997). Port Phillip Bay Coastal Values Assessment. Physical Characteristics and Environmental Values Part A. Report to Parks Victoria. Gutteridge Haskins & Davey Pty Ltd, Melbourne.

Gomon, M. F., Glover, J. C. M. & Kuiter, R. H. (Eds) (1994). The Fishes of Australia's South Coast. State Print, Adelaide. 992 pp.

Gregr, E. J. & Trites, A. W. (2001). Predictions of critical habitat for five whale species in the waters of coastal British Columbia. *Canadian Journal of Fisheries and Aquatic Science* **58**, 1265-1285.

Gunthorpe, L. & Hamer, P. (Eds) (1998a). Corner Inlet Fish Habitat 1998. Compiled by the Fish Habitat Assessment Group. Fisheries Victoria Assessment Report No. 20. pp. 50. Fisheries Victoria, East Melbourne.

Gunthorpe, L. & Hamer, P. (Eds) (1998b). Western Port Fish Habitat 1998. Compiled by the Fish Habitat Assessment Group. Fisheries Victoria Assessment Report No. 21. pp. 58. Fisheries Victoria, East Melbourne.

Gunthorpe, L., Hamer, P. & Walker, S. (1998). Bay and Inlets Scalefish Fisheries Review. Influence of Environmental and Habitat Features on Scalefish Catches from Victorian Bays and Inlets. Volume 1: Lifecycles and Habitat Requirements of Selected Victorian Fish Species. pp. 88. Marine and Freshwater Resources Institute, Queenscliff.

Hall, D. N. (1984). The Coorong: biology of the major fish species and fluctuations in catch rates 1976-1983. *SAFIC* **8**, 3-17.

Hall, D. N. & MacDonald, C. M. (1986). Commercial Fishery Situation Report: Net and line fisheries of Port Phillip Bay, Victoria, 1914 - 1984. pp. 121. Department of Conservation, Forests and Lands, Fisheries Division. Marine Fisheries Report, East Melbourne.

Hamer, P., Jenkins, G. & Welsford, D. (1997). Sampling of newly-settled snapper, *Pagrus auratus*, and identification of preferred habitats in Port Phillip Bay - a pilot study. pp. 53. Marine and Freshwater Resources Institute, Queenscliff.

Hamer, P. A. & Jenkins, G. P. (2004). High levels of spatial and temporal recruitment variability in the temperate sparid *Pagrus auratus*. *Marine and Freshwater Research* **55**, 663-673.

Harris, G., Batley, G., Fox, D., Hall, P., Jernakoff, P., Molloy, R., Murray, A., Newell, B., Parslow, J., Skyring, G. & Walker, S. (Eds) (1996). Port Phillip Bay Environmental Study Final Report. CSIRO Publishing, Canberra.

Harris, J. A. (1968). The yellow-eye mullet. Age structure, growth rate and spawning cycle of a population of yellow-eye mullet, *Aldrichetta forsteri* (Cuv. and Val.) from the Coorong lagoon, South Australia. *Transactions of the Royal Society of South Australia* **92**, 37-50.

Harris, J. E., Hinwood, J. B., Marsden, M. A. H. & Sternberg, R. W. (1979). Water movements, sediment transport and deposition, Western Port, Victoria. *Marine Geology* **30**, 131-161.

Harris, J. E. & Robinson, J. B. (1979). Circulation in Western Port, Victoria, as deduced from salinity and reactive silica distributions. *Marine Geology* **30**, 101-116.

Hart, P. & Purser, G. (1995). Effects of salinity and temperature on eggs and yolk sac larvae of the greenback flounder (*Rhombolsolea tapirina* Gunther, 1862). *Aquaculture* **136**, 221-230.

Hart, P. R., Hutchinson, W. & Purser, G. (1996). Effects of photoperiod, temperature and salinity on hatchery reared larvae of the greenback flounder (*Rhombosolea tapirina* Gunther, 1862). *Aquaculture* **144**, 303-311.

Hilborn, R. and Walters, C. (1992) Quantitative fisheries stock assessment. Choice dynamics and uncertainty. Routledge, Chapman and Hall, London.

Hindell, J. S. & Jenkins, G. P. (2004). Spatial and temporal variability in the assemblage structure of fishes associated with mangroves (*Avicennia marina*) and intertidal mudflats in temperate Australian embayments. *Marine Biology* **144**, 385–396.

Hindell, J. S., Jenkins, G. P. & Keough, M. J. (2000a). Evaluating the impact of predation by fish on the assemblage structure of fishes associated with seagrass (*Heterozostera tasmanica*) (Martens ex Ascherson) den Hartog, and unvegetated sand habitats. *Journal of Experimental Marine Biology and Ecology* **255**, 153-174.

Hindell, J. S., Jenkins, G. P. & Keough, M. J. (2000b). Variability in abundance of fishes associated with seagrass habitats in relation to diets of predatory fishes. *Marine Biology* **136**, 725-737.

Hobday, D. K. (1992). Abundance and distribution of pilchard and Australian anchovy as prey species for the little penguin *Eudyptula minor* at Phillip Island, Victoria. *Emu* **91**, 318-341.

Hobday, D. K., Officer, R. A. & Parry, G. D. (1999). Changes to demersal fish communities in Port Phillip Bay, Australia, over two decades, 1970-91. *Journal of Marine and Freshwater Research* **50**, 397-407.

Hoedt, F. E. & Dimmlich, W. F. (1994). Diet of subadult Australian salmon, *Arripis truttaceus*, in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* **45**, 617-623.

Hope Black, J. (1971). Benthic Communities. *Memoirs of the National Museum of Victoria* **32**, 129-170.

Hosmer, D. W. J. & Lemeshow, S. (Eds) (1996). Applied logistic regression. John Wiley and Sons, New York.

Hutchins, B. & Swainston, R. (Eds) (1986). Sea fishes of southern Australia. Swainston Publishing, Perth. 437 pp.

Hyndes, G. A., Kendrick, A. J., MacArthur, L. D. & Stewart, E. (2003). Differences in the species and size-composition of fish assemblages in three distinct seagrass habitats with differing plant and meadow structure. *Marine Biology* **142**, 1195-1206.

Isaak, D. J. & Huber, W. A. (1997). Integrating new technologies into fisheries science: the application of geographic information systems. *Fisheries* **22 (1)**, 6-10.

Jenkins, G. P. (1986). Composition, seasonality and distribution of ichthyoplankton in Port Phillip Bay, Victoria. *Journal of Experimental Marine Biology and Ecology* **37**, 507-520.

Jenkins, G. P. & Black, K. P. (1994). Temporal variability in settlement of a coastal fish (*Sillaginodes punctata*) determined by low-frequency hydrodynamics. *Limnology and Oceanography* **39 (7)**, 1744-1754.

Jenkins, G. P., Black, K. P. & Hamer, P. A. (2000). Determination of spawning areas and larval advection pathways for King George whiting in south-eastern Australia using otolith microstructure and hydrodynamic modelling. I. *Marine Ecology Progress Series* **199**, 231-242.

Jenkins, G. P., Black, K. P., Wheatley, M. J. & Hatton, D. N. (1997a). Temporal and spatial variability in recruitment of a temperate, seagrass-associated fish is largely determined by physical processes in the pre- and post-settlement phases. *Marine Ecology Progress Series* **148**, 23-35.

Jenkins, G. P. & Hamer, P. A. (2001). Spatial variation in the use of seagrass and unvegetated habitats by post-settlement King George whiting (*Percoidei: Sillaginidae*) in relation to meiofaunal distribution and macrophyte structure. *Marine Ecology Progress Series* **224**, 219-229.

Jenkins, G. P., Keough, M. J. & Hamer, P. A. (1998a). The contribution of habitat structure and larval supply to broad-scale recruitment variability in a temperate zone, seagrass-associated fish. *Journal of Experimental Marine Biology and Ecology* **226**, 259-278.

Jenkins, G. P. & May, H. M. A. (1994). Variation in settlement and larval duration of King George whiting, *Sillaginodes punctata* (*Sillaginidae*), in Swan Bay, Victoria, Australia. *Bulletin of Marine Science* **54**, 281-296.

Jenkins, G. P., May, H. M. A., Wheatley, M. J. & Holloway, M. G. (1997b). Comparison of fish assemblages associated with seagrass and adjacent unvegetated habitats of Port Phillip Bay and Corner Inlet, Victoria, Australia, with emphasis on commercial species. *Estuarine, Coastal and Shelf Science* **44**, 569-588.

Jenkins, G. P. & Sutherland, C. R. (1997). The influence of habitat structure on nearshore fish assemblages in a southern Australian embayment: colonisation and turnover rate of fishes associated with artificial macrophyte beds of varying physical structure. *Journal of Experimental Marine Biology and Ecology* **218**, 103-125.

Jenkins, G. P., Watson, G. F. & Hammond, L. S. (1993). Patterns of utilisation of seagrass (*Heterozostera*) dominated habitats as nursery areas by commercially important fish. pp. 100. Victorian Institute of Marine Sciences, East Melbourne.

Jenkins, G. P., Watson, G. F., Hammond, L. S., Black, K. P., Wheatley, M. J. & Shaw, C. (1996a). Importance of shallow water, reef-algal habitats as nursery areas for commercial fish from southeastern Australia. pp. 165. Department of Zoology, University of Melbourne and Victorian Institute of Marine Sciences, East Melbourne.

Jenkins, G. P., Welsford, D. C., Keough, M. J. & Hamer, P. A. (1998b). Diurnal and tidal vertical migration of pre-settlement King George whiting Sillaginodes punctata in relation to feeding and vertical distribution of prey in a temperate bay. *Marine Ecology Progress Series* **170**, 239-248.

Jenkins, G. P. & Wheatley, M. J. (1998). The influence of habitat structure on nearshore fish assemblages in a southern Australian embayment: comparison of shallow seagrass, reef-algal and unvegetated sand habitats, with emphasis on their importance to recruitment. *Journal of Experimental Marine Biology and Ecology* **221**, 147-172.

Jenkins, G. P., Wheatley, M. J. & Poore, A. G. B. (1996b). Spatial variation in recruitment, growth and feeding of post-settlement King George whiting, *Sillaginodes punctata*, associated with seagrass beds of Port Phillip Bay, Australia. *Canadian Journal of Fisheries and Aquatic Science* **53**, 350-359.

Jones, G. K., Hall, K. M. & Staniford, A. (1990). The South Australian marine scalefish fishery: stock assessment, economics, management. South Australian Department of Fisheries Green Paper.

Kailola, P. J., Williams, M. J., Stewart, P. C., Reichelt, R. E., McNee, A. & Grieve, C. (Eds) (1993). Australian Fisheries Resources. Bureau of Resource Sciences, Department of Primary Industries and Energy, and Fisheries Research and Development Corporation, Canberra, Australia.

Kemp, Z. & Meaden, G. (2002). Visualization for fisheries management from a spatiotemporal perspective. *ICES Journal of Marine Science* **59**, 190-202.

Klumpp, D. W. & Nichols, P. D. (1983). A study of food chains in seagrass communities II. Food of the Rock Flathead, *Platycephalus laevigatus* Cuvier, a major predator in a *Posidonia australis* seagrass bed. *Australian Journal of Marine and Freshwater Research* **34**, 745-754.

Kuiter, R. H. (Ed.) (1993). Coastal fishes of south-eastern Australia. Crawford House Press, Bathurst. 437 pp.

Kurth, D. (1957). An investigation of the greenback flounder, *Rhombosolea tapirina* Gunther. Thesis, University of Tasmania.

Last, P. R., Scott, E. O. G. & Talbot, F. H. (Eds) (1983). Fishes of Tasmania. Tasmanian Fisheries Development Authority, Hobart. 563 pp.

LCC (1993). Marine and Coastal Special Investigation: Descriptive Report. Land Conservation Council, Melbourne.

Lenanton, R. C. J. (1977). Aspects of the ecology of fish and commercial crustaceans of the Blackwood River estuary, Western Australia. Fisheries Research Bulletin 19. pp. 72. Western Australian Department of Fisheries and Wildlife, Perth.

Lenanton, R. C. J. (1982). Alternate non-estuarine nursery habitats for some commercially and recreationally important fish species of south-western Australia. *Marine and Freshwater Research* **33**, 881-900.

Longmore, A. R. (1997). Analysis of water quality in Western Port, 1973-1977 in relation to protection of beneficial uses. Marine and Freshwater Resources Institute, Queenscliff.

Longmore, A. R., Cowdell, R. D. & Flint, R. (1996). Nutrient status of the waters of Port Phillip Bay. Technical Report No. 24 Port Phillip Bay Environmental Study. Marine and Freshwater Researches Institute, Queenscliff.

MacDonald, C. M. (1982). Life history of snapper *Chrysophrys auratus* (Bloch and Schneider, 1801) in Australian waters. pp. 16. Victorian Department of Conservation, Forests and Lands, East Melbourne.

MacDonald, C. M., Winstanley, R. H. & Hall, D. N. (1995). Commercial Fishery Situation Report; Net Fishing in Mallacoota Inlet, 1916 - 1993. pp. 60.

Mace, P. M. (1997). Developing and sustaining world fisheries resources: the state of the science and management. In 'Developing and sustaining world fisheries resources. The state of science and management. 2nd World Fisheries Congress.' (Eds D. A. Hancock, D. C. Smith, A. Grant & J. P. Beumer). CSIRO Publishing, Melbourne.

Marrs, S. J., Tuck, I. D., Atkinson, R. J. A., Stevenson, T. D. I. & Hall, C. (2002). Position data loggers and logbooks as tools in fisheries research: results of a pilot study and some recommendations. *Fisheries Research* **58**, 109-117.

Marsden, M. A. H. & Mallett, C. W. (1974). Morphology and sediment distribution, Western Port Bay, Victoria. pp. 33. Geology Department, School of Earth Sciences, University of Melbourne, Melbourne.

Maurstad, A. (2002). Fishing in murky waters - ethics and politics of research on fisher knowledge. *Marine Policy* **26**, 159-166.

May, H. M. A. & Jenkins, G. P. (1992). Patterns of settlement and growth of juvenile flounder, *Rhombosolera tapirina*, determined from otolith microstructure. *Marine Ecology Progress Series* **79**, 203-214.

Meaden, G. J. (1999). Application of GIS to Fisheries Management. In 'Marine and Coastal Geographic Information Systems.' (Ed. D. W. D. Bartlett). pp. 205-226. Taylor and Francis, London.

Meaden, G. J. & Do Chi, T. (1996). Geographical Information Systems: Applications to Marine Fisheries. pp. 335. FAO, Rome.

Monaco, M. E. & Christensen, J. D. (1997). Coupling Species Distributions and Habitat. In 'Changing Oceans and Changing Fisheries: Environmental Data for Fisheries Research and Management.' (Eds G. W. Boehlert & J. D. Schumacher). NOAA National Marine Fisheries Service, Silver Spring, MD.

Monaco, M. E., Weisberg, S. B. & Lowery, T. A. (1998). Summer Habitat Affinities of Estuarine Fish in US Mid-Atlantic Coastal Systems. *Fish. Manag. Ecol* **5**, 161-171.

Neira, F. J., Coutin, P., Morrison, S., and Hall., K. (1997). Pilchard 1996. Bay and Inlet Stock Assessment Group. Marine and Freshwater Resources Institute, Queenscliff.

Neira, F. J. & Tait, S. E. (1996). Ichthyoplankton survey in Port Phillip Bay (Victoria) 1995-96: preliminary results. Marine and Freshwater Resources Institute, Queenscliff.

Nicholls, A. G. (1973). Growth in the Australian 'salmon' *Arripis trutta* (Bloch and Schneider). *Australian Journal of Marine and Freshwater Research* **24**, 159-174.

Norcross, B. L., Blanchard, A. & Holladay, B. A. (1999). Comparison of models for defining nearshore flatfish nursery areas in Alaskan waters. *Fisheries Oceanography* **8**, 50-67.

Norcross, B. L., Holladay, B. L. & Muter, F.-J. (1995). Nursery area characteristics of pleuronectids in coastal Alaska, USA. *Netherlands Journal of Sea Research* **34**, 161-175.

Norcross, B. L., Muter, F.-J. & Holladay, B. A. (1997). Habitat models for juvenile pleuronectids around Kodiak Island, Alaska. *Fishery Bulletin* **95**, 504-520.

Odum, R. H., Ford, W. M., Edwards, J. W., Stihler, C. W. & Menzel, J. M. (2001). Developing a habitat model for the endangered Virginian northern flying squirrel (*Glaucomys sabrinus fuscus*) in the Allegheny Mountains of West Virginia. *Biological conservation* **99**, 245-252.

Officer, R. A. & Parry, G. D. (1997). Food webs of demersal fish in Port Phillip Bay. CSIRO Port Phillip Bay Environmental Study Technical Report No. 36. Marine and Freshwater Resources Institute, Queenscliff.

Officer, R. A. & Parry, G. D. (2000). Effects of season, size, depth and time of day on diets of demersal fish in Port Phillip Bay. Report No.11. pp. 68. Marine and Freshwater Resources Institute, Queenscliff.

Parry, G. D., Hobday, D. K., Currie, D. R., Officer, R. A. & Gason, A. S. (1995). The distribution, abundance and diets of demersal fish in Port Phillip Bay. CSIRO Port Phillip Bay Environmental Study Technical Report 21. pp. 107. Marine and Freshwater Resources Institute, Queenscliff.

Paulin, C. D. (1990). *Pagrus auratus,* a new combination of the species known as "snapper" in Australasian waters (Pisces: Sparidae). *New Zealand Journal of Marine and Freshwater Research* **24**, 259-265.

Paxton, J. R., Hoese, D. F., Allen, G. R. & Hanley, J. E. (Eds) (1989). Zoological catalogue of Australia: Pisces. Petromyzontidae to carangidae. AGPS, Canberra. 664 pp.

Pearce, J. & Ferrier, S. (2001). The practical value of modelling relative abundance of species for regional conservation planning: a case study. *Biological Conservation* **98**, 33-43.

PMA (1987). Sediments of Port Phillip Bay. pp. 47. Marine Model Laboratory, Port of Melbourne Authority, Melbourne.

Pulliam, H. R. (2000). On the relationship between niche and distribution. *Ecology Letters* **3**, 349-361.

Quinn, G. P. & Keough, M. J. (Eds) (2002). Experimental design and data analysis for biologists. Cambridge University Press, 537 pp.

Ramm, D. C. (1986). An ecological study of the ichthyoplankton and juvenile fish of the Gippsland Lakes, Victoria. Thesis, University of Melbourne

Rigby, B. A. (1984). The ecology of fish inhabiting estuarine seagrass habitats in the Gippsland Lakes, Victoria. Ph.D. Thesis, University of Melbourne, Melbourne.

Robertson, A. I. (1977). Ecology of juvenile King George whiting *Sillaginodes punctatus* (Cuvier and Valenciennes) (Pices: Perciformes) in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* **28**, 35-43.

Robertson, A. I. (1982). Population dynamics and feeding ecology of juvenile Australian salmon (*Arripis trutta*) in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* **33**, 369-375.

Roloff, G. J. & Kernohan, B. J. (1999). Evaluating reliability of habitat suitability index models. *Wildlife Society Bulletin* **27**, 973-985.

Roob, R., Morris, P. & Werner, G. (1998). Victorian Marine Habitat Database: Seagrass of Corner Inlet and Nooramunga. Marine and Freshwater Resources Institute, Queenscliff.

Rubec, P. J., Bexley, J. C. W., Norris, H., Coyne, M. S., Monaco, M. E., Smith, S. G. & Ault, J. S. (1999). Suitability modeling to delineate habitat essential to sustainable fisheries. *American Fisheries Society Symposium* **22**, 108-113.

Rubec, P. J., Christensen, J. D., Arnold, W. S., Norris, H., Steele, P. & Monaco, M. E. (1998). GIS and modeling: coupling habitats to Florida fisheries. *Journal of Shellfish Research* **17**, 1451-1457.

Shapiro, M. A. (1975). Western Port Bay Environmental Study 1973-1974. pp. 581. Ministry for Conservation, Melbourne.

Shaw, M. & Jenkins, G. P. (1992). Spatial variation in feeding, prey distribution and food limitation of juvenile flounder *Rhombosolea tapirina* Gunther. *Journal of Experimental Marine Biology and Ecology* **165**, 1-21.

Smith, D. C. & MacDonald, C. M. (1997). King George whiting 1996. Compiled by the Bay and Inlet Fisheries Stock Assessment Group. Fisheries Victoria Assessment Report No. 15. pp. 18. Fisheries Victoria, East Melbourne.

Soniat, T. M. & Brody, M. S. (1988). Field validation of a habitat suitability index model for the American oyster. *Estuaries* **11**, 87-95.

Stanley, C. A. (1978). Area of distribution, movements, age composition and mortality rates of the Australian salmon population in Tasmania, Victoria and New South Wales. *Australian Journal of Marine and Freshwater Research* **29**, 417-433.

Stanley, C. A. (1980). Australian Salmon. Fisheries Situation Report No. **5**. pp. 11.

Stanley, C. A. & Malcolm, W. B. (1977). Reproductive cycles in the eastern subspecies of the Australian salmon, *Arripis trutta marginata* (Cuvier and Valenciennes). *Australian Journal of Marine and Freshwater Research* **28**, 287-301.

Starr, R. M. & Fox, D. S. (1997). Can fishery catch data supplement research cruise data? A geographical comparison of research and commercial catch data. In 'Developing and sustaining world fisheries resources. The state of science and management. 2nd World Fisheries Congress.' (Eds D. A. Hancock, D. C. Smith, A. Grant & J. P. Beumer). CSIRO Publishing, Melbourne.

Stoner, A. W., Manderson, J. P. & Pessutti, J. P. (2001). Spatially explicit analysis of estuarine habitat for juvenile winter flounder: combining generalized additive models and geographic information systems. *Marine Ecology Progress Series* **213**, 253-271.

Tabachnick, B. G. & Fidell, L. S. (Eds) (1989). Using multivariate statistics. Harper and Row Publishers, New York. 746 pp.

Tanaka, M. (1985). Factors affecting the inshore migration of pelagic larval and demersal juvenile red sea bream *Pagrus major* to a nursery ground. *Trans. Am. Fish. Soc.* **114**, 471-477.

Terrell, J. W. (Ed.) (1984). Proceedings of a workshop on fish habitat suitability index models. Biological Report 85(6). pp. 393. US Department of the Interior, US Fish and Wildlife Service, Division of Biological Services, Washington D.C.

Terrell, J. W., McMahon, T. E., Inskip, P. D., Raleigh, R. F. & Williamson, K. L. (1982). Habitat suitability models: Appendix A. Guidelines for riverine and lacustrine applications of fish HSI models with the habitat evaluation procedures. pp. 54. US Department of the Interior, Fish and Wildlife Service, Washington DC.

Thomas, J. A. & Bovee, K. D. (1993). Application and testing of a procedure to evaluate transferability of habitat suitability criteria. *Regulated Rivers Research and Management* **8**, 285-294.

Thomson, J. M. (1957). The food of Western Australian estuarine fishes. Fisheries Research Bulletin No. 7. pp. 16. Western Australian Department of Fisheries, Fisheries Bulletin, Perth.

Vadas, R. L. J. & Orth, D. J. (2001). Formulation of habitat suitability models for stream fish guilds: do the standard methods work? *Transactions of the American Fisheries Society* **130**, 217-235.

Williams, A. & Bax, N. J. (2001). Delineating fish-habitat associations for spatially based management: an example from the south-eastern Australian continental shelf. *Marine and Freshwater Research* **52**, 513-536.

Wilson, G. (Ed.) (1986). Snapper. When, where and how to catch them. A guide to Port Phillip and Westernport Bay. Geoff Wilson, Geelong.

Winstanley, R. (1983). The food of snapper *Chrysophrys auratus* in Port Phillip Bay, Victoria. pp. 14. Victorian Department of Conservation, Forests and Lands, Fisheries and Wildlife Service, Melbourne.

WPRPCC (1992). Western Port Bay Strategy: A Strategy for the Protection and Development of Western Port. pp. 151. Western Port Regional Planning and Coordination Committee, Cranbourne.

Yugovic, J., Jaremovic, R., Costello, C. & Meredith, C. (1993). Land Conservation Council Marine and Coastal Special Investigation: Terrestrial Component of Biological Resources. Biosis Research Pty Ltd, Hawthorn.

Zeiler, M. (Ed.) (1999). Modeling Our World. The ESRI Guide to Geodatabase Design. ESRI Press, California.

Zheng, X., Pierce, G. J. & Reid, D. G. (2001). Spatial patterns of whiting abundance in Scottish waters and relationships with environmental variables. *Fisheries Research* **50**, 259-270.

11. Appendix 1: Intellectual Property

Intellectual property arising from this project resides in the following components of the project:

1. Habitat suitability models from fishery independent data analysis or Port Phillip Bay and Western Port (Section 5).
2. ArcView Habitat Suitability Modelling Application – customised modelling interface for ArcView 3.3 and Spatial Analyst (Section 7).
3. Habitat suitability models from fishery dependent data analysis for Port Phillip Bay (Section 6).

12. Appendix 2: Staff

The following PIRVic staff participated in this study.

Mr David Ball

Dr Liz Morris

Dr Jeremy Hindell

Mr Sean Blake

Mr Allister Coots

13. Appendix 3: Temperature and Salinity Data Analysis

13.1. Port Phillip Bay Data

We had access to the following three datasets that included salinity and temperature for Port Phillip Bay:

1. Environmental Protection Authority (EPA) fixed site water quality monitoring data. Includes six fixed-site monitoring points sampled seasonally and an additional 28 beach-monitoring sites sampled mainly in Spring, Summer and Autumn (Figure 13.1).
2. PIRVic fixed-site monitoring data. Includes six fixed-site monitoring sites (Figure 13.1) sampled seasonally for seven years (Longmore *et al*. 1996).
3. Water quality transects undertaken as part of the Port Phillip Bay Environmental Study (Longmore *et al.* 1996b). Measures included nutrients, salinity and temperature readings taken monthly along continuous random transects throughout the Bay during the period 1993 – 1995.

In order to analyse salinity and temperature variations in Port Phillip Bay it was necessary to divide the Bay into regions or zones. We elected to use the segments defined by the State Environment Protection Policy (SEPP) - Waters of Port Phillip Bay 1975 (Figure 13.1). These represent environmental zones within the Bay based on hydrodynamic and water quality influences and also correspond to the distribution of fixed-site water quality monitoring undertaken by the EPA.

13.1.1. Analysis of Fixed-site Monitoring

The EPA and PIRVic fixed-site monitoring data were combined for this analysis to provide an improved spatial coverage from which to describe patterns in temperature and salinity across Port Phillip. In order to test whether salinity and temperature differed between these fixed sites we used a two-factor ANOVA with site and season as fixed factors. In this case we did not analyse the data by segment because not all segments included a fixed monitoring site, but all four seasons were included in this analysis and the range of salinity and temperature measured at each site was also calculated. We used mean values at each site for each year and the range measured in each season in each year as replicates for this analysis.

In order to test for differences between segments, we pooled all of the fixed-site data and the EPA beach monitoring data. Three of the segments did not include winter data so we excluded all winter data from this analysis. There were considerable differences in the numbers of samples that contributed to the summary values in each segment, as there were only two sites in each of the Altona and Central segments (Figure 13.1), so these mean values will be less precise than those for other segments. Replicates for seasons were the mean value for the season for each year and each annual value for season represents an independent replicate in this two-factor ANOVA. Segment and season were treated as fixed factors.

13.1.2. Analysis of Transect Dataset

The transect dataset consisted of salinity and temperature readings taken monthly along random transects throughout the Bay during 1993 - 1995. For the purposes of our analysis we split the Bay into the EPA segments and calculated a mean value per segment per transect. Data has also been coded by season to see whether any differences between segments were dependent on season.

There were 25 transects and all segments, except Corio, had 25 replicate mean values. The random transects passed through the Corio segments less frequently than the other segments, so we had to omit the Corio segment from the analysis because there were not enough replicates for each season.

This dataset was considered separately to the fixed-site data due to the different nature of the data collection procedures and the fact that we were able to include all four seasons in the analysis.

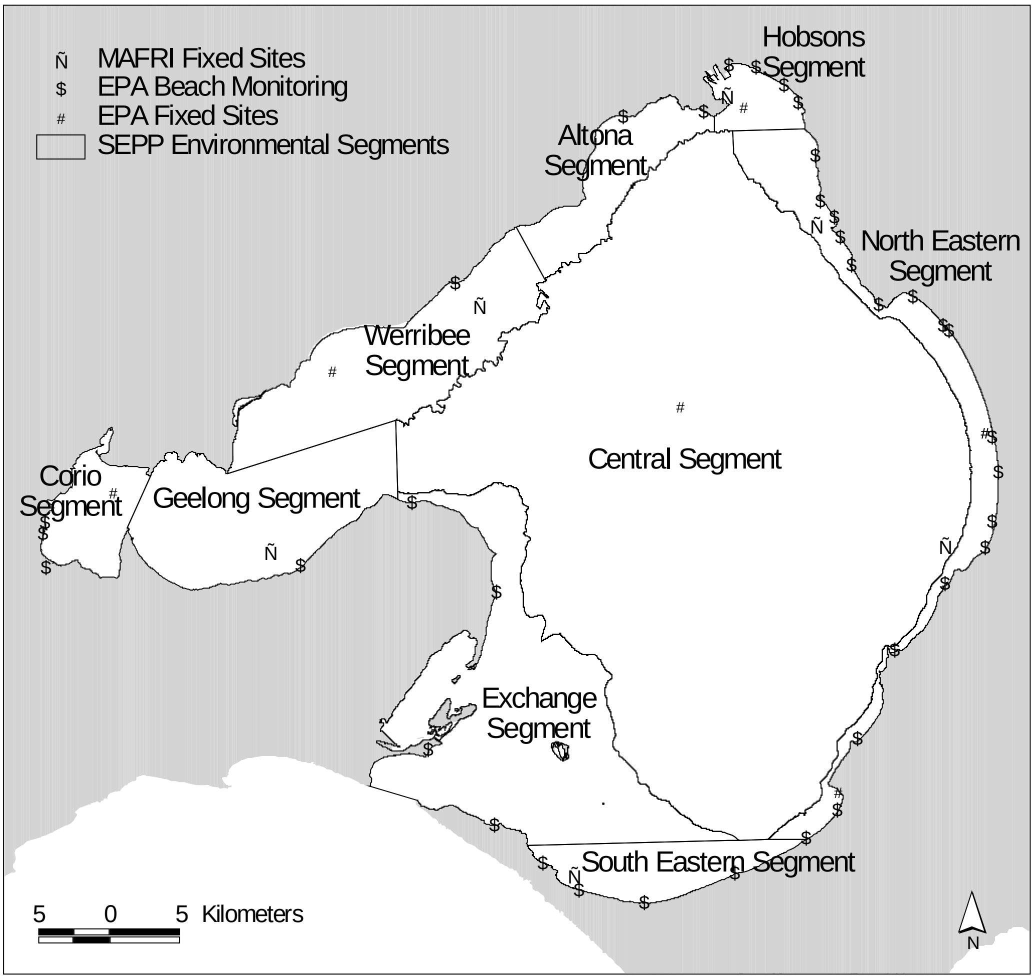


Figure 13.1. Port Phillip environmental segments (SEPP - The Waters of Port Phillip Bay 1975) and water quality monitoring sites. Sites identified as MAFRI fixed sites are now PIRVic sites.

13.2. Western Port Data

The Western Port Environmental Study (Shapiro 1975) divided the bay into six zones which were considered to represent the natural subdivisions of the Bay and detailed sampling was carried out over a four-year period. Initially, 48 stations were sampled for a range of water quality parameters including salinity and temperature, but this expanded to 62 stations and 26 shore locations (Harris *et al*. 1979). This data has only recently been analysed statistically by Longmore (1997) who suggested that for water quality monitoring, only two segments should be used. The majority of the freshwater input is in the north eastern part of the Bay, although the Bass River that enters the south east of the Bay also accounts for a reasonable proportion of the freshwater input (Harris & Robinson 1979). During summer there is very low rainfall in the Western Port catchment and the combination of low freshwater inputs and high temperatures can result in high evaporation rates and hypersaline waters in the shallow northeastern region (Harris and Robinson 1979). Temperatures can also be more variable in these shallow areas when compared to the more stable Bass Strait waters (Shapiro 1975).

Recent data for Western Port was only available from the fixed-site monitoring undertaken by the EPA. This dataset consists of three sites sampled during the years 1990 – 2000 (Hastings, Barralliar Island and Corinella). While data collection was seasonal, there were too many missing data points to include season in the analysis. Instead, a mean value for both temperature and salinity was extracted for each year and these values comprised the replicates in a one-way analysis of variance to test for differences between sites. The range of temperature and salinity recorded at each site for each year was also analysed using the yearly range as replicates in the analysis.

Where appropriate, data were transformed to meet the assumptions of the ANOVA (ie. normality and homogeneity of variance) for all of the analyses reported here.

13.3. Results

13.3.1. Port Phillip Bay Temperature

Analysis of EPA and PIRVic fixed-site data

There were significant differences in the mean temperature between sites ($F_{11,276}$=4.11, P<0.0001) and seasons ($F_{3,276}$=775, P<0.0001), but there was not a significant site by season interaction ($F_{27,276}$=1.22, P=0.2020). While the mean temperature differed by 8°C between summer and winter, differences in mean temperatures between sites differed by only 1.6°C. Despite the relatively large differences in mean temperatures experienced seasonally in the Bay, the lack of a significant interaction between site and season indicates that these seasonal differences in mean temperature are showing the same pattern at all of the fixed monitoring sites (Figure 13.2).

The range of temperatures were significantly different between sites ($F_{11,223}$=8.11, P<0.0001) and seasons ($F_{3,223}$=62.13, P<0.0001), but there was no significant site by season interaction ($F_{27,223}$=0.72, P=0.8699). Differences between mean temperature range experienced at the different sites were small in magnitude (maximum difference of 2.2°C). The smallest mean range in temperatures was observed in winter and the largest mean range of temperatures was observed in autumn (Figure 13.3). The seasonal differences in temperature ranges were consistent at all sites.

Analysis of pooled data-sets by segment (excluding winter)

Mean temperature recorded from each segment varied inconsistently between seasons ($F_{16,760}$=3.39, P<0.0001), but mean temperatures were highest in summer and lowest in spring (Figure 13.4). Differences between sites were small in each season, with the most pronounced differences between the Altona and Central segments in spring and autumn. The pattern of mean temperatures across segments differed slightly in summer compared to the other seasons, with the Exchange segment having a lower temperature compared to other segments in summer (Figure 13.4). The Exchange segment is subject to more influence from Bass Strait and the coastal waters of Victoria do not show the same large seasonal fluctuation as the Bay water. Mean temperatures also differed by a greater amount between spring and summer in the Werribee and Central segments, but the magnitude of these differences were small (Figure 13.4) and unlikely to be of biological importance. Furthermore, the general pattern of seasonal differences was very similar in all segments (Figure 13.4).

The range of temperatures also differed between segments, with the patterns of these differences dependent on season ($F_{16,662}$ =4.44, P<0.001). Temperatures varied by a greater amount in summer (up to 9°C at Altona) and differences between segments were of greater magnitude in summer, particularly for the Altona and Central segments, which had the highest (7.7°C) and lowest (3.26°C) mean temperature ranges respectively (Figure 13.5). In autumn, patterns in the average range of temperatures were similar to those observed in spring. The Geelong and Werribee segments experienced relatively high fluctuations in temperatures (5.77°C and 5.15°C respectively) in all three seasons (Figure 13.5).

Analysis of transect dataset

There was a seasonal difference in mean temperatures throughout the Bay ($F_{3,165}$=263.13, P<0.001), with temperatures in summer and autumn considerably higher than winter and spring. There was no difference in mean temperatures between segments ($F_{7,165}$=0.37, P=0.92), nor was there a significant segment by season interaction ($F_{21,165}$=0.44, P=0.986). Temperature range (square root transform) did differ between segments ($F_{7,165}$=18.12, P<0.001) as well as season ($F_{3,165}$=8.19, P<0.001) and there was no indication that differences in temperature range between segments were dependent on season ($F_{21,165}$=1.36, P=0.144). Temperature ranges experienced within the segments were greatest in the Exchange segment, but the difference between maximum and minimum temperatures experienced on any one transect was less than 2°C and unlikely to be of any biological importance (Table 13.1).

Table 13.1. Mean values per segment for temperature and the range of this parameter measured in each segment. Data are presented as mean values plus or minus one standard deviation.

Segment	Mean temperature (+/-s.d.)	Temperature range (sqrt +/- s.d.)
Geelong	15.53 +/-4.06	0.77 +/-0.23
Corio	17.38 +/-4.90	0.64 +/-0.28
Werribee	15.43 +/-4.09	0.99 +/-0.24
Altona	15.42 +/-4.02	0.73 +/-0.26
Hobsons	15.61 +/-4.02	1.12 +/-0.40
Northeast	15.97 +/-3.80	0.85 +/-0.36
Southeast	15.39 +/-3.35	0.79 +/-0.27
Exchange	15.10 +/-2.97	1.42 +/-0.48
Central	15.61 +/-3.85	1.30 +/-0.30

13.3.2. Port Phillip Bay Salinity

Analysis of EPA and PIRVic fixed-site data

Mean salinity differed between site ($F_{11,276}$=31.8, P<0.001) and between season ($F_{3,276}$=18.32, P<0.0001) but there was not a significant site by season interaction ($F_{33,276}$=1.07, P=0.366). The Williamstown and Hobsons Bay sites had the lowest mean salinity (31.26 ppt and 32.87 ppt) and Sandringham, the northeast site and the Wooley Reef site also had slightly lower salinity than the other sites (33.09 ppt, 33.87 ppt and 33.9 ppt respectively). The highest mean salinity was recorded from the Corio site (35.38 ppt) and so despite the significant differences between sites, the magnitude of these differences was small (Figure 13.6).

Mean salinity was lowest in spring and highest in autumn but the difference between seasons was only 1.29 ppt. The pattern of seasonal differences in mean salinity was consistent across all sites (Figure 13.6).

The patterns in the ranges of salinity experienced were dependent on site and season ($F_{33,224}$=1.92, P=0.0031). While for the majority of sites the greatest temperature ranges were experienced in summer, at the Williamstown, Sandringham and Wooley Reef sites the greatest temperature ranges were experienced in spring (Figure 13.7).

Analysis of pooled data-sets by segment (excluding winter).

There was a significant interaction in mean salinity values between segment and season ($F_{16,724}$=1.85, P=0.023) and while interactions make the interpretations of main effects difficult, mean salinity did tend to be lower in spring than the other seasons and the Hobsons Bay and the Northeastern segments appeared to experience lower salinity than the other segments. While the pattern of seasonal differences seemed very similar at all sites (Figure 13.8) it was more marked at the Hobsons and Northeastern segments. Overall however, differences in mean salinity recorded across all sites and seasons were small (Figure 13.8).

The patterns of difference in the ranges of salinity (log transformed) experienced were also dependent on both segment and season ($F_{16,648}$=1.93, P=0.016). The Hobsons Bay segment experienced the greatest range in salinity (Figure 13.9), which is not surprising as the Yarra River discharges into this segment. The

Northeastern and the Altona segments experienced greater ranges in salinity than the other segments. The Hobsons Bay and Northeastern segments differed from the other segments in that the salinity range in spring was more similar to that experienced in summer (Figure 13.9). This pattern was evident in the Central segment but there was a relatively low salinity range recorded from this segment overall.

Analysis of transect data-set

Mean salinity differed between segments and seasons ($F_{7,165}$=16.36, P<0.001 and $F_{3,165}$=8.41, P=0.002) but the differences between segments did not depend on the season ($F_{21,165}$=0.83 P=0.961). The Hobsons Bay segment had the lowest mean salinity and the Corio segment had the highest, although the magnitude of these differences was only around 4 ppt (Table 13.2). Examination of the residuals indicated that the assumption of homogeneity of variance was not met, with variance increasing as mean salinity decreased. This is not surprising, as areas within the influence of freshwater inputs such as Hobsons segment are more likely to experience high variability in salinity regimes.

The analysis of the square-root transformed ranges of salinity also showed that there were significant differences in the range of salinity experienced in each segment ($F_{7,165}$=20.93, P<0.001) and season ($F_{3,165}$=3.80, P=0.011), but there was not a significant segment by season interaction ($F_{21,165}$=0.77, P=0.75). This further emphasises the point that segments with lower mean salinities, e.g. the Hobsons Bay segment, also experienced a greater salinity range than all other segments.

Table 13.2. Mean values per segment for salinity and the range of these parameters experienced in each segment. Data is presented as mean values plus or minus standard deviations.

Segment	Mean salinity (+/- s.d.)	Salinity range (sqrt +/- s.d.)
Geelong	34.20 +/-0.69	0.53 +/-0.17
Corio	35.28 +/-0.72	0.49 +/-0.62
Werribee	33.87 +/-0.96	0.90 +/-0.70
Altona	33.54 +/-1.43	0.76 +/-0.66
Hobsons	31.41 +/-2.57	2.38 +/-1.11
Northeast	33.17 +/-1.42	1.04 +/-0.47
Southeast	34.45 +/-0.47	0.82 +/-0.23
Exchange	34.67 +/-0.42	1.10 +/-0.39
Central	33.86 +/-0.75	1.31 +/-0.69

13.3.3. Western Port Temperature

There was no difference in the mean temperature ($F_{2,30}$=0.03, P=0.97) recorded at each site (Table 13.3) and no difference in the range of temperatures experienced at each site ($F_{2,30}$=0.94, P=0.40).

Table 13.3. Mean temperature values recorded from three sample sites in Western Port.

Site	Mean temperature (+/- s.d.)	Mean temperature range (+/- s.d.)
west	16.45 +/-1.59	9.62 +/-1.84
north	16.51 +/-1.66	10.5 +/-2.14
east	16.34 +/-1.62	10.8 +/-2.21

13.3.4. Western Port Salinity

Salinity did not vary significantly between sites ($F_{2,30}$=1.7, P=0.2) but there was a difference in the range of salinities recorded at each site ($F_{2,30}$=5.3, P=0.011). The eastern site experienced the greatest mean range in salinity, the western site experienced the lowest mean range in salinity and the northern site was intermediate between the east and the west (Tukeys test P<0.05, Table 13.4). While differences in mean salinity ranges experienced at each site do not seem particularly large, it is important to remember that they are averaged over an entire year and at times of high rainfall are likely to be pronounced.

Table 13.4. Mean salinity values recorded from the three sample sites in Western Port

Site	Mean salinity (+/- s.d.)	Mean salinity range (+/- s.d.)
west	35.48 +/-0.64	2.03 +/-1.0
north	35.26 +/-0.67	3.25 +/-1.6
east	34.83 +/-1.11	4.32 +/-2.2

13.4. Discussion

13.4.1. Port Phillip Bay

The analysis of the datasets for salinity indicated that there were differences between segments in the mean salinity experienced in each segment, but the differences in mean values tended to be small and well within the tolerance ranges identified for each species investigated in this study (Section 4). Of more interest was the difference in the ranges of salinity experienced between segments and sites, as areas with more variable salinity may provide a more stressful environment for fish.

Early life history stages may be particularly affected by changes in salinity due to their reduced mobility compared to adults and the fact that they are often found in very shallow areas. Pulses of freshwater are less likely to affect the larger life stages of bottom dwelling fish as they can move away from an area and also the changes in salinity are unlikely to affect the deeper habitats. As a consequence we coded SEPP segments (Figure 13.1) as being ‘low’, ‘medium’ or ‘high’ salinity variability for use in the habitat suitability modelling for juvenile species (Figure 3.9). The only segment coded ‘high’ was Hobsons Bay and the only segment that was coded medium was the Northeastern segment, which includes several large permanent watercourses. The remaining segments were coded as ‘low’ salinity variability. Surprisingly, there was no evidence to suggest that the Werribee segment should be coded as high or medium despite the freshwater discharges, which include two rivers as well as outfall drains from the Western Treatment Plant.

There was some evidence that temperature also differed across the Bay, but while seasonal differences were relatively large they appeared to happen consistently across the whole Bay. Similarly, there was some evidence of differences in ranges of temperatures between sites, but the magnitude of differences was very small and unlikely to be of biological importance. For this reason we did not include temperature as a GIS layer or a predictor in the habitat suitability modelling process for Port Phillip Bay.

13.4.2. Western Port

The results recorded here are consistent with those recorded by Longmore (1997) and reflect the fact that the majority of freshwater input occurs in the north and east of the bay. The net circulation around French Island is in a clockwise direction and as a consequence freshwater discharging from the northern rivers will move primarily down the eastern arm (Harris and Robinson 1979).

The analysis suggested that Western Port could be divided into three segments; an eastern segment with a high salinity variability, a northern segment with a medium salinity variability and a western/bay mouth segment with low salinity variability. However, we had insufficient fisheries independent data to effectively derive habitat suitability indices based on salinity variability as a model parameter (Section 5.2.2).

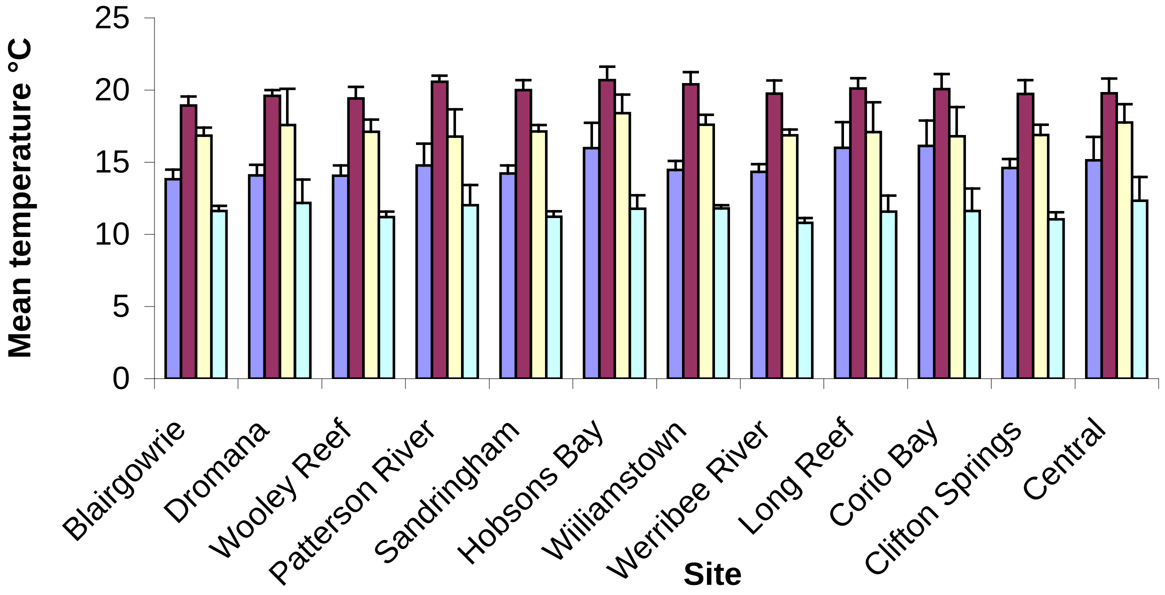


Figure 13.2. Mean temperatures recorded from the Port Phillip Bay EPA and PIRVic fixed site monitoring in each of the four seasons.

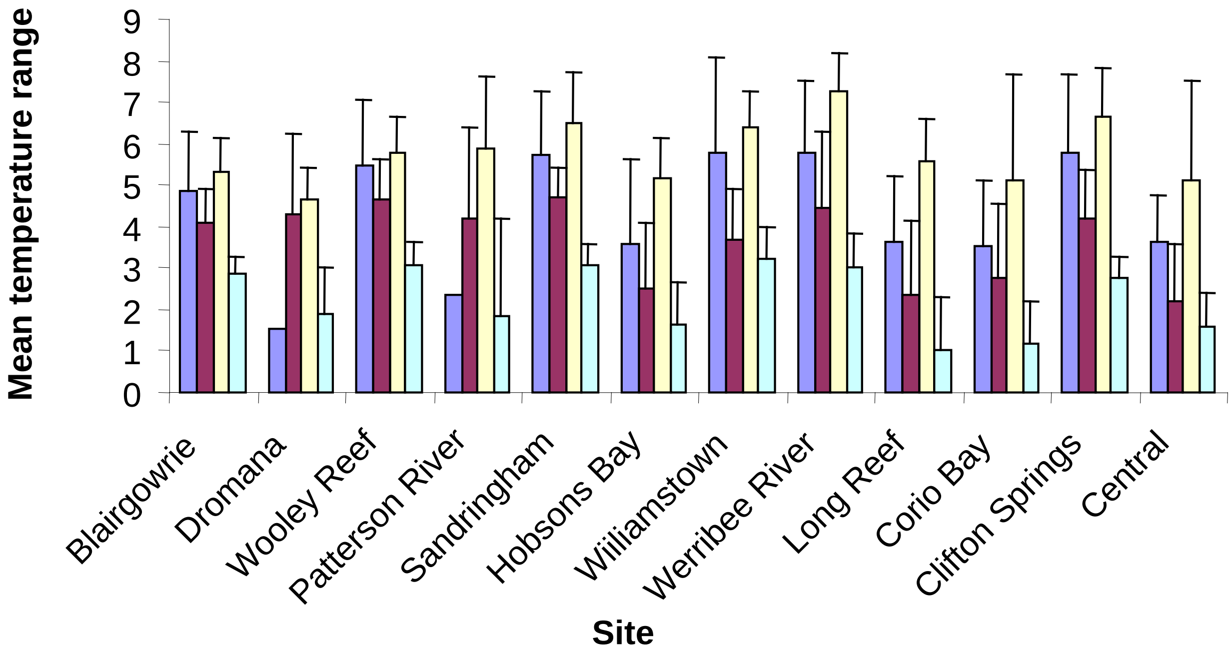


Figure 13.3. Mean temperature ranges recorded from the Port Phillip Bay EPA and PIRVic fixed site monitoring in each of the four seasons.

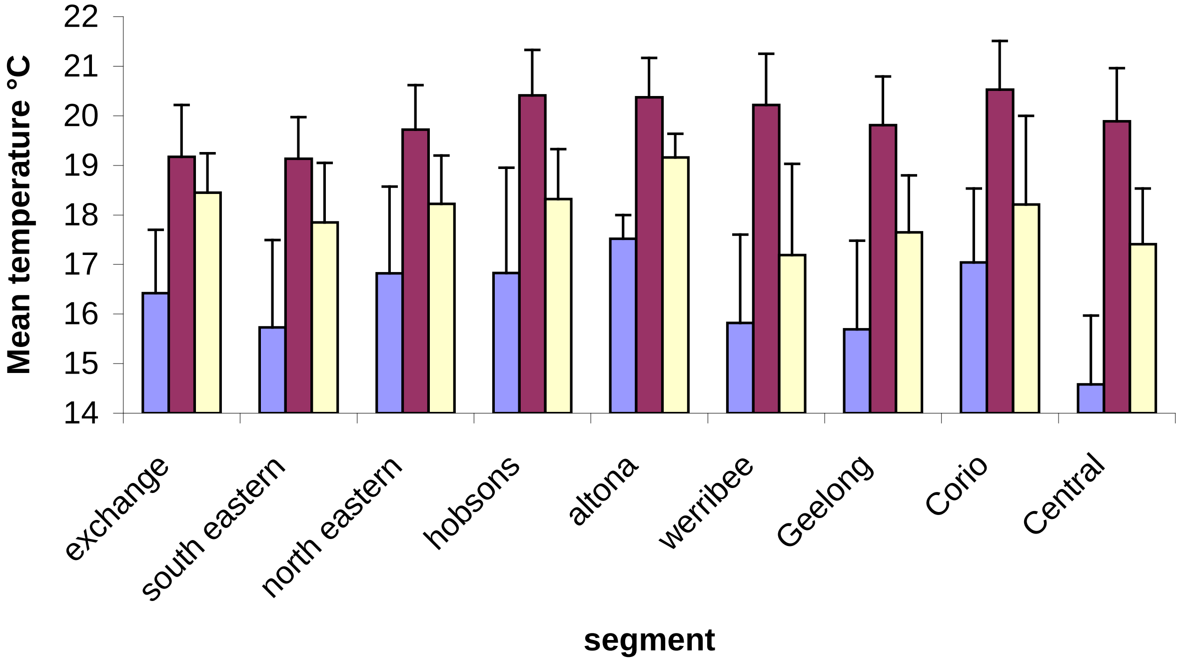


Figure 13.4. Port Phillip Bay mean temperatures recorded from each of the 9 segments in spring, summer and autumn.

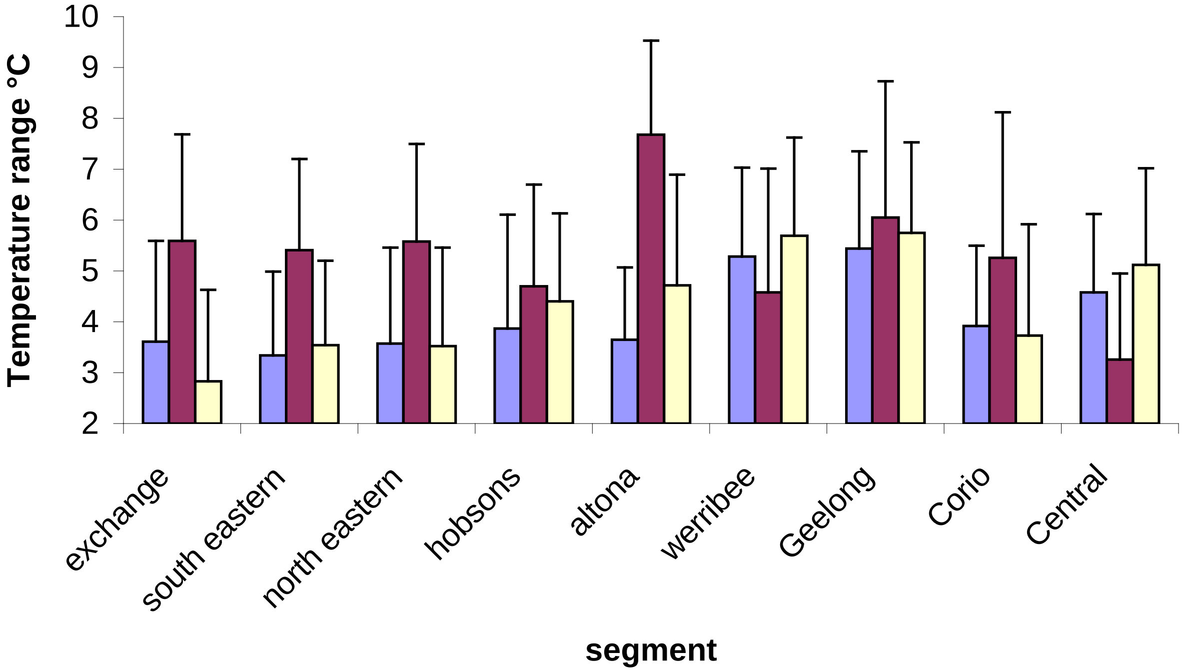


Figure 13.5. Port Phillip Bay mean temperature ranges recorded from each of the 9 segments in spring, summer and autumn.

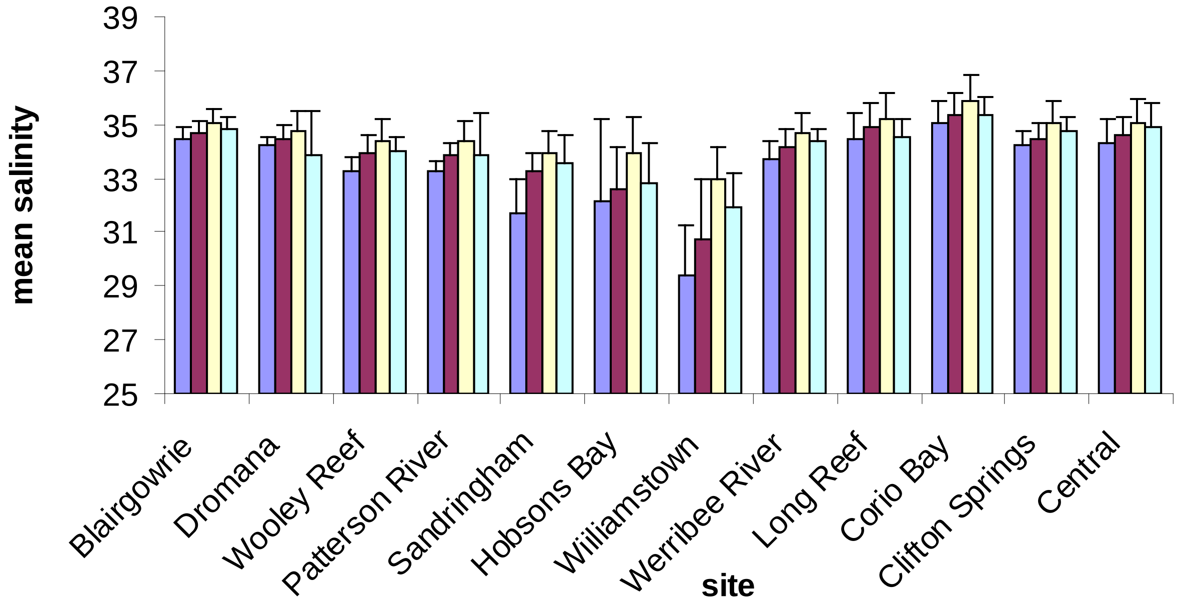


Figure 13.6. Mean salinity recorded from the Port Phillip Bay EPA and PIRVic fixed site monitoring in each of the four seasons (salinity is ppt).

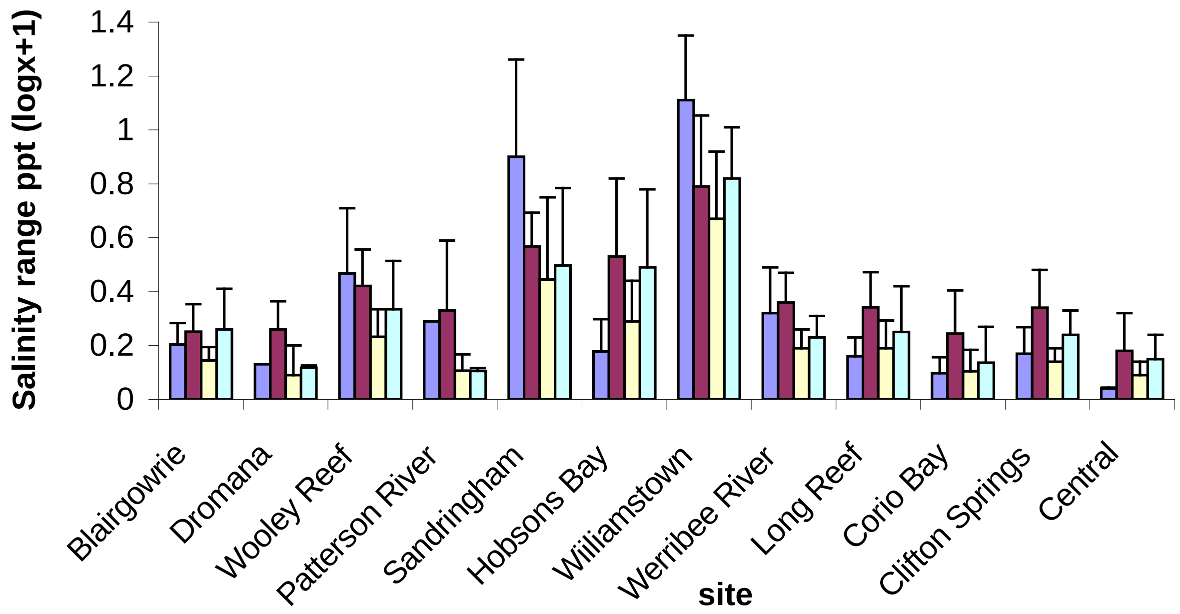


Figure 13.7. Mean salinity ranges recorded from the Port Phillip Bay EPA and PIRVic fixed site monitoring in each of the four seasons.

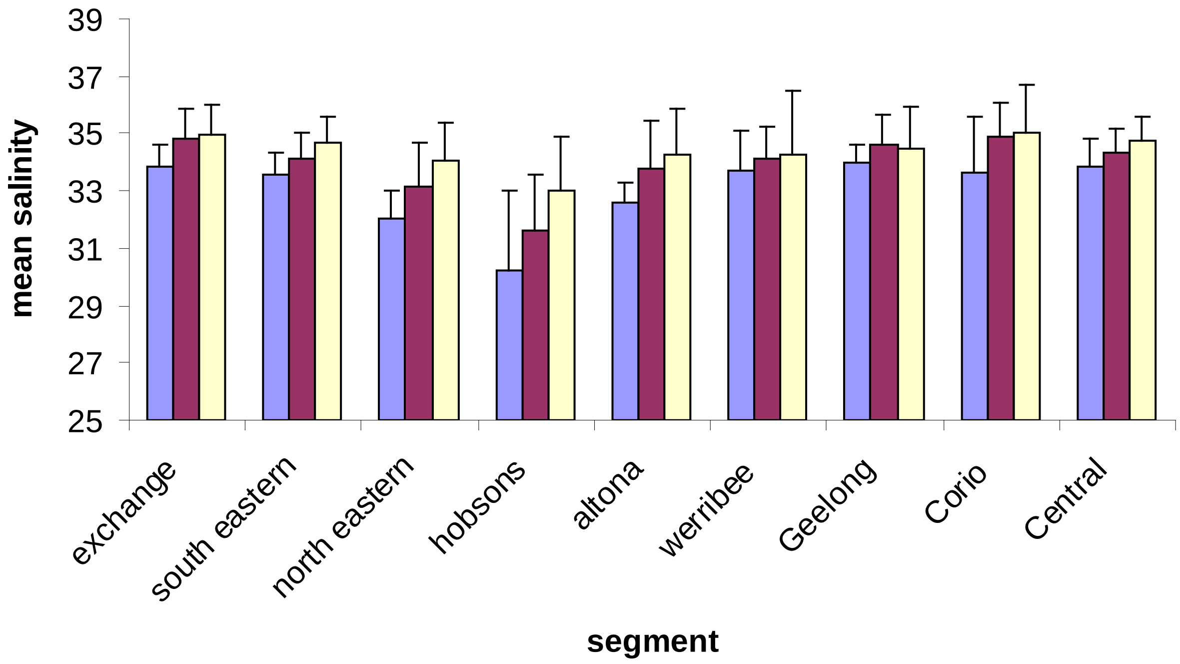


Figure 13.8. Port Phillip Bay mean salinity recorded in each of the nine segments in spring, summer and autumn (salinity is ppt).

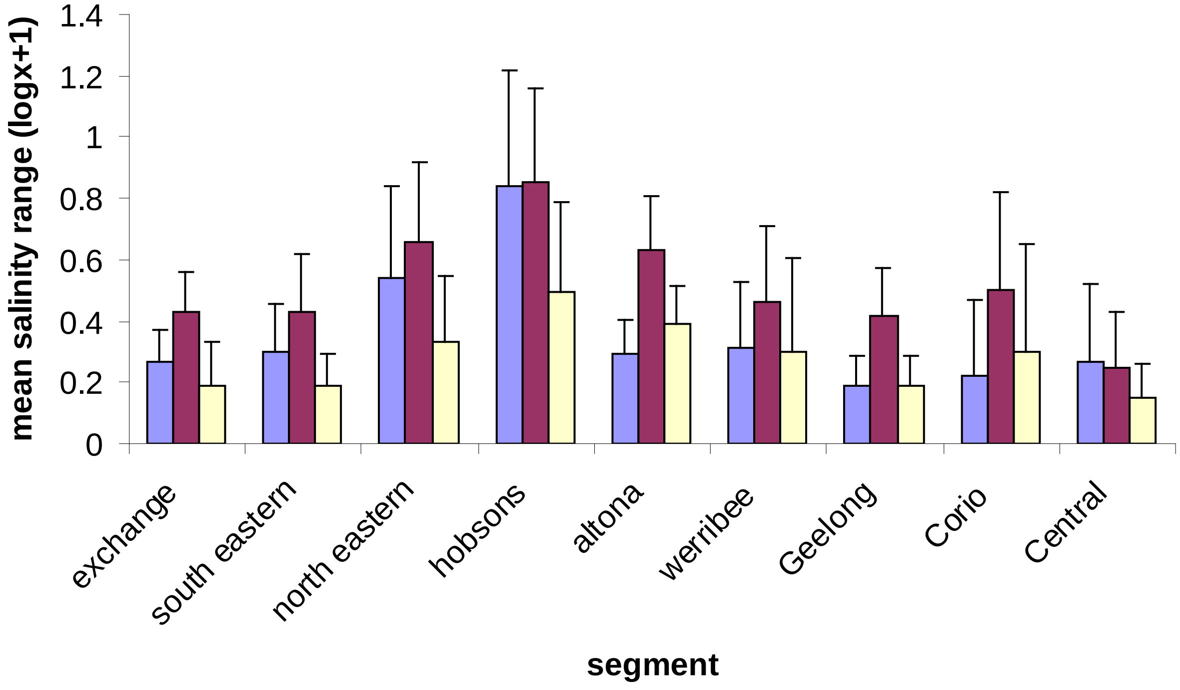


Figure 13.9: Port Phillip Bay mean salinity range recorded in each of the nine segments in spring, summer and autumn (salinity is ppt).

14. Appendix 4: Fisheries Independent Data Summary

Table 14.1. Port Phillip Bay data summary tables. (Jenkins *et al*. 1993; Parry *et al*. 1995; Jenkins *et al*. 1996a; Hamer *et al*. 1997; Officer & Parry 1997; Jenkins & Wheatley 1998; Hobday *et al*. 1999)

Sand flathead >15 cm

Category	Total area sampled (m^2)	Total fish caught	Total no. shots	Total no. presence records	Total absence records
Sediment					
Clay	1656134	27890	128	115	13
Sand Silt Clay	6711052	50617	452	287	165
Sand Clay	2576624	17635	201	137	64
Sand	1951701	5445	132	84	48
Coarse Sand	1171151	3012	80	54	26
Total	**14,066,662**	**104,599**	**993**	**677**	**316**
Depth					
0-2 m	0	0	0	0	0
3-5 m	19435	1	4	1	3
6-10 m	3888996	6904	267	161	106
11-15 m	3964346	24971	298	210	88
16-20 m	3318399	31783	256	181	75
21-30 m	2875486	40940	168	124	44
Total	**14,066,662**	**104,599**	**993**	**677**	**316**

Sand flathead <15 cm

Category	Area sampled (m^2)	Total fish caught	Total no. shots	Total no. presence records	Total no. absence records
Sediment					
Clay	35507	16	36	10	26
Sand silt clay	86175	46	94	24	70
sand clay	58600	45	57	24	33
sand	26340	10	33	6	27
coarse sand	15235	9	20	5	15
Total	**221,857**	**126**	**240**	**69**	**171**
Depth					
0-2 m	0	0	0	0	0
3-5 m	1941	1	3	1	2
6-10 m	69670	18	72	11	61
11-15 m	76518	58	88	33	55
16-20 m	51772	33	56	14	42
21-30 m	21956	16	21	10	11
Total	**221,857**	**126**	**240**	**69**	**171**

Sand flathead juveniles

Category	Area sampled (m^2)	Total fish caught	Total no. shots	Total no. presence records	Total no. absence records
Sediment					
Clay	0	0	0	0	0
Sand silt clay	0	0	0	0	0
sand clay	43050	18	557	13	544
sand	26958	1	495	1	494
coarse sand	0	0	0	0	0
Undefined	5004	0	72	0	72
Total	**75,012**	**19**	**1,124**	**14**	**1,110**
Depth					
0-2 m	7650	0	102	0	102
3-5 m	67362	19	1022	14	1008
6-10 m	0	0	0	0	0
11-15 m	0	0	0	0	0
16-20 m	0	0	0	0	0
21-30 m	0	0	0	0	0
Total	**75,012**	**19**	**1,124**	**14**	**1,110**
Substrate type/biota					
Bare	25929	19	387	14	373
Seagrass	46479	0	661	0	661
Reef	2304	0	72	0	72
Undefined	300	0	4	0	4
Total	**75,012**	**19**	**1,124**	**14**	**1,110**

King George whiting

Category	Total area sampled (m²)	total fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
Clay	1451253	449	70	15	55
Sand silt clay	6070317	326	289	24	265
sand clay	2231057	29	106	9	97
sand	1829533	311	85	9	76
coarse sand	1076892	9	49	5	44
Total	**12,659,052**	**1,124**	**599**	**62**	**537**
Depth					
0-2 m	0	0	0	0	0
3-5 m	17494	1	1	1	0
6-10 m	3627565	986	168	39	129
11-15 m	3572789	127	169	14	155
16-20 m	2710722	6	131	5	126
21-30 m	2730482	4	130	3	127
Total	**12,659,052**	**1,124**	**599**	**62**	**537**

King George whiting juveniles

Category	Total area sampled (m²)	total fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
Clay	0	0	0	0	0
Sand silt clay	0	0	0	0	0
sand clay	43825	3116	595	234	361
fine sand	48083	3305	841	309	532
medium sand	4700	1098	94	41	53
coarse sand	0	0	0	0	0
Undefined	2304	685	72	33	39
Total	**98,912**	**8,204**	**1,602**	**617**	**985**
Depth					
Intertidal	7650	309	102	41	61
0-1 m	86512	7882	1405	574	831
1-2 m	4750	13	95	2	93
Total	98,912	8,204	1,602	617	985
Substrate type/biota					
bare near seagrass	33129	4168	531	256	275
seagrass	53679	3328	805	321	484
reef	2304	685	72	33	39
bare alone	9500	16	190	4	186
undefined	300	7	4	3	1
Total	**98,912**	**8,204**	**1,602**	**617**	**985**

Greenback flounder

Category	total area sampled (m²)	total no. fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
Clay	1451253	168	70	43	27
Sand silt clay	6070317	151	289	53	236
sand clay	2231057	71	106	25	81
sand	1829533	43	85	10	75
coarse sand	1076892	8	49	6	43
Total	**12,659,052**	**441**	**599**	**137**	**462**
Depth					
0-2 m	0	0	0	0	0
3-5 m	17494	0	1	0	1
6-10 m	3627565	178	168	40	128
11-15 m	3572789	97	169	30	139
16-20 m	2710722	25	131	16	115
21-30 m	2730482	141	130	51	79
Total	**12,659,052**	**441**	**599**	**137**	**462**

Greenback flounder juveniles

Category	Total area sampled (m²)	total no. fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
clay	0	0	0	0	0
sand silt clay	0	0	0	0	0
sand clay	43525	540	591	88	503
fine sand	48383	666	845	182	663
medium sand	4700	7	94	6	88
coarse sand	0	0	0	0	0
undefined	2304	0	72	0	72
Total	**98,912**	**1,213**	**1,602**	**276**	**1,326**
Depth					
Intertidal	7650	260	102	33	69
0-1 m	86512	899	1405	223	1182
1-2 m	4750	54	95	20	75
Total	**98,912**	**1,213**	**1,602**	**276**	**1,326**
Substrate type/biota					
bare near seagrass	40129	983	531	188	343
seagrass	53679	51	805	31	774
reef	2304	0	72	0	72
bare alone	2500	178	190	56	134
undefined	300	1	4	1	3
Total	**98,912**	**1,213**	**1,602**	**276**	**1,326**

Snapper

Category	area sampled (m^2)	total no. fish caught	total no. shots	Total no. presence records	total no. absence records
Sediment					
Clay	1451253	531	70	35	35
Sand silt clay	6070317	2454	289	105	184
sand clay	2231057	909	106	35	71
sand	1829533	54	85	6	79
coarse sand	1076892	371	49	9	40
Total	**12,659,052**	**4,319**	**599**	**190**	**409**
Depth					
0-2 m	0	0	0	0	0
3-5 m	17494.22	0	1	0	1
6-10 m	3627565	915.9957	168	34	134
11-15 m	3572789	2204.998	169	65	104
16-20 m	2710722	686.0001	131	35	96
21-30 m	2730482	511.995	130	56	74
Total	**12,659,052**	**4,319**	**599**	**190**	**409**

Snapper juveniles

Category	area sampled (m^2)	total no. fish caught	total no. shots	Total no. presence records	total no. absence records
Sediment					
Clay	18114	9	19	5	14
Sand silt clay	53230	207	55	33	22
sand clay	25456	224	27	23	4
sand	16744	94	18	11	7
coarse sand	9255	38	10	9	1
Total	**122,799**	**572**	**129**	**81**	**48**
Depth					
0-2 m	0	0	0	0	0
3-5 m	1024	8	1	1	0
6-10 m	38145	95	39	21	18
11-15 m	53966	333	57	38	19
16-20 m	21349	116	23	15	8
21-30 m	8313	20	9	6	3
Total	**122,797**	**572**	**129**	**81**	**48**

Southern calamari

Category	Total area sampled (m²)	total no. fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
clay	1451253	490	70	50	20
sand silt clay	6070317	1959	289	212	77
sand clay	2231057	304	106	59	47
sand	1829533	467	85	56	29
coarse sand	1076892	516	49	38	11
Total	**12,659,052**	**3,736**	**599**	**415**	**184**
Depth					
0-2 m	0	0	0	0	0
3-5 m	17494	3	1	1	0
6-10 m	3627565	1104	168	109	59
11-15 m	3572789	893	169	103	66
16-20 m	2710722	624	131	98	33
21-30 m	2730482	1112	130	104	26
Total	**12,659,052**	**3,736**	**599**	**415**	**184**

Rock flathead

Category	area sampled (m²)	total no. fish caught	total no. shots	total no. presence records	total no. absence records
Sediment					
clay	1451253	10	70	6	64
sand silt clay	6070317	4	289	4	285
sand clay	2231057	1	106	1	105
sand	1829533	2	85	1	84
coarse sand	1076892	0	49	0	49
Total	**12,659,052**	**17**	**599**	**12**	**587**
Depth					
0-2 m	0	0	0	0	0
3-5 m	17,494	0	1	0	1
6-10 m	3,627,565	14.00112	168	9	159
11-15 m	3,572,789	3.000229	169	3	166
16-20 m	2,710,722	0	131	0	131
21-30 m	2,730,482	0	130	0	130
Total	**12,659,052**	**17**	**599**	**12**	**587**

Table 14.2. Western Port data summary tables

* Substrate type/biota data came from gill net data which includes pooled data so no figures available for number of shots or presence/absence data

Sand flathead*

Category	total area sampled (m²)	total fish caught	total no. shots	presence	absence
Depth					
15-20 m	120982	12	3	3	0
10-15 m	1017572	79	21	9	12
2-10 m	512337	5	11	5	6
0-2 m	13013	2	21	2	19
Total	**1,663,904**	**98**	**56**	**19**	**37**

Category	total averaged areas sampled (m²)	total fish caught
Substrate type/biota		
Channel	600	1150
Mangrove	160	0
Seagrass	900	225
Bare	1360	6347
Total	**3,020**	**7,722**

Calamari

Category	total area sampled (m²)	total fish caught	total no. shots	presence	absence
Depth					
15-20 m	120982	2	3	1	2
10-15 m	1017572	240	21	13	8
2-10 m	512337	187	11	8	3
0-2 m	13013	11	21	4	17
Total	**1,663,904**	**440**	**56**	**26**	**30**

Snapper

Category	total area sampled (m²)	total fish caught	total no. shots	presence	absence
Depth					
15-20 m	120982	2	3	2	1
10-15 m	1017572	35	21	10	11
2-10 m	512337	0	11	0	11
0-2 m	13013	0	21	0	21
Total	**1,663,904**	**37**	**56**	**12**	**44**

King George whiting*

Category	total averaged areas sampled (m^2)	total fish caught
Substrate type/biota		
Channel	600	150
Mangrove	160	10
Seagrass	900	225
Bare	1360	227
Total	**3,020**	**612**

Rock flathead*

Category	total averaged areas sampled (m^2)	total fish caught
Substrate type/biota		
Channel	600	1150
Mangrove	160	0
Seagrass	900	6300
Bare	1360	1247
Total	**3,020**	**8,697**

Yellow-eye mullet*

Category	total averaged areas sampled (m^2)	total fish caught
Substrate type/biota		
Channel	600	1450
Mangrove	160	950
Seagrass	900	82950
Bare	1360	61541
Total	**3,020**	**146,891**

Western Port juvenile data from all seine datasets.

Sand flathead

Category	total area sampled (m^2)	total no. fish caught	total no. shots	presence	absence
Substrate type/biota					
Channel	5833	13	83	10	73
Mangrove	1559	0	36	0	36
Seagrass	7871	1	112	1	111
Bare	11117	31	172	20	152
Total	**26,380**	**45**	**403**	**31**	**372**
Depth					
Intertidal	3118	0	72	0	72
0-2 m	15953	16	227	12	215
2-5 m	7309	29	104	19	85
Total	**26,380**	**45**	**403**	**31**	**372**

King George whiting

Category	total area sampled (m^2)	total no. fish caught	total no. shots	presence	absence
Substrate type/biota					
Channel	5833	2	83	1	82
Mangrove	1559	6	36	6	30
Seagrass	7871	29	112	13	99
Bare	11117	29	172	18	154
Total	**26,380**	**66**	**403**	**38**	**365**
Depth					
Intertidal	3118	13	72	10	62
0-2 m	15953	51	227	27	200
2-5 m	7309	2	104	1	103
Total	**26,380**	**66**	**403**	**38**	**365**

Rock flathead

Category	total area sampled (m^2)	total no. fish caught	total no. shots	presence	absence
Substrate type/biota					
Channel	5833	8	83	7	76
Mangrove	1559	0	36	0	36
Seagrass	7871	17	112	10	102
Bare	11117	9	172	7	165
Total	**26,380**	**34**	**403**	**24**	**379**
Depth					
Intertidal	3118	1	72	1	71
0-2 m	15953	25	227	16	211
2-5 m	7309	8	104	7	97
Total	**26,380**	**34**	**403**	**24**	**379**

Yellow-eye mullet

Category	total area sampled (m^2)	total no. fish caught	total no. shots	presence	absence
Substrate type/biota					
Channel	5833	0	83	0	83
Mangrove	1559	2	36	1	35
Seagrass	7871	6	112	3	109
Bare	11117	13	172	3	169
Total	**26,380**	**21**	**403**	**7**	**396**
Depth					
Intertidal	3118	2	72	1	71
0-2 m	15953	19	227	6	221
2-5 m	7309	0	104	0	104
Total	**26,380**	**21**	**403**	**7**	**396**

Greenback flounder

Category	total area sampled (m^2)	total no. fish caught	total no. shots	presence	absence
Substrate type/biota					
Channel	5833	2	83	2	81
Mangrove	1559	2	36	2	34
Seagrass	7871	1	112	1	111
Bare	11117	18	172	12	160
Total	**26,380**	**23**	**403**	**17**	**386**
Depth					
Intertidal	3118	8	72	7	65
0-2 m	15953	13	227	8	219
2-5 m	7309	2	104	2	102
Total	**26,380**	**23**	**403**	**17**	**386**

Table 14.3. Corner Inlet data summary

Sand flathead

Category	area sampled (m²)	total fish caught	total no. shots	presence	absence
Depth					
intertidal	0	0	0	0	0
0-2 m	2,473	0	2	0	2
2-5 m	9,219	9	10	5	5
5-10 m	31,199	102	29	19	10
10-15 m	16,248	75	17	12	5
15-20 m	5,804	15	6	3	3
20-30 m	1,023	1	1	1	0
Total	**65,966**	**202**	**65**	**40**	**25**

Juvenile data

King George whiting

Category	area sampled (m²)	total fish caught	total no. shots	presence	absence
Substrate type/biota					
mangrove	1,559	0	36	0	36
seagrass	14,250	5	190	4	186
bare	11,009	71	162	21	141
Total	**26,818**	**76**	**388**	**25**	**363**
Depth					
intertidal	3,118	0	72	0	72
0-2 m	23,250	76	310	25	285
2-5 m	450	0	6	0	6
Total	**26,818**	**76**	**388**	**25**	**363**

Greenback flounder

Category	area sampled (m²)	total fish caught	total no. shots	presence	absence
Substrate type/biota					
mangrove	1,559	15	36	7	29
seagrass	14,250	11	190	9	181
bare	11,009	144	162	44	118
Total	**26,818**	**170**	**388**	**60**	**328**
Depth					
intertidal	3,118	53	72	16	56
0-2 m	23,250	117	310	44	266
2-5 m	450	0	6	0	6
Total	**26,818**	**170**	**388**	**60**	**328**

Yellow eye mullet

Category	area sampled (m^2)	total fish caught	total no. shots	presence	absence
Substrate type/biota					
mangrove	1,559	20	36	1	35
seagrass	14,250	30	190	7	183
bare	11,009	35	162	6	156
Total	**26,818**	**85**	**388**	**14**	**374**
Depth					
intertidal	3,118	36	72	2	70
0-2 m	23,250	49	310	12	298
2-5 m	450	0	6	0	6
Total	**26,818**	**85**	**388**	**14**	**374**

Rock flathead

Category	area sampled (m^2)	total fish caught	total no. shots	presence	absence
Substrate type/biota					
mangrove	1,559	0	36	0	36
seagrass	14,250	35	190	28	162
bare	11,009	26	162	17	145
Total	**26,818**	**61**	**388**	**45**	**343**
Depth					
intertidal	3118	0	72	0	72
0-2 m	23250	61	310	45	265
2-5 m	450	0	6	0	6
Total	**26,818**	**61**	**388**	**45**	**343**

Yank flathead

Category	area sampled (m^2)	total fish caught	total no. shots	presence	absence
Substrate type/biota					
mangrove	1559	0	36	0	36
seagrass	14250	2	190	2	188
bare	11009	45	162	36	126
Total	**26,818**	**47**	**388**	**38**	**350**
Depth					
intertidal	3118	0	72	0	72
0-2 m	23250	47	310	38	272
2-5 m	450	0	6	0	6
Total	**26,818**	**47**	**388**	**38**	**350**

15. Appendix 5: Fishery Dependent Data Summary

Table 15.1. Summary of commercial fishery catch records used in habitat suitability modelling.

Season	Year	Gear Code*	Species	Number Of Fishing Blocks	Days	Hours	Shots	Hook-lifts	Hook-hours
Autumn	1998	AL	Australian salmon	1	1	NA	NA	0	4
Spring	1998	AL	Australian salmon	1	51	NA	NA	45	622
Summer	1998/99	AL	Australian salmon	1	33	NA	NA	0	450
Spring	1999	AL	Australian salmon	1	29	NA	NA	0	233
Summer	1999/00	AL	Australian salmon	3	22	NA	NA	8	280
Autumn	2000	AL	Australian salmon	1	22	NA	NA	0	176
Summer	2000/01	AL	Australian salmon	2	16	NA	NA	0	186
Autumn	2001	AL	Australian salmon	1	15	NA	NA	26	202
			AL Sub-total	11	189	NA	NA	79	2153
Autumn	1998	HS	Australian salmon	12	265	1862	431	NA	NA
Winter	1998	HS	Australian salmon	11	337	2216	581	NA	NA
Spring	1998	HS	Australian salmon	11	358	2745	567	NA	NA
Summer	1998/99	HS	Australian salmon	16	453	3355	1142	NA	NA
Autumn	1999	HS	Australian salmon	11	277	1966	417	NA	NA
Winter	1999	HS	Australian salmon	11	342	2440	526	NA	NA
Spring	1999	HS	Australian salmon	16	454	3179	704	NA	NA
Summer	1999/00	HS	Australian salmon	15	486	3329	771	NA	NA
Autumn	2000	HS	Australian salmon	12	428	3135	680	NA	NA
Winter	2000	HS	Australian salmon	12	259	1804	406	NA	NA
Spring	2000	HS	Australian salmon	11	313	2131	457	NA	NA
Summer	2000/01	HS	Australian salmon	10	346	2036	450	NA	NA
Autumn	2001	HS	Australian salmon	15	516	3249	732	NA	NA
			HS Sub-total	163	4834	33446	7865	NA	NA
Autumn	1998	MN	Australian salmon	11	370	2007	475	NA	NA
Winter	1998	MN	Australian salmon	9	523	3109	652	NA	NA
Spring	1998	MN	Australian salmon	8	256	1265	293	NA	NA
Autumn	1999	MN	Australian salmon	13	496	2717	566	NA	NA
Winter	1999	MN	Australian salmon	10	454	2176	545	NA	NA
Spring	1999	MN	Australian salmon	8	256	1394	291	NA	NA
Autumn	2000	MN	Australian salmon	11	285	1681	295	NA	NA
Winter	2000	MN	Australian salmon	7	263	1570	291	NA	NA
Spring	2000	MN	Australian salmon	7	160	854	173	NA	NA
Summer	2000/01	MN	Australian salmon	1	10	80	13	NA	NA
Autumn	2001	MN	Australian salmon	8	143	902	162	NA	NA
			MN Sub-total	93	3216	17755	3756	NA	NA

Table 15.1 continued

Winter	1998	AL	King George whiting	1	7	NA	NA	0	104
Spring	1998	AL	King George whiting	1	51	NA	NA	45	622
Summer	1998/99	AL	King George whiting	3	95	NA	NA	31600	59691
Autumn	1999	AL	King George whiting	4	31	NA	NA	14	442
Winter	1999	AL	King George whiting	1	22	NA	NA	44	328
Spring	1999	AL	King George whiting	1	29	NA	NA	0	233
Summer	1999/00	AL	King George whiting	4	28	NA	NA	22	388
Autumn	2000	AL	King George whiting	1	9	NA	NA	0	156
Spring	2000	AL	King George whiting	1	5	NA	NA	4	78
Summer	2000/01	AL	King George whiting	1	11	NA	NA	0	159
Autumn	2001	AL	King George whiting	1	15	NA	NA	26	202
			AL Sub-total	19	303	NA	NA	31755	62403
Autumn	1998	HS	King George whiting	19	371	2548	598	NA	NA
Winter	1998	HS	King George whiting	16	406	2712	692	NA	NA
Spring	1998	HS	King George whiting	21	419	3202	663	NA	NA
Summer	1998/99	HS	King George whiting	20	492	3578	1207	NA	NA
Autumn	1999	HS	King George whiting	19	447	3136	708	NA	NA
Winter	1999	HS	King George whiting	19	416	2745	640	NA	NA
Spring	1999	HS	King George whiting	18	470	3290	759	NA	NA
Summer	1999/00	HS	King George whiting	16	490	3354	778	NA	NA
Autumn	2000	HS	King George whiting	17	461	3350	756	NA	NA
Winter	2000	HS	King George whiting	15	261	1876	426	NA	NA
Spring	2000	HS	King George whiting	16	358	2406	563	NA	NA
Summer	2000/01	HS	King George whiting	14	392	2343	544	NA	NA
Autumn	2001	HS	King George whiting	16	539	3385	809	NA	NA
			HS Sub-total	226	5522	37924	9145	NA	NA
Autumn	1998	MN	King George whiting	23	479	2679	620	NA	NA
Winter	1998	MN	King George whiting	14	658	3955	845	NA	NA
Spring	1998	MN	King George whiting	13	361	2110	516	NA	NA
Summer	1998/99	MN	King George whiting	3	47	263	51	NA	NA
Autumn	1999	MN	King George whiting	19	552	3040	653	NA	NA
Winter	1999	MN	King George whiting	18	576	2899	692	NA	NA
Spring	1999	MN	King George whiting	15	346	1890	383	NA	NA
Autumn	2000	MN	King George whiting	15	354	2146	371	NA	NA
Winter	2000	MN	King George whiting	16	441	2661	484	NA	NA
Spring	2000	MN	King George whiting	12	232	1265	263	NA	NA
Summer	2000/01	MN	King George whiting	1	10	80	13	NA	NA
Autumn	2001	MN	King George whiting	12	189	1173	208	NA	NA
			MN Sub-total	161	4245	24161	5099	NA	NA

Table 15.1 continued

Autumn	1998	AL	Snapper	9	112	NA	NA	42200	121000
Winter	1998	AL	Snapper	1	2	NA	NA	400	1200
Spring	1998	AL	Snapper	21	306	NA	NA	94897	400440
Summer	1998/99	AL	Snapper	22	436	NA	NA	152110	614842
Autumn	1999	AL	Snapper	18	277	NA	NA	78158	390506
Spring	1999	AL	Snapper	18	211	NA	NA	60800	201632
Summer	1999/00	AL	Snapper	23	275	NA	NA	81014	264693
Autumn	2000	AL	Snapper	18	201	NA	NA	57200	214076
Spring	2000	AL	Snapper	15	138	NA	NA	49400	173267
Summer	2000/01	AL	Snapper	17	231	NA	NA	76600	230869
Autumn	2001	AL	Snapper	12	137	NA	NA	40000	126470
			AL Sub-total	174	2326	NA	NA	732779	2738995
Autumn	1998	HS	Snapper	8	255	1749	427	NA	NA
Winter	1998	HS	Snapper	8	301	1942	494	NA	NA
Spring	1998	HS	Snapper	13	374	2846	609	NA	NA
Summer	1998/99	HS	Snapper	17	472	3402	1187	NA	NA
Autumn	1999	HS	Snapper	14	420	2925	676	NA	NA
Winter	1999	HS	Snapper	9	312	2351	496	NA	NA
Spring	1999	HS	Snapper	10	369	2710	597	NA	NA
Summer	1999/00	HS	Snapper	14	379	2887	639	NA	NA
Autumn	2000	HS	Snapper	12	405	3012	687	NA	NA
Winter	2000	HS	Snapper	9	204	1544	350	NA	NA
Spring	2000	HS	Snapper	15	336	2276	541	NA	NA
Summer	2000/01	HS	Snapper	14	340	2230	476	NA	NA
Autumn	2001	HS	Snapper	13	431	2695	650	NA	NA
			HS Sub-total	156	4598	32569	7830	NA	NA
Autumn	1998	MN	Snapper	5	318	1673	407	NA	NA
Winter	1998	MN	Snapper	6	521	2983	635	NA	NA
Spring	1998	MN	Snapper	6	238	1303	285	NA	NA
Summer	1998/99	MN	Snapper	9	113	660	141	NA	NA
Autumn	1999	MN	Snapper	5	173	846	216	NA	NA
Winter	1999	MN	Snapper	8	412	1952	516	NA	NA
Spring	1999	MN	Snapper	9	287	1558	329	NA	NA
Summer	1999/00	MN	Snapper	3	35	199	45	NA	NA
Autumn	2000	MN	Snapper	8	223	1250	231	NA	NA
Winter	2000	MN	Snapper	8	332	1961	360	NA	NA
Spring	2000	MN	Snapper	6	139	699	169	NA	NA
Summer	2000/01	MN	Snapper	1	10	80	13	NA	NA
Autumn	2001	MN	Snapper	3	84	534	103	NA	NA
			MN Sub-total	77	2885	15698	3450	NA	NA

Table 15.1 continued

Autumn	1998	HS	Flounder	12	294	2045	458	NA	NA
Winter	1998	HS	Flounder	14	379	2639	662	NA	NA
Spring	1998	HS	Flounder	11	352	2757	564	NA	NA
Summer	1998/99	HS	Flounder	19	489	3566	1204	NA	NA
Autumn	1999	HS	Flounder	15	402	2901	660	NA	NA
Winter	1999	HS	Flounder	17	404	2728	632	NA	NA
Spring	1999	HS	Flounder	15	442	3114	729	NA	NA
Summer	1999/00	HS	Flounder	16	490	3354	778	NA	NA
Autumn	2000	HS	Flounder	15	448	3257	731	NA	NA
Winter	2000	HS	Flounder	15	275	1895	441	NA	NA
Spring	2000	HS	Flounder	15	353	2382	556	NA	NA
Summer	2000/01	HS	Flounder	13	377	2233	496	NA	NA
Autumn	2001	HS	Flounder	17	560	3527	847	NA	NA
			HS Sub-total	194	5265	36397	8760	NA	NA
Autumn	1998	MN	Flounder	6	345	1961	435	NA	NA
Winter	1998	MN	Flounder	7	538	3281	653	NA	NA
Spring	1998	MN	Flounder	7	229	1405	299	NA	NA
Summer	1998/99	MN	Flounder	7	104	549	110	NA	NA
Autumn	1999	MN	Flounder	8	374	2081	462	NA	NA
Winter	1999	MN	Flounder	5	265	1347	282	NA	NA
Spring	1999	MN	Flounder	6	221	1287	241	NA	NA
Summer	1999/00	MN	Flounder	2	6	17	6	NA	NA
Autumn	2000	MN	Flounder	3	105	637	108	NA	NA
Winter	2000	MN	Flounder	3	117	759	122	NA	NA
Spring	2000	MN	Flounder	4	71	364	73	NA	NA
Summer	2000/01	MN	Flounder	5	32	217	35	NA	NA
Autumn	2001	MN	Flounder	8	171	1086	189	NA	NA
			MN Sub-total	71	2578	14991	3015	NA	NA
Autumn	1998	HS	Yellow-eye mullet	16	338	2287	556	NA	NA
Winter	1998	HS	Yellow-eye mullet	13	381	2502	661	NA	NA
Spring	1998	HS	Yellow-eye mullet	13	362	2807	564	NA	NA
Summer	1998/99	HS	Yellow-eye mullet	19	489	3566	1204	NA	NA
Autumn	1999	HS	Yellow-eye mullet	14	379	2688	585	NA	NA
Winter	1999	HS	Yellow-eye mullet	12	373	2525	581	NA	NA
Spring	1999	HS	Yellow-eye mullet	15	441	3112	729	NA	NA
Summer	1999/00	HS	Yellow-eye mullet	12	411	2814	627	NA	NA
Autumn	2000	HS	Yellow-eye mullet	9	385	2858	624	NA	NA
Winter	2000	HS	Yellow-eye mullet	12	251	1751	395	NA	NA
Spring	2000	HS	Yellow-eye mullet	11	316	2182	496	NA	NA
Summer	2000/01	HS	Yellow-eye mullet	12	359	2121	468	NA	NA
Autumn	2001	HS	Yellow-eye mullet	17	554	3495	822	NA	NA
			HS Sub-total	175	5039	34706	8313	NA	NA

Table 15.1 continued

Autumn	1998	MN	Yellow-eye mullet	16	444	2408	557	NA	NA
Winter	1998	MN	Yellow-eye mullet	11	607	3636	770	NA	NA
Spring	1998	MN	Yellow-eye mullet	10	368	2045	524	NA	NA
Summer	1998/99	MN	Yellow-eye mullet	1	17	113	27	NA	NA
Autumn	1999	MN	Yellow-eye mullet	14	510	2764	589	NA	NA
Winter	1999	MN	Yellow-eye mullet	12	535	2652	643	NA	NA
Spring	1999	MN	Yellow-eye mullet	10	296	1581	335	NA	NA
Autumn	2000	MN	Yellow-eye mullet	9	270	1618	279	NA	NA
Winter	2000	MN	Yellow-eye mullet	8	315	1840	339	NA	NA
Spring	2000	MN	Yellow-eye mullet	7	150	762	163	NA	NA
Summer	2000/01	MN	Yellow-eye mullet	1	10	80	13	NA	NA
Autumn	2001	MN	Yellow-eye mullet	8	154	946	173	NA	NA
			MN Sub-total	107	3676	20445	4412	NA	NA
			Total	1627	44676	268093	61643	764613	2803551

* Gear codes: AL = all lines and hooks, MN = all mesh nets, HS = all haul seines.