Murray Cod Modelling to Address Key Management Actions

Final report for Project MD745
CONTENTS

List of tables and figures .............................................................................................................. v
Acknowledgements .......................................................................................................................... vii
Executive Summary ............................................................................................................................ viii

1 Overview of Project Brief ............................................................................................................ 1
  1.1 Project Objectives ......................................................................................................................... 1
  1.2 Project Approach and Methodology ............................................................................................ 1

2 Methodology ..................................................................................................................................... 3
  2.1 Overview of project methodology ............................................................................................... 3
    2.1.1 Murray cod biology and ecology ............................................................................................ 3
    2.1.2 A review of approaches for testing alternate management options ........................................ 3
  2.2 Workshop 1: Specialist workshop on Murray cod modelling ........................................................ 3
  2.3 Key management questions .......................................................................................................... 4
    2.3.1 Habitat and flow management questions .................................................................................. 4
    2.3.2 Trophic interaction management questions .............................................................................. 4
    2.3.3 Fishery management questions ............................................................................................... 4
    2.3.4 The modelling brief ................................................................................................................ 4
  2.4 Workshop outcomes .................................................................................................................... 5

3 Development of population models for Murray cod ................................................................. 6
  3.1 Life cycle ..................................................................................................................................... 6
  3.2 Life cycle graphs, stage and age classified matrix models for Murray cod .................................. 6
  3.3 Characteristic equations .............................................................................................................. 10
  3.4 Stochasticity .............................................................................................................................. 10
  3.5 Density dependence .................................................................................................................... 11

4 Data assessment .............................................................................................................................. 14
  4.1 Fecundity rates ........................................................................................................................... 14
  4.2 Survival and transition rates ....................................................................................................... 15
    4.2.1 Other survival parameters ..................................................................................................... 17
  4.3 Variable growth .......................................................................................................................... 17
  4.4 Expressions of risk and scenario ranking ..................................................................................... 20

5 Management scenarios ................................................................................................................... 22
  5.1 Workshop 2: Specialist workshop on Murray cod modelling ...................................................... 22
  5.2 Modelling brief .......................................................................................................................... 22
  5.3 Fishery management questions .................................................................................................. 23
  5.4 Habitat and flow management questions ..................................................................................... 24
  5.5 Other issues ................................................................................................................................ 25
  5.6 Some scenario examples ............................................................................................................. 26
    5.6.1 Modelled Current Victorian regulations ................................................................................. 26
    5.6.2 Modelled Slot size regulations of 60–100cm ....................................................................... 27
    5.6.3 Cumulative threats – multiple modelled impacts ................................................................. 28
  5.7 ESSENTIAL – modelling framework ............................................................................................ 30

6 Discussion and conclusion ............................................................................................................. 31
Appendix 1: A review of the ecological knowledge of Murray cod

A1.1 Taxonomy
A1.2 Distribution
  A1.2.1 Natural distribution
  A1.2.2 Introductions
  A1.2.3 Population decline
A1.3 Conservation Status
A1.4 Social issues
A1.5 Ecology
  A1.5.1 Habitat
  A1.5.2 Diet
  A1.5.3 Behaviour
  A1.5.4 Age and Growth
  A1.5.5 Reproduction
  A1.5.6 Recruitment
  A1.5.7 Migration and movements
  A1.5.8 Habitat selection and movement
A1.6 Threats
  A1.6.1 Flow Regulation
  A1.6.2 Habitat degradation
  A1.6.3 Reduced water quality
  A1.6.4 Barriers
  A1.6.5 Alien Species
  A1.6.6 Exploitation
  A1.6.7 Stocking and Translocations
  A1.6.8 Genetic Issues
  A1.6.9 Diseases
  A1.6.10 Climate Change

Appendix 2: A review of approaches for testing alternate management options

A2.1 Methods
A2.2 Results and Discussion
  A2.2.1 Literature search
  A2.2.2 General modelling approaches
  A2.2.3 Features of models used to test management and policy options
  A2.2.4 Examples of models used to test management hypotheses
A2.3 Conclusion

Appendix 3: Trophic interaction model for Murray cod

Appendix 4 The addition of longevity and fecundity data to the Murray cod model

A4.1 Introduction
A4.2 Methods and Results
  A4.2.1 Otoliths
  A4.2.2 Gonads
A4.3 Conclusion
A4.4 How this changes the model
Life history cycle of female Murray cod with associated estimated time transitions for each stage of development. Depending on the conditions, eggs may hatch in 6–16 days after being laid, whereas it takes up to 5 years to pass through the juvenile stage. 

Figure 2: Life cycle graph for the Murray cod with an annual time step, where: transition to the juvenile stage, $F$, represents egg, larval and fingerling or young of the year survival, juveniles surviving and remaining juveniles, $P_1$, juveniles surviving and maturing, $G$, and adults surviving, $P_2$. 

Figure 3: Life cycle graph with age-structure for juvenile female Murray cod with no age-structure for adults. 

Figure 4: Part of an age structured life cycle graph for female Murray cod. 

Figure 5: Spawner-recruits relationship, blue line with density shape parameter 1 and black line with density shape parameter 10. 

Figure 6: In circumstances where the population is under the carrying capacity (20,000 adults) and with increasing density over time, the density dependent factor acts to decrease survival proportionally. The black line is the density dependent factor for one year olds and the blue line is the density dependent factor for two year olds. 

Figure 7: In circumstances where the population is over the carrying capacity (20,000 adults) and with decreasing density over time, the density dependent factors act to reduce survival dramatically and then increase over time. The black line is the density dependent factor for one year olds, the blue line is the density dependent factor for two year olds, brown three and four year olds, and green five, six and seven year olds. 

Figure 8a: Survival rate estimates from the analysis of mark-recapture data for the given size class. 

Figure 8b: Fishing rate estimates from the analysis of mark-recapture data for the given size class. 

Figure 9: Parametric analysis of age data to fit a survival curve. 

Figure 10: Observed variation length at age. Red line is the mean estimate of length given age and the blue lines represent upper and lower plausible bounds. 

Figure 11: Some examples of the probability of being in a size class, given age. 

Figure 12: Minimum population size risk curves (risk increases as the risk curve shifts to the left and risk decreases as risk curves shift to the right). 

Figure 13: Minimum population size risk curves for a modelled Murray cod population with an average carrying capacity of 20,000 adults, where the green line is the no fishing risk curve; the blue line is the risk curve for the former regulations with a take rate of 10%; and the red line is the risk curve for the former regulations with a take rate of 20%.
Figure 14: Minimum population size risk curves for a modelled Murray cod population with an average carrying capacity of 20,000 adults, where the green line is the no-fishing risk curve; the blue line is the risk curve for the former regulations with a take rate of 10%; and the red line is the risk curve for the former regulations with a take rate of 20%. Dashed lines are the corresponding risk curves associated with increasing the minimum size to 60 cm and protecting all fish over 100 cm.

Figure 15: Figure 15: Minimum population size risk curves for a modelled Murray cod population for scenarios 1–6: 1) dashed blue line; 2) orange line; 3) brown line; 4) mauve line; 5) olive line; and 6) yellow line.

Figure A1.1: Components of the life cycle of Murray cod. SWH = structural woody debris; CVD variation in depth; OHV = overhanging vegetation; DNB = distance to bank (after Koehn 2006).

Figure A1.2: Timing of key components of the life cycle of Murray cod. Dashed line indicates that access is dependent on flows; dotted line indicates that some fish remain in the lake. Arrows indicate extended periods for some fish (after Koehn 2006).

Figure A2.1: Alternate research strategies when using models a) general exploration of ecological phenomena b) testing management options.

Figure A2.2: Steps in the assessment and management of a population. The block letters and solid arrows indicate the usual approach. Italic letters indicate the passive adaptive approach and the open arrows indicate the active adaptive approach (after Collie and Walters 1993).

Figure A3.1: A schematic Murray cod food web. Arrows denote energy fluxes and their direction. Only Murray cod is in the >90 cm size group. Murray cod, trout cod and carp are in the 60–90 cm size group. The 30–60 cm, the 10–30 cm and the <10 cm size groups include trout cod, golden perch, and silver perch. In addition, Murray cod, carp and redfin are in the 30–60 cm size group and Murray cod, gudgeons, smelt, hardyheads and gobies are in the <10 cm size group. Secondary producers are zooplankton and macroinvertebrates.

Figure A3.2: A signed digraph of the schematic Murray cod food web. Links terminating in an arrow denote positive effects, while links terminating in a filled circle denote negative effects. Links that connect a variable to itself denote self-regulation (negative) feedbacks. 1, >90 cm size group; 2, 60–90 cm size group; 3, 30–60 cm size group; 4, juvenile Murray cod; 5, 10–30 cm size group; 6, <10 cm size group; 7, secondary producers; 8, primary producers.

Figure A4.1: Relationship between Murray cod total length and estimated age. The red squares indicate the taxidermist data and the blue triangles data from the lower Murray River (courtesy Brenton Zampatti, SARDI).

Figure A4.2: A sectioned otolith from an Ovens River Murray cod (780 mm long) estimated at 12+ years old.

Figure A4.3: Ovary (left) from a large Murray cod with 110,000 eggs. Photo to right is microscopic image of the mature eggs.
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EXECUTIVE SUMMARY

This project has developed a population model for Murray cod to assess impacts of threats and recovery options. The model and management scenarios to be explored were developed through a highly inclusive and consultative workshop process that involved modellers, fish ecologists, fisheries scientists and management personnel. It was agreed that a stochastic population model for Murray cod would be the most applicable for the key management actions currently needed.

It was determined that the modelling should investigate three particular general management areas: fishery management; habitat and flow management; and trophic interactions. While the first two areas are covered extensively, trophic interactions have only been dealt with in an introductory manner due to a lack of available data and information. Discussion on this aspect does, however, provide a basis for which this area can further be explored.

Structured population models provide a quantitative link between the individual and the population, with the model being built around a simple description of the Murray cod life cycle. It incorporates demographic and environmental stochasticity, density dependence and variable growth and was built following an assessment of the available data, in particular: fecundity, growth and survival rates. Outputs from the model are expressed in terms of risk to the population. The model structure includes a wide range of parameters that address particular population impacts and allow management scenarios to be explored for both site specific applications as well as for the larger scale. Parameters in the model include a suite of recreational fishing regulations; stocking rates; habitat changes; fish kills; thermal pollution; larval mortality due to weirs; and larval losses via irrigation off-takes.

Modelled management scenarios for Murray cod indicate that the risk to populations can be reduced substantially by appropriate changes to management actions. In particular, changes to the size limits on angler take can have a major impact on population persistence. The implementation of a slot size that protects both smaller and larger fish reduced population risk considerably. Our results support recent changes toward introduction of a legal take slot size of 60–100cm by some states. The recognition of the cumulative impacts of co-occurring threats is important when assessing risk to populations. When operating together they can contribute to impact substantially on risk and population numbers. Importantly, the inclusion of these parameters in the model allows these threats to be assessed on a site by site basis.

Active adaptive policy appears crucial in managing stock-recruitment systems. This involves a range of stakeholders and relationships, many of which have been facilitated for Murray cod through the formation of the Recovery Team and the Murray cod Taskforce and through this modelling process. Management arrangements need to be further formalised, with consideration given to formal reviews of progress. The outputs from modelling management scenarios, together with additional data collection and inputs will provide a solid basis for improved management of Murray cod populations which will assist in ensuring its sustainability. Changes in angling regulations provide an opportunity to test predicted outcomes through appropriate monitoring.
1. OVERVIEW OF PROJECT BRIEF

The Native Fish Strategy for the Murray-Darling Basin provides a response to the key threats to its native fish populations [Murray-Darling Basin Commission 2004a]. These include flow regulation, habitat degradation, lowered water quality, man made barriers to fish movement, the introduction of alien fish species, fisheries exploitation, the spread of diseases and translocation and stocking of fish. Native fish populations in the Basin’s rivers have declined under these threats with experts estimating that current population levels are about 10 per cent of those at pre-European settlement [Murray-Darling Basin Commission 2004a]. The goal of the Native Fish Strategy is to rehabilitate native fish communities in the Basin back to 60 per cent of their estimated pre-European settlement levels after 50 years of implementation. The Native Fish Strategy has been developed and will be implemented within the context of the Murray-Darling Basin Commission’s Integrated Catchment Management Policy to ensure that the Basin sustains viable fish populations and communities throughout its rivers. This policy reflects a current commitment by the community and governments to manage and use the resources of the Basin in an ecologically sustainable manner. The Native Fish Strategy also calls for further research on native and alien fish ecology, especially work that will improve management actions.

This project seeks to develop a population model for Murray cod to assess impacts of threats and recovery options. In particular this work would help clarify alternative options for management in relation to size and bag limits and potential recovery times from over fishing and fish kills. This will also help identify the type of monitoring necessary to support changing management actions as well as assessing populations under threat.

1.1 Project Objectives

The objectives of this project are to:

- Develop a computer model (or models) to represent the population dynamics of Murray cod populations under alternative management options.
- Develop various management scenarios in relation to size and bag limits and potential recovery times from over fishing, fish kills and other management or environmental scenarios which may affect Murray cod populations.
- Document the findings of this work, and the implications for developing management options for Murray cod and the research on Murray cod biology and ecology required for improving the model (or models).

1.2 Project Approach and Methodology

The project tasks to be developed by the consultant as part of the bid development process to address the key objectives above, include:

1. Participate in an inception meeting to understand the Commission’s (as represented by the Project Steering Committee) rationale for the project and its expectations in terms of approach, inputs required and outcomes sought;

2. Review and summarise the relevant scientific, management, angler and aquaculture literature on:
   - Murray cod biology and ecology;
   - Management options for Murray cod and similar fish in the Murray-Darling Basin and elsewhere; and,
   - Population and other [climate and GIS] models for fish or other fauna which will allow alternative management options to be tested;

3. Develop conceptual models of Murray cod biology and ecology and identify information gaps required to populate these models;

4. Identify data availability and quality and fill information gaps from literature review or a workshop with specialists to develop algorithms for each aspect of the life history which affect distribution and population levels of Murray cod, including but not limited to:
- Age-growth models
- Stock recruitment models
- Population models

Document the limitations, data gaps and degree of uncertainty as a result;

5. Choose or develop an appropriate model, set of models or model structure to accurately represent Murray cod ecology and environmental heterogeneity of the Murray-Darling Basin Rivers;

6. Develop a range of management scenarios or strategies which affect Murray cod reproduction and survival and flow management strategies;

7. Meet with Commission’s Project Steering Committee to review data, models and the range of management scenarios to test;

8. Apply the model or set of models to test the management scenarios;

9. Prepare a draft report, distribute to project steering committee members and meet to discuss the report. The Commission will provide feedback on the draft report in order to develop a final report. This feedback will be based upon the comments from the project steering committee, Murray-Darling Basin Commission staff and partners, and, potentially, an independent reviewer;

10. Finalise the report (addressing feedback) which documents the findings of this project, and implications thereof, for management scenarios to ensure Murray cod populations remain viable across their current range within the Murray-Darling Basin.

This project will help meet objective 1 of the National Murray Cod Recovery Plan (National Murray cod Recovery Team 2007): Determine the distribution, structure and dynamics of Murray cod populations across the Murray-Darling Basin. This is more explicitly defined in Action 1.12.: Develop appropriate decision support tools and models that allow the future management actions for Murray cod to be evaluated within a risk management framework. This is a ‘High’ priority action for the recovery plan.
2. METHODOLOGY

2.1 Overview of project methodology

Population models vary on a continuum from simple to complex, depending on the management question to be addressed and the availability of information to parameterise them. Given the variety of possible management concerns outlined in the project brief, and the range of models available to consider, it was decided that 3 initial steps were required.

1. A project inception meeting to understand the Commission’s rationale for the project and its expectations in terms of approach, inputs required and outcomes sought.
2. A review of the literature on Murray cod will be conducted to ensure an up to date ecological understanding and appropriate data sources.
3. A workshop to address uncertainty in: biological and ecological knowledge; data; management; environmental; resource exploitation/extraction; and ecological theory.

Reviews of relevant literature were undertaken for this project and are presented in the Appendices.

2.1.1 Murray cod biology and ecology

The biology and ecology of Murray cod have been studied over many years [see Appendix 2]. Much of the life history of Murray cod is now reasonably well understood, however, there remain some key data gaps that require further research. For example, reproductive capacity is not well understood for large fish, and specifically, there are no absolute quantitative relationships between size or age of a fish and their reproductive capacity. Determination of the proportion of adults that make reproductive inputs to a population is not known. While spawning is known to occur regularly, the key drivers to recruitment remain somewhat uncertain. The ecology of adult Murray cod in terms of habitat selection and movements is also reasonably well understood, although this is less certain for juvenile fish. Both intra- and inter-specific competition for resources have not been explored and ecosystem carrying capacities are not known.

2.1.2 A review of approaches for testing alternate management options

Population modelling can be a valuable tool in assessing the management of a species of interest [see Appendix 3], particularly when detailed information is lacking and attaining such information may take years or decades (Christensen and Walters 2004). Models explore the effects of life history such as migratory behaviour (Jager 2006a; b), growth rate (Bearlin et al. 2002) or environmental variability (Brown and Walker 2004) on persistence. This information can be used to choose between management options when they are explicitly tested and can also indicate new management options not previously considered. In addition, sensitivity analysis can identify the life history stages critical for population growth and can guide conservation actions which concentrate on these stages (Kareiva et al. 2000).

The review of modelling approaches and consultation with management agencies identified inclusive workshops as the most appropriate approach for the development of the model, the management scenarios and its testing. As a consequence, two workshops were undertaken: the first to develop the modelling approach and management scenarios; and the second to refine the scenarios and test the applicability of the model and its efficacy of use by managers.

2.2 Workshop 1: Specialist workshop on Murray cod modelling

A workshop was held in May 2007 at the Arthur Rylah Institute for Environmental Research [Department of Sustainability and Environment: Heidelberg] to determine the best approach for developing a population model(s) for the management of Murray cod. The aim of the workshop was to bring together a range of technical experts and jurisdictional representatives [SA, QLD, VIC, NSW, ACT and Commonwealth] to determine the key management actions to address the sustainable management of Murray cod as well as the knowledge requirements necessary to develop the appropriate model(s) to assess the key management actions.

The specific workshop objectives were to address uncertainty in: biological and ecological knowledge; data; management; environmental; resource exploitation and/or extraction; and ecological theory.
The workshop was attended by: Paul Brown (DPI, Victoria), Changhao Jin (DSE, ARI, Victoria), John Koehn (DSE, ARI, Victoria), Mark Lintermans (MDBC, ACT), Simon Nicol (SPC, New Caledonia), Roger Pech (Landcare Research, New Zealand), Bill Phillips (Mainstream Consulting), David Ramsey (DSE, ARI, Victoria), Stuart Rowland (DPI, NSW), Andrew Sanger (DPI, NSW), Charles Todd (DSE, ARI, Victoria), Terry Walker (DPI, Victoria), Karen Weaver (DPI, Victoria), Glenn Wilson (UNE, NSW), Ross Winstanley (Angler Representative, Vic), Qifeng Ye (SARDI, SA) and Brenton Zampatti (SARDI, SA).

2.3 Key management questions
The workshop identified the 3 general management areas that the management model was required to address:
1. Habitat and flow management
2. Trophic interaction management
3. Fishery management

2.3.1 Habitat and flow management questions
Specific issues were discussed relating to habitat and flow management. These included changes that occur to a population with:
- an increase in habitat;
- an improvement to riparian vegetation;
- a reduction of the impacts of cold water pollution;
- provision of environmental flows at differing times, durations, etc.;
- the installation of fishways along the Murray River; and
- the establishment of Habitat Management Areas.

2.3.2 Trophic interaction management questions
Specific issues were discussed relating to trophic interaction management, including:
- whether increasing the Murray cod population (an apex predator) would alter other native and introduced fish populations;
- whether differing rates of stocking would alter the natural population;
- whether reducing redfin and carp populations would alter the cod population; and
- similarly for other invasive species.

2.3.3 Fishery management questions
Specific issues were discussed relating to fishery management, including the changes that occur to a population with:
- changes to the minimum legal length;
- changes to/setting the maximum legal length;
- changes to the bag or possession limits;
- variations in fishing effort/rate;
- alterations to the timing or duration of closed seasons;
- reducing illegal take;
- banning set lines; and
- establishment of ‘no take’ areas.

2.3.4 The modelling brief
A workshop recommendation was that modelling address, and develop where possible, specific scenarios to account for:
- impacts of latitude/climate on populations (spatial variation)
- impacts of differing recreational fishery ‘rules’ in each jurisdiction
• minimum and maximum sizes, bag/possession limits, closed seasons, etc.
• factoring in the impacts of illegal take
• factoring in the impacts of stocking programs
• trophic interactions (interspecific and intraspecific).
• habitat restoration impacts (including flows)
• management units – spatial considerations

2.4 Workshop outcomes

Much discussion was given to uncertainty in: biological and ecological knowledge; data [especially take rates]; management; environmental changes; resource exploitation/extraction; ecological theory; and modelling approaches.

While knowledge is rarely perfect, management decisions are constantly made with imperfect knowledge on the relevant subject. Models are a simplification of reality and are therefore incomplete and imprecise, however they are very useful in the decision making process due to the ability to identify important timeframes, data requirements, key sensitivities and whether a management decision is robust when allowing for uncertainty. Modelling also assists in identifying conflicts and emerging issues, testing hypotheses and evaluating management actions prior to implementation. See Appendix 2 for a literature review of population and other models used to test alternate management options and thus aid management and conservation of fish.

Three possible modelling approaches were proposed as useful frameworks in which management questions could be tested. These include stock assessment, population viability analysis and trophic models. All of which were considered plausible contenders to assess key management questions.

General consensus was achieved on the need for the model to include population structure and the ability to assist in ranking management actions for their impact on population persistence. Population Viability Analysis style modelling [stochastic population model] uses the life history of the organism of interest to structure the model, and can include age-size structure while providing output on comparative risks of management actions [considered to be highly desirable]. Scenarios addressing habitat and flow management questions can be suitably included as well as fishery management questions [including stock recruitment questions]. There exists some data to parameterise a stochastic population model [such as estimated survival] from age data and some fecundity data. However, quantitative information on early life history, [egg, larval and young of the year survival] can only be inferred. If suitable data exist, spatial variation can be represented either through different survival or growth patterns or structurally via a metapopulation construction. There is no information on variability in survival or fecundity or what might drive this variation. Trophic interaction questions can be modelled, however limited quantitative information is available to parameterise a trophic interaction model which are generally highly data demanding to parameterise appropriately. Furthermore interactions are not well understood. In addition, depending on the management question/action being considered, a trophic interaction model may be of limited applicability. For specific management questions relating to trophic interactions it is appropriate to use a trophic web model. Given the limitations, the effect of perturbations in a trophic web model can be predicted using a number of techniques such as loop analysis [Dambacher et al. 2002; 2003] or fuzzy logic in fuzzy cognitive maps [Ramsey and Veltman 2005]. The trophic interaction model can be used to identify key knowledge gaps and research needs. The exploration of a qualitative trophic web interaction model and assessment of the feasibility for its use in resource management for Murray cod is undertaken in Appendix 3.

In summary it was agreed that a single species stochastic population model for Murray cod would be broadly applicable to the key management actions currently available. A range of likely or possible management scenarios impacting on cod biology and ecology have been identified and the uncertainties associated with each management scenario have been broadly documented. Additional data holdings have been identified, as well as gaps in the data sets that limit the modelling approaches. Finally, it was agreed that a risk based framework for ranking management actions/scenarios was the appropriate output from the model or models.
3. DEVELOPMENT OF POPULATION MODELS FOR MURRAY COD

3.1 Life cycle

Individual organisms are born, grow, mature, reproduce and ultimately die. The likelihood that any one of these events occurs within a particular time period depends upon the environment that the individual inhabits and the evolutionary adaptation of the individual to its environment. Life cycle analysis descriptively translates the individual to the population level. The likelihoods that determine the population-level rates of birth, growth, maturation, fertility and mortality, are collectively described as the vital rates, and it is these vital rates that determine the dynamics of a population. Structured population models provide a quantitative link between the individual and the population, built around a simple description of the life cycle.

The population projection matrix is a $k \times k$ matrix $A$ (also called a square matrix) whose elements are made up of the vital rates describes the population projection matrix. The state of the population at time $t$ is described by the vector $n(t)$ (a single column matrix), whose entries $n_i(t)$ give the numbers of individuals in each age or stage class described by the life cycle graph. If the elements of the projection matrix $A$ do not change with time, then the population dynamics can be simply represented by the linear model

$$ n(t+1) = An(t) \quad (1) $$

If the population dynamics vary with time (environmental variation) and vary with the population itself (density dependence or negative feedback through overuse of a resource), then the population dynamics are represented by a system of inhomogeneous nonlinear equations, for which no analytic solution may exist,

$$ n(t+1) = A_{nt} n(t) \quad (2) $$

However, the analytic solutions to equation (1) provide useful information about some of the population dynamics arising from equation (2). Eigenvalues provide the complete dynamic information from the solution to a set of static, algebraic equations, such as equation (1). Given the $k \times k$ matrix $A$ and a non-zero population vector $n(t)$ then there exists a scalar $\lambda$ such that $An(t) = \lambda n(t)$, where $\lambda$ known as the eigenvalue. As the matrix $A$ is a $k \times k$ matrix then there exist $k$ values for $\lambda$, of which the largest positive value is equivalent to the finite rate of increase (or growth rate of increase) of the population model. The characteristic equation of the matrix $A$ is

$$ \det (A - \lambda I) = 0 \quad (3) $$

See Manly (1990), Saila et al. (1991), Burgman et al. (1993), and Caswell (2001) for greater detail on population modelling using matrix modelling techniques particularly for wildlife management.

3.2 Life cycle graphs, stage and age classified matrix models for Murray cod

Murray cod have an estimated life span of approximately 50 to 60 years with a maximum recorded weight of 113.5 kg (Allen, 1989). Age at sexual maturity is thought to be between 4 and 7 years for females and 3 and 6 years for males (Koehn and O’Connor 1990) although Rowland (1998a; b) reports all females to be sexually mature by age 5. Murray cod lay demersal, adhesive eggs on hard substrates with larger females producing more than 60 000 eggs (Cadwallader and Backhouse 1983), and possibly up to 200 000 (Lake, 1967a). Murray cod eggs begin to hatch around six days after spawning; however, in lower temperatures eggs begin to hatch around ten days after spawning (Rowland 1988b) and continue to hatch for a further six days (Cadwallader and Gooley 1985). Murray cod larvae are well developed upon hatching and have the capacity to both feed and move immediately. After approximately another 20 days the egg sac is consumed and larvae are now considered juvenile fish. There are four life stages of Murray cod as shown in the life cycle in Fig. 1.
3. Development of population models for Murray cod

Figure 1: Life history cycle of female Murray cod with associated estimated time transitions for each stage of development. Depending on the conditions, eggs may hatch in 6–16 days after being laid, whereas it takes up to 5 years to pass through the juvenile stage.

Even though there are different temporal scales associated with each stage of development, the appropriate unit of time to consider for a population model of Murray cod is annual time steps, particularly given that breeding/recruitment is an annual event. The structure of the model depends on both biological and ecological understanding and available data. For example if no specific adult age survival rates exist then, in keeping with the principle of parsimony, adult survival would be treated the same no matter what age. A stage-structured model with annual time steps is depicted in Fig. 2.

Figure 2: Life cycle graph for the Murray cod with an annual time step, where: transition to the juvenile stage, $F$, represents egg, larval and fingerling or young of the year survival, juveniles surviving and remaining juveniles, $P_1$, juveniles surviving and maturing, $G$, and adults surviving, $P_2$.

The associated projection matrix corresponding to Fig. 2. is:

$$
\begin{pmatrix}
P_1 & F \\
G & P_2
\end{pmatrix}
$$

where $P_1$ is the proportion of juveniles that survive and do not mature, $G$ is the proportion of juveniles that survive and mature, $P_2$ is the proportion of adults that survive and $F$, the fecundity rate, is the proportion of eggs, larvae and fingerlings that survive. However, the matrix in equation (4) does not satisfy the modelling brief, particularly with regard to modelling the impacts of a recreational fishery, as it presents some difficulties in terms of summarising recruitment as the number of eggs produced varies significantly between different sized and aged fish.
There is a long history of age-structured models being used to facilitate the analysis of exploited fish populations (Megrey 1989), and in more recent years including length-based age-structure (Fournier et al. 1998; Hampton and Fournier 2001). In developing an age-structured model for Murray cod, both the life history and management questions were used to guide the process. A simple age-structured model for female Murray cod, with adult survival not based on age, is presented in Fig. 3.

![Life cycle graph with age-structure for juvenile female Murray cod with no age-structure for adults.](image)

The life cycle graph shown in Fig. 3 only requires 6 rates to be estimated, age specific survival for juveniles, survival for adults and fecundity or recruitment to one year olds. The corresponding projection matrix is

$$
\begin{pmatrix}
0 & 0 & 0 & 0 & F \\
G_1 & 0 & 0 & 0 & 0 \\
0 & G_2 & 0 & 0 & 0 \\
0 & 0 & G_3 & 0 & 0 \\
0 & 0 & 0 & G_4 & P \\
\end{pmatrix}
$$

(5)

This construct has similar limitations to equation (4). For example, some fishing regulations are expressed as specific components of the population, (eg. size limits) and this structure does not allow for specific exploitation scenarios to be examined.

From limited data, and through analysing variable growth, 98% of Murray cod 25 and older are greater than 1m in length [S. Morison and G. Gooley unpubl. data]. Female fish greater than 1m are thought to make a significant contribution to reproductive capacity of the population, where anecdotal evidence points to larger numbers of eggs being produced by large fish (200 000 eggs, Lake 1967a; >120 000 eggs, J.D. Koehn and I. Stewart unpubl. data). An age-structured model for female Murray cod requires the adult stage to be disaggregated into the age classes. However, given the maximum possible age as well as possible management scenarios about fishing regulations it was decided to disaggregate adults from ages 5 to 24 and leave adults 25 years old and older aggregated as the final stage. The associated life cycle graph is presented in Fig. 4 and is represented by the projection matrix in equation (6).
Figure 4: Part of an age structured life cycle graph for female Murray cod.
The age-structured matrix requires estimates of age-based survival rates and age-based fecundity or recruitment to one year olds. Provided size-at-age and fecundity-at-age are known then this construct is able to address most of the key management questions and in particular can be used to examine the specific scenarios relating to fishing regulations and the potential changes to the regulations.

3.3 Characteristic equations

Solving the characteristic equation provides the dominant eigenvalue, which is equivalent to the geometric growth rate. The characteristic equation for the projection matrix in equation (6) is

\[ \lambda^{25} - P_{25} \lambda^{24} - \lambda^{20} F_1 \prod_{j=4}^{24} G_j + \sum_{j=5}^{25} (F_{j-1} P_{25-j} - F_{j} G_{j-1}) \lambda^{24-j+1} \prod_{j=4}^{24-j+1} G_j = 0 \]  

(7)

3.4 Stochasticity

Stochastic population modelling uses Monte Carlo simulation where random numbers are generated from distributions describing variation in parameters. The purpose is to determine how random variation, lack of knowledge, or error affects the sensitivity, performance, or reliability of the predictions [Wittwer 2004]. Monte Carlo simulation is categorised as a sampling method as the inputs are randomly generated from probability distributions to simulate the process of sampling from an actual population [Wittwer 2004]. Including mechanistic descriptions of demographic and environmental variation into an underlying projection matrix construct produces a stochastic population model. Demographic stochasticity is modelled by allowing for variation in the survival and reproduction of individuals [Akçakaya 1991] and is incorporated by using a binomial distribution to model the number of individuals surviving between consecutive time steps, and a Poisson distribution to model recruitment (Todd et al. 2005). Environmental stochasticity is modelled by randomly selecting survival and fecundity rates from specified distributions for each time step (Todd and Ng 2001).

A variate \( X \) with parameters \( a, b \) and \( c \) that has a given parametric distribution, Dist, is denoted \( X \sim \text{Dist}(a,b,c) \) [Berry and Lindgren 1996] where some or all of the parameters may be omitted [Evans et al. 1993]. In the following equations, the expressions a Bin \((N,G)\) and Poisson \((GN)\) refer to random variates where the random variate Bin \((N,G)\) has a binomial distribution with survival \( G \) of \( N \) individuals, e.g. Bin \((N,G)\) = \( X \sim \text{Bin}(N,G) \), and the random variate Poisson \((GN)\) has a Poisson distribution with a mean and variance \( GN \), e.g. Poisson \((GN)\) = \( Y \sim \text{Poi}(GN) \).
3.5 Density dependence

There are difficulties in detecting density-dependent relationships (Hassell 1986; Gaston and Lawton 1987; Rose and Cowan 2000; Yearsley et al. 2003) even when appropriate data are available (Hilborn and Walters 1992; Burgman et al. 1993; Maunder 1997; for example, long time series data), let alone when appropriate data are limited or non-existent. However, the effects of density-dependence may still be included in the model through exploration of the species’ life history (Gaston and Lawton 1987; Hilborn and Walters 1992; Burgman et al. 1993; Rose and Cowan 2000). Todd et al. (2004) explored a number of density dependent constructs for trout cod. Contest competition (the unequal division of resources) is typically modelled by the Beverton–Holt function (Ricker 1975; Table 2) and is compensatory in that rates of increase in recruitment will diminish as the population increases and will always allow some recruitment. Scramble competition (the equal division of resources) is typically modelled by the Ricker function (Ricker 1975; Table 2), however, it is over compensatory, and may lead to recruitment failure for large population sizes. A top down approach to modelling density dependence occurs where resources are allocated to older fish first, and allowances made for small juvenile fish to exploit alternative resources. Scramble competition is likely to be the incorrect mechanism in which to model recruitment, particularly as larvae are the most vulnerable life stage with little mobility and are unlikely to be able to access resources equally in a heterogeneous environment. The exploration undertaken by Todd et al. (2004) concluded that the Ricker function did not capture the dynamics thought to be associated with recruitment in trout cod. Given that trout cod are closely related to Murray cod (trout cod was only identified as a separate species to Murray cod in 1972: Berra and Weatherly 1972), and that similar ecological drivers are likely to affect recruitment for both species, the Ricker function was considered inappropriate for modelling density-dependence.

The Beverton–Holt function is generally applied to affect the strength of recruitment only (one year olds in an age-structured model) and in stage-structured modelling may not capture other levels of density-dependence relating to the interaction of subsequent stages (older age classes). Applying the Beverton-Holt function to larvae produces sufficient negative feedback to constrain population growth. The strength of the negative feedback can be modified to yield either relatively constant recruitment or pulse like recruitment with a number of years between recruitment events (Fig. 5). Such characteristics are desirable when modelling regional differences between the lower reaches of the Murray River and other areas (Brenton Zampatti pers comm.).

A ceiling cut-off to each age class was implemented in the top down approach considered by Todd et al. (2004). That is, once a preset limit was reached for any given age class the remaining fish were ‘removed’. A more realistic approach, however, would be to proportionally decrease survival as density increases above an allocated level. This allows any given age class to temporarily rise above the allocated resources without being cut-off by a fixed ceiling. Additionally, bottlenecks may occur dynamically (i.e. one year a bottleneck may occur in three year olds and the next year it may be two year olds), where the impact of density is most intensely directed. That is, the bottleneck is not necessarily programmed by the construct itself but is a function of the age structure and vital rates. Moreover, the model can include competition between age classes. For example, if there has been a strong recruitment to 1 year old fish in one year, then in the next year that cohort will impact on the number of new 1 year old fish, whereas 1 and 2 year old fish exert little negative feedback on 3 year old fish. Another example might be that 3 and 4 year old fish overlap in their resource requirements, but do not overlap with 5 year old fish.
The following is some example code:

\[
TempSum = N_{25+}(t) + \sum_{s} N_{s}(t)
\]

\[
ddFactor_{5}(t) = \begin{cases} CC < TempSum & CC/TempSum \\ CC \geq TempSum & 1 \end{cases}
\]

\[
N_{5}(t+1) = Bin(N_{4}(t), \ ddFactor_{5}(t) \times G_{i}(t))
\]

\[
TempSum = N_{25+}(t) + \sum_{s} N_{s}(t)
\]

\[
ddFactor_{4}(t) = \begin{cases} CC < TempSum & CC/TempSum \\ CC \geq TempSum & 1 \end{cases}
\]

\[
N_{4}(t+1) = Bin(N_{3}(t), \ ddFactor_{4}(t) \times G_{i}(t))
\]

\[
N_{3}(t+1) = Bin(N_{2}(t), \ ddFactor_{3}(t) \times G_{i}(t))
\]

Note that the indices on the sum function are different for \(ddFactor_{5}(t)\) compared to \(ddFactor_{4}(t)\).

To capture similar resource requirements as well as likely competition amongst fish of similar age and size, fish aged 15–25 were grouped together to impact on each other and younger fish, fish aged 11–14 were grouped together to impact on each other and younger fish, fish aged 8–10 were grouped together, fish aged 5–7 were grouped together and fish aged 3 and 4 were grouped together. See Figs. 6 and 7 for some deterministic examples of changing density and changing density dependent factors.

In a very confined space such as a river channel, there is likely to be density-dependence occurring at different ages and sizes. Large Murray cod have no natural fish predators and will feed on smaller Murray cod and any other species, similarly mid-sized cod will also feed on smaller fish but may also be out competed by larger Murray cod. To capture density dependence at all levels, both the Beverton-Holt function to larval production and the top-down proportional change to age specific survival rates were applied in the Murray cod population model.

Figure 5: Spawner – recruits relationship, blue line with density shape parameter 1 and black line with density shape parameter 10.
3. Development of population models for Murray cod

**Figure 6:** In circumstances where the population is under the carrying capacity (20,000 adults) and with increasing density over time (red line) the density dependent factor acts to decreases survival proportionally, time units being years. The black line is the density dependent factor for one year olds and the blue line is the density dependent factor for two year olds.

**Figure 7:** In circumstances where the population is over the carrying capacity (20,000 adults) and with decreasing density over time (red line) the density dependent factors act to reduce survival dramatically and then increase over time. The black line is the density dependent factor for one year olds, the blue line is the density dependent factor for two year olds, brown three and four year olds, and green five, six and seven year olds.
4. DATA ASSESSMENT

4.1 Fecundity rates

Data on the fecundity of Murray cod have changed little since the early 1990’s. Studies by Rowland (1988a; b) and Koehn and O’Connor (1990) form the best knowledge on fecundity (Table 1). Additional data for large Murray cod are currently being collected (J.D. Koehn and I. Stewart unpubl. data) and may be incorporated at a later date. Some data have been collected by DPI Victoria on size at sexual maturity and when they become available it may also be included at a later date. Other studies, such as the rehabilitation of the Murray River (Hume to Yarrawonga reach) will be analysing fish frames collected from some fishers, and where possible, fecundity data will be collected (J. Lyon pers. comm.). This remains an area of poor understanding for Murray cod and some simple studies would greatly improve our understanding of Murray cod fecundity. Assuming a one to one sex ratio, the following fecundity estimates are expressed as female eggs only (Table 1).

Table 1: Age based fecundity estimates, expressed as female eggs only, for Murray cod estimated from Koehn and O’Connor (1990): feci is the number of eggs produced per female fish in each age class using the construct of equation 6: m = mean; Sd = standard deviation; CV = coefficient of variation.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>m</th>
<th>Sd</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>fec5</td>
<td>3000</td>
<td>1500</td>
<td>50%</td>
</tr>
<tr>
<td>fec6</td>
<td>5000</td>
<td>2400</td>
<td>48%</td>
</tr>
<tr>
<td>fec7</td>
<td>7000</td>
<td>3200</td>
<td>46%</td>
</tr>
<tr>
<td>fec8</td>
<td>9000</td>
<td>4000</td>
<td>44%</td>
</tr>
<tr>
<td>fec9</td>
<td>12000</td>
<td>5300</td>
<td>44%</td>
</tr>
<tr>
<td>fec10</td>
<td>16000</td>
<td>6900</td>
<td>43%</td>
</tr>
<tr>
<td>fec11</td>
<td>20000</td>
<td>8400</td>
<td>42%</td>
</tr>
<tr>
<td>fec12</td>
<td>25000</td>
<td>10500</td>
<td>42%</td>
</tr>
<tr>
<td>fec13</td>
<td>30000</td>
<td>12600</td>
<td>42%</td>
</tr>
<tr>
<td>fec14</td>
<td>34000</td>
<td>13900</td>
<td>41%</td>
</tr>
<tr>
<td>fec15</td>
<td>38000</td>
<td>15600</td>
<td>41%</td>
</tr>
<tr>
<td>fec16</td>
<td>41000</td>
<td>16800</td>
<td>41%</td>
</tr>
<tr>
<td>fec17</td>
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<td>17600</td>
<td>41%</td>
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<td>18500</td>
<td>41%</td>
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<td>fec19</td>
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<td>48000</td>
<td>19200</td>
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<td>40%</td>
</tr>
<tr>
<td>fec22</td>
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<td>19600</td>
<td>40%</td>
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<tr>
<td>fec23</td>
<td>49000</td>
<td>19600</td>
<td>40%</td>
</tr>
<tr>
<td>fec24</td>
<td>49000</td>
<td>19600</td>
<td>40%</td>
</tr>
<tr>
<td>fec25+</td>
<td>50000</td>
<td>20000</td>
<td>40%</td>
</tr>
</tbody>
</table>

The parameter feci is the number of eggs produced per fish in age class i and forms part of the parameterisation of Fi in equation 7, such as Fi = feci × egg survival × larval survival × fingerling survival.
4. Data assessment

4.2 Survival and transition rates

Mark-recapture is considered to be the best method for the estimation of the parameters required for structured population models (White and Burnham 1999; Todd et al. 2001). There is only one known mark-recapture data set for Murray cod in the Murray-Darling Basin (Freshwater Ecology ARI, unpublished data). The data may be analysed to parameterise a size based model, although given the analytic construct required there may be insufficient data and inherent biases. The mark-recapture data allow for the estimation of the average fishing impact on survival as well as the associated average fishing rates in each size class (see Fig. 8a and 8b for an example). These data have been collected from a specific location and may not be appropriate for basin wide decision making or scenario testing, however it does provide quantitative estimates of size based fishing rates which is important in identifying plausible ranges of fishing rates.

Figure 8a: Survival rate estimates from the analysis of mark-recapture data for the given size class.

![Survival rate estimates](image)

Figure 8b: Fishing rate estimates from the analysis of mark-recapture data for the given size class.

![Fishing rate estimates](image)
Figure 9: Parametric analysis of age data to fit a survival curve.

Table 2: Age specific survival for Murray cod estimated from age data, CV's are postulated.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>mean</th>
<th>sd</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>$G_1$</td>
<td>0.4790</td>
<td>0.0958</td>
<td>20.0%</td>
</tr>
<tr>
<td>$G_2$</td>
<td>0.5846</td>
<td>0.0877</td>
<td>15.0%</td>
</tr>
<tr>
<td>$G_3$</td>
<td>0.6552</td>
<td>0.0983</td>
<td>15.0%</td>
</tr>
<tr>
<td>$G_4$</td>
<td>0.7054</td>
<td>0.1058</td>
<td>15.0%</td>
</tr>
<tr>
<td>$G_5$</td>
<td>0.7431</td>
<td>0.0743</td>
<td>10.0%</td>
</tr>
<tr>
<td>$G_6$</td>
<td>0.7722</td>
<td>0.0772</td>
<td>10.0%</td>
</tr>
<tr>
<td>$G_7$</td>
<td>0.7954</td>
<td>0.0795</td>
<td>10.0%</td>
</tr>
<tr>
<td>$G_8$</td>
<td>0.8144</td>
<td>0.0611</td>
<td>7.5%</td>
</tr>
<tr>
<td>$G_9$</td>
<td>0.8301</td>
<td>0.0623</td>
<td>7.5%</td>
</tr>
<tr>
<td>$G_{10}$</td>
<td>0.8434</td>
<td>0.0633</td>
<td>7.5%</td>
</tr>
<tr>
<td>$G_{11}$</td>
<td>0.8547</td>
<td>0.0427</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{12}$</td>
<td>0.8646</td>
<td>0.0432</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{13}$</td>
<td>0.8731</td>
<td>0.0437</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{14}$</td>
<td>0.8807</td>
<td>0.0440</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{15}$</td>
<td>0.8874</td>
<td>0.0444</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{16}$</td>
<td>0.8934</td>
<td>0.0447</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{17}$</td>
<td>0.8988</td>
<td>0.0449</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{18}$</td>
<td>0.9037</td>
<td>0.0452</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{19}$</td>
<td>0.9081</td>
<td>0.0454</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{20}$</td>
<td>0.9121</td>
<td>0.0456</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{21}$</td>
<td>0.9158</td>
<td>0.0458</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{22}$</td>
<td>0.9192</td>
<td>0.0460</td>
<td>5.0%</td>
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<tr>
<td>$G_{23}$</td>
<td>0.9224</td>
<td>0.0461</td>
<td>5.0%</td>
</tr>
<tr>
<td>$G_{24}$</td>
<td>0.9253</td>
<td>0.0463</td>
<td>5.0%</td>
</tr>
<tr>
<td>$P_{25+}$</td>
<td>0.9375</td>
<td>0.0469</td>
<td>5.0%</td>
</tr>
</tbody>
</table>
Age data obtained through analysing otoliths can be used to generate estimates of age specific survival (Fig. 9 and Table 2). Survival rates are calculated as the ratio between consecutive predicted relative frequencies, for example \( \text{Nfreq(Age}_4) = 0.1468 \) and \( \text{Nfreq(Age}_3) = 0.2081 \) and therefore the proportion of three year old fish that survive to become four year old fish is 0.7054. Coefficients of variation are inestimable through this technique, and have been assumed to decrease with age (Table 2). Age data are available from numerous researchers around the Murray-Darling Basin. So far this study has obtained age data from 220 Murray cod collected from the Murray River and tributaries (Morrison and Gooley unpublished data). Other age data were obtained from SARDI (Qifeng Ye and Brenton Zampatti unpublished data) and taxidermists (Appendix 5).

4.2.1 Other survival parameters

Once eggs are laid they must survive to hatch into larvae. Larvae must survive to become free swimming young-of-the-year juvenile fish (fingerlings) and fingerlings must survive to become one year olds in the case of the aged based model or survive to enter the first size class in the size based model. In the case of the aged based model, this leaves 3 survival parameters to be estimated; egg, larval and fingerling survival, where \( F_i = G_{x_i}G_{y_i}G_{f_i}G_{e_i}c_i \). Todd et al. (2004) estimated egg survival to be 0.5 for trout cod and in the absence of any other data it is reasonable to assume the same for Murray cod, i.e. \( G_x = 0.5 \) [annual survival rate]. There are no means by which to estimate larval and fingerling survival directly, however, solving the characteristic equation 3.9 and substituting in all other parameter estimates (Tables 2 and 3) provides a growth rate as a function of the two survival rates combined, i.e. \( G_xG_0 \). The population growth rate for Murray cod is unknown, however it is reasonable to examine three cases of the product \( G_xG_0 \): 1) \( G_xG_0 = 0.0005 \); 2) \( G_xG_0 = 0.0015 \); and \( G_xG_0 = 0.0025 \). case 1) returns a lower growth rate, \( \lambda = 1.0910 \); case 2) a moderate growth rate, \( \lambda = 1.1981 \); and case 3) a higher growth rate, \( \lambda = 1.2630 \). Todd et al. (2004) postulated that larvae were the most vulnerable life stage and that fingerling survival was likely to be an order of magnitude higher than larval survival for trout cod. Estimates of larval and fingerling survival for Murray cod in the absence of external mortality impacts such as thermal pollution, are presented in Table 3. Particular scenarios that may impact on either egg or larval survival, or both, will reduce the underlying growth rate and may produce growth rates less than 1. These scenarios will be individually manipulated within the model.

### Table 3: Postulated larval and fingerling survival for Murray cod under three alternative growth rates.

<table>
<thead>
<tr>
<th>Case</th>
<th>Growth rate (( \lambda ))</th>
<th>( G_L )</th>
<th>( G_0 )</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>( \lambda = 1.0910 )</td>
<td>0.0071</td>
<td>0.0707</td>
</tr>
<tr>
<td>2</td>
<td>( \lambda = 1.1981 )</td>
<td>0.0122</td>
<td>0.1225</td>
</tr>
<tr>
<td>3</td>
<td>( \lambda = 1.2630 )</td>
<td>0.0158</td>
<td>0.1581</td>
</tr>
<tr>
<td>CV</td>
<td>50%</td>
<td></td>
<td>25%</td>
</tr>
</tbody>
</table>

4.3 Variable growth

Murray cod exhibit variable growth (Fig. 10). Additionally, mark-recapture data indicate inconsistent growth between years. Variable growth can be accounted for directly in a size based model through the estimate of transition rates from the mark-recapture data, however this would restrict the model’s applicability to the mid-Murray. It is also possible to account for variable growth in an age based model by including a plausible range of length given age or a size distribution for each age. Wan et al. (1999) developed a more flexible growth model than the standard von Bertalanffy growth function.

\[
\text{Length} = W_f - \left( W_f - W_0 \right) / \left( c \left( 1 - W_0 / W_f \right) \left( 1 - \exp \left( k \text{Age} \right) \right) + \exp \left( k \text{Age} \right) \right)
\]

(8)

where \( W_f, W_0 \) and \( k \) are equivalent to \( L_{\text{inf}}, t_0 \) and \( k \) in the von Bertalanffy growth curve and the parameter \( c \) provides a point of inflexion and therefore slightly more elastic growth. Fitting the growth model (Wan et al., 1999) to the age-length data to obtain parameter estimates for the model: \( W_f = 150; W_0 = 6; c = -103 \); and \( k = 0.0011 \) [see Fig. 8].
Figure 10: Observed variation length at age. Red line is the mean estimate of length given age and the blue lines represent upper and lower plausible bounds.

Selecting $k$ from a bounded distribution between the upper and lower plausible bounds allows lengths to be derived given age: $k = 0.0006 + 0.0015 \times \text{Beta}(4.2, 8)$, where Beta($a$, $b$) is a random deviate drawn from a beta distribution with shape parameters $a$ and $b$ (see Evans et al. 1993). By producing 1000 lengths for each age allows probabilities to be generated for assigning size to an age and therefore the expected size distribution for each age class. This approach is useful in assessing the impact of the removal of certain size class of fish. For example, a five year old fish has probability of being in size class 40-45cm of 0.02 (Pr(40-45|5)=0.019); 45-50cm is Pr(45-50|5) = 0.112; 50-55cm is Pr(50-55|5) = 0.249; 55-60cm is Pr(55-60|5) = 0.304; 60-65cm is Pr(60-65|5) = 0.220; 65-70cm is Pr(65-70|5) = 0.082; 70-75cm is Pr(70-75|5) = 0.013; and the probability for a five year old fish being in any of the other size classes is 0 (see Table 4 and Fig. 11).
<table>
<thead>
<tr>
<th>Age</th>
<th>30-40</th>
<th>40-45</th>
<th>45-50</th>
<th>50-55</th>
<th>55-60</th>
<th>60-65</th>
<th>65-70</th>
</tr>
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<tbody>
<tr>
<td>2</td>
<td>0.568</td>
<td>0.050</td>
<td>0.004</td>
<td>0.000</td>
<td>0.000</td>
<td>0.000</td>
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</tr>
<tr>
<td>3</td>
<td>0.385</td>
<td>0.338</td>
<td>0.203</td>
<td>0.060</td>
<td>0.007</td>
<td>0.000</td>
<td>0.000</td>
</tr>
<tr>
<td>4</td>
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<td>0.249</td>
<td>0.304</td>
<td>0.220</td>
<td>0.082</td>
</tr>
<tr>
<td>6</td>
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<td>0.000</td>
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<td>0.090</td>
<td>0.218</td>
<td>0.291</td>
<td>0.246</td>
</tr>
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4.4 Expressions of risk and scenario ranking

The model uses a Monte Carlo simulation technique where the user determines the number of iterations produced. Typically, in order to examine the consequences of a potential management actions, each scenario is run (iterated) a minimum of 1,000 times. The purpose of the large number of iterations is to sample sufficiently from the parameter distributions so that a full exploration of the variation of the distribution is undertaken and the likelihood of extreme events can be examined (Ferson et al. 1989; Burgman et al. 1993). The data generated from the simulation can be represented as probability distributions (or histograms) or converted to error bars, reliability predictions, tolerance zones, and confidence intervals (Wittwer 2004).

<table>
<thead>
<tr>
<th>Size classes</th>
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<th>75-80</th>
<th>80-85</th>
<th>85-90</th>
<th>90-95</th>
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<td>0.000</td>
<td>0.002</td>
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</table>

Figure 11: Some examples of the probability of being in a size class, given age, where the unit of length is centimetres.
Recording the minimum population size from each iteration or trajectory and then graphing the associated normalised cumulative frequency distribution produces a graph of probabilities versus population size, the minimum population size risk curve. These represent both the chances of extinction (probability of falling to zero) and the chances of falling below some non-zero population threshold (Burgman et al. 1993). Additionally, risk curves can be readily compared and assessed in terms of increasing or decreasing risk by a shift to the left or right respectively of the minimum population size risk curve (Fig. 12). A method for quantifying changes in risks is to calculate the average minimum population size for each curve and compare these values (McCarthy 1995; McCarthy and Thompson 2001; Todd et al. 2002; 2004).

**Figure 12: Minimum population size risk curves (risk increases as the risk curve shifts to the left and risk decreases as risk curves shift to the right).**

Given that one of the objectives of the project is to examine a number of management scenarios, it is useful to report on the statistics of: risk curves associated with the distribution of the minimum population size (may be specific elements of the population such as fish aged 5–9, 10–14, 15–19, or fish aged 20 or older); the average minimum population size; the absolute difference in the average minimum population size and the percentage change in average minimum population size.
5. MANAGEMENT SCENARIOS

5.1 Workshop 2: Specialist workshop on Murray cod modelling

A second workshop on Murray cod modelling was held on April 18th 2007. The participants were managers/policy makers from around the Murray-Darling basin with the focus of the workshop on management outcomes. The invited participants were: Matt Barwick, Mark Lintermans [ACT], Glenn Wilson [NSW/QLD], Ross Winstanley [VIC], Andrew Sanger [NSW], Cameron Westaway [NSW], Travis Dowling [VIC], Julia Smith [VIC], Karen Weaver [VIC], Gary Backhouse [VIC], Peter Kind [QLD] and Anita Ramage [QLD]. South Australian representatives were unable to attend on April 18th, on May 2nd, Charles Todd met with Qifeng Ye, Jason Higham and Brenton Zampatti and held a mini workshop at SARDI, in South Australia.

A presentation was made to the workshop of the development of the management model and the key areas of uncertainty that remain. It was concluded from the workshop that the Murray cod management model adequately captures:

- Density-dependence (depending on parameter setting the model can generate consistent recruitment as observed in the mid-Murray reaches [and tributaries] and pulse type recruitment observed in the lower-Murray);
- The variety of current recreational fishing regulations and has the capacity to alter these regulations (options include slot size and closures – temporary or permanent);
- A variety of fishing rates that can be applied to two size classes, fish below 1m and fish above 1m, by selecting a choice of fishing rates or applying a set of specific fishing rates from the analysis of mark-recapture data collected from the Murray below Yarrawonga;
- The choice of stocking fingerlings or 1 year old fish and the number of years of stocking;
- Thermal pollution scenarios; and
- Modelling habitat change as either an increase or decrease to the available habitat, the year to implement such changes as well as the length of time that the change will remain.

The general conclusion from the workshop was that the software developed met the expectations of the project brief. Modifications/improvements were suggested at the workshop which included: accounting for mortalities of fish released after being angled; some fish kill scenarios to examine both severity and likely time to recovery from when fish kills occur; and changing the thermal pollution section to ‘impacts on early life stages’ that can not only capture thermal impacts but mortalities due to weirs and water off takes such as channels and pumps.

5.2 Modelling brief

The first Murray cod modelling workshop identified 3 general management areas that a management model should address:

1. Fishery management
2. Habitat and flow management
3. Trophic interaction management (Appendix 3)

A wide range of parameters have been included in the model to allow for the greatest site specific application. The parameters in the model are listed below and will be afforded greater explanation for their use in a user manual which will now be produced to accompany the model software.
5.3 Fishery management questions

Specific issues discussed relating to fishery management included changes that may occur with:

- changing the minimum legal length;
- changing/setting the maximum legal length;
- changing the bag or possession limits;
- a range of fishing effort/rate;
- adjusting the closed seasons;
- reducing illegal take;
- outlawing set lines;
- having ‘no take’ areas.

Model parameters

**Fishing**

- Fishing/no fishing
- Regulations
- Minimum size
- Maximum size
- Change regulations during simulation
- Change regulations
- New minimum size
- New maximum size
- Year to change fishing
- Break between implementing new regulations
- Time step to collect statistics from
- Take rate
- Rates for legal fish to 1m
- Rates for 1m+
- Catch and release mortality rate
- Illegal fishing [Yes/No]
- Rate below minimum size
- Rate above maximum size
- Catch and release mortalities

**Stocking**

- Stocking/no stocking
- Fingerlings
- 1 year olds
- Number of fish to be released annually
- Stocking duration

Angling regulations (bag and size limits) are key fishery management options for Murray cod. Angling regulations for Murray cod, however, vary for each State and Territory across the species’ range [Table 5] and changes to the fishing minimum and maximum legal size lengths are all allowed within the model. Fishing rates determine the take and hence contribute to overall population mortality. Fishing take rates of >15% per year have been estimated for the Murray River below Yarrawonga [Nicol et al. 2005] and the Mullaroo Creek [Saddlier et al. 2007]. Further analysis indicates that for certain size classes, rates may be as high as 35% [Todd unpubl. data].
Consequently, a range of plausible take rates of 0–30% are included. An option of no take is included as a base for the model and also represents the option for a reserve or no take zone. Additional impacts on adult fish can come from illegal fishing (above bag limit, take of undersize fish and take by illegal methods) and from post-release mortalities from catch and release fish. Anglers frequently catch, and then release Murray cod either because the fish is under the legal size limit or they do not wish to keep the fish. Often the estimate of fishing effort only includes the number of fish harvested and the voluntary release of fish is ignored (Clark 1983). Hooking mortality for the released fish, however, may be considerable (Clark 1983; Muoneke and Childress 1994; Bettoli and Osborne 1998) and needs to be considered in the overall mortality estimates for the population. Stocking of hatchery produced fish is another management option that is widely used to establish or enhance stocks for angling (Harris 2005), but the success of which has rarely been assessed.

### Table 5: Angling regulations for Murray cod for each State and Territory across the species’ range. (Information from PIRSA 2008, Fisheries Victoria 2008, NSW DPI 2008, DPIF Qld 2008).

<table>
<thead>
<tr>
<th>Regulation</th>
<th>SA</th>
<th>Vic</th>
<th>NSW</th>
<th>ACT</th>
<th>Qld</th>
</tr>
</thead>
<tbody>
<tr>
<td>Closed season</td>
<td>1 Aug-31 Dec</td>
<td>1 Sept-30 Nov</td>
<td>1 Sept-30 Nov</td>
<td>1 Sept-30 Nov</td>
<td>1 Sept-30 Nov</td>
</tr>
<tr>
<td>Bag/possession limit</td>
<td>1</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Minimum size limit</td>
<td>60 cm</td>
<td>50 cm</td>
<td>55* cm</td>
<td>50 cm</td>
<td>60 cm</td>
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<tr>
<td>Maximum size limit</td>
<td>100 cm</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other conditions</td>
<td>Boat limit 2</td>
<td>Only 1 &gt; 75 cm</td>
<td>Only 1 &gt; 100 cm</td>
<td></td>
<td>110 cm</td>
</tr>
</tbody>
</table>

* * to be increased to 60 cm after 1.12.2008.

5.4 **Habitat and flow management questions**

Specific issues discussed relating to habitat and flow management included the changes that may occur to a population with:

- an increase or decrease in habitat;
- an improvement to riparian vegetation;
- reducing the impacts of cold water pollution;
- providing environmental flows at differing times, durations, etc;
- fish ways being installed along the Murray River; and
- the establishment of Habitat Management Areas.

### Model parameters

**Habitat Change**

- Habitat change selection
- Decrease/increase
- Enter the proportional change as a percentage
- Enter a time step to implement changes

While the importance of habitat to fish populations is well recognised, the impact of a habitat change on a fish population is often difficult to predict. Modelling direct impacts of habitat changes on Murray cod populations is uncertain due to a lack of knowledge of the direct relationships between habitats and population numbers and density. Such habitat changes can occur through removal or addition, impoundment of waters by weirs or reductions in flows, including the effects of drought or climate change. Modelling such scenarios was included by estimating potential changes in population carrying capacity. The change in carrying capacity was assumed
to be either detrimental or beneficial equally to all life stages. It was presumed that a proportional decrease in overall habitat may result in a similar proportional decrease in population although there is usually little definitive data to prove such a change. Changes to habitat quality through changes to a particular habitat parameter (eg. wood, water velocity, depth, water quality) are even more difficult to quantify. If no quantitative measurements of habitat change can be measured (eg. an area reduction), changes may be considered in a less quantitative way using our knowledge of the species to inform the direction and proportion of the change. Projected flow data for the future years may be useful for examining the impacts of predicted climate change.

5.5 Other issues

Specific issues discussed relating to habitat and flow management included the changes that may occur to a population with:

Fish Kills
- Percentage loss
- Year fish kill begins
- Period of increased probability of fish kill
- Probability of fish kill

Impacts to early life stages
- Thermal pollution
- Thermal effects yes/no/spawning failure
- Degree of impact
- Larval mortality due to weirs (undershot or overshot, etc)
- Larval loss into irrigation off-takes

Common parameters
- Average adult population size
- Initial adult population size
- Density dependence
- Beverton-Holt shape

A number of significant fish kills have occurred in recent years that have impacted important Murray cod populations (Koehn 2005b; Sinclair 2005a; Lugg 2000). Such kills have effected populations to varying degrees (Koehn 2005b) and can be modelled as instantaneous percentage reduction in overall population. A range of other threats that can increase the likelihood of mortality for Murray cod have been identified (Koehn 2005; National Murray cod Recovery team 2007) and have the potential to operate at particular sites. These threats have been included as it is important to recognise that when these threats operate concurrently, their cumulative effects can pose significant impacts on populations. The impacts of changed thermal regimes downstream of dams have already been explored in other studies, effecting spawning, egg and larval survival and growth rates (Todd et al. 2005; Sherman et al. 2007). Two additional impacts on larvae may occur during the period when they are drifting in the river channel (peak of November: Koehn and Harrington 2006). The passage of larvae over weirs can be cause considerable larval mortalities (Baumgartner et al. 2006). Passage through undershot weirs can result in deaths of $52 \pm 13\%$ of Murray cod larvae, while passage over overshot weirs can result in mortalities of $11 \pm 5\%$ (Baumgartner et al. 2006). Larvae can also be lost from the main river channel in irrigation waters that may be extracted. While quantification of these losses has not been undertaken, it is considered that they may be considerable (Koehn et al. 2004b; King and O’Connor 2007). The ability for these threats and other management options to operate together allows cumulative impacts to be explored.
The survival and fecundity rates are specified such that, in the absence of threatening processes, the growth rate is greater than 1, i.e. $\lambda = 1.2$. This implies that populations will continue to grow without constraint. In section 3.5, density-dependence was discussed with the application of a Beverton-Holt function to larval development and a top down density dependent factor proportionally reducing age specific survival rates. Specifying the Beverton-Holt shape parameter determines the strength of density-dependence applied to larvae. Specifying the average adult population size sets the level above which the density dependence mechanisms exert the strongest negative feedback. A low average population size models a small population and a large value models a larger population. The initial value allows the option of beginning the population at a size other than the average population size.

Sensitivity analysis is an important part of the decision making process. Ferson and Ginzburg (1996) introduced a taxonomy of uncertainty for use when dealing with models of stochastic processes. Their system included four kinds of uncertainty:

- Structural uncertainty;
- Parameter uncertainty;
- Dependency uncertainty; and
- Shape uncertainty (about a distribution).

These four kinds of uncertainty can be characterised in two ways. They can either be instances of epistemic uncertainty resulting from incomplete information about the system in question (ignorance) or the underlying or inherent stochasticity in the system (variability) (Ferson and Ginzburg, 1996). For example, structural uncertainty may be what type of density-dependence should be used to model recruitment: a question of ignorance; or parameter uncertainty may arise from how much recruitment varies from year to year. Ferson and Ginzburg’s (1996) approach was motivated by model-building for ecological risk assessment, and it effectively summarises the kinds of things that models (and modellers) need to consider when formulating responses to environmental problems and management questions. As a rule of thumb, the four kinds of uncertainty provide the example set over which sensitivity analyses should be undertaken. The sensitivity analysis options (including changes to density dependence) allow for the exploration of parameter, dependency and shape uncertainty. In practice, undertaking sensitivity analysis requires a methodological approach to making parameter changes, then recording the metric about which the sensitivity was assessed. For example, one management metric may be the number of fish older than 20 years of age (representing most fish over 1 metre). The metric should be linked to the decision making process, that is, that metric can be used in the examination of a variety of different management actions. If undertaking sensitivity analysis leads to conflicting interpretations about which action is more effective, then it becomes important to understand what has contributed to this sensitivity. There are three metrics that are being used to determine sensitivity to management actions and they are 20 yr olds (plus), fish in the size class 60-90cm, and total adults in terms of the average minimum population size; the absolute difference in the average minimum population size and the percentage change in average minimum population size.

5.6 Some scenario examples

The following are some examples of how to use and interpret some of the output generated by the Murray cod stochastic population model. Each model ran 1 000 iterations of 50 time steps, using an average carrying capacity of 20 000 adult Murray cod and an initial population size of 20 000 fish so that the unimpacted scenario was well away from a zero population size or extinction.

5.6.1 Modelled Current Victorian regulations

The current Victorian regulations allow for fish to be kept if they are 50 cm in length or longer. There are no protections for fish over 1 metre (100 cm) other than to limit the number of fish over 75 cm that one angler may have. The take rate has been measured to be up to 30% in some reaches of the Murray River (Todd unpubl. data). The minimum population size (MPS) risk curves in Fig. 13, represent the probability that the population will fall to or below a given threshold population size at least once over the 50 times steps. The green line represents the MPS risk curve associated with no fishing (Fig. 13). Introducing a take rate of 10% increases the risks by shifting the risk curve to the left, represented by the blue line (Fig. 14). A take rate of 20% adds more risk as can be seen with the red line shifted even further to the left (Fig. 13).
5. Management scenarios

5.6.2 Modelled Slot size regulations of 60–100cm

It has been proposed by some states to introduce a ‘slot-size’, where fish can only be legally taken when between certain upper and lower size limits. The risk curves in Fig. 14 are associated with the original curves found in Fig. 13, and additionally a ‘slot-size’ scenario has also been modelled where fish under 60 cm and over 100 cm are protected. The blue dashed line is the risk curve associated with a 10% take and ‘slot-size’ scenario and the red dashed line is the risk curve associated with a 20% take and ‘slot-size’ scenario. Both these scenarios reduce risk by moving the risk curve to the right.

Figure 13: Minimum population size risk curves for a modelled Murray cod population with an average carrying capacity of 20,000 adults, where the green line is the no fishing risk curve; the blue line is the risk curve for the former regulations with a take rate of 10%; and the red line is the risk curve for the former regulations with a take rate of 20%. 
5.6.3 Cumulative threats – multiple modelled impacts

Threatening processes do not generally happen in isolation and act cumulatively on population processes. In Fig. 15, the scenarios are:

1. the long-dashed blue line is the risk curve associated with a 10% take and ‘slot-size’ 60–100cm scenario as in Fig. 14;
2. the orange line is the risk curve associated with a 10% take and ‘slot-size’ and 15% mortality due to catch and release;
3. the brown line is the risk curve associated with a 10% take and ‘slot-size’ and 15% mortality due to catch and release and 20% larval loss associated with an overshot weir;
4. the mauve line is the risk curve associated with a 10% take and ‘slot-size’ and 15% mortality due to catch and release and 20% larval loss associated with an overshot weir and 20% larval loss due to irrigation off take;
5. the olive line is the risk curve associated with a 10% take and ‘slot-size’ and 15% mortality due to catch and release and 20% larval loss associated with an overshot weir and 20% larval loss due to irrigation off take and a 20% reduction in the carrying capacity; and
6. the yellow line is the risk curve associated with a 20% take and ‘slot-size’ and 15% mortality due to catch and release and 20% larval loss associated with an overshot weir and 20% larval loss due to irrigation off take and a 20% reduction in the carrying capacity.
As threatening processes are included in the model they shift the risk curve further to the left, where some have a greater impact than others. A method for quantifying the cumulative threatening processes is to calculate the average minimum population size for each of the risk curves (Table 6). The average minimum population size can be used to quantify the differences between risk curves. For example a population facing the single threat associated with scenario 1 is expected to have approximately 3075 more fish than the same population with multiple threats as in scenario 5, \((8289.43 - 5214.52 = 3074.91)\).

Table 6: Average minimum population size associated with the specified scenario

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Average minimum population size</th>
</tr>
</thead>
<tbody>
<tr>
<td>No fishing</td>
<td>12433.10</td>
</tr>
<tr>
<td>1</td>
<td>8289.43</td>
</tr>
<tr>
<td>2</td>
<td>7094.13</td>
</tr>
<tr>
<td>3</td>
<td>6789.24</td>
</tr>
<tr>
<td>4</td>
<td>6381.52</td>
</tr>
<tr>
<td>5</td>
<td>5214.52</td>
</tr>
<tr>
<td>6</td>
<td>2664.32</td>
</tr>
</tbody>
</table>

Given the number of parameters included in the model, the number of different management scenarios that can be considered, and the different characteristics of each management site, there are a multitude of scenario combinations that can be explored. It is not the intention of this report to provide a large number of such examples, especially as they could be misinterpreted as ‘answers’ to particular management issues. The scenarios given above provide examples of how management options may be assessed using the model and it is important that each management decision is worked through the model on a site by site basis to ensure that all threats are considered.
5.7 ESSENTIAL – modelling framework

The modeling of all scenarios was undertaken using the software package ESSENTIAL [Todd and Lovelace 2009]. ESSENTIAL is a highly flexible stochastic modelling platform that allows both expert model development as well as general use by way of access to a limited suite of parameters. Data can be accessed on all parameters over all time steps and iterations. A specific application of ESSENTIAL is being developed to provide the Murray Darling Basin Commission with a stand alone Murray cod Management Model.
6. DISCUSSION AND CONCLUSION

The management scenarios modelled were established through a highly consultative workshop approach and included various options for changes in angling regulations in relation to size and bag limits and the potential impacts of changes to habitats. A no take scenario which equates to a fishery closure was used as the base model. Modelled management scenarios for Murray cod indicate that the risk to populations can be reduced substantially by the appropriate changes to management actions. In particular, changes to the size limits on angler take can have a major impact on population persistence. The implementation of a slot size that protects both smaller and larger fish reduced population risk considerably. Our results support recent changes toward introduction of a legal take slot size of 60–100 cm by some States. Whilst habitat changes are difficult to quantify, it was illustrated that reductions in carrying capacity can place additional risk on populations, particularly when combined with angler take. Importantly, the cumulative impacts of less recognised threats such as thermal pollution, fish kills and mortalities to larvae over weirs and losses into irrigation off-takes can be explored and need to be recognised as having the potential to contribute significantly to mortalities at certain sites.

The cumulative impacts of multiple threats on fish populations are rarely considered. Their potential cumulative or compounding affects, however can be greater than some of the more recognised impacts such as angler take. While some impacts, such as cold water releases have been explored for Murray cod (Todd et al. 2005; Sherman et al. 2007), many others have not been considered. Not all of these impacts have a linear response and some such as thermal impacts may over-ride all others. For example, if water temperatures do not reach the recognised spawning temperature of 15°C (Humphries 2005; Koehn and Harrington 2006), then spawning and hence recruitment may not occur. Impacts on larvae, such as temperature, damage passing through weirs, and diversions into irrigation channels, only occur through the period when larvae are drifting (peak in November in the Murray River: Koehn and Harrington 2006). They can, however, have major impacts on long-term population viability. Such impacts are not constant and may be quite site specific. The inclusion of the ability to test such cumulative impacts in the model are useful to illustrate how protection at these vulnerable and less obvious life stages, from less well-recognised threats, can still have major effects on the species’ conservation and the number of fish available to anglers.

There is an increasing trend for catch-and-release fishing, including for Murray cod. While this may initially reduce overall take, it may still contribute significantly to overall mortality. In a review of catch and release studies in North America, Bartholomew and Bohnsack (2005) reported an average catch and release mortality of 18%, although this varied greatly by species and within species. Significant mortality factors include: anatomical hooking location; use of natural bait; hook removal; hook type (J or circle, barbed, barbless); deeper depth of capture; warmer water temperatures; and extended playing and handling times. While mortality distributions varied between species in that study, they were, however, similar for salmonids, marine and freshwater species. Results from that work also suggested that many of the reported mortality rates were underestimates of actual mortality (especially for marine species) as they rarely included predation during capture or after release. While this factor may be less relevant to large Murray cod it may have a greater impact on undersize fish. There is also a lack of data concerning any cumulative effect of multiple hookings. Hence, the reporting of removal rates may not reflect the total mortality or impacts of fishing because they do not reflect mortality for the released fish and this needs to be considered in overall mortality estimates for the population (Bartholomew and Bohnsack 2005).

While there is considerable anecdotal information about illegal removal of Murray cod in certain areas, there is little quantification of the amount of illegal removal and what impact this may have on the population (Joy Sloan, Victorian Fisheries, pers. comm.). Such illegal removal can be in the form of fish which do not comply with legal size or bag limits or those which may be taken by illegal methods such as nets or set-lines. It has been recognised that removal of undersized Murray cod, especially 45–50 cm fish, does occur (Nicol et al. 2005). Removal of fish, either by legal or illegal means, impacts on a population’s resilience and increases the risk of population decline. While sometimes difficult to quantify, all removal of fish must be taken into account in the management of Murray cod populations and, consequently, any action that reduces the illegal removal of fish will reduce the risk to the population.
Management of recreational angling for Murray cod is not uniform across jurisdictions. South Australia has recently implemented a minimum legal size of 60 cm and a maximum size of 100 cm (slot size 60-100 cm). Modelling results from this study would support the introduction of this slot size limit as it reduces population risk (Figure 15). While recent changes to New South Wales regulations also include this slot size, it still allows the take of one fish above 75 cm. This does not ensure compliance with the scenario modelled in this study and will have additional risk. Site specific increase in take can also occur if angling becomes more restricted in a nearby area. The recent changes to both South Australia and New South Wales regulations may transfer some angling pressure to Victorian waters in border areas. To avoid this there is a need for uniformity in regulations to be achieved as quickly as possible. Such issues could be avoided through a more uniform multi-state approach to Murray cod management.

The recognition of the cumulative impacts of co-occurring threats is important when assessing risk to populations. There are several threats to Murray cod which are not well recognised (Koehn 2005b) and to date have received relatively little recognition in population management (eg. larval loss in irrigation off-takes). When operating with other threats, however, they increase the risk of a low population size. The example provided in section 5.6.3 highlights their cumulative impacts, and Table 6 indicates how the results from the model may be used to compare these impacts. Importantly, the inclusion of the relevant parameters in the model allows these threats to be assessed on a site by site basis.

At first glance, it could be perceived that the management of the recreational fishery for Murray cod and the species’ conservation may be in conflict. However, assessment of this through the recovery planning process, which involved managers, scientists and stakeholders from both conservation and recreational fishery sectors indicated that there was general commonality between objectives (National Murray Cod Recovery Team 2007). Moreover, the methods used for fishery and conservation planning contain essentially similar elements and could be formally incorporated into a more comprehensive approach to the management of Murray cod. Fisheries and conservation management are both characterised by conflicting objectives, multiple stakeholders with divergent interests and high levels of uncertainty about the dynamics of the resources (Smith et al. 1999; Pressey et al. 2007). Predicting the results of any management action is uncertain because the dynamics of the ecosystems and their processes are complex and poorly understood, with limited understanding of individual threats, combinations of threats, new threats and the dynamic nature of threats, ecosystem and anthropogenic changes (Sainsbury et al. 2000; Pressey et al. 2007).

Management-Strategy-Evaluations (MSE) is a process used in commercial fisheries to assess the consequences of a range of management options and trade-offs [Smith et al. 1999]. It develops a clearly defined set of management objectives, a set of performance criteria, a set of management strategies or options and a means of calculating the performance criteria for each objective. Objectives rely on simulation testing of the whole management process using performance measures derived from operational objectives [Sainsbury et al. 2000]. MSE deals explicitly with uncertainty and seeks to identify trade-offs among management objectives and evaluate the consequences of alternative strategies or decisions, rather than trying to seek ‘optimal’ solutions. It seeks to provide decision makers with information on which to base management choices and usually involves methods such as modelling to test management scenarios [Smith et al. 1999]. MSE incorporates many of the concepts and techniques of adaptive management [AM] that have been developed as a framework for addressing the general problems of management under uncertainty [Walters 1986; Walters and Hilborn 1978; Smith and Walters 1981]. AM and Adaptive Environmental Assessment and Management (Holling 1978) brings to management the philosophy of scientific method where management actions are implemented in a well-defined framework for goal-setting, monitoring and evaluation of outcomes [Walters 1986; Clark et al., 1995]. It aims to deal explicitly with uncertainty through a process of identification and analysis of critical aspects of management [Walters and Hilborn 1978; Bearlin et al. 2002].

Models play an important role in AM and MSE processes and aid the development of cooperative and quantitative approaches to management [Starfield 1997]. Modelling within such processes allow simulation of: the observation or monitoring process; scientific assessment or data analysis; how the results of the data analysis will be used for management purposes and the implementation of management decisions [Sainsbury et al. 2000]. In applying a precautionary approach, such as may be needed for threatened species, this allows different levels of precaution to be examined [Sainsbury et al. 2000]. Management under uncertainty necessitates the importance of feedback between the rate of exploitation and learning about the stock dynamics [Smith 1993]. Uncertainty in these processes can occur through: model error (eg. inappropriate function choice), process noise (eg. recruitment variability), data error (eg. ageing), bias in estimators, lack of contrast in data, or failure to achieve the managed target [Smith 1994; Bearlin et al. 2002]. Scientific input into stock
6. Discussion and conclusion

Assessment and MSE remains a key requirement for an effective management system (Smith et al. 1999; Prager and Rosenberg 2008). Difficulties in stock-recruitment relationships and optimal policies can be due to parameter uncertainty and well as environmental variation. Much of our understanding of commercial fish stocks dynamics has come from information generated by the process of harvesting those stocks (Smith and Walters 1981). Such collection obviously does not occur for threatened species and there is a need for collection of data comparable to that which would normally be collected as harvest for commercial fisheries. In the case of Murray cod, some data may be obtained from recreational anglers. The collection of these data needs to be undertaken in a scientific manner, designed to answer management questions, possibly supplemented by scientific data collection. The involvement in monitoring and data collection by this important stakeholder group importantly helps to ensure ‘ownership’ of any stock/conservation assessment and subsequent management decisions.

There are a series of data gaps that have been identified in the process of this study that, if filled through appropriate monitoring and data collection, would reduce the uncertainty of the model and its outputs. The size of the resource and absolute catch/take rates are unknown, particularly on a regional or site basis. Catch and release mortality rates for different size classes and the cumulative effect of multiple captures need quantification. Data on age-size, size distributions and size-maturity relationships inform important parameters in the model. Not all of these relationships are well understood although it would be relatively simple to improve our understanding of these relationships. For example, fish carcases were not retained from fish kills and, as a consequence, an opportunity to gain valuable scientific data on the age-structure of the Murray cod populations was lost (Koehn 2005b). Such data could have been used to inform the model and to plan restorative actions. Similarly, such data could also be collected from the carcases of fish that are kept by anglers. This needs to be undertaken in a scientific manner on a regional basis. The recent change in angling regulations provides an opportunity to test predicted outcomes through appropriate monitoring.

Murray cod is a threatened species which needs immediate management actions in order to rehabilitate populations. Like many other fish species it is subject to habitat changes and other threats, and, as a popular angling species, it is subject to take. All of these impacts may increase the risk of population reductions or extinction and need to be monitored regularly so that appropriate changes to management actions can be quickly adapted if needed. The methodology presented here offers a formalised, rational, modelling approach which can form the basis for the assessment and prioritisation of management options for Murray cod to minimise the risk to populations. Such modelling also highlights data gaps and monitoring requirements and can become an integral part of the conservation and fishery management process and provides a tool for exploring the outcomes of management scenarios at both the regional and local scale. The modelling process has helped facilitate interagency Murray cod management and emphasises the need for coordination between fishery managers, water and environmental protection/conservation agencies (Southwick and Loftus 2003). Deliberate, active adaptive policy appears crucial in managing stock-recruitment systems (Smith and Walters 1981) not only to commercial species but also to threatened species recovery (Bearline et al. 2002). It involves a range of stakeholders including: scientists, industry and conservation NGO’s (Smith et al. 2001). These relationships have been facilitated for Murray cod through the formation of the Recovery Team and the Murray cod Taskforce (Lintermans and Phillips 2005) and this modelling process. Management needs to be further formalised with consideration given to formal review of progress (Prager and Rosenberg 2008). The outputs from modelling these scenarios, together with additional data collection and inputs (eg. creel surveys) will provide a solid basis for improved management of Murray cod population which can ensure its sustainability. Changes in angling regulations provide an opportunity to test predicted outcomes through appropriate monitoring.
7 REFERENCES


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References


References


APPENDIX 1: A REVIEW OF THE ECOLOGICAL KNOWLEDGE OF MURRAY COD

A1.1 Taxonomy

Murray cod is one of four taxa within the endemic Percichthyid genus Maccullochella. The other representatives are the Mary River cod Maccullochella peelii mariensis (endemic to the Mary River system in south-eastern Qld), the trout cod Maccullochella macquariensis (occurring in the Murray and Murrumbidgee River systems in NSW, ACT, and Vic) and the eastern freshwater cod Maccullochella iker (occurring in coastal rivers of north-eastern NSW) (Harris and Rowland 1996). All species and sub-species of Maccullochella are considered threatened nationally; trout cod, Mary River cod and eastern freshwater cod are Endangered, while Murray cod is Vulnerable (EPBC Act).

Murray cod is most closely related to Mary River cod, the two taxa being considered subspecies. There are, however, other taxonomic reviews underway which may further clarify relationships. Murray cod is similar in appearance to trout cod, which has resulted in some confusion in the identification and taxonomic status of both species, especially as the range of both species overlapped historically, to a large degree. Although trout cod was first described in 1829 and Murray cod in 1838, it was not until 1972 that the two species were confirmed as being distinct, separate species (Berra and Weatherley 1972). Hybridisation between Murray cod and trout cod has been reported in Cataract Dam (Nepean River New South Wales) where both species were introduced (Wajon 1983; Harris and Dixon 1988), in the Murray River downstream from Yarrawonga Weir, and in a fish hatchery (Douglas et al. 1995). The occurrence of hybrids in the Murray River is one of very few cases where hybridisation has been reported in freshwater fish in natural wild situations in Australia.

A1.2 Distribution

A1.2.1 Natural distribution

The Murray cod is endemic to the Murray-Darling River system in south-eastern Australia, including South Australia, Victoria, New South Wales, Australian Capital Territory and Queensland (Harris and Rowland 1996). The species occurred throughout almost the entire system, with the exception of some of the upper reaches of tributaries. Murray cod still occurs throughout almost all of its historic range, although with some localised extinctions in several upper tributaries.

A1.2.2 Introductions

The Murray cod has been successfully bred in hatcheries for many years, and both hatchery-bred and wild-caught fish have been widely translocated and stocked within and outside its natural range (Lintemans 2005; Pierce 1990; Rowland 1989). Murray cod populations in some areas, particularly in lakes and impoundments, are maintained by stockings of hatchery-bred fish. Translocations into areas outside its natural range have resulted in many extra populations becoming established.

A1.2.3 Population decline

The Murray cod remains widely distributed throughout the Murray-Darling River system with only a small decline in total range, although it has undergone an extensive decline in abundance since European settlement of Australia, especially in the last 70 years (Cadwallader and Gooley 1984; Harris and Gehrke 1997; Rowland 2005), and there have been some recent localised extinctions (Koehn et al. 1995). An indication of the extent of the decline can be gauged from historical and anecdotal records, and from data provided by commercial fisheries. Early records dating from the early times of European settlement indicate that Murray cod were abundant, and of large size, in the Murray-Darling River system. The explorer John Oxley, in 1817, recorded that the Lachlan River ‘is rich in the most excellent fish, procurable in the utmost abundance. One man in less than an hour caught eighteen large fish, one of which was a curiosity from its Immense size and beauty of its colours…It weighed an entire 70 pounds…Most of the other fish taken this evening weighed from fifteen to thirty pounds each’ (cited in Rowland 2005). There are similar enthusiastic historical reports of captures of abundant
large cod from other rivers in the system (Rowland 1989, 2005). The commercial fishery for cod developed in the mid to late 1800s, and early fishery reports noted the large numbers and size of cod present. Catches apparently peaked in the early 1900s, then declined, then reached a smaller peak in the early 1950s, when up to 150 tonnes in South Australia and 140 tonnes in New South Wales was caught per year (Dakin and Kesteven 1938; Rowland 1989; Kailola et al. 1993; Ye et al. 2000). Catches declined steeply soon after, but continued at a very low level for about another 40 years. There was also a large reduction in both the number of commercial fishers and the area available for commercial fishing, and the last commercial wild fishery for Murray cod finally closed in 2003.

A1.3 Conservation Status

The Murray cod is listed as Vulnerable under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). It is currently considered threatened in Victoria, where it has been assessed as Endangered (DSE 2003) and is listed under the Flora and Fauna Guarantee Act 1988 (FFG Act). The species is also a component of the ‘Lowland riverine fish community of the southern MDB’, a Listed threatened community under the FFG Act. In NSW, the Murray cod is also a member of three listed ‘Endangered Ecological Communities’ under the Fisheries Management Act 1994: (1) the ‘Aquatic ecological community in the natural drainage system of the lower Murray River catchment’, (2) the ‘Aquatic ecological community in the natural drainage system of the lowland catchment of the Darling River’ and (3) ‘Aquatic ecological community in the natural drainage system of the lowland catchment of the Lachlan River’.

Murray cod has suffered a substantial decline in abundance throughout the Murray-Darling River system since European settlement of Australia, although its natural distribution remains largely unchanged. The recommendation for listing Murray cod under the EPBC Act concluded that the species had declined substantially in numbers, with an estimated historic decline of at least 30% in numbers within the last 50 years, and an estimated maximum Extent of Occurrence of 660 km², within which there has been substantial loss and degradation of habitat (TSSC 2001). More generally, native fish populations in the Murray-Darling River system are estimated to have declined to about only 10% of their pre-European abundance (MDBC 2004a). A review of the status of Murray cod in 2001 prior to its listing under the EPBC Act concluded that ‘persistence of the species does not appear to be of immediate concern but the integrity of wild populations and of the ecosystems which support them are seriously threatened’ (Kearney and Kildea 2001).

A1.4 Social issues

Murray cod is an iconic species, and has significant economic, cultural, recreational and environmental value for Australians (Koehn 2005a; Rowland 2005; Sinclair 2005a, 2005b). Indeed, it has been described as the ‘flagship freshwater fish for all of Australia’ (Kearney and Kildea 2001). Many Australians hold passionate views about Murray cod, be they angler, scientist, riverside resident, environmentalist or water manager. For instance, public reaction to a major fish kill in the Goulburn River (Vic) in 2004 led to the Victorian Government establishing an environmental audit of the whole river downstream of Lake Eildon, the first of its kind in Victoria (Sinclair 2005a).

Murray cod traditionally was a major part of the diet of aboriginal tribes living adjacent to inland waters, and an important cultural icon for these tribes (Lawrence 1971; Ramsay Smith 1930). Early European settlers also used Murray cod as a food source, and the species was once common enough to support commercial fisheries throughout its range. As one of the largest freshwater fish in Australia, the Murray cod generates considerable public interest because of its size and association with the Murray-Darling River system. It is a highly sought after, freshwater angling species, with an estimated 566,000 anglers fishing in the Murray-Darling Basin during 2000/01, with 22% targeting Murray cod (Park et al. 2005). There are significant social and economic benefits of recreational fishing for cod to local communities. It is also important in aquaculture, as a food fish and for stocking for recreational angling. These qualities of Murray cod transform its significance from being merely ‘a fish’ to being an important component of Australian folklore and cultural heritage (Koehn 1994). The species also provides a significant way for the community to connect to the river environment. The management of Murray cod populations and their riverine habitats become the management of a part of Australian cultural heritage (Sinclair 2005a). As such, many Australians have a stake in the sustainable management of Murray cod populations and their habitats.
Appendix 1: A review of the ecological knowledge of Murray Cod

A1.5 Ecology

Although Murray cod is a well known species, it is only recently that the study of its ecology in the wild has occurred, with most information derived from studies of captive animals, or anecdotally from natural history observations (Koehn and O’Connor 1990). Murray cod can grow to a substantial size, reputedly to 1.8m in length (Whitley 1955) and 113.6kg in weight (Noble, in Rowland 1989). Most specimens currently taken are less than 5kg in weight, and fish greater than 1m and 40kg are rarely seen now (Lintermans 2007).

A1.5.1 Habitat

The Murray cod occurs in a range of flowing and standing waters, from small, clear, rocky streams on the inland slopes and uplands of the Great Diving Range, to the large, turbid, meandering slow-flowing rivers, creeks, anabranches, lakes and larger billabongs of the inland plains of the Murray-Darling Basin. Within these broad habitat types, Murray cod are usually found associated with complex structural cover such as large rocks, large snags, smaller structural woody habitat, undercut banks and over-hanging vegetation (Dakin and Kesteven 1938; Lake 1967b; Langtry in Cadwallader 1977; Cadwallader 1979; Cadwallader and Backhouse 1983; Harris and Rowland 1996; Koehn 1997, 2006; Rowland 1988a, 2005). The species frequents the main river channel as well as larger tributaries and anabranches (which are important habitats) and is considered a ‘main channel specialist’ (Humphries et al. 2002). It will use floodplain channels when these are inundated (Koehn 1997, 2006; Koehn and Harrington 2005), but the use of the floodplain proper by adults, juvenile or larvae appears limited (Koehn and Harrington 2005, 2006; King and Koehn unpubl. data). While nursery habitats for post-larval fish have not been identified, juveniles less than one year old have been found in main river channels where it appears they settle at a late larval stage (Koehn and Harrington 2005).

A1.5.2 Diet

Murray cod are the top-order or apex aquatic predator in the Murray-Darling River system (Rowland 2005, Ebner 2006, Baumgartner 2007) and are therefore likely to have a profound impact on food chains and on the aquatic community. Indeed, its impact is likely to be so substantial that it lead Kearney and Kildea (2001) to state: ‘The ecological significance of the Murray cod on the Murray-Darling system can be argued to be more complex and profound than that for any single terrestrial animal, except humans’.

As an apex predator, Murray cod feed mainly on fish and large crustaceans (Ebner 2006, Baumgartner 2007). They are carnivorous, with a diet including a wide variety of aquatic organisms such as spiny crayfish, yabbies and shrimps, as well as fish including Goldfish, Redfin, Carp, Bony Herring (and occasionally, Silver Perch and Golden Perch). Aquatic insects and bivalve molluscs are also taken (Cadwallader and Backhouse 1983; Harris and Rowland 1996; Rowland 1988a). Diet changes with age and size. Murray cod larvae consume zooplankton, particularly cladocerans and copepods (Koehn unpubl.; Rowland 1992), and begin feeding on aquatic insects at 15–20 mm in length (Kailola et al. 1993). Adult cod consume larger prey items, occasionally including vertebrates such as frogs, reptiles and birds (Rowland 1988a).

A1.5.3 Behaviour

Murray cod are most active during spring and early summer and appear to be more active at night (Koehn unpubl. data). During the day they normally seek shelter around logs and other debris, the resting places appearing to form the focal point of their territories (Kailola et al. 1993; Harris and Rowland 1996; Koehn 1997). Young Murray cod become territorial and behave aggressively towards other cod from 40–50 mm in length, and adults are considered solitary and highly territorial (Cadwallader 1979; Cadwallader and Backhouse 1983; Cadwallader and Gooley 1985), although anglers report the capture of several similar size cod from the one location, indicating aggregations may occur (Kearney and Kildea 2001).

A1.5.4 Age and Growth

Age and growth rates have been documented for several lake and river populations (Anderson et al. 1992; Gooley 1992; Rowland 1985, 1988a, 1998b). Newly hatched larvae are 6–9 mm in length, have a large yolk sac, and begin feeding about 10 days after hatching at about 20°C. Growth rate varies considerably between locations and seasons, and is influenced by temperature, habitat and food availability. In New South Wales rivers, growth has been estimated as 23 cm and 0.2 kg, 35 cm and 0.8 kg, 50 cm and 2.0 kg, 58 cm and 3.5 kg, and 64 cm and 5.0 kg after one to five years respectively. At over five years of age, fish grow at between...
1.0–2.5 kg per year, with fish from impoundments tending to grow faster than fish from rivers, while Murray River fish are heavier per unit length than those from the Darling River (Rowland 1988a). Murray cod reach sexual maturity at 4–6 years of age (occasionally earlier in some populations) and at minimum weights of about 2 kg for females and 0.7 kg for males (Cadwallader and Gooley 1984; Gooley et al. 1995; Rowland 1988b). In southern waters, feeding activity (and therefore growth rate) is reduced by low water temperatures during winter, and fish probably mature later and at a larger size than fish in more northern waters (Glen Wilson, UNE, unpublished data). The Murray cod is among the most long-lived Australian freshwater fish, with a 1.4 m long, 43 kg fish being 47 years old (Anderson et al. 1992). A 1.27 m fish collected from the Murray River in 1996 downstream of Yarrawonga was aged at 49 years (Greg Sharp, DPI, pers. comm.). It is possible that some of the largest specimens taken in the past may have been much older, with an age estimate of the largest cod ever caught (113.6 kg) being 74–114 years old (Rowland 1988a).

### A1.5.5 Reproduction

Aspects of the reproductive biology of Murray cod have been reported from a range of hatchery studies and some recent studies and observations in the wild (Lake 1959, 1967a; b; Langtry, in Cadwallader 1977; Cadwallader et al. 1979; Cadwallader and Gooley 1984; Rowland 1983a,b, 1985, 1988b; Gooley et al. 1995; Humphries 2005; Koehn and Harrington 2005, 2006). The species has an annual reproductive cycle and a relatively short, defined breeding season. In captivity, Murray cod form pairs and spawn in spring-summer, in response to rising water temperatures of 16.5–23.5°C, with most spawning at around 20°C (Cadwallader et al. 1979; Cadwallader and Gooley 1984; Gooley et al. 1995; Rowland 1985, 1998a). In the wild, spawning has been shown to occur at temperatures as low as 15°C (Humphries 2005, Koehn and Harrington 2006). Reproduction appears to be largely dependent upon water temperature, with flooding or a rise in water level apparently not required to initiate spawning (Rowland 1983a,b, 1988b; Cadwallader and Gooley 1985). Spawning in rivers has been shown to occur regularly each year despite a range of flow conditions (Humphries 2005, Koehn and Harrington 2006). Spawning commences in early spring in the northern part of its range, but may not commence until late spring or early summer in the southern part of its range (Rowland 1988b). In a study of the biology of Murray cod in Lake Charlegrark (Vic), the smallest ripe male was found to be 5–6 years old, 440 mm long and 1.4 kg in weight, while the smallest ripe female 6–7 years old, 485 mm long and 2.3 kg in weight (Cadwallader and Gooley 1984).

The number of eggs laid is generally related to the size of the female, with females 2–3 kg producing 6–10,000 eggs, at 5 kg producing around 40,000 eggs, at 23 kg around 90,000 eggs, 110,000 eggs at 33 kg (Stuart and Koehn unpubl. data), and up to 200,000 eggs in very large fish (Kailola et al. 1993; Lake 1959, 1967a; Rowland 1988b). The importance of large fish to the reproductive outputs of the population is now being recognised (Stuart and Koehn 2007) and additional age-length-fecundity data is being collected to attempt to further develop these relationships to refine population models and management actions. The eggs are 2.5–3 mm in diameter, swelling to 3–4 mm in diameter when water-hardened, adhesive, demersal, opaque and pale amber in colour (Dakin and Kesteven 1938; Lake 1967a; Cadwallader and Backhouse 1983; Rowland 1983a,b). The average diameter of the yolky part of the egg is 2.5 mm and there is usually one large and many small oil globules present. Eggs are laid on a hard substrate such as large structural woody habitat, rocks and clay surfaces, while in ponds and dams, captive cod have spawned inside hollow objects such as concrete pipes and metal drums, on fallen timber and directly on the substrate (Cadwallader et al. 1979; Cadwallader and Gooley 1984; Gooley et al. 1995; Rowland 1988a). Murray cod will excavate saucer-shaped depressions in the substrate, with bigger fish creating bigger depressions. It is not certain if these are only resting sites or are used for spawning. The finding of large depressions in the substrate of natural waters inhabited by Murray cod has led to the belief that cod use the depressions as spawning sites, similar to the nests made by Freshwater Catfish Tandanus tandanus. Large numbers of these depressions were seen on mud banks in the Murrumbidgee River in the late 1940s in October, and it was speculated that these were Murray cod ‘egg pans’ (Langtry, in Cadwallader 1977). The eggs are typically deposited in a layer one egg thick, and the area covered by an egg mass laid on the substrate in a small dam at Snobs Creek research facility measured about 45 cm by 35 cm (Cadwallader and Backhouse 1983). The eggs are guarded by the male fish and hatch after 4–13 days, depending on temperature, with hatching occurring within 5–7 days at temperatures of 20–22°C, and being mostly completed 2–3 days after commencement (Cadwallader et al. 1979; Cadwallader and Gooley 1984; Kailola et al. 1993; Rowland 1988b, 1998a, 2005). Spawning generally occurs in a single event, although multiple spawnings and the spawning of male fish with multiple partners have been recorded (Brett Ingram, DPI, and Steve Thurston, DPI NSW, pers. comm.).
Appendix 1: A review of the ecological knowledge of Murray Cod

Larval Murray cod initially remain near the spawning site, usually forming large clumps. After several days the clumps disperse, and the larvae have a nocturnal drifting stage, where they rise in the water column and drift with the current (Humphries et al. 2002; Koehn and Harrington 2005, 2006), which probably aids in dispersal away from the spawning site. Drifting larvae range from 9.5-15mm and may occur over up to 10 weeks with a peak in abundance in November. Variability in abundance has been best accounted for by three variables – year, day length and flow in the previous seven days (Koehn and Harrington 2006).

A1.5.6 Recruitment

While spawning in Murray cod apparently does not require flooding, recruitment success appears to be strongly linked to river flow, with good year classes in some rivers coinciding with a rise in water level or flooding at or soon after spawning (Rowland 2005; Ye et al. 2000). Recruitment success is likely to be linked to timing, duration, water quality, and especially temperature of the flows and in addition flooding in spring appears to provide optimum conditions for survival and recruitment of larvae and juveniles in rivers (Kearney and Kildea 2001; Rowland 1985, 1989, 1998a). King et al. (2007) found increased recruitment of G0 Murray cod in the year following flooding, although distinct correlations between flows and year classes in the mid reaches of the Murray River are less certain (Nicol and Koehn, unpubl. data).

Prior to 2003, data on the stock status of Murray cod in SA reaches of the Murray River were derived from commercial fisheries data. This fishery was discontinued in 2003 and since this time, data have been collected using fishery independent methods. Length-frequency data from commercial fishers pre 2003 indicate that strong recruitment of Murray cod in the SA reaches of the Murray River last occurred in 1994 (Ye and Zampatti 2007). The data also indicate that a low level of recruitment may have occurred in 1998 and 2000. Recruitment in these years was associated with instream and overbank discharges in the river of approximately 30,000 to 100,000 ML/d. Fishery independent data collected from 2005 onwards indicate that minimal Murray cod recruitment has occurred since 2000 with the majority of fish (collected by electrofishing, drum netting and gill netting) being greater than 700 mm in length. Nevertheless, current research in flowing anabranch habitats indicates that these regions may provide a base level of Murray cod recruitment during years of sustained low or uniform (entitlement) flows in the South Australian reach of the Murray River.

A1.5.7 Migration and movements

The Murray cod have been considered to be a generally non-migratory and sedentary species (Cadwallader and Backhouse 1983; Reynolds 1983; Kailola et al. 1993; Humphries et al. 1999 Cadwallader 1977). This is generally so for part of the year when movement is limited and site fidelity high. Both lake and river fish have been shown to undertake substantial long-distance movements prior to spawning (Koehn 1997, 2006; Koehn and Nicol 1998). Fish tagged in Lake Mulwala (Murray River NSW) moved up to 100 km into the Murray and Ovens River systems prior to spawning, before returning to their original territory after several weeks. Homing occurred for about two-thirds of fish. Upstream movements may coincide with rising water levels, although some movement occurred without flooding. Individual fish commenced upstream movement from late winter to late spring, so not all fish in the same population move at the same time. Variation in movement patterns occurred between individual fish and larger movements for river fish were restricted to fish >65cm in length. Lake fish also moved further upstream than river fish (Koehn 2006). Several land-locked, lake-dwelling populations of Murray cod are known, so a spawning migration is not essential for spawning.

A1.5.8 Habitat selection and movement

Habitat selection and movements of the various life stages of Murray cod can be summarised in the Figs A1.1 and A1.2. Both adult and juvenile (age-0 onwards) Murray cod have been found to select similar habitats associated with variation in depth, structured woody habitat and overhanging vegetation, and were generally lower in depths and water velocities and closer to the banks (Fig. A.1, Koehn 2006). Larger fish were not more likely to be captured in deeper water and flood plain use was limited (Koehn 2006). While Murray cod movements have been found to be localised for part of the year, movement increased greatly during late winter and spring prior to spawning (Koehn 2006). The upstream movements were then followed by a downstream return to original locations (Fig. A.1 and Fig. A.2). Upstream migration undertaken by adults may compensate for downstream drift of larvae (Koehn 2006).
Figure A1.1: Components of the life cycle of Murray cod. SWH = structural woody debris; CVD variation in depth; OHV = overhanging vegetation; DNB = distance to bank (after Koehn 2006).

Figure A1.2: Timing of key components of the life cycle of Murray cod. Dashed line indicates that access is dependent on flows; dotted line indicates that some fish remain in the lake. Arrows indicate extended periods for some fish (after Koehn 2006).
A1.6 Threats

The threats to Murray cod have been summarised in several recent publications (Koehn 2005b; Lintermans et al. 2005; Rowland 2005; TSSC 2001). The Murray cod has declined throughout the Murray-Darling River system since European settlement, from causes including habitat loss and degradation, pollution, barriers to fish passage, flow regulation, cold water releases and fishing.

Environmental changes are probably the main cause of the substantial decline in abundance of Murray cod. Rowland (2005) suggested the decline of Murray cod in NSW had different causes at different stages. Overfishing by the commercial fishery between the late 1800s and 1930s caused the initial decline, then chemical pollution from agriculture in the early 1900s, predation by and competition from Redfin Perch Perca fluviatilis in the 1950s and 1960s; and reduced survival and recruitment of larvae and juveniles due to the effects of river regulation since the 1950s. Current recreational fishing pressure in some areas may be leading to unstable population structure (Nicol et al. 2005), while major kills of adult Murray cod, apparently from poor water quality, are still occurring in core parts of its range (Koehn 2005b). Management issues such as competing interests for water, a lack of ‘ownership’ of and lack of effective legal responsibility for, Murray cod, and the inadequate response to fish kills, especially in identifying causes and initiating remedial action, are also considered threats (Koehn 2005b; Sinclair 2005a). The lack of key biological information such as assessment of recruitment, mortality of different age classes, and the level of take, especially of large adults is hindering effective management for recovery (Koehn 2005b).

Reductions in native fish populations result from the interactions between inadequate flows, poor water quality, poor habitat and predation (Cottingham et al. 2001). In some cases, the actual threat may have ceased (e.g. commercial fishing), but its consequences are still being felt. In other cases, such as river regulation, the threat is sustained and on-going. Other threats are erratic and episodic, such as deteriorating water quality and indirectly from fires (greater erosion clogging rivers) causing fish kills. The cumulative impact of many small or low risk threats (e.g. fish kills, angler take, low water temperatures or lack of flooding reducing breeding success) can combine to further reduce population numbers and increase localised extinction risk through population fragmentation and incremental loss. Isolated populations are most at risk, and fragmentation of habitat reduces likelihood of recolonisation. Population fragmentation and incremental population loss decreases the chance of being able to recolonise after catastrophic events. Deviations from sustainable population structures such as through the disproportionate loss of breeding adults, for example, can add risk to long-term population viability. The Murray cod is a slow-growing, long-lived territorial predatory species. For a species with these life cycle characteristics, localised extinctions may continue to occur after the primary cause of decline has ceased to operate.

The major current and suspected threats impacting on Murray cod are detailed as follows.

A1.6.1 Flow Regulation

Flow regulation occurs where water is impounded and removed, or removed directly, from a river system. Many rivers in the Murray-Darling Basin have dams and weirs that regulate flow, and a substantial amount of water is abstracted from the Murray River system annually (10,800 GL/year) (Lintermans and Phillips 2004), through collection in impoundments, diversion through irrigation channels and direct pumping from rivers, largely for agricultural use. Flow regulation has greatly altered the natural flow regime of rivers. The consequences of this impoundment and removal of water from the river systems includes a reduction in flow rate and volume, extended periods of critical low flows and no flow, loss of flow variation and seasonality and loss of low to medium flood events. Upstream from the dam wall, there is permanent flooding, reduced flow and high water. In extreme cases the natural flow regime is now reversed, with low winter flows in rivers as water is contained within impoundments, and high flows in summer as water is released for irrigation. River regulation has also altered both the quality and availability of floodplain habitats such as backwaters and billabongs, due to reduced flooding.

The impact of river regulation and altered flow regimes is implicated in the decline of many Murray-Darling River system fish species (Murray-Darling Basin Commission 2004a). Most debates regarding the importance of floods and a natural flow regime for native fish involves the contribution flow makes to conditions that enhance recruitment. While the applicability of the Flood Pulse Concept (Junk et al. 1989) to Australia fishes, including Murray cod, has recently been questioned (Humphries et al. 1999), it has been suggested that recruitment success of Murray cod is directly linked to river flow, with a rise in water temperature and flood events being
key triggers for spawning and survival of young fish (Kearney and Kildea 2001; Ye et al. 2000; Rowland 1998). Reduc-
tions in flooding may be a major cause for the decline of Murray cod as a result of changes in suitable con-
ditions for spawning and larval recruitment (Rowland 1989). The impact on the native fish community in the 
Murray-Darling River system is thought to have been substantial (Murray-Darling Basin Commission 2004a).
Reduced flooding also affect the ability of fish to migrate, especially those species that undertake pre- or post-
spawning movements. Reduced flooding reduces the amount of habitat available, especially for the smaller 
species. Cooler water temperatures downstream from dams may inhibit spawning and slow growth (Koehn 
2001). Dams and weirs also act as barriers to fish movement. The potential for direct loss of native fish into 
irrigation channels and through pumps is unknown, but could potentially be relatively high (Koehn et al. 2004b; 
Koehn 2005a; Lintermans and Phillips 2004). This is a view supported by a preliminary investigation of the movements of tagged fish in Lake Nagambie (Goulburn River, Vic) (T. Ryan, pers. comm.). Despite the use of 
fish exclusion devices elsewhere in the world to prevent fish loss to irrigation systems, and the heavy reliance on irrigation water in the Murray-Darling Basin, no exclusion devices have been fitted to irrigation off takes to prevent fish loss (Blakeley 2004). River regulation has played a significant role in the decline of 
Murray cod since the mid-1950s as the optimum conditions for survival of Murray cod are much less frequent (Rowland 1985, 1989).

A1.6.2 Habitat degradation

Habitat degradation comes about through a variety of causes. Desnagging involves the removal, lopping or 
realignment of this structural woody habitat, to facilitate navigation, improve water flow, mitigate floods and 
protect assets such as bridges from flood damage due to debris jams forming. Murray cod are dependent on large structural woody habitat (snags: fallen tree trunks and branches, particularly River Red Gum Eucalyptus camaldulensis) for habitat and shelter. The removal of woody habitat has been widespread in Murray-Darling Basin rivers, particularly in lowland reaches over a large number of years (Gippel et al. 1992; Mudie 1961; Phillips 1972; Treadwell et al. 1999). Desnagging has undoubtedly reduced or destroyed prime habitat for adult Murray cod, and has also led to fragmentation of remaining available habitat. While desnagging as a regular activity has now largely ceased (except for specific instances where infrastructure such as bridges may be at risk), there is still considerable manipulation of snags through realignment, lopping and other river ‘improvement’ activities, and timber is continually removed from dry floodplain channels that are used by cod when the channels carry water (S. Nicol DSE-ARI pers. comm.). The cumulative effects of many manipulations over time is probably quite substantial, and the long-term affects of widespread desnagging may still be impacting Murray cod populations. Reinstatement of woody habitat is now recommended as a priority action for river restoration (Murray-Darling Basin Commission 2004a), and our understanding of its effects and fish-habitat relationships is increasing (Nicol et al. 2002).

Increased siltation through runoff after events such as land clearing and wildfires can have a major effect on isolated or stocked populations. In upland cod populations where cover is often provided by boulder or other hard substrate diversity and snags are naturally less abundant, sedimentation removes significant cover. Extensive wildfires in south-eastern Australia in the summer of 2003 burnt through several areas in the ACT and Victoria, and large amounts of sediment are now flowing into streams. An extensive fish kill occurred in the Buckland and Ovens Rivers (Vic) in March 2003 (J. Lyon DSE-ARI pers. comm.) after heavy rains fell over the fire area and washed enormous amounts of sediment and ash into the system. The infilling of undulations and holes by sedimentation may also impact on cod habitats and could blanket spawning substrates. Deposited sediments may also affect the abundance of food items such as plankton and insects associated with aquatic vegetation. Removal of riparian vegetation leads to reduced shelter, food and timber input into rivers and causes bank instability, leading to erosion and increased sedimentation. River regulation can also reduce habitat availability. Reductions in riparian vegetation result in reduced organic inputs including woody habitat (Hynes 1970). Incremental changes to habitats and changes to ecosystem processes, such as changes in overall river productivity (perhaps caused by a change in water temperature or nutrients trapped by dams) (McCully 1996), can indirectly and gradually affect fish populations.
A1.6.3 Reduced water quality

Reduced water quality can be caused by altered flows through diversion, impoundment or sustained dry periods reducing run-off. Consequences include excessively raised or lowered water temperatures, reduced dissolved oxygen levels, concentration of nutrients and environmental contaminants. Nutrient run-off from urban and agricultural areas can cause increased growth of phytoplankton, initiating plankton blooms and reducing oxygen levels. Fish kills can result from these conditions, and have become a depressingly regular feature in recent years. There were at least 21 fish kills in the Goulburn-Broken catchment (Vic) alone from 1998–2004 (ECOS 2004) while in New South Wales there were at least 34 fish kills per year between 1986–1996, with the real figure estimated to exceed 60–80 per year (Lugg 2000). Recent major fish kills involving Murray cod occurred in Broken Creek (Vic) in 2002, Ovens River (Vic) in 2003, Goulburn River (Vic) and Darling River (NSW) in 2004 (data from Koehn 2005b). At least 3000 adult Murray cod were killed in the Darling River kill, described as ‘the biggest cod kill in history’ (Sinclair 2005a). Suspended sediment, low oxygen levels, herbicides and altered water temperatures have all been suggested as possible causes of recent kills of thousands of native and introduced fish species, including large numbers of Murray cod (Koehn 2005b). These kills have probably been the result of a number of factors, exacerbated by extremely low (or no) flows, or sudden releases from dams of high temperature and low dissolved oxygen water, and have highlighted the fact that water quality problems remain a threat to this species. Modelling of the impact of the Darling River fish kill on the Murray cod population and options for management indicates that it will take decades for the cod population to recover, and will be extremely costly (Koehn 2005b). However recent compilation of survey data indicates populations in the lower Darling River appear to be in good condition (Gilligan, NSW DPI, pers. comm.).

While such kills provide a graphic reminder of the critical impact of water quality changes, non-critical changes are more common and may have greater overall impacts. High turbidity and salinity may also have adverse physiological or behavioural effects. Stratification may occur in pools due to temperature or salinity gradients, resulting in de-oxygenated, saline bottom layers (Anderson and Morison 1989). Increased salinity in the Murray-Darling Basin is a major problem causing extensive degradation in some areas. Salinity levels in the rivers and lakes vary widely, and the adults of many native fish species have at least a short-term tolerance to moderate to high salinity levels. However, early life history stages (e.g. eggs, larvae) are more sensitive to elevated salinity levels, and the long-term effects of sub-lethal levels of salinity on all life stages are unknown. Chotipuntu (2003) predicted that salinities above 0.34g/L would result in significant impacts on Murray cod. Elevated salinity levels may also affect food sources such as invertebrates, algae and macrophytes, consequently affecting habitat complexity and quality.

Water released from the bottom of large reservoirs may be up to 7–12°C cooler than ambient river water temperatures, especially over summer (Cadwallader 1978). Cold-water pollution from low-level releases from dams has been estimated to impact on at least 2800 km of waterways in the Murray-Darling Basin (Ryan et al. 2003), and this impact has been significant on species such as Golden Perch and Murray cod (Murray-Darling Basin Commission 2004a; Ryan et al. 2003). Reduced water temperatures may impair spawning, egg and larval survival, swimming speeds, feeding and growth rates, and favour potential predators and competitors such as the introduced Redfin Perch. Murray cod spawn at around 20°C. Juvenile Murray cod held at 24°C grew almost twice as long and 3.5 times as heavy as fish held at 13°C over a 3-month period (Ryan et al. 2003). Cold water pollution from low level outlets on dams may lead to localised extinctions downstream of large dams where water consistently fails to reach temperatures required for spawning of some species such as Golden Perch and Murray cod. The Goulburn River downstream from Lake Eildon is considered heavily stressed (EPA 2004), with cold water pollution a major contributor. Murray cod have become locally extinct in the Mitta Mitta River downstream of Lake Dartmouth since construction of the dam, most likely due to cold water releases (Koehn et al. 1995, Todd et al. 2005). It is unknown whether recent reports of captures of Murray cod in the lower Mitta Mitta River are from resident fish of fish migrating from Lake Hume. Water temperature and population modelling indicate that cold water releases from Lake Hume are likely to have detrimentally affected Murray cod populations in the Murray River downstream and remediation of this threat is likely to assist in rehabilitation of Murray cod populations (Sherman et al. 2007).

The input of pollutants and toxins to rivers may directly poison fish. Heavy metal poisoning from the Captains Flat mines caused the local extinction of Murray cod from the Molonglo River in the ACT (Lintermans 2002). Declines and local extinctions in northern NSW in the early 1900s have been linked to regular fish kills caused by agricultural chemicals (Rowland 2005). Herbicide use is widespread in the irrigation channel system in Victoria to keep them free of weeds, but this causes regular fish kills, including of Murray cod [EPA 2004; Sinclair 2005a], which may be a substantial threat given the magnitude of fish loss to irrigation channels.
Impacts of lesser known chemicals such as hormones from sewage effluents and their impact on fish breeding and sex ratios are unknown.

### A1.6.4 Barriers

Barriers to fish movements include dams, weirs, culverts, levee banks and areas of unsuitable habitat, high flow or turbulence. There are more than 3600 structures that can impede fish movements in the Murray-Darling Basin (Murray-Darling Basin Commission 2004a). Such barriers limit the ability of migratory fish species to complete their life cycle, and, even for non-migratory species, can limit the ability to colonise or recolonise suitable habitat, and can reduce gene flow by fragmenting populations. Barriers may also cause physical injury and/or mortality to drifting eggs and larvae, and may cause premature settling out in low flow areas immediately above barriers, subjecting them to unsuitable conditions reducing survival. Barriers have been recognised as a major threatening process operating throughout the Murray-Darling River system (Murray-Darling Basin Commission 2004a), and in many coastal waterways in eastern and southern Australia. Recent research has provided a greater understanding of the movement of Murray cod, with larvae having a nocturnal downstream drifting stage and some adult cod making substantial upstream and downstream movements of several hundred kilometres (Koehn 1997; Koehn and Nicol 1998; Humphries et al. 2002; King et al. 2003). Barriers may have a major impact on cod populations, interfering with pre and post spawning movements, and fragmenting and isolating populations from one another, leading to problems such as genetic drift and loss of genetic variability. A major program is underway in the Murray River system to facilitate fish passage past barriers, which should be of substantial benefit to the native fish of the Basin, including Murray cod. However, fishways facilitate predominantly upstream movement, and downstream movement may be a problem (Lintermans and Phillips 2004).

### A1.6.5 Alien Species

Eleven alien fish species are now established in the Murray-Darling River system (Murray-Darling Basin Commission 2004a), with Carp *Cyprinus carpio*, Redfin Perch *Perca fluviatilis*, Goldfish *Carassius auratus* and Eastern Gambusia *Gambusia holbrooki* the most widespread. Any impact on Murray cod from these alien species is likely to occur through a range of mechanisms including predation, competition, habitat alteration and spread of diseases and parasites. Carp receive a considerable amount of public attention and are often blamed for many of the ills of the river, such as poor water quality. Recent reviews of carp introduction and impact (Koehn et al. 2000; Koehn 2004) indicate that the Carp is a typical invasive species, which is resilient and well-adapted to exploiting riverine environments that are already degraded. Carp now comprise a majority of the fish biomass in the Basin (Harris and Gehlke 1997), and may comprise up to 90% of the fish biomass at some locations in the Murray River. In the recent Pilot Sustainable Rivers Audit, Carp were the most widespread species recorded, occurring at 63 of the 92 assessment sites across four river valleys (Murray-Darling Basin Commission 2004c). Recent surveys in NSW indicate that Carp now inhabit about 77% of NSW waterways, and a further 2% are also likely to be infested (Graham et al. 2005). The species has continued to disperse throughout the inland waterways; in the Murray-Darling Basin only some upper catchment areas are free of Carp. Despite public opinion, there is no scientific evidence that increases in carp have affected cod numbers (Koehn et al. 2000). At high densities Carp may increase turbidity and reduce aquatic vegetation through their feeding habits, reducing habitat for native species.

There is some correlation between high numbers of alien fish, especially Carp and Redfin Perch, and low numbers of native fish including Murray cod (Rowland 2005). The recent apparent increases in cod number in NSW coincide with historically low numbers of Carp and Redfin Perch. Predation by and competition with Redfin Perch in the 1950s and 1960s may have been a contributing factor to the decline of Murray cod in the southern part of Murray-Darling Basin during that time (Rowland 2005). Although Carp may compete with Murray cod for space, there is no evidence for any other form of competition between Murray cod and Carp, and young Carp may provide a source of food for Murray cod. Effects of other species that can reach very high densities, such as Eastern Gambusia and Oriental Weatherloach *Misgurnus anguillicaudatus*, are not known. Serious predation by Brown Trout *Salmo trutta* and Rainbow Trout *Oncorhynchus mykiss* on Murray cod is considered unlikely due to limited overlap in the habitats of these species (Koehn 2005a). Alien species are also suspected of introducing a number of parasites and diseases to Australia [see diseases section below]. However, while the impact of alien species is probably substantial, in some instances it can be difficult separating this from the other threatening process operating, especially the impact of flow regulation and the consequences for native fish habitats.
Appendix 1: A review of the ecological knowledge of Murray Cod

A1.6.6 Exploitation

A1.6.6.1 Commercial Fishing

Exploitation of Murray cod has occurred through both commercial and recreational fishing. The species was once common enough to support commercial fisheries, based mainly in Murray and Murrumbidgee rivers, which developed in the mid to late 1800s (Dakin and Kesteven 1938; Kailola et al. 1993; Kearney and Kildea 2001; Reid et al. 1997; Rowland 1985, 1989, 2005; Ye et al. 2000). The big operators used paddle-steamers as fishing boats, and Murray cod dominated the catch. Total catch peaked in the early 1900s, but by the 1930s had declined to unprofitable levels for the big operators, although a number of smaller operators continued fishing (Pollard and Scott 1966; Whitley 1937). There was a smaller peak in the fishery in the 1950s, when almost 300 tonnes of Murray cod per year was caught in NSW and SA, followed by a sharp decline in the commercial catch and a major decline in abundance of cod between 1955 and 1964 in NSW and SA (Reynolds 1976; Rowland 2005). Commercial fishing continued for another 40 years, but the catch declined to less than 10 tonnes/year in NSW in the 1990s. Concern over declining native fish stocks led to the closure of the commercial fisheries by 2003. Murray cod populations would have been very susceptible to commercial fishing on this scale, and the early decline was caused primarily by over fishing (Reid et al. 1997; Rowland 1989, 2005). There has been a noticeable recovery in year classes of Murray cod and other native fish species since the cessation of commercial activity in South Australia (Ye pers. comm.).

A1.6.6.2 Recreational Fishing

There remain a range of issues concerning recreational angling to ensure fishery sustainability and secure conservation objectives. There will be a continuation or potential enhancement of existing protection measures, including current regulations for size and bag/possession limits, seasonal closures of waters to fishing, provision of advice to anglers through signage and to recreational fishing guides, and patrols and inspections by fisheries officers to check compliance with regulations. There is already a major shift in angler attitudes, with improving angler ethics and conservation sentiment towards the Murray cod (Harris 2005). The Murray cod is a highly prized species among recreational anglers, and surveys suggest a high level of compliance with fishing regulations, and a growing trend among some anglers to practice catch and release (Park et al. 2005). The National Recreational and Indigenous Fishing Survey (Henry and Lyle 2003) estimated a release rate of 77.6% for Murray cod.

Murray cod is considered a premier freshwater angling species, and there is heavy recreational fishing pressure in virtually all of its range which has replaced the commercial fishery catch (Kearney and Kildea 2001). Estimated total legal catch of Murray cod is considerable. Data from the National Recreational and Indigenous Fishing Survey, in a 12 month period from March 2000 estimated that 106,000 cod weighing 216 tonnes were caught and retained, while another 368,000 cod were caught and released (Park et al. 2005). The previous commercial catch has now largely been replaced by increased take by recreational anglers. An expanding recreational fishery was probably responsible for a decline of cod numbers in central and northern NSW rivers in the 1970s and 1980s (Rowland 2005), while Harris (2005) cited over-harvesting is one of the current ‘greatest burdens’. The heavy fishing pressure on some sections of the Murray River is likely to be impacting on population structure of Murray cod (Nicol et al. 2005). In the reach between Tocumwal and Yarrawonga, angler take of fish at 50 cm (the minimum legal size) and above was an estimated 32%, and fish larger than 50 cm are rare, given the large number of fish in smaller size classes. In Lake Mulwala (a large impoundment on the Murray River upstream from Yarrawonga), considered a premier Murray cod fishery, fish caught in fishing tournaments averaged only 46–50 cm in length, with few fish taken in larger size classes (Park 2005). Although the Lake Mulwala population was considered to be self-sustaining (Park 2005), the removal of such a high proportion of size classes above 50 cm (likely to be of prime breeding age) may have severe impacts on population structure, and may not be sustainable for some populations, leading to population instability or crashes (Nicol et al. 2005). Recent radiotracking programs for Murray cod have indicated high numbers may be taken by anglers e.g. 19% of tagged fish (3 of 16 fish) taken from the Macintyre River near Goondiwindi (Andrew Berghius, DPI, pers. comm.) and 15% of tagged fish (5 of 32 fish) taken in the first year of monitoring in the Mullaroo Creek (Steve Saddlier, DSE-ARI, pers. comm.). This information comes from verified angler returns. The numbers of other fish potentially lost through illegal take is unknown.

There is also concern that the current minimum size of 50 cm does not allow cod to reach breeding age and breed at least once before being at the risk of capture by anglers and removed from the population. Some females reach maturity at age 4, but most reach maturity at age 5 (Koehn and O’Connor 1990), while some fish
at age 3 and 4 (generally pre-breeding age) are 50 cm long or larger (Anderson *et al.* 1992, Nicol *et al.* 2005), which suggests that 50 cm minimum size may be too small for some fish to breed. Some experienced fishers do target larger fish in the 20–40 kg size range (Rowland 2005), although an increasing number of these fish are released after capture.

High fishing pressure with minimum size limits can also apply selection pressure by favouring fish that mature at a smaller size, thus driving the population to ‘dwarfism’ (Conover and Munch 2002). The implications of this potential change on Murray cod is unknown.

The maintenance of adequate breeding stock is essential for the continuation of ‘wild stock’ populations and such populations are at greater risk with lower adult numbers and potentially lower fecundities from younger adult fish. Many populations however, especially in impoundments, are managed as ‘put and take’ fisheries where not all stocked fish may be required or expected to become sexually mature. The objectives for such populations should be made explicit with different objectives set for conservation management actions. While catch and release is becoming more commonly practiced by anglers, fish damage and mortalities associated with such releases remain unknown and should be determined so that any impacts on populations can be assessed. Poaching is still a serious problem in some areas (Kearney and Kildea 2001), and regular compliance patrols are still required. Due to the methods used to illegally harvest fish, particularly for commercial gain, take is non-discriminatory at both species and size levels. Death rates associated with these measures can be quite high, resulting in wastage. Regulating take means that size of fish, number of fish and gear used can be managed to minimise impacts on populations.

There were few regulations for recreational fishing for Murray cod prior to 1992, but all jurisdictions now have regulations governing cod fishing, including size and bag limits, and closed seasons (Lintermans 2005). It would be valuable to have a consistent regulatory regime between States and Territories to minimise confusion regarding regulations. The high release rate (77%) of Murray cod caught by anglers suggests good compliance with the legal minimum size (Park *et al.* 2005), and there is a growing trend among some anglers to practice catch and release. While many anglers do observe the fishing regulations and release undersize fish, Murray cod are quite sensitive to handling, and are very susceptible to fungal infections when handling removes skin mucous and scales. The impact of angling capture and release on the survival rate of released Murray cod is not known, but studies on other recreational freshwater fish species indicate post-capture mortality rates between 1% and 70% of released fish (Muoneke and Childress 1994). The effects of catch and release on future breeding success of captured Murray cod is also unknown.

**A1.6.6.3 Illegal Fishing**

Poaching of Murray cod and capture by illegal methods, including wire traps, set and cross lines, was considered to be a threat to some populations as long ago as the 1950s (Langtry, in Cadwallader 1977). Fisheries officers in South Australia and New South Wales report detecting hundreds of illegal traps each year (pers. comms., cited in Kearney and Kildea 2001). Illegal fishing methods, especially using drum nets, often target fish during the breeding season when they are more vulnerable, through increased activity associated with spawning such as pre-spawning movement, and large catches are taken in NSW through illegal fishing (Rowland 2005). The current illegal catch has not been quantified but is estimated to be very high, perhaps as high as or higher than the recreational fishery (Kearney and Kildea 2001, Lintermans and Phillips 2005). Take by illegal methods, especially wire traps, is indiscriminate and highly injurious to cod and other non-target species. Illegal, unreported and unregulated fishing is a significant threat to the sustainability of native freshwater fish resources including Murray cod, and also disadvantages legitimate users and the wider community. Notwithstanding the ban on commercial fishing and perhaps because of it, the illicit market demand for wild caught cod remains strong.

**A1.6.7 Stocking and Translocations**

Stocking and translocation of fish has been credited with the re-establishment of cod populations in several upper tributaries in northern NSW, after major declines and some local extinctions in the early 1900s (Rowland 2005). Principle concerns relating to stockings and translocations include the establishment of populations outside of their natural range, and the implications of release of hatchery produced fish, which have a limited genetic base, into natural systems.
Appendix 1: A review of the ecological knowledge of Murray Cod

A1.6.7.1 Translocations

Murray cod have been historically translocated into many areas, both within and outside their natural range, the latter translocations resulting in the establishment of several additional populations, that may be a threat to the fish and large invertebrate fauna of these areas, especially in the Cooper Creek system (J. Pritchard DSE-ARI pers. comm.). In some recent cases, Murray cod have been ‘rescued’ from lakes and rivers drying up and released into other waters, usually with little thought to any impact on fish populations in the receiving waters.

A1.6.7.2 Stocking

Stocking programs are primarily undertaken for recreational angling opportunities, not conservation. Stocking is a management option often suggested to reinforce reduced fish populations, and has been adopted as a management tool by fisheries agencies and angler organisations. Stocking of Murray cod fingerlings from hatcheries is currently an important management tool used to supplement or create cod fisheries across the Murray-Darling Basin, and an estimated 1,000,000 cod are stocked throughout the Basin each year, mostly in impoundments rather than rivers (Lintermans et al. 2005), although some stocking of weirs occurs. The majority of stockings have occurred in Victoria and New South Wales. The total number of Murray cod stocked prior to 2004 by State is: Victoria 2.89 million, NSW 2.85 million, ACT 405,160 and Qld 320,000 (Lintermans et al. 2005).

Stocking is often perceived as a ‘panacea’ to declining fish populations (Harris 2003), as it provides an easy management option that may result in deferring more difficult, expensive and controversial, but more effective management options. The effectiveness of Murray cod stocking has not been quantified, and while it is probably most effective in impoundments, it is riverine populations that are under threat (Koehn 2005a). Stocking can provide some positive consequences such as the recolonisation of areas affected by threatening processes in the past, where those processes have ceased to be detrimental on fish populations and areas have been rehabilitated. There are also positive social and economic benefits associated with stocking Murray cod.

Stocking is generally not a long-term conservation solution, as it may be ‘masking’ the true status of the species, and mask natural population recruitment levels. Its necessity highlights the fact that populations may not be sustainable under current exploitation rates or habitat conditions (Koehn 2005a; Lintermans et al. 2005). Stocking may also direct efforts away from more difficult but fundamental habitat improvement/threat amelioration activities that are necessary to achieve sustainable population levels without artificial enhancement.

A1.6.8 Genetic Issues

A major problem with translocation and stocking occurs through loss of genetic integrity and fitness from wild populations, and shifts in genotype due to swamping of remnant populations with hatchery-bred fish, often from a much narrower genetic base. Hatchery-produced fish from a narrow genetic base may adversely impact genetic diversity of wild populations, especially if hatchery fish ‘swamp’ remnant wild populations. The genetic diversity of Murray cod released from Victorian hatchery stockings in 2001/2 was found to be not representative of natural populations, with only 6 of 11 haplotypes present (Bearlin and Tikel 2003). Such genetic restriction may be more severe in non-government hatcheries, and would be further exacerbated by line-breeding for aquaculture for human consumption. There is currently no monitoring of genetic stocks of hatchery fish.

Genetic research is underway to develop genetics models that will evaluate the impacts of various hatchery and stocking practices for Murray cod (Brett Ingram, DPI, pers. comm.). As Murray cod are widely produced in hatcheries and stocked for recreational angling, and the genetic influence of these hatchery stocked fish has been considered a potential threat to the species (National Murray cod Recovery Team 2007). Murray cod populations in the Murray-Darling Basin are largely panmictic (one large population experiencing extensive gene flow) with most catchments being genetically similar (Rourke 2007). There are three exceptions to this, in that the Lachlan, Macquarie and Gwydir Rivers contained genetically distinct populations. These catchments all have large wetlands at their downstream reaches that may be responsible for this genetic differentiation. The Border Rivers (Beardy, Dumaresq, Namoi and Macintyre Rivers) also present a distinct cluster which may need to be considered separately (Rourke 2007).

The development and implementation of quality assurance and accreditation schemes for hatcheries in each State and Territory would help ensure that stockings of hatchery produced fish into the wild will not adversely affect the genetic diversity of natural populations and prevent the introduction of unwanted biological material into the wild. A Hatchery Quality Assurance Program (Rowland and Tully 2004) has been developed for NSW.
The collection of wild fish as broodstock is also an important issue that requires clear policy to ensure it is undertaken in a sustainable manner.

A1.6.9 Diseases

Very little is known about the prevalence and impact of diseases on Murray cod. However, most naturally occurring pathogens are unlikely to be a problem except to injured fish, or where water quality deteriorates and fish become highly stressed. Fungal infections occur on fish subject to rough handling on capture, and may reduce survival of fish released after angling capture. The major concern probably relates to those exotic diseases introduced to Australia with imported fish, and have found their way into the environment. Diseases and pathogens of potential major concern include the Epizootic Haematopoietic Necrosis (EHN) virus, Viral Encephalopathy and Retinopathy (VER), Goldfish Ulcer Disease (GUD), Asian Fish Tapeworm Bothrioccephalus acheilognathus and the parasitic copepod Anchorworm Lernaea cyprinacea. The introduced Redfin Perch carries EHN (Langdon et al. 1986), to which Murray cod and other native species such as Silver Perch and Macquarie Perch are highly susceptible [Langdon 1989; Langdon et al. 1986; Langdon et al. 1987; Murray-Darling Basin Commission 2004a]. A Murray-Darling Basin Commission project is currently underway investigating the susceptibility of native fish species to EHN and its epidemiology in the wild.

A new iridovirus has been detected in cultured Murray cod in Victoria but has not yet been detected in wild fish [Prof. Richard Whittington, pers. comm.; unpubl. data]. The abundance of some alien fish such as Carp and Eastern Gambusia may act as source for introduced pathogens such as Anchorworm and Asian Fish Tapeworm. Ectoparasitic protozoans including Chilodonella species, Ichthyophthirius species, Myxosoma species and Trichodina species are widespread and can be problematic in fish culture conditions [Ashburner and Ehl 1973; Langdon 1989; Langdon et al. 1986; Langdon et al. 1987; Rowland and Ingram 1991], but their occurrence or impact in the wild is unknown. Chilodonella infestation has killed adult trout cod kept at a hatchery [Ingram and Rimmer 1992] and has been suggested as a threat to wild populations [Douglas et al. 1994]. There is the potential to introduce disease to wild populations through the release of hatchery-bred fish. All hatcheries breeding Murray cod need to comply with National Policy for the Translocation of Live Aquatic Organisms guidelines [MCFFA 1999], requiring disease screening prior to release.

A1.6.10 Climate Change

The threat posed by climate change (‘global warming’) will potentially have significant and far-reaching impacts on the Murray-Darling River system. The consequences for much of south-eastern Australia (including the Murray-Darling Basin) are predicted to be an overall reduction in rainfall, less winter/spring rainfall, possibly increased summer rainfall, more frequent and increased length of dry periods, and an increase in the extent and frequency of extreme rainfall events. The potential increases in temperatures (both minimum and maximum) will also increase evaporation rates, so not only will less rainfall, with less runoff, but more surface water, especially from lakes and impoundments, will be lost to evaporation. All of this means less water in the rivers, especially at crucial times such as the spring-early summer breeding period for species such as Murray cod, and over summer. Such conditions are likely to potentially increase pressure on many native fish including Murray cod, through reduced flows and increasingly stressed rivers, with a much higher risk of fish kills during summer. During periods of drought where fish retreat to permanent water refugia, angling pressure may become focussed on these areas. Investigations of scenarios for freshwater fish from climate change and reductions in river discharge found that both could reduce freshwater biodiversity and have implications for survival of species [Xenopoulos et al. 2005].
Quantitative modelling is increasingly used as a tool to conceptualise ecological processes and provide a framework for hypothesis testing and the examination of management scenarios (Hilborn and Mangel 1997). The difficulty in building ecological models has been the trade-off between realism, generality and precision (Levins 1966). More recently, numerical methods allow inductive exploration of asymptotic behaviour (the position of maxima and minima) and transient behaviour (changes over time) of models as well as perturbation analysis (analysing the impact of disturbances or other forcing factors) (Odenbaugh 2005). It is reasonable to argue that describing any empirical relationship (e.g. a straight line relationship between food intake and growth in fish of a certain cohort) is a model. However the distinction is made here between describing a straight line univariate relationship and more complex multivariate or multi-component model more often referred to as imodelling in the literature. The strengths of modelling as opposed to describing empirical relationships, alone, lie in 1) the explicit nature of assumptions; and 2) recognition of the degree uncertainty affects inference, 3) sensitivity analysis to describe values of particularly influential parameters at which large or biologically important changes in response variables occur. These influential parameters must either be determined accurately or management alternatives have to be examined to decrease sensitivity.

Population and trophic modelling are powerful tools to analyse and describe fish ecology and are increasingly being used as a tool that can be used to aid management decisions regarding fish stocks and fish conservation. Although, these models are rarely used to directly test proposed or incumbent management actions or policy initiatives. More often they stop one step short and answer general ecological questions relating to general ecological threats or environmental variability. While the process of modelling is similar between both these types of studies, and they may both provide management recommendations, there are important differences, namely in defining management and policy options to be tested a priori rather than referring to management implications of model outputs post priori (Fig. A2.1). Where studies do test management options directly Adaptive Management is often used as a framework whereby each management scenario is presented as an alternative research hypotheses to be tested. Adaptive management provides a process to explicitly incorporate management into research and the analysis of data collected from research fed back into management (Fig. A2.2). Active adaptive management is seen as superior to more ad hoc methods of managing threatened or over-exploited species (Collie and Walters 1993; Beartin et al. 2002).

There is a need to assess the relative importance of management imperatives in driving population and trophic modelling investigations. This paper synthesizes inferences drawn from investigations that directly test alternative management or policy scenarios for fish management and conservation in freshwater and marine ecosystems.
Figure A2.1: Alternate research strategies when using models a) general exploration of ecological phenomena b) testing management options.

Figure A2.2: Steps in the assessment and management of a population. The block letters and solid arrows indicate the usual approach. Italic letters indicate the passive adaptive approach and the open arrows indicate the active adaptive approach (after Collie and Walters 1993).
Appendix 2: A review of approaches for test alternate management options

A2.1 Methods

We initially wanted to know the level of freshwater fish research in this area. We searched the Web of Science for topic “fish” and “model**” (not “marine”, “ocean”, or “sea”) and each of the following: “trophic model”, “ecosystem model”, “population model”, “stochastic population model” to ascertain the degree to which population and trophic models were used in fish biological investigations. To compare the number of investigations that were related to management with those that were not, we then applied the same search and added “manage**”. In order to assess how many of the management studies directly tested management options we searched within “manage**” for topics: “adaptive management”, “option**”, scenario**, and “alternate”. Acknowledging that much of the development of population and trophic modelling has occurred in marine fisheries studies, we also searched under “Carl Walters” as an author who has made a significant contribution to research on integrating ecosystem and population modelling and fisheries management over the past four decades as well as “A.E. Punt” who has contributed to population modelling in Australian marine fisheries.

A2.2 Results and Discussion

A2.2.1 Literature search

There were 9353 mainly non-marine papers investigating fish using models of some description. Of these 1512 (16%) mentioned population models 56(1%) investigated stochastic population models, 379 (4%) investigated ecosystem models and 338 (4%) investigated trophic models.

831 (9%) of the fish modelling papers were management related and of these 280 (34%) were population model-related, with 18 (2%) stochastic population modelling papers. 118 (14%) investigated ecosystem models and 49 (6%) investigated trophic models. Of the 168 trophic and ecosystem modelling papers 21 featured both ecosystem and trophic models. A further 121 papers were found in searches for ‘adaptive management’, ‘alternate’, ‘option**’ and ‘scenario’ combined.

Very few fish modelling papers were integrated with management; however when management was part of the investigation, population and ecosystem models were often used. More than double the proportionate number of population and ecosystem modelling papers were represented in the management category compared to the non-management category. Trophic models had equally low representation in both categories. Stochastic population modelling papers were rare but better represented in papers discussing management.

By assessing titles of management-related studies only, then reading abstracts of those studies that appeared likely to explicitly test management and policy options only nine papers were found that tested management options in freshwater systems.

A2.2.2 General modelling approaches

When quantitative models are used to directly test actual or proposed management scenarios in aquatic ecosystems they are heuristic: using often inaccurate or little information to develop rules or algorithms regarding the relationships between parameters (predictors) and response variables such as fish population size as well as finding the most likely response to different model inputs [scenarios]. Model estimates are not precise [within ±10% of true values]. Rather, model estimates show trends in response variables, directions and time scales of expected changes such that a management response can be determined (Walters et al. 2000). With little information, the algorithms, equations and parameter estimates may be inaccurate and so discrimination between hypotheses or management alternatives is usually qualitative rather than quantitative.

In each ecological discipline, modelling has developed in different ways with different modelling techniques being used to answer what are essentially similar questions. Heuristic population and trophic modelling have dominated fish ecology, having developed in an attempt to find solutions to over-exploitation of marine fisheries. Models include single-species models that simulate populations through time e.g (Bearlin et al. 2002; Sabo 2005) and energy budget models or bioenergetic models (e.g. Ecosim and Ecopath, Christensen and Walters 2004) that describe the flow of energy between components of the ecosystem. Holistic models incorporate features of both. These models can all be considered aggregated or p-state models (McDermot and Rose 2000), in which population abundances or biomass are represented as state variables and interspecific interactions are represented with relatively few, lumped parameters. In advocating the use of individual based or i-state models, McDermot and Rose (2000) present a set of limitations of aggregated modelling approaches:
1. budget models often assume equilibrium conditions, rather than temporally dynamic populations;
2. the lumped parameters to represent species interaction effects in coupled single-species models may lack biological meaning;
3. representing the mean attributes of fish populations (as in holistic models) may be misleading because the atypical individual, not the average individual survives;
4. for all of the aggregated approaches, population parameters with strong biological linkages (such as mortality rates) are difficult to estimate reliably; and
5. individual-level behaviours, such as movement, choice of feeding patch, and vulnerability to gape-limited predators (which can be important to fish population dynamics) are difficult to represent realistically in aggregated models.

One aggregated bioenergetic modelling approach that has been used extensively in marine systems is Ecosim with Ecopath (EwE), with 2400 registered users in 120 countries and leading to in excess of 150 publications as of 2004 (Christensen and Walters 2004). EwE combines software for ecosystem trophic mass balance (biomass and flow) analysis (Ecopath) with a dynamic modelling capability (Ecosim) for exploring past and future impacts of fishing and environmental disturbances [Christensen and Walters 2004]. We mention it briefly here because while we were unable find an example of it being used to test management options in freshwater systems, it seems to provide potential to be used in this way.

Population modelling approaches may be single species (Bearlin et al. 2002; Sabo 2005), multispecies (McDermot and Rose 2000; Marttunen and Vehanen 2004), continuous or discrete, deterministic or stochastic (Bearlin et al. 2002) or both (Sabo 2005). It is possible to model whole populations or individuals and metapopulations at regional or patch scales. Population modelling includes population viability analysis (Jager 2006a; b), virtual population analysis (Marttunen and Vehanen 2004), and matrix population analysis (Kareiva et al. 2000).

### A2.2.3 Features of models used to test management and policy options

#### Simulation

Whether aggregated or individual-based models were used to test management options, simulation was a feature in all studies reviewed here. By using simulation modelling rather than on ground experimentation, substantial short term cost and impacts on a number of users of the resource or surrounding habitat are initially avoided while building evidence in support of alternate management or policy options (Collie and Walters 1993).

#### Parameters

The models were built using a number of parameters. Typically bioenergetic models had many more parameters than population models. The objective of parsimony is not applied to bioenergetic models. Often parameters in bioenergetic models were divided into compartments. Each compartment represented a part of the ecosystem which often had an internal closed dynamic (sets of equations describing the variability in parameters and response variables over time) and then multiple relationships describing interactions between components. In order to create full stochastic models, each parameter requires a coefficient of variation and distribution. Most modelling exercises are deterministic due to the lack of data on variation even for very well studied systems (e.g. Kareiva et al. 2000).

Typical parameters of populations models include:
- Population growth rate
- Recruitment rate
- Mortality or survival rates of different cohorts
- Immigration rates
- Emigration rate
- Duration of larval stage
- Growth
- Sex ratio
- Average size or age at maturity
- Relationship between fecundity and size or age
- Measure of density effects
Appendix 2: A review of approaches for test alternate management options

The physiological parameters in standard bioenergetics models may be large and may include:

- Production
- Respiration
- Biomass accumulation
- Egestion
- Excretion
- Decomposition
- Conversion efficiency
- Consumption
- Handling time
- Food intake per unit biomass
- Ecotrophic efficiency
- Individual growth
- Mortality
- Recruitment
- Immigration
- Emigration

A2.2.4 Examples of models used to test management hypotheses.

Nine studies explicitly tested management options for freshwater fish using modelling. All of these studies used simulation modelling rather than experimental management and most then recommended the implementation of management options based on simulation. None of the papers outlined an experimental design to test the options indicated by simulation modelling. Management options explored for single species included both augmentation of threatened species (Bearlin et al. 2002; Jager 2006) and control of pest species (Brown and Walker 2004; Sabo 2005). Multiple species models and bioenergetic models were used to explore tradeoffs between different fish species of interest to different stakeholders (Marttunen and Vehanen 2004) or ecosystem and social impacts (McDermot and Rose 2000; Guneralp and Barlas 2003).

A2.3 Conclusion

Population modelling can be a valuable tool in species conservation or invasive species control when detailed information is lacking and attaining such information may take years or decades (Christensen and Walters 2004). Models explore the effects of life history such as migratory behaviour (Jager 2006a; b), growth rate (Bearlin et al. 2002) or environmental variability (Brown and Walker 2004) on persistence. This information can be used to choose between management options when they are explicitly tested and can also indicate new management options not previously considered. In addition, sensitivity analysis can identify the life history stages critical for population growth and can guide conservation actions which concentrate on these stages (Kareiva et al. 2000).
Table A2.1: Details of models used to test management options for fish exploitation and conservation in freshwater systems

<table>
<thead>
<tr>
<th>Model output/variables</th>
<th>Models Used</th>
<th>Tested management options</th>
<th>Sensitivity analysis</th>
<th>Conclusions (quoted directly from papers)</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Population size</td>
<td>Discrete time stochastic population model</td>
<td>Compare effects of stocking rates and alternative growth rates in success of reintroduction (recruitment). Also monitoring effort and management risk profile.</td>
<td>Alternative growth rates</td>
<td>Under low risk an 8 year reintroduction performed best but as tolerance of risk increased shorter (higher numbers for each reintroductions were preferred. Monitoring was so uncertain that there was almost no ability to detect success or failure.</td>
<td>Bearlin, et al. (2002)</td>
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<tr>
<td>pseudo extinction, biomass %change min, biomass %change max</td>
<td>Stochastic simulation (PVA)</td>
<td>Compare effects of daughterless carp, spawning sabotage, unselective fishing versus selective for mature age fish.</td>
<td>Tested effects of climate and hydrological variance and scale and shape parameters if stock recruitment relationship.</td>
<td>Fast growing short lived populations may respond to daughterless carp or spawning sabotage whereas longer lived populations may respond better to removal-type management options. Unselective removal by poisoning or trapping all age classes increased pseudo-extinctions while selective removal only reduced biomass to 60% below pre management biomass where reductions to €10% are required to establish stable low growth rate [Thresher 1997].</td>
<td>Brown and Walker (2004)</td>
</tr>
<tr>
<td>Model carp yield under regulated and unregulated conditions. Chla, Crayfish population, fishing workforce, macrophyte cover, water chemistry</td>
<td>Dynamic [Bioenergetic]</td>
<td>1. Dam vs. no dam on lake. 2. Regulation – compare water levels. 3. Irrigation region extended 4. Increased agricultural yield 5. Collapse of crayfish due to disease (real vs. no collapse [hypothetical] and reintroduction of crayfish [mg option]). 6. Macrophytes clearance vs. no clearance</td>
<td>No</td>
<td>1–3. Dam has largest ecosystem effect. 4. Increase in agricultural production has no ecosystem effect and positive social effect. 5. Reintroduction of crayfish fail due to disease. 6. Macrophyte clearance causes dominance of algae [i.e., algal blooms], high turbidity, the loss of pike [fishery] and the establishment of a stable carp population.</td>
<td>Guneralp and Bartas (2003)</td>
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<tr>
<td>1. probability of downstream migration 2. population size 3. metapopulation status</td>
<td>Population viability analysis</td>
<td>Compared screening size at turbine and fishway for downstream passage. Compare value of upstream and downstream passage.</td>
<td>Density dependent effects of upstream passage.</td>
<td>Status of fish without upstream passage is higher in river configurations with high interspersion of source and sink segments and a source segment far upstream. Therefore the configuration of dams can influence restoration efforts. Upstream passage alone did not change status of fish needed to be augmented by protecting fish from turbines through screening or downstream passage.</td>
<td>Jager (2006a)</td>
</tr>
<tr>
<td>1. population size 2. metapopulation status</td>
<td>Population viability analysis</td>
<td>6 scenarios: 1. baseline: wide screening, 2. narrow screening, 3. downstream passage, 4. translocation with wide screen, 5. translocation with narrow screen 6. translocation with downstream passage.</td>
<td>Effort vs quota based capture</td>
<td>Translocation did not have clear benefits unless accompanied by slowing emigration (narrow screening of downstream passage) or allowing safe emigration (providing downstream passage). Translocation had highest benefits when the sink populations was in a long river segment [not habitat limited] and positioned upstream where the population could then be a source for multiple short segments [habitat limited] down stream.</td>
<td>Jager (2006b)</td>
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### Appendix 2: A review of approaches for test alternate management options

<table>
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</thead>
<tbody>
<tr>
<td>Rate of population decline</td>
<td>Age structured matrix model</td>
<td>1. Eliminate migration mortality (all fishing etc). 2. Reduced harvest rated from 50% to 10%</td>
<td>Wide range of parameter values tested.</td>
<td>Elimination on migration mortality does not reverse decline of stocks. Without the options 2-4, populations would have declined at a rate of 50-60% and gone to extinction. First year mortality to be reduced by 11% and ocean estuarine mortality by 9% to reverse population decline Conclusions are robust to a wide range of parameter values.</td>
<td>Kareiva, et al. (2000)</td>
</tr>
<tr>
<td>Average catch Average size of each species. Employment rate of fisherman. Cash flow in region.</td>
<td>VPA Bioenergetics Multispecies</td>
<td>1. Recreational fishery – to increase the catch and size of predatory fish and their using gill net size restriction 2. Combined fishery no gill net restriction and 3. Commercial fishery – maximize the vendace catch by professional fishermen with no gillnet restriction. Each scenario had alternate stocking rates for 4 species.</td>
<td>Tested natural mortality and changes in minimum and maximum mortality of stocked individuals.</td>
<td>Combined scenario (2) has most positive impact on cashflow. The results also showed that fishing regulations for gill nets of 27–50 mm improved the catch and average size of predatory fish about 30%– 40%. Reduce stocking rate of species displaying density dependent yield. Adjust stocking rate of predatory fish to abundance of its primary prey species.</td>
<td>Marttunen and Vehanen (2004)</td>
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<td>Predicted annual species biomasses, planktivore biomass, and total zooplankton consumption over time. Lowered zooplankton consumption implies less algae.</td>
<td>Individual based generic fish food web simulator (Bioenergetic)</td>
<td>1. compare different stocking regimes, 2. compare low and high interannual variation 3. minimum fish size 4. fish die-off event</td>
<td>No</td>
<td>Neither periodic nor continuous stocking generated the desired responses in the long-term either having little effect on zooplankton consumption or resulting in significant changes in piscivore mean lengths or in the composition of the fish community. Continuous high level stocking resulted in the desired lowered zooplankton consumption Under enhancement and the one species die-off, the low variability simulation returned to baseline more quickly. Higher Daphnia turnover rates and lower predator effectiveness characteristic of the high variability calibration resulted in faster planktivore numerical response rates and slower piscivore numerical response rates leading to one species extinction and replacement by another. The greater difference between prey and predator response rates increased the time lags of predator responses and resulted in increased variability.</td>
<td>McDermot and Rose (2000)</td>
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<tr>
<td>Quasi extinction threshold (variation in prey abundance) and prey carrying capacity</td>
<td>Deterministic consumption Stochastic and deterministic growth</td>
<td>Compared “do nothing” with proportional, trigger and total eradication of introduced predator.</td>
<td>Tested introduced predator process noise attack rate, harvest intensity on variation in the abundance of native prey populations.</td>
<td>Variability in predator growth rates has largest impact on prey quasi extinction and prey carrying capacity. Threshold trigger harvest is best management option (= to eradication) and better than proportion harvest of pest predator. The efficacy of all three management actions varies with the attack rate of the predator, and thus the stability of the deterministic dynamics. The predator can essentially be ignored during years of low abundance as long as numbers are reduced to some pre-defined threshold during outbreak years. By harvesting only in years when predator abundance is high, conservation organizations can potentially reduce management effort (and potentially costs) without sacrificing efficacy.</td>
<td>Sabo (2005)</td>
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<td>Model output (response variables)</td>
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<td>Conclusions (quoted directly from papers)</td>
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<td>Proportional response of egg and larval replicates; post-spawning survival; The average female adult population size quasi-extinction risk</td>
<td>Stochastic population model.</td>
<td>Did not compare management scenarios but models were provided that &quot;can help identify the magnitude of the change required to ameliorate altered thermal regimes for the benefit of a freshwater fish population.&quot;</td>
<td>Sensitivity to changes in carrying capacity and survival rates.</td>
<td>A sudden drop in egg and larval survival is likely to occur between 12.8°C and 16°C.</td>
<td>Todd, et al. (2006)</td>
</tr>
<tr>
<td>Proportional response of egg and larval post-spawning survival; average minimum population size</td>
<td>One-dimensional hydrodynamic reservoir model and stochastic population model.</td>
<td>Compared business-as-usual (Base) with mitigation of cold water pollution using selective withdrawal (SW) scenarios.</td>
<td>Sensitivity to changes in carrying capacity, spawning Schedule, and initial population size.</td>
<td>SW was predicted to increase the average minimum population size of early-season spawners by 360%, 240% and 110% over the business-as-usual case for spawning commencing on 11 October, 26 October and 10 November, respectively. For mid-season spawners, the corresponding increases were 200%, 70% and 25%. If the population are early spawners, the threshold population is predicted to increase by a factor varying from 2 to 4 depending on spawning commencement date. If they are mid season spawners threshold population is likely to increase by a factor of 0.25.</td>
<td>Sherman, et al. (2007)</td>
</tr>
<tr>
<td>Catch_il and Likelihood of depletion to minimum [250t]</td>
<td>Schaefer production model and the non-bootstrap ad hoc tuned VPA.</td>
<td>Compared performance of Schaefer production model with the non-bootstrap ad hoc tuned VPA.</td>
<td>Sensitivity to the standard deviation of extent of observational Error the catch-at-age for each fishery.</td>
<td>The results indicate that, even though yields from the fishery are likely to be low, the benefits of conducting annual surveys exceed the costs. The value of collecting age-composition data is less clear. Management procedures based on Virtual Population Analysis achieve more variable catches and are less likely to satisfy AFMA performance criteria than management procedures based on a Schaefer production model, but former achieve higher levels of &quot;guaranteed&quot; catch for the industry.</td>
<td>Punt and Smith (1999)</td>
</tr>
<tr>
<td>Spawner biomass</td>
<td>Multi-fleet stock assessment model based on the 'integrated analysis' approach</td>
<td>Compared catch limits of 1000 to 20000t per annum over the next 20 years in 'spawning' sub-fishery that operates on the spawning stock in winter and 'non-spawning' subfisheries that includes all other catches in the south-eastern fishery.</td>
<td>Looked at the sensitivity of the model to abundance, egg production estimates, discard (smaller fish discarded)</td>
<td>The model indicated that the 'spawning' and 'nonspawning' subfishery have notably different vulnerability patterns [Fig. 4] and that vulnerability drops off with length for the 'non-spawning' subfishery. Catch rates of blue grenadier were influenced by the year-class strength and the different populations harvested. The probability of dropping below 40% of base virgin (female) spawner biomass over the next five years is less than 10% for catches &lt;-10 000 t for all situations except that in which length-at-age follows a von Bertalanffy growth curve. In contrast, if a 20-year time horizon is considered, a 10000 t catch limit leads to a &gt;50% probability of dropping below 40% the virgin (female) spawner biomass if the 'halve egg estimate' situation is correct.</td>
<td>Punt, et al. (2001)</td>
</tr>
</tbody>
</table>
### Appendix 2: A review of approaches for test alternate management options

<table>
<thead>
<tr>
<th>Model output (response variables)</th>
<th>Models Used</th>
<th>Tested management options</th>
<th>Sensitivity analysis</th>
<th>Conclusions (quoted directly from papers)</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biomass</td>
<td>Multispecies virtual population analysis and Ecosim</td>
<td>Management options not tested. Compared modelling methods in explaining variation in biomass.</td>
<td>No</td>
<td>Ecosim outputs closely reproduced MSVPA biomass estimates and catch data for sprat (Sprattus sprattus), herring (Clupea harengus), and cod (Gadus morhua), but only after making adjustments to cod recruitment, to vulnerability to predation of specific species, and to foraging times. Fishing, the chief source of mortality for cod and herring, and cod reproduction, as driven by oceanographic conditions as well as unexplained variability, were also key structuring forces of models. Biomass changes at higher trophic levels caused by fishing, trophic interactions, and/or variable recruitment, influenced lower trophic levels, similar to trophic cascades observed in some freshwater-lake food webs. Biomass changes at higher trophic levels caused by fishing, trophic interactions, and/or variable recruitment, influenced lower trophic levels, similar to trophic cascades observed in some freshwater-lake food webs. (Harvey et al. 2003)</td>
<td>[Harvey et al. 2003]</td>
</tr>
<tr>
<td>CPUE and fishing mortality</td>
<td>Age structured production</td>
<td>Management options not tested. Compared fishing gear</td>
<td>No</td>
<td>Despite low productivity of the Pacific ocean tuna fisheries have sustained large yields. Blue Marlin is declining the most at 20% 1950's levels. Tuna on the other hand do not decrease below 40-50% 1950s levels. Albacore has been at low levels but has increased to historical levels. Bigeye and yellow fin tuna are experiencing increasing productivity. (Cox et al. 2002)</td>
<td>[Cox et al. 2002]</td>
</tr>
<tr>
<td>CPUE and fishing mortality</td>
<td>Multispecies model using Ecospath and Ecosim</td>
<td>Management options not tested. Relative contribution of fishing and trophic impacts on tuna dynamics.</td>
<td>No</td>
<td>Decline in predation mortality owing to depletion of predators was greatest for small yellow fin tuna and may account for apparent increases in biomass. For other Tunas predicted changes in predation mortality rates were small or overwhelmed by large increases in fishing mortality. (Cox et al. 2002)</td>
<td>[Cox et al. 2002]</td>
</tr>
</tbody>
</table>
APPENDIX 3: TROPHIC INTERACTION MODEL
FOR MURRAY COD

Trophic components and interactions in the Murray-Darling River ecosystem are essential to all biological processes, such as energy transfer, organic matter breakdown and population dynamics. They affect how fish communities change and respond to changes. Trophic interactions are likely to be an important driver of Murray cod populations. A trophic interaction model helps to place the management of Murray cod in an ecosystem context.

Body size is central to the structure and function of food webs (Elton 1927; Cohen et al. 2003; Woodward et al. 2005). Many biological properties of individuals including metabolic rate, growth rate and productivity, natural mortality and lifespan, and spatial niche are correlated with body size (Peters 1983; Elser et al. 1996). Furthermore, body size contributes to determining predator-prey interactions (Woodward and Hildrew 2002). These interactions are fundamental to population dynamics as all organisms are prey at least at some stage of their life cycle. Predators are typically larger than their prey because prey size is limited by the allometric diameter of predator’s mouth. The significance of size-based predation and the large scope for fish growth means that body size is often a better indicator of trophic level than species identity. Because it captures so many aspects of ecosystem functioning, body size can therefore be used to synthesize a suite of co-varying traits into a single dimension. In this perspective, we examine trophic components and interactions of the food web associated with Murray cod, based on the assumption that body size rather than taxonomy is the principle descriptor of an individual’s role in the food web.

Adult Murray cod can grow to a large size (up to 180cm total length TL and 113.5 kg) and have a large mouth gape (Harris and Rowland 1996). Murray cod is classified as an apex aquatic predator (Rowland 2005; Ebner 2006; Baumgartner 2007). We constructed a size-structured food web of the trophic interactions associated with Murray cod (Fig. 1). The size-structured food web is characterized by larger predators eating smaller prey, with similar-sized predators occupying almost the same position within the food web, regardless of their species identity. This takes into account predatory interactions, the changes in these interactions with body size, and the life stages of Murray cod. In this construction of the food web model, we separate juvenile Murray cod from other fish in the 10–30 cm size group. Different models could be constructed by separating juvenile Murray cod from the 30–60 cm and the <10 cm size group.

Qualitative modelling provides a means to confront complexity in ecosystems, and is especially useful where trophic components and interactions of the food web are known but not quantified. To understand the response of the fish in one size group to changes of the fish in other size groups, qualitative analysis is an appropriate starting point. Loop analysis, one technique of qualitative modelling systematically developed by Levins (1974; 1975), consists of the analysis of signed digraphs [directed graphs] representing whether increases in one variable induce qualitative increases or decreases in other variables, or leave them unchanged. In this study, we use the matrix algebra method for loop analysis formulated by Dambacher et al. (2002; 2003). This method is able to predict the direction or sign of response to perturbations rather than the magnitude of response. It is based on the relative degree of ambiguity of a response, as defined by the countervailing number of complementary feedback cycles contributing to the response. The certainty of response predictions is scaled by the ratio of the net and absolute number of complementary feedback cycles.
Figure A3.1: A schematic Murray cod food web. Arrows denote energy fluxes and their direction. Only Murray cod is in the >90 cm size group. Murray cod, trout cod and carp are in the 60–90 cm size group. The 30–60 cm, the 10–30 cm and the <10 cm size groups include trout cod, golden perch, and silver perch. In addition, Murray cod, carp and redfin are in the 30–60 cm size group and Murray cod, gudgeons, smelt, hardyheads and gobies are in the <10 cm size group. Secondary producers are zooplankton and macroinvertebrates.
Fig. A3.2: A signed digraph of the schematic Murray cod food web. Links terminating in an arrow denote positive effects, while links terminating in a filled circle denote negative effects. Links that connect a variable to itself denote self-regulation (negative) feedbacks. 1, >90 cm size group; 2, 60–90 cm size group; 3, 30–60 cm size group; 4, juvenile Murray cod; 5, 10–30 cm size group; 6, <10 cm size group; 7, secondary producers; 8, primary producers.
From the schematic Murray cod food web in Fig. A3.1, we construct a signed digraph (Fig. A3.2). The resulting qualitatively specified community matrix is:

$$
\begin{bmatrix}
-1 & 0 & 0 & 1 & 1 & 1 & 1 & 0 \\
0 & -1 & 0 & 1 & 1 & 1 & 1 & 0 \\
0 & 0 & -1 & 0 & 0 & 1 & 1 & 0 \\
-1 & -1 & 0 & -1 & 0 & 1 & 1 & 0 \\
-1 & -1 & 0 & -1 & 1 & 1 & 1 & 0 \\
-1 & -1 & -1 & -1 & -1 & 1 & 1 & 1 \\
0 & 0 & 0 & 0 & 0 & 0 & -1 & 0
\end{bmatrix}
$$

whose elements represent positive [+1], negative [-1] and null [0] effects. The net and absolute number of complementary feedback cycles contributing to response in system variables can be calculated from the qualitatively specified community matrix using matrix algebra.

The net number and sign of complementary feedback cycles is detailed in the adjoint matrix:

$$
\begin{bmatrix}
a_{11} & a_{12} & a_{13} & a_{14} & a_{15} & a_{16} & a_{17} & a_{18} \\
a_{21} & a_{22} & a_{23} & a_{24} & a_{25} & a_{26} & a_{27} & a_{28} \\
a_{31} & a_{32} & a_{33} & a_{34} & a_{35} & a_{36} & a_{37} & a_{38} \\
a_{41} & a_{42} & a_{43} & a_{44} & a_{45} & a_{46} & a_{47} & a_{48} \\
a_{51} & a_{52} & a_{53} & a_{54} & a_{55} & a_{56} & a_{57} & a_{58} \\
a_{61} & a_{62} & a_{63} & a_{64} & a_{65} & a_{66} & a_{67} & a_{68} \\
a_{71} & a_{72} & a_{73} & a_{74} & a_{75} & a_{76} & a_{77} & a_{78} \\
a_{81} & a_{82} & a_{83} & a_{84} & a_{85} & a_{86} & a_{87} & a_{88}
\end{bmatrix}
= 
\begin{bmatrix}
9 & -5 & -3 & 1 & 1 & 3 & 0 & 6 \\
-5 & 9 & -3 & 1 & 1 & 3 & 0 & 6 \\
1 & 1 & 9 & -3 & -3 & 5 & 0 & 10 \\
-3 & -3 & 1 & 9 & -5 & -1 & 0 & -2 \\
-3 & -3 & 1 & -5 & 9 & -1 & 0 & -2 \\
1 & 1 & -5 & -3 & -3 & 5 & 0 & -4 \\
0 & 0 & 0 & 0 & 0 & 0 & 0 & 14 \\
0 & 0 & 0 & 0 & 0 & 0 & 14 & 28
\end{bmatrix}
$$

The off-diagonal element $a_{ij}$ ($i \neq j$) of the adjoint matrix predicts the direction or sign as well as the relative magnitude of the response of size group $i$ to positive input to size group $j$.

The absolute number of complementary feedback cycles is detailed in the absolute-feedback matrix:

$$
\begin{bmatrix}
17 & 13 & 11 & 17 & 17 & 11 & 0 & 86 \\
13 & 17 & 11 & 17 & 17 & 11 & 0 & 86 \\
11 & 11 & 53 & 11 & 11 & 9 & 0 & 106 \\
17 & 17 & 11 & 17 & 13 & 11 & 0 & 86 \\
17 & 17 & 11 & 13 & 17 & 11 & 0 & 86 \\
11 & 11 & 9 & 11 & 11 & 9 & 0 & 62 \\
0 & 0 & 0 & 0 & 0 & 0 & 0 & 62 \\
86 & 86 & 106 & 86 & 86 & 62 & 62 & 574
\end{bmatrix}
$$
Dividing the absolute value of each element of the adjoint matrix by each corresponding element of the absolute-feedback matrix yields the weighted-predictions matrix:

\[
\begin{bmatrix}
    w_{11} & w_{12} & w_{13} & w_{14} & w_{15} & w_{16} & w_{17} & w_{18} \\
    w_{21} & w_{22} & w_{23} & w_{24} & w_{25} & w_{26} & w_{27} & w_{28} \\
    w_{31} & w_{32} & w_{33} & w_{34} & w_{35} & w_{36} & w_{37} & w_{38} \\
    w_{41} & w_{42} & w_{43} & w_{44} & w_{45} & w_{46} & w_{47} & w_{48} \\
    w_{51} & w_{52} & w_{53} & w_{54} & w_{55} & w_{56} & w_{57} & w_{58} \\
    w_{61} & w_{62} & w_{63} & w_{64} & w_{65} & w_{66} & w_{67} & w_{68} \\
    w_{71} & w_{72} & w_{73} & w_{74} & w_{75} & w_{76} & w_{77} & w_{78} \\
    w_{81} & w_{82} & w_{83} & w_{84} & w_{85} & w_{86} & w_{87} & w_{88}
\end{bmatrix}
\begin{bmatrix}
    0.53 & 0.38 & 0.27 & 0.06 & 0.06 & 0.27 & 1 & 0.07 \\
    0.38 & 0.53 & 0.27 & 0.06 & 0.06 & 0.27 & 1 & 0.07 \\
    0.09 & 0.09 & 0.17 & 0.27 & 0.27 & 0.56 & 1 & 0.09 \\
    0.18 & 0.18 & 0.09 & 0.53 & 0.38 & 0.09 & 1 & 0.02 \\
    0.18 & 0.18 & 0.09 & 0.38 & 0.53 & 0.09 & 1 & 0.02 \\
    0.09 & 0.09 & 0.56 & 0.27 & 0.27 & 0.56 & 1 & 0.06 \\
    1 & 1 & 1 & 1 & 1 & 1 & 0.23 \\
    0 & 0 & 0 & 0 & 0 & 0 & 0.23 & 0.23 & 0.05
\end{bmatrix}
\]

Each element of this matrix scales the probability for sign determinacy of response predictions in the adjoint matrix. It has been shown that the prediction weight of 0.5 is a general threshold for sign determinacy (Dambacher et al. 2003). Above 0.5 the probability of predicting the correct sign of response is generally greater than 90%.

The qualitative trophic interaction model predicts a neutral effect on each other between secondary producers and all the groups with higher trophic levels. This prediction having the prediction weight of 1 is expected to be completely reliable in terms of their response sign or direction. The only other predictions with high sign determinacy other than self effects are that a sustained increase in fish abundances in <10 cm size group results in an increase in fish abundances in 30–60 cm size group (corresponding to \( a_{36} = 5 \) in the adjoint matrix with the prediction weight \( w_{36} = 56 \)) and, conversely, a sustained increase in fish abundances in 30–60 cm size group results in a decrease in fish abundances in <10 cm size group (corresponding to \( a_{36} = -5 \) in the adjoint matrix with the prediction weight \( w_{63} = 0.56 \)). One of the implications of these predictions is that an increase in alien fish, especially carp and redfin perch, with their size between 30 cm and 60 cm may reduce the abundance of young Murray cod <10 cm.

The trophic interaction model developed here makes a start on the modelling to support ecosystem-based fish management as an alternative approach to single-species fish management. Improving modelling trophic interactions for Murray cod requires enhancing knowledge, understanding and synthesis of trophic components and interactions in the Murray-Darling Basin ecosystem. Quantitative information on biological rates, strengths of trophic interactions and fishing and environmental effects are crucial for quantifying forcing factors. More broadly, a comprehensive ecosystem model for Murray cod should provide full coverage of ecosystem components [such as predators, prey, habitat and flow] and interactions linking these components, and the integration of physical and biological processes at different scales.
APPENDIX 4: THE ADDITION OF LONGEVITY AND FECUNDITY DATA TO THE MURRAY COD MODEL

A4.1 Introduction

Murray cod Maccullochella peeli peeli is a large, iconic Australian freshwater fish that is also a popular angling species. Like many native fish species in the Murray-Darling Basin, Murray cod populations have suffered extensive declines and was listed as Vulnerable under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act).

The objectives of this Appendix are:

- to determine whether large female and male Murray cod are capable of producing viable gametes for spawning.
- to assess the longevity and absolute fecundity of female Murray cod.
- to investigate whether large female Murray cod could potentially contribute to improved spawning success with larger diameter egg sizes.
- to recommend management actions to sustain Murray cod populations based on their longevity and breeding potential.
- to build greater certainty into age-length relationships for large (>1m) Murray cod.

A4.2 Methods and Results

To date, otoliths and gonads from 86 frozen Murray cod, of legal size, have been donated by four taxidermy companies in Victoria. The Victorian/NSW cod samples were kept by anglers from the Murray (50 fish), Murrumbidgee (8), Goulburn (5 fish), Edwards (4 fish) and Ovens rivers (3) fish. The remaining fish were caught in the Wakool, Lachlan, Darling rivers and smaller tributaries. The fish ranged in size from 670 to 1380 mm long (3.3 to 39.1 kg). The gonads and otoliths were dissected from the frozen cod samples.

A4.2.1 Otoliths

Otoliths were processed and aged following the methods of Anderson (1992) and a broad range of ages, 4 to 44 years, were estimated from the Victorian/NSW dataset (see Figure 1). Data were also provided by SARDI (courtesy Brenton Zampatti) for 28 large Murray cod from the lower Murray in South Australia. The fish were collected from fish kills in 2008, from Lake Bonney and the Lock 3 area. The South Australian Murray cod ranged from 14-44 years and ranged in size from 980 to 1290 mm, however, total weight and sex data were unavailable for most of these fish. For all Murray cod, the growth rate was slow and variable, with fish from South Australia appearing older for their size in the larger size groups.
The gonads of the Murray cod donated by Victorian taxidermists were collected from a wide range of seasons but with few female fish in spawning condition. This was largely due to the closed season negating the possibility of such samples. Nevertheless, six female fish were examined with ripening ovaries. The gonads were macroscopically staged (after Rowland 1998) and their total weight recorded. The samples were dissected and the diameter of a sub-sample of eggs recorded via a microscope and digital software. These gonads could not be submitted for histology as they had been frozen by the anglers.

One female fish (1140 mm, 33 kg, 23+ years old) had 1.037 kg of ripe eggs (shown below) which equates to a fecundity estimate of almost 110,000 eggs – a much higher estimate than previous studies had shown. This is important information for the Murray cod model. In addition, one fish (1160 mm, 26+ years) was examined with calcified blockage within the ovaries. These phenomena have been noted previously but there appeared to be no indication it was associated with age. None of the large fish 30+ years had calcified ovaries and the largest study fish (1380 mm, 39.1 kg) was female.
Appendix 4: The addition of longevity and fecundity data to the Murray cod model

Fig. A4.3: Ovary (left) from a large Murray cod with 110,000 eggs. Photo to right is microscopic image of the mature eggs.

A4.3 Conclusion
The present project has provided new data associating the age of Murray cod with their total fecundity and spawning condition. Female fish in spawning condition are difficult to collect but our evidence has demonstrated a much higher fecundity than previously documented. There also appears to be no indication of senescence in the small number of large female fish examined that were over 30 years old. The variability in growth, with large fish not necessarily being old fish, also makes the assumption of senescence difficult to demonstrate. Importantly, the fecundity estimates and the potential for differential growth rates in SA and Vic/NSW have implications for updating the outputs of the Murray cod model.

A4.4 How this changes the model
The model changes are numerous although model output has not change significantly. Due to a better understanding of the age-length relationship, it has been decided to increase the age classes modelled from 15 to 25. The final age class will be for 25 year plus fish. This ensures that most fish (98%) in the final age/stage class will be greater than 1m long. Also the fecundity schedule was changed to reflect the fecundity information contained in this report. This report has been updated to reflect both the structural and parameter changes brought about by this report.
MURRAY-DARLING BASIN AUTHORITY

Murray Cod Modelling to Address Key Management Actions

Final report for Project MD745