Assessing the potential for restocking the western school prawn *Metapenaeus dalli* in a temperate Australian estuary

A thesis presented on partial fulfilment of the degree of

Bachelor of Science Honours

School of Veterinary and Life Sciences

Murdoch University, Western Australia

Andrew David Broadley, BSc., M. Info.Tech.
Declaration

I declare that this thesis is my own account of my research and contains as its main content work which has not previously been submitted for a degree at any tertiary education institution.

……………………………………..

Andrew Broadley

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THESIS TITLE: Assessing the potential for restocking the western school prawn *Metapenaeus dalli* in a temperate Australian estuary

AUTHOR: Andrew BROADLEY

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Abstract
The population dynamics of restocking involves trade-offs between different release scenarios and the conservation of wild stocks and evaluating these trade-offs is an important component of restocking programs. This study used length frequency data and mixture analysis to estimate growth, mortality and reproductive parameters for the western school prawn *Metapenaeus dalli* in the Swan-Canning Estuary. These biological parameters provide the first quantitative estimates of growth and mortality for *M. dalli* and were used as inputs into a bioeconomic model developed with the *EnhanceFish* software used to evaluate the effectiveness of different stocking levels and sizes-at-release. This research had two primary objectives:

1) To use the data from a spatially and temporally comprehensive sampling program for *M. dalli* in the Swan-Canning Estuary to determine the biological parameters for growth, mortality and size at maturity; and

2) Develop a bioeconomic model using the *EnhanceFish* software to evaluate the potential increase in population biomass from restocking with different numbers of released prawns and different sizes-at-release.

Monthly length frequency data demonstrated that *M. dalli* exhibits strong seasonal growth and reproductive cycles. Growth was estimated using Somers (1988) seasonal adaptation of the von Bertalanffy growth curve. This analysis demonstrated that females grew significantly larger (*L*∞ = 33.72 mm CL, *k* = 1.06) than males (*L*∞ = 24.20 mm CL, *k* = 0.97). Gravid females, or females carrying a spermatophore, were only found between November and April, when surface and bottom water temperatures were between 20 and 27 ºC. Pauly’s (1983a, b, 1984) seasonal catch-curve revealed little difference in total instantaneous mortality between female and males (*Z* = 0.048 and 0.047 week⁻¹, respectively). Whereas Pauly’s (1990) non-
seasonal catch-curve showed a larger range in mortality between females (0.067 week\(^{-1}\)) and males and (0.052 week\(^{-1}\)). Since recreational fishing effort and thus fishing mortality is very low, these estimates provide a close approximation for natural mortality (\(M\)) in the population. The patterns of growth and reproduction were compared with those recorded in this system ≈ 30 years ago by Potter et al. (1986), who found a similar seasonal pattern of growth. However, the maximum size of prawns recorded in that study were about 10% smaller and the estimates of \(L_\infty\) about 20% smaller than those from the current sampling. These differences indicate that the far greater fishing pressure in this earlier period may have selectively removed larger prawns from the population.

The release scenarios evaluated by the bioeconomic model varied from 650,000 to 5 million prawns (i.e. 650,000, 1 million, 2 million and 5 million) and the size-at-release from 1 mm carapace length (CL) to 10 mm CL. The greatest potential returns were obtained when 5 million prawns were released at a size of 10 mm CL, however, such an aquaculture effort would require substantial capital expenditure in order to produce the required number of juveniles. The model results also highlighted that density-dependent processes are likely to be important and decrease the relative effectiveness of releases. At the current low population level and without any restocking, the population biomass was projected to remain virtually unchanged over a five-year period. The model results also highlight the need for empirical information on natural morality, the influence of density on growth and survival and how time-at-release might influence survival. These results improve our biological understanding of \(M.\ dalli\) and will be used to help plan any future restocking of this species in the Swan-Canning Estuary.
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1. Introduction

The global marine fishery catch has not increased since the mid-1990s, despite substantial advances in the effectiveness of fishing effort through technological change (Pauly et al., 2002; FAO, 2012). At the same time, reports on changes in the abundance of top-level predators and ecosystem function due to industrial-level fishing have been documented (Myers and Worm, 2003; Baum and Worm, 2009). Worm et al. (2009) reported examples of some fisheries recovering in developed countries such as the United States, Australia and New Zealand. A more recent study identified five major geographic areas of focus for fisheries conservation, namely the west coast of Mexico, north east and north west Africa, south east Africa and Indonesia, and highlighted the increasing exploitation rates, high biodiversity and poor management capacity in these regions in contrast to Europe, North America and Oceania (Worm and Branch, 2012). To reduce the impact of fisheries, Pauly et al. (2002) suggested that fishing capacity needs to be greatly reduced and significant spatial closures employed throughout the world’s oceans to recover fisheries. However, this places significant economic and social hardship on businesses and communities that previously had open access to these fishing zones (Mascia et al., 2010; Bennett and Dearden, 2014).

An alternative potential management intervention for rebuilding stocks and increasing production is the use of restocking or stock enhancement (Bell et al., 2005, 2008; Leber, 2013). While restocking and stock enhancement both involve the release of aquaculture-raised young “fish” into natural systems, the status of the wild stock and purpose of the release differ significantly between restocking and stock enhancement. Thus, restocking involves the release of cultured juveniles to restore
over-exploited or severely depleted wild populations and it attempts to achieve long-term benefits by recovering the stocks spawning biomass (Bell et al., 2005, 2008; Leber, 2013). In contrast, stock enhancement seeks to improve fishing yield by increasing recruitment when the spawning biomass has not been depleted. Enhancement programs often pursue returns on investment in shorter time frames than restocking releases to demonstrate a positive cost-benefit to key stakeholders (Bell et al., 2005, 2008; Leber, 2013).

1.1 Fishery dynamics
Fisheries are dynamic systems made of complex interactions between biological, ecological, economic and social characteristics and the success of a release program depends on how well these characteristics are understood and how the interdependencies between them are managed (Blankenship and Leber, 1995; Lorenzen et al., 2010). Key biological attributes of fishery systems include the population dynamics and life history traits of the target species for release (Leber, 2013). The interaction between the wild and released populations and their marine habitat are critical ecological characteristics (Loneragan et al., 2013a). The economic feasibility and costs and benefits of a release program should be assessed on a regular basis to ensure program outcomes are beneficial to key stakeholders (Lorenzen et al., 2010). Understanding the social well-being and equity amongst stakeholders and their influence on the outcomes of the release program is also fundamental to its success (Johnston et al., 2010).

1.2 Population dynamics
Variations in life history traits of the target population (e.g. growth, natural mortality, recruitment, maximum age and size, age and size at first maturity, fecundity and predator-prey relationships) determine the type of release program required to meet
fishery management objectives (Bell et al., 2005). The post-release survival of the released fish can be improved by establishing baseline information for size-at-release, habitat for release, the scale of release, and the timing and acclimatisation period in the release habitat. In some cases, negative effects of hatchery releases have become evident as the scale and frequency of the releases increases, e.g. an increase in mortality induced through releasing large numbers of young that compete with the wild populations, i.e. density-dependent mortality (Lorenzen et al., 2010; Kitada, 2014; Laugen et al., 2014).

Modelling or simulating population dynamics provides some insight into how the wild and hatchery populations will react to various restocking or enhancement scenarios (Ye et al., 2005; Loneragan et al., 2006). The precision of the estimates from these theoretical experiments with models is determined by the availability and accuracy of the estimates of parameters used in the models and whether the model provides a realistic representation of the enhancement system. Under these circumstances, the models are able to produce realistic results and can be applied at a much lower cost than field based trials (Lorenzen, 2005; Ye et al., 2005; Medley and Lorenzen, 2006). Furthermore, developing the models provides a framework for integrating the knowledge from different disciplines and engaging with the stakeholders of the release program (Loneragan et al., 2004, 2006; Lorenzen et al., 2010).

1.3 Measuring success
An important component of evaluating the success of release programs is determining the extent to which they meet fishery management goals and how the costs of the programs compare with those of more traditional management tools,
such as reductions in fishing effort through spatial and temporal closures or changes in size and bag limits. The degree of success or failure of a release program is measured by a set of criteria created during the early stages of the release program, ideally prior to the start of large scale releases (Lorenzen et al., 2010). Quantitative success criteria provide a means to gauge a program’s progress toward meeting fishery goals without the subjectivity of stakeholder perceptions associated with qualitative methods (Grafton, 2006; Hoggarth, 2006; Lorenzen et al., 2010). Measures of success vary between commercial and recreational fisheries, but can include the number of released individuals recaptured, an increase in the catch per unit effort, an increase in total yield or value of the fishery and enhancing the quality of the fishing experience.

The growth and mortality of the wild population and the released individuals are important parameters for the models used in evaluating enhancement programs such as EnhanceFish (Medley and Lorenzen, 2006) and the model developed by Ye et al. (2005) for the brown tiger prawn fishery in Exmouth Gulf Western Australia. It is also important to understand how changes in the model parameters may affect the predicted returns from enhancement or restocking. This can be achieved through simulations and drawing parameters from a defined distribution around the chosen value. For example, Ye et al. (2005) found that the model predictions of the numbers and value of prawns captured following enhancement were most sensitive to variation in density-dependent mortality and the immediate post-release mortality.
1.4 The western school prawn (*Metapenaeus dalli*)

The genus *Metapenaeus* is one of 25 extant genera belonging to the Penaeidae, which comprises 28 species that occur exclusively throughout the inshore and estuarine waters of the Indo-West Pacific (Dall, 1990; Degrave et al., 2009). Within this region, species from this genus contribute to important commercial and recreational fisheries and aquaculture production (Dichmont et al., 2006; Kompas et al., 2010; Montgomery, 2010).

The only Metapenaeid species that occurs in south-western Australia is the western school prawn *Metapenaeus dalli* (Racek, 1957). The species occurs along the west coast of Australia from Darwin in the north to Cape Naturaliste in the south and also in Java, Indonesia (Grey et al., 1983). While this species is found in inshore marine waters < 30 m in latitudes above 31° S, it is only found in estuaries below this latitude and is believed to complete its entire life cycle within these systems (Potter et al., 1986, 1989). Thus, it is classified as a solely estuarine species (Potter et al., 2015).

Potter et al. (1986) used hand trawl nets and otter trawls to investigate the biology of this species in the Swan-Canning Estuary. Their data, collected about 30 years ago, provided an understanding of the growth and timing of reproduction in the estuary (November to April) but did not estimate the parameters of growth or mortality for the population. More recently, the stages of development from egg to postlarvae have been described, with *M. dalli* taking about 15 days to reach the postlarval stage.

Both *M. dalli* and the western king prawn *Penaeus latisulcatus* were the focus of a small commercial and iconic recreational fishery in the Swan-Canning Estuary. The
commercial fishery catch peaked at 15 tonnes in 1959, but declined afterwards leading to its closure in the mid-1970s (Smith et al., 2006). At its peak, the recreational prawning involved over 50,000 people in the Perth metropolitan area and became an iconic pastime, particularly during the Christmas period (Smithwick et al., 2011). Catch rates in the recreational fishery also declined, with the last significant catches recorded in the late 1990s. The reasons for the decline are unclear, however, it is likely due to a combination of overfishing, changing environmental conditions and recruitment failure (Smith et al., 2006, 2007). Smith et al. (2007) concluded that, on the basis that populations of *M. dalli* had not recovered, despite the cessation of commercial fishing and the massive reduction in recreational effort and catch, that, “The Swan-Canning population of *M. dalli* has limited capacity to recover from depletion because it is a small, discrete breeding stock and reliant on self-replenishment”. Thus, given the long-term recruitment failure a restocking program was likely to be best suited to increase the population size of *M. dalli* in the Swan-Canning Estuary by bypassing the recruitment bottleneck during the high mortality stage of larvae through to juveniles.

In response to the depleted status of the *M. dalli* population in the Swan-Canning Estuary a trial integrated restocking program was initiated with three major components:

1) development of methodologies to spawn, culture and release large numbers of postlarval prawns;

2) evaluation of the population in the estuary and development of optimal release strategies;
3) engagement of the community in collecting broodstock from the estuary and providing information on recreational catches.

The research in this thesis builds on the second component, which attempts to understand the status of the stocks and evaluate the potential returns from restocking.

1.5 Objectives of the Thesis

This thesis had two overall objectives:

1) To use the data from the systematic, intensive sampling program of *M. dalli* in the Swan-Canning Estuary to determine the biological parameters for growth, mortality and size at maturity (Chapter 2); and

2) Develop a bioeconomic model using the *EnhanceFish* platform to evaluate the potential increase in population biomass from restocking with different numbers of released prawns and different sizes-at-release. The bio-economic model was also used to evaluate how the population would change over time if no restocking program was carried out (Chapter 3).
2. Estimation of Growth and Reproductive Parameters of the Western School Prawn (*Metapenaeus dalli*) in the Swan-Canning Estuary

2.1 Abstract
This study used length frequency data and mixture analysis to estimate growth, mortality and reproductive parameters for the western school prawn *Metapenaeus dalli* in the Swan-Canning Estuary. The monthly length frequency data, collected from hand trawl and otter trawl sampling on the new moon throughout the estuary, over 12 lunar cycles, indicated strong seasonal growth and reproductive cycles. Most growth occurs during the warmer months of October-April, with little to no growth in the colder months from May-September. The Somers’ (1988) seasonal adaption of the von Bertalanffy growth model was used to estimate the growth parameters of female and male *M. dalli*: females grew significantly larger (*L*∞ = 33.72 mm CL, *k* = 1.06) than males (*L*∞ = 24.20 mm CL, *k* = 0.97). Gravid females, or females carrying a spermatophore, were only found from November to April, when surface and bottom water temperatures were between 20 and 27 °C. Both seasonal and non-seasonal length converted catch curves were used to estimate the rate of total mortality for males and females. The seasonal model revealed little difference between female and male total instantaneous mortality (*Z* = 0.048 and 0.047 week⁻¹, respectively). The non-seasonal model showed a larger range in mortality between females (0.067 week⁻¹) and males (0.052 week⁻¹). Since fishing mortality is very low, these estimates provide a close approximation for natural mortality *M* in the population. The estimation of growth parameters for *M. dalli* in the Swan-Canning Estuary are the first to be based on rigorous quantitative sampling of the system. The patterns of growth and reproduction were compared with those from Potter et al. (1986), who found a similar seasonal pattern of growth but the maximum size of *M. dalli* were about 10% smaller between 1977 to 1982. The smaller size of prawns
from the historical study may be due to less intense sampling with otter trawls in the
deep water or recreational fishing, which was much more intense than currently.
These results improve our scientific understanding of *M. dalli* and will be used to
help restock this species in the Swan-Canning Estuary.

2.2 Introduction
The biological properties of a fish population are a reflection of the growth, mortality
and reproduction of individuals within the population (Pope et al., 2010; Haddon,
2011). These processes influence the abundance, sex ratio, length and age structure
and spatial distribution of the population. Biotic and abiotic effects such as a species
life-history strategy (e.g. r or K strategist), predation, competition, food availability,
water temperature and oxygen depletion also influence the dynamic behaviour of a
fish population (Winemiller, 2005; Pope et al., 2010). Estimating a population’s key
parameters (e.g. growth, mortality and reproduction) quantitatively is a critical step
in understanding its current state and provides a basis for future fishery management
decisions (Haddon, 2001; Pope et al., 2010).

2.1.1 Biology of penaeid prawns
Members of the family Penaeidae are found in offshore, inshore and estuarine
habitats in temperate and tropical latitudes (∼ 40° S to 40° N) around the world (Dall,
1990; Bailey-Brock and Moss, 1992). The penaeid go through a number of distinct
developmental stages after hatching: the planktonic larval stages of nauplius,
protozoea, and mysid before assuming the benthic behaviour of postlarvae, juveniles
and adults (Dall, 1990). However, the habitat chosen for a particular life stage varies
considerably between individual species and life stages i.e. pelagic or benthic,
estuarine, inshore or offshore (Dall, 1990). Migration between habitats is an
important population characteristic as biotic and abiotic stressors influence the
dynamic properties of these populations (García, 1988; Bailey-Brock and Moss,
1992; Winemiller, 2005; Pope et al., 2010).

The genus *Metapenaeus* is one of 25 extant genera belonging to the Penaeidae
family. This genus comprises 28 species, which occur exclusively throughout the
inshore and estuarine waters of the Indo-West Pacific, extending from south-western
Africa, throughout Asia and into Oceania (Dall, 1990; Degrave et al., 2009). Prawns
in the genus *Metapenaeus* contribute to important commercial and recreational
fisheries throughout the Indo-West Pacific region (Dall, 1990). For example, in
subtropical and temperate New South Wales an average of 1,410 tonnes of prawns,
worth over $18 million, were caught annually between 2004 to 2009, in inshore and
estuarine environments with *Metapenaeus macleayi* and *Metapenaeus bennettiae*
comprising 54% and 32% of the total catch by weight and value, respectively
(Montgomery, 2010). Prawns, predominantly *Penaeus plebejus* and *M. macleayi*,
inhabiting the estuaries in this region of Australia are also exploited by recreational
fishers who remove ≈ 4,700 tonnes annually (Montgomery, 2010).

As noted in Chapter 1, the western school prawn *M. dalli* is an estuarine species in
the Swan-Canning Estuary (Potter et al., 1986, 2015) (Fig. 2.1a). In contrast, the
western king prawn *Penaeus latisulcatus*, also found in the Swan-Canning Estuary, is
thought to have a similar life cycle to those of many other penaeids, with spawning
and the release of eggs taking place in the marine environment and the postlarvae
and juvenile stages occupying either inshore coastal environments or estuaries (e.g.
Dall, 1990; Potter et al., 1991; Bailey-Brock and Moss, 1992) (Fig. 2.1b). This
species has been classified as marine estuarine-opportunist (Potter et al., 2015). It lives its early life stages in protected inshore coastal habitats. However, it enters the estuary in large numbers as a juvenile then migrates back to its near shore habitat as an adult (García, 1988; Dall, 1990; Bailey-Brock and Moss, 1992; Potter et al., 2015).

Figure 2.1 The lifecycle of (a) marine estuarine-opportunist (*Penaeus latisulcatus*) and (b) solely estuarine (*Metapenaeus dalli*) (Figures drawn by Dr James Tweedley).

Both *M. dalli* and *P. latisulcatus* were the focus of a small commercial and iconic recreational fishery in the Swan-Canning Estuary in south-western Australia. However, the commercial fishery ceased in the 1970s and the recreational fishery declined greatly in catch and effort over 20 years ago (Smith et al., 2007, see also Chapter 1). The reasons for the decline are unclear, however, it is likely due to a combination of overfishing, changing environmental conditions and recruitment failure (Smith et al., 2007).

2.1.2 Estimating age, growth and mortality

The pioneering work by early and more recent fishery scientists (e.g. Baranov, 1918; Thompson and Bell, 1934; Le Cren, 1951; Beverton and Holt, 1957; Ricker, 1958;
Gulland and Holt, 1959; Hoenig and Hanumara, 1982; Pauly, 1984; Hilborn and Walters, 1992; Quinn and Deriso, 1999) has led to a variety of quantitative methods used to investigate and assess the dynamic behavior of fish populations. The study of a population’s length or age composition through time provides the data to quantitatively evaluate its dynamic recruitment, growth and mortality processes (Caddy, 1996; Quinn and Deriso, 1999; Gayanilo et al., 2005a; Haddon, 2011). In general, typical stock assessment methods are based on a time-series of estimated biomass and catches, together with data on the age structure, growth, mortality and reproduction to estimate the current fishing mortality rates and spawning stock biomass. Age-structured models are difficult to implement for crustaceans because of the difficulty in ageing them and length-based methods of assessment are used for prawns, crabs and lobster.

A variety of computationally intensive statistical techniques are now common place in fishery science (e.g. Quinn and Deriso, 1999; Kirkwood et al., 2001; Gayanilo et al., 2005a; Haddon, 2011), which has led to the evolution of using length frequency data as a relatively cost-effective alternative to tag-recapture studies or direct age determination (i.e. fish otolith or scale analysis) (Pauly and Morgan, 1985,; Pilling et al., 2008). Length frequency distributions from samples of a population can be grouped into cohorts using model based clustering techniques (e.g. mixture models) (Macdonald and Pitcher, 1979; Benaglia et al., 2009). The distribution of these cohorts provides an indicator of the relative age structure of the population and following the changes in size and abundance over time allows the population’s recruitment, growth and mortality to be estimated (Pauly and Morgan, 1985,; Caddy, 1996; Quinn and Deriso, 1999; Gayanilo et al., 2005a; Haddon, 2011).
2.1.3 Objectives of chapter two

A comprehensive sampling program of *M. dalli* was initiated in 2013, as part of a major initiative to investigate the feasibility of restocking prawns in the Swan-Canning Estuary (Chapter 1). This program collected prawns from the shallow, nearshore waters and the deeper, offshore waters, every month for a 12-month period in 2013/14.

The overall objective of this Chapter was to use the data from the systematic, intensive sampling program of *M. dalli* in the Swan-Canning Estuary to determine the biological parameters for growth, mortality and size at maturity. A variety of models were used to explore and estimate quantitatively for the first time, the growth and instantaneous total mortality of *M. dalli* in the Swan-Canning Estuary from the recent intensive sampling program. Because of the significant seasonal changes in water temperature in south-western Australia, seasonal growth models were compared with a non-seasonal model to determine whether they provided a better description of *M. dalli* growth.

Growth curves were fitted to the historical data collected 30 years ago and the parameters for growth and mortality were estimated to make comparisons between the biological parameters of the population from the past with those from the recent sampling program. The current and historical time epochs differ greatly in the biomass of the population, catch and intensity of fishing effort; in the historical data (1978 to 1982, Potter et al., 1986), recreational catches and fishing effort, and by inference the population biomass, were much higher than the current time (2013 to 2014). The research in this Chapter also expands on previous work by Potter et al.
(1986) to compare the timing of reproduction and develop a maturity schedule for *M. dalli* in the Swan-Canning Estuary.

2.3 **Material and methods**

2.3.1 **Study area**

The Swan-Canning Estuary is located in the metropolitan Perth region of southwestern, Western Australia (Fig. 2.2). The climate in this region is Mediterranean with dry, hot summers and cool, wet winters (BOM, 2014). The estuary is a permanently open system with a long narrow channel (lower estuary) that leads into two large central basins (middle estuary) and two main tributary rivers of the Swan and Canning (upper estuary) (Loneragan et al., 1989; Kanandjembo et al., 2001). The longest tributary, the Swan River, extends 25 km upstream from the middle estuary to the confluence of Ellen Brook. The tidal section of the Canning River reaches 11 km from its entry point to the middle estuary (Canning Bridge), upstream to Kent Street Weir (Kelsey et al., 2010).

The surrounding Swan-Canning catchment covers an area of approximately 2,090 km² and includes Perth city and most of the greater Perth metropolitan area (OAG, 2014). The catchment area is heavily urbanised around the lower and mid regions of the Swan-Canning Estuary. The upper regions of the catchment have large areas of agriculture, light industry and nature reserves. In recent decades, nutrients and other pollutants from the catchment and urban surrounds have flowed into the Swan-Canning Estuary causing eutrophication, cyanobacterial blooms and sediment loading (Kelsey et al., 2010; OAG, 2014).
2.3.2 Rainfall and water quality data for the Swan-Canning Estuary

Rainfall data for the Perth metropolitan station were obtained from the Bureau of Meteorology Climate Data Online web site for the months of this study from September 2013 until October 2014. Additionally, salinity, temperature and dissolved oxygen data were obtained from the Western Australian Department of Water, Water Information Reporting system for the same period. Each of the water quality sampling sites were selected according to their location, in a proximity central to each of the sampling regions, and the availability of data.

2.3.3 Sampling procedure

The Swan-Canning Estuary was divided into five sampling regions (Fig. 2.2): Lower Melville Water (LM), Upper Melville Water (UM), Lower Canning Estuary (LC), Upper Canning Estuary (UC) and the Middle Swan Estuary (MS). These regions cover an area of around 35.1 km², of which 15.5 km² are in shallow waters (i.e. < 2 m) and 19.6 km² in deep water (i.e. 2 to 17 m). Prawns were sampled at night at 20 shallow sites using a hand trawl net and 16 deep sites using a small otter trawl net, between October 2013 and September 2014. Sampling was completed during the new moon phase, approximately every 29 days, when the moon provided about 10% of the illumination of a full moon.
Figure 2.2 The location of the deep (otter trawl = ●) and shallow (hand trawl = ○) sites with lines separating each of the 5 sampling regions.

The shallow water sites were sampled using a 4 m wide hand trawl constructed of 9 mm mesh throughout. The width of the hand trawl net during trawling was approximately 2.85 m but varied slightly amongst sites depending on factors such as condition of the substratum, presence of submerged obstacles and localised wind and wave conditions. Two replicate trawls, each covering a distance of 200 m and a swept area of 570 m$^2$, were carried out at each site. The total area covered by hand trawling of the 20 sites on each new moon was 22,800 m$^2$. The efficiency of the hand trawl net has not been estimated.

Prawns were sampled in deeper water using a 2.6 m wide otter trawl net, with 25 mm mesh in the body, and 9 mm mesh in the cod end. The net was towed at a speed of 1.62 knots for five minutes, covering a distance of approximately 250 m. Two trawls were completed at each site, each covering a swept area of approximately 650 m$^2$. The total swept area covered by trawls at the 16 deep water sites on each complete
sampling of the new moon was 20,800 m$^2$. The sampling efficiency of the otter trawl net for catching *M. dalli* has not been determined.

After each trawl, prawns were collected, placed on ice and returned to the laboratory for identification and measurement. When prawn catches were large, prawns were identified in the field, sexed and measured (see below) and returned to the water.

A number of length measurements were recorded (to the nearest 0.01 mm) for each *M. dalli* using digital vernier callipers. These measurements included the total length (tip of the rostrum to the tip of the telson), body length (orbital indent to the tip of the telson), carapace length 1 (orbital indent to the posterior edge of the carapace) and carapace length 2 (tip of the rostrum to the posterior edge of the carapace) (Fig. 2.3a). The wet weight (nearest 0.01 g), and sex of the prawn were also recorded. Females were identified by presence of a thelycum (Fig. 2.3c), and males by the presence of a petasma (Fig. 2.3b). Individuals without a thelycum or petasma were recorded as unknown sex. Female prawns were also inspected to determine if they were gravid (i.e. had large green ovaries, as described by Tuma, 1967) and possessed a spermatophore.
Figure 2.3 Photos of the western school prawn, *Metapenaeus dalli*, showing (a) the length measurements recorded for each prawn, (b) petasma of the male prawn, (c) thelycum of the female prawn and (d) gravid condition of female. Photos (a) to (c) by Dr James Tweedley, Murdoch University and (d) by the Swan River Trust.
2.3.4 Length-weight relationship

Initially, the relationship between carapace length (subsequently referred to as CL) and weight, in grams, was evaluated with a non-linear least squares (NLS) model in R (R Core Team, 2014) using the equation: \( W = aL^b \) (where \( W \) = weight, \( L \) = Length, \( a \) and \( b \) are model parameters) (Le Cren, 1951) for females and males separately. However, a subsequent review of the residuals revealed their variance was increasing with size thus indicating multiplicative error. Therefore, a linear model using the log transformed length-weight equation: \( \log(W) = \log(a) + b\log(L) \), was computed (Le Cren, 1951). A plot of residuals from the log transformed data showed improved homoscedasticity. Consequently, this model was chosen to best evaluate and derive the length-weight relationships (see 2.4.2). A bias correction factor for back-transforming mean weight values for a given length was calculated using:

\[
\frac{e^{-s^2_{Y|X}}}{2},
\]

where \( s^2_{Y|X} \) is the mean square error from the linear model (Ogle, 2014).

2.3.5 Process and data preparation

The length frequency data from hand and otter trawls were used to estimate growth and mortality. The process of data preparation and the estimation procedures are summarised in Figure 2.4. Growth was estimated from the pooled data from both sampling methods after adjusting for swept area (see below) by mixture analysis and modal progression (Fig. 2.4). The historical data of Potter et al. (1986) were also used to estimate growth from modal progression.

The catchability of *M. dalli* using the hand and otter trawls were assumed to be equal, and the length frequency data from the otter trawls were scaled up by a factor
of 1.096 according to the ratio of the swept area of the two methods. The unknown sex group (subsequently referred as the juvenile group) were allocated equally to each of the female and male groups. The CL measurements from these data were then allocated to 1 mm size classes.

2.3.6 Estimation of growth

2.3.6.1 Identifying cohorts

Monthly histograms of the weighted 1 mm CL data were created in R and reviewed visually to gain an understanding of changes in *M. dalli* length frequency distribution over time and identify potential modal groups (cohorts). Finite mixture analysis was conducted in R using the Mixtools package (Benaglia et al., 2009) (Fig. 2.4). Starting values for the mean, standard deviation (SD) and weighting for each potential mixture component (cohort) in a monthly sample were estimated from the histograms visually. A two step iterative process was employed to generate normal distributions for each of the components using the Expectation-Maximisation (EM) algorithm from the Mixtools package (Benaglia et al., 2009). Hypothesis testing ($a = 0.05$) using 1,000 bootstrap replicates was used to produce a likelihood ratio statistic for the null hypothesis of a $k$ component fit versus an alternative hypothesis of $k+1$ (up to a maximum of 10) components for each monthly sample (Benaglia et al., 2009). The resultant outputs of this process were optimised estimates of the mean, SD, weighting and number of prawns in each mixture component for each monthly sample.
Figure 2.4 Flowchart showing the high level processes used to generate growth parameters, mortality estimates, maturity and timing of reproduction of *Metapenaeus dalli*. EM = Expectation-Maximisation algorithm; LCC = length converted catch curves.
2.3.6.2 Analysis of historical data

Data on the biology of *M. dalli* in the Swan-Canning Estuary collected by Potter et al. (1986) between 1978 and 1981 were analysed to estimate growth rates for comparison with the current study. Potter et al. (1986) used a similar sized mesh in the otter trawl net (25 mm) and a slightly larger mesh (19 mm) in the hand trawl compared to those used in the current study. The means and SD of the female and male CL frequency distributions were visually estimated from Figure 3 in Potter et al. (1986). These data were used to reproduce Potter et al. (1986) modal progression graph in R.

2.3.6.3 Parameter estimation from length frequencies

The methods outlined in this section were applied only to the 2013/14 data set because the original length frequency data were not available from the historical study. The primary purpose of estimating parameters from length frequencies was to create a set of robust starting values for the non-linear models (see 2.3.5.4).

The weighted 1 mm size class length frequency data from the current study were grouped by sampling month and analysed using the Length Frequency Data Analysis 5 (LFDA 5) (Kirkwood et al., 2001) and Fisheries Stock Assessment Tools II (FiSAT II) packages (Gayanilo et al., 2005a). In LFDA 5, the Hoenig and Hanumara (1982) and Pauly et al. (1992) seasonal version of the von Bertalanffy growth function (VBGF) were fitted to the length frequency data using the ELEFAN (Pauly, 1985,) method. A score grid search in LFDA 5 provided the initial parameter estimates, which were then optimised and plotted using the automatic maximisation process. Similarly, in FiSAT II, the ELEFAN I routine was used to directly fit a seasonal
VBGF (Pauly et al., 1992), by using a response surface analysis and then plotting and optimising the fit by eye.

2.3.6.4 Parameter estimation from modal progression

The same method for estimating growth parameters by fitting a non-linear least squares (NLS) model to data derived from modal progression was used for the 2013/14 and historical data sets. For the historical data, the modal progression graph recreated in R was used to model and estimate historical (1978 to 1981) growth parameters (Fig. 2.4).

For the 2013/14 data, the means and SD from the finite mixture analyses were plotted separately for female and male prawns in a 12 monthly time series. Modal progression was used to identify cohorts by tracking each point through time and visually observing its position relative to adjacent cohorts.

The modal progression analysis for the 2013/14 data identified distinct cohorts growing through the 12 sampling periods. To aid model fitting, these data were rearranged to a relative time scale spanning 24 months. In effect, prawns in the 0+ class represent growth in the first 12 months and those in the 1+ class are in their second year of growth. This step was not required for the historical data because the study undertaken by Potter et al. (1986) covered four years.

The Somers (1988) seasonally oscillating adaption of the VBGF was applied to estimate growth parameters. The Somers adaptation is:

\[ L(t) = L_\infty \left\{ 1 - e^{-[K(t-t_0)+ S(t)-S(t_0)]} \right\}, \]

with \( S(t) = (CK/2\pi)\sin2\pi(t - t_s) \),
and \( S(t_0) = (CK/2\pi) \sin2\pi(t_0 - t_s) \),

where \( L(t) \) is the average length at time \( t \), \( L_\infty \) is the asymptotic length, \( k \) is the rate at which the model reaches asymptotic length, \( t_0 \) is the theoretical time where the average length is 0. The functions \( S(t) \) and \( S(t_0) \) generate the seasonal oscillation of the growth curve: \( C \) controls the amplitude of the growth oscillation during the winter period (if \( C=1 \) growth stops or if \( C=0 \) there is no seasonal oscillation), \( t_s \) is the start of the curved portion of the first growth oscillation.

The Somers (1988) model was fitted in R using the FSA (Ogle, 2014) and Minpack (Elzhov et al., 2013) package. This package provided an implementation of the Somers (1988) growth function and the Minpack package was used to implement an NLS function using a modified Levenberg-Marquardt algorithm which supports lower and upper parameter constraints. The starting values for the NLS model \((L_\infty, k, t_0, C \text{ and } t_s)\) for 2013/14 and historical data were estimated by averaging the results of \( L_\infty, k, t_0, C \text{ and } t_s \) from the length frequency analysis using LFDA 5 and FiSAT II. The following parameters were constrained to optimise model fitting: \( C \) between 0 and 1, \( t_0 \) between -1 and 0 and \( t_s \) between -1 and 1.

The NLS model assumes that the data are homoscedastic and the errors are normally distributed. These assumptions were validated by, 1) plotting the residuals and fitted values for each model and visually verifying the distribution of the plotted points, 2) creating histograms of the residuals and visually checking the distribution for symmetry around the midpoint. A bootstrapping technique using 1,000 resampled data sets was used to create 95\% confidence intervals (CI) for each of the estimated parameters.
2.3.6.5  Test for seasonality

The growth increments identified in the modal progression analysis were analysed using a Gulland and Holt (1959) plot in FiSAT II (Fig. 2.5). The main purpose of this process was to evaluate statistically the strength of the seasonal oscillation for the 2013/14 data. A regression analysis of the growth rate and mean length was conducted by setting $L_{\infty}$ (forced Gulland and Holt plot) to the average $L_{\infty}$ derived from the length frequency analysis. FiSAT II then generated a plot of the residuals (Fig. 2.5), as percentage deviation against time of year, followed by a t-test used to determine the significance of separating the residuals into two separate groups (Gayanilo et al., 2005b). This analysis showed that the residuals during the winter months were, in all cases, except one $\leq 0$ (% deviation) for both female and male prawns but not in summer (Fig. 2.5) and that the seasonal growth oscillation term was significant.

![Figure 2.5](image.png)

**Figure 2.5** Plot of residuals from a Gulland and Holt plot showing percentage deviation against time. The red line indicates a strong and significant difference between seasonal growth oscillations for both (a) female (b) male.

2.3.7  Estimation of mortality

Instantaneous total mortality ($Z$) was estimated separately for female and male prawns using the weighted 1 mm size class length frequency data from the 2013/14
data set (Fig. 2.4). Catch curve regression was implemented in R using length-converted catch curves (LCC) (Pauly, 1983a, b, 1984) and seasonal length-converted catch curves (SLCC) (Pauly, 1990). The SLCC approach was also adopted to investigate whether Pauly’s (1990) conclusion that the LCC method tends to overestimate $Z$ when a seasonal oscillation is present.

The LCC method was implemented with:

$$\log \left( \frac{N_i}{\Delta t_i} \right) = a + b \times t_i,$$

with $t_i = t_0 - \left( \frac{1}{k} \right) \times \log \left( 1 - \frac{i}{L_\infty} \right)$,

where $N$ is the number of $M. dalli$ in length class $i$, $\Delta t$ is the time it takes prawns to grow through length class $i$, $t$ is the relative age at the mid-length of class $i$ (calculated using the inverse von Bertalanffy growth equation), and the absolute value of $b$ becomes an estimate of $Z$.

Implementation of the SLCC method followed the steps described by Pauly (1990) and is defined in the equation: $\log(N) = a + b \times \hat{t}$, where $N$ is the number of prawns in a pseudo cohort, $\hat{t}$ is the relative age of prawns in that cohort (relative to the scale used to define the pseudo cohorts, i.e. a 2 year period) and $Z$ is derived from the absolute value of $b$. There are three main steps used in this study to implement Pauly’s (1990) approach in R: i) defining the boundaries of pseudo cohorts by creating 12 growth curves at 3 monthly intervals from right to left along the x axis, ii) summing all prawns ($N$) between the growth curve boundaries (the pseudo cohorts), iii) plotting the natural logarithm of $N$ against relative age intervals. Finally, similar to the LCC method the slope of the descending portion of the curve was calculated to provide an estimate of $Z$. 

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The LCC and SLCC methods make the following assumptions: a) the *M. dalli* population is closed i.e., there is no immigration or emigration, b) $Z$ is constant and does not change relative to age or size, c) the vulnerability of prawns to capture is constant, and d) the sample is not biased.

2.3.8 Time and size at maturity

Macroscopic identification of gravid prawns was used in this study as a proxy to identify sexual maturation of female *M. dalli*. The maturity schedule assumes there is a difference between morphologically mature prawns (i.e. prawns that have grown to 21 mm CL, the size at which 100% of the population is considered mature, and will cycle between ovigerous and non-ovigerous stages of maturation), and functionally mature prawns (i.e. ovigerous individuals, currently appearing gravid).

The relationships between size and the presence of a spermatophore and gravid ovaries were examined for female prawns by using data containing all prawns for each sampling month when gravid prawns were present (November to April). A histogram of gravid prawns was constructed using 1 mm CL classes to identify the smallest and largest length class containing mature prawns. Prawns between these length classes represent an approximate proportion of the female population shifting from an immature to a mature state. This transition was evaluated with logistic regression using a logit transformation in a general linear model (GLM) using R and the formula:

$$\log \left( \frac{p}{1-p} \right) = a + b_1X$$
where: \( p \) is proportion mature and \( 1 - p \) is proportion immature; \( a \) and \( b_1 \) are model parameters and \( X \) is the CL. The CL where 50\% (CL\(_{50}\)) and 90\% (CL\(_{90}\)) of the female prawn population are gravid was calculated using:

\[
x = \log \left( \frac{p}{1 - p} \right) - a \frac{b_1}{b_1}
\]

where \( p \) is 0.5 (50\% mature) and 0.9 (90\% mature), \( a \) and \( b_1 \) are model parameters. Confidence intervals for CL\(_{50}\) and CL\(_{90}\) were created by bootstrapping 1,000 samples using the bootCase function from the Car package in R (Ogle, 2014).

### 2.4 Results

#### 2.4.1 Weather and aquatic environmental data

The total rainfall for the 12 months from October 2013 to September 2014 was 599 mm (Fig. 2.6), 22\% below the long-term mean of 732 mm per annum (BoM, 2014). The two months preceding the start of sampling, August and September 2013, were the wettest (311 mm in total) recorded in 58 years (BOM, 2014). Throughout the sampling period 90\% of the rainfall occurred during May to September 2014 (Fig. 2.6) (BOM, 2014). The average maximum air temperatures between October 2013 and September 2014 were close to the long-term average, except for November 2013, (29.1 °C) and August 2014 (21.5 °C), which were 2.6 °C and 2.4 °C, respectively, warmer than the average (BOM, 2014).
Figure 2.6 Monthly rainfall (mm) for the Perth metropolitan region during the sampling period from October 2013 to September 2014. Data obtained from the Bureau of Meteorology (BOM, 2014).

The seasonal patterns of change in water temperature were similar in the five regions of the Swan-Canning Estuary, with surface temperature values ranging from a minimum of 11.2 °C in the Lower Canning (LC) during June 2014 to a maximum of 27.3 °C in the Upper Canning (UC) in February 2014 (Fig. 2.7a). The lowest bottom temperature was 14.4 °C in August 2014 in the Middle Swan (MS) and the highest was 26.5 °C in the UC in February 2014 (Fig. 2.7b). The lowest range in surface temperature (13.7 to 25.3 °C) was recorded in LM, and the highest range in the UC (12.8 to 27.3 °C). Bottom temperatures varied less than those of the surface water with the greatest range in the UC (15.6 to 26.5 °C).

Surface salinity ranged from 2.5 in the UC during August 2014 to 37.3 in the LC during April 2014 (Fig. 2.7c). The lowest bottom salinity was 3.1 in the UC in October 2013 and the highest was 37.4 in LC during April 2014 (Fig. 2.7d). The lowest variation in surface salinity was found at LM (8.5 to 36.8) and salinities in
this region remained greater than 20 between June and September 2013, when salinities in all other regions declined to < 15.

The lowest concentration of surface dissolved oxygen (DO) was 5.8 mg L$^{-1}$ (April 2014, UC) with a maximum value 14.3 mg L$^{-1}$ (October 2013, LC) (Fig. 2.7e). In contrast, the bottom DO was often < 6 mg L$^{-1}$ and declined to values of < 4 mg L$^{-1}$ in the UC, with a minimum value of 1.2 (Fig. 2.7f). The maximum bottom DO 7.9 mg L$^{-1}$ was at LM during July and August 2014 (Fig. 2.7f).
2.4.2 Length-weight relationship

The length-weight relationships for female and male *M. dalli* were (see Figure 2.8b and d):

Female \( \log(W) = -6.29 + 2.68 \log(CL) \), \( R^2 = 0.98, n=1721 \)

and

Male \( \log(W) = -6.78 + 2.89 \log(CL) \), \( R^2 = 0.98, n=1394 \).

Alternatively, on the original scale, with bias correction:

Female \( W = (0.0019CL^{2.68}) \times 1.0067 \)

and

Male \( W = (0.0011CL^{2.89}) \times 1.0058 \).

**Figure 2.8** The relationship between carapace length and wet weight for (a) female and (c) male *M. dalli* and the log transformed relationships with line of best-fit for (b) females and (d) males.
2.4.3 Identifying cohorts and examining growth

A total of 4,698 *M. dalli* (2,535 females, 2,163 males) were caught during the 12 months of this study, 588 in the hand trawl net and 4110 in the otter trawl. The smallest catch of females (n=43) and males (n=34) occurred in October 2013. The largest female catch was 416 prawns in June 2014 while that of males was 326 prawns in December 2013. The female prawns ranged in CL from 6.5 to 30.5 mm and the males from 6.5 to 24.1 mm CL. The smallest prawn, 3.3 mm CL was caught in April 2014.

Three main cohorts (A, B and C, with mean sizes of 8.4, 13.1 and 16.1 mm CL, respectively) were evident in the CL distribution for female prawns in October 2013, with a fourth cohort (D, mean = 24.7 mm CL) that was present in October and December only (Fig. 2.9a). Cohort D is represented by a relatively small number of individuals (Female n=10 and male n=11). The three main cohorts (A, B and C) could be easily followed October to March when they had reached a mean size of 22.8, 26 and 28.4 mm CL, respectively (Fig. 2.9a). Only small numbers of large female prawns (< 20) were present until August (cohorts A and C) and September (cohort B). A new cohort of female prawns (E) with a mean size of 10 mm CL appeared in March and this group could be followed until September when the mean size was 12 mm CL (Fig. 2.9a).
Figure 2.9. Monthly carapace length (CL) frequency histograms for (a) female and (b) male *Metapenaeus dalli* in 1 mm length classes from hand trawls and otter trawls (n). Normal distributions (blue line) were fitted to identify the mean, SD, weighting and number of prawns in each cohort (labeled A to F).
A single main cohort of males (A) was evident from October and easily followed through to September and range in mean size from 13.0 to 19.6 mm CL (Fig. 2.9b). A smaller cohort (B) was only present from October to December (mean of 18.6 to 20.7 mm CL). Cohort C was identified in November (mean = 11.5 mm CL) and was only able to be tracked until February where it reached a mean size of 13 mm CL. A new group of male prawns also appeared in March (D) and was evident until September when it had achieved a mean size of 11 mm CL (Fig. 2.9b). Two smaller cohorts E and F appeared in April (mean = 4.7 mm CL) and May (mean = 7.6 mm CL) and were present until September and August where they reach a size of 5.9 and 8.0 mm CL, respectively (Fig. 2.9b).

Figure 2.10 The mean carapace length (± 1 SD) for the cohorts identified in the analysis of the carapace length frequency distributions in Figure 2.9 of (a) female and (b) male Metapenaeus dalli in each month between October 2013 and September 2014.
The mean carapace length of female cohorts A, B and C increased at \( \approx 2 \text{ mm CL month}^{-1} \) between October and March (Fig. 2.10a). The growth of male prawns in cohorts A, B and C over the same time period was much slower, \( \approx 1 \text{ mm month}^{-1} \) (Fig. 2.10b). The mean carapace length of the cohorts of female (E and F) and male prawns (D, E and F) changed very little during the late autumn and winter months (May to September), increasing by \(< 2 \text{ mm CL} \) during these five months (females, cohort E, Fig. 2.10a).

The projected growth curves from the 12 months of this study to a two-year period (see Methods and Fig. 2.4b) showed a highly seasonal pattern of growth (Fig. 2.11). This pattern was also observed when the seasonal growth model was fitted to the historical data collected between 1977 and 1982 (Fig. 2.11). Fitting the growth data with non-linear least squares (NLS) growth models gave similar values of \( k \) for the current and historical curves for females (1.06 and 1.05, respectively) and males (0.97 and 1.01, respectively). However, the asymptotic mean carapace length (\( L_\infty \)) for females (33.7 mm CL) and males (24.2 mm CL) in the 2013/14 data set were much longer than those for the historical values (females = 28.0 mm CL, males = 20.0 mm CL).

The von Bertalanffy growth parameters from the four different methods of estimation gave estimates of \( L_\infty \) that varied by 2.9 mm CL for females (\( \approx 9\% \)) and 0.8 mm CL for males (\( \approx 3\% \)). The estimates of \( k \) varied by less than 8\% for females and 3\% for males (Table 2.1).
Figure 2.11 Growth models for the (a) female and (b) male *Metapenaeus dalli* for the 2013/14 data the (c) female and (d) male for the historical data using Somers’ (1988) seasonal adaption of the von Bertalanffy growth model.
Table 2.1 von Bertalanffy growth parameters for female and male *Metapenaeus dalli* using Length Frequency Data Analysis 5 (LFDA) (Potter et al., 1986), Fisheries Stock Assessment Tools II (FiSAT II) (Kirkwood et al., 2001) and the non-linear least squares (NLS) function from the Minpack package (Gayanilo et al., 2005a). The 95% CIs were estimated for the NLS method only. NGT=Non growth time within a year.

<table>
<thead>
<tr>
<th>Data Set/Sex</th>
<th>Method</th>
<th>( L_\infty ) (CI)</th>
<th>( k ) (CI)</th>
<th>( t_0 ) (CI)</th>
<th>( t_s )</th>
<th>C</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013/14</td>
<td>Female</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>LFDA (Hoenig)</td>
<td>31.0 (CI)</td>
<td>1.04</td>
<td>-0.77</td>
<td>0.20</td>
<td>0.90</td>
</tr>
<tr>
<td></td>
<td>LFDA (Pauly)</td>
<td>30.80 (CI)</td>
<td>0.99</td>
<td>-0.81</td>
<td>0.16</td>
<td>NGT=0</td>
</tr>
<tr>
<td></td>
<td>FiSAT II</td>
<td>32.0 (CI)</td>
<td>0.98</td>
<td>N/A</td>
<td>-0.10</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td>NLS</td>
<td>33.72 (30.9-34.0)</td>
<td>1.06 (0.91-1.20)</td>
<td>-0.16 (-0.22-0.11)</td>
<td>-0.20</td>
<td>0.99</td>
</tr>
<tr>
<td>Male</td>
<td>LFDA (Hoenig)</td>
<td>24.10 (CI)</td>
<td>0.98</td>
<td>-0.82</td>
<td>0.25</td>
<td>0.80</td>
</tr>
<tr>
<td></td>
<td>LFDA (Pauly)</td>
<td>24.80 (CI)</td>
<td>0.95</td>
<td>-0.82</td>
<td>0.24</td>
<td>NGT=0</td>
</tr>
<tr>
<td></td>
<td>FiSAT II</td>
<td>24.90 (CI)</td>
<td>0.98</td>
<td>N/A</td>
<td>-0.10</td>
<td>0.89</td>
</tr>
<tr>
<td></td>
<td>NLS</td>
<td>24.20 (22.9-26.9)</td>
<td>0.97 (0.74-1.10)</td>
<td>-0.29 (-0.34-0.24)</td>
<td>-0.22</td>
<td>0.88</td>
</tr>
<tr>
<td>Historical</td>
<td>Female</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>NLS</td>
<td>27.99 (26.4-29.8)</td>
<td>1.05 (0.87-1.26)</td>
<td>-0.16 (-0.20-0.12)</td>
<td>-0.14</td>
<td>0.89</td>
</tr>
<tr>
<td></td>
<td>Male</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>NLS</td>
<td>19.97 (18.8-21.7)</td>
<td>1.01 (0.80-1.20)</td>
<td>-0.26 (-0.50-0.19)</td>
<td>-0.06</td>
<td>1.0</td>
</tr>
</tbody>
</table>

2.4.4 Mortality

The instantaneous total mortality (Z) for female *M. dalli* estimated from the slope of the relationship between estimated abundance and relative age ranged from 2.48 year\(^{-1}\) (SLCC) to 3.46 year\(^{-1}\) (LCC) (Figure 2.12a and b, Table 2.2). The values of Z estimated for males varied less than those for females, ranging from 2.45 year\(^{-1}\) (SLCC) to 2.69 year\(^{-1}\) (LCC) (Fig. 2.12c and d, Table 2.2). The equivalent weekly rates for mortality in female were 0.048 and 0.067 week\(^{-1}\) and for males were 0.047 and 0.052 week\(^{-1}\) (Table 2.2).

Table 2.2 Instantaneous mortality (Z) and 95% confidence intervals (CI) for female and male *Metapenaeus dalli* estimated from 12 months of sampling from September 2013 to October 2014 using both the Length Converted Catch Curves (LCC) and Seasonal Length Converted Catch Curves (SLCC) methods (Pauly, 1990).

<table>
<thead>
<tr>
<th>Sex</th>
<th>Method</th>
<th>Z year(^{-1}) (CI)</th>
<th>Z week(^{-1}) (CI)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Female</td>
<td>LCC</td>
<td>3.46 (3.03-3.89)</td>
<td>0.067 (0.058-0.075)</td>
</tr>
<tr>
<td></td>
<td>SLCC</td>
<td>2.48 (1.70-3.17)</td>
<td>0.048 (0.033-0.061)</td>
</tr>
<tr>
<td>Male</td>
<td>LCC</td>
<td>2.69 (2.00-3.38)</td>
<td>0.052 (0.038-0.065)</td>
</tr>
<tr>
<td></td>
<td>SLCC</td>
<td>2.45 (1.03-3.87)</td>
<td>0.047 (0.020-0.074)</td>
</tr>
</tbody>
</table>
Figure 2.12 Non-seasonal length converted catch curves for (a) female (c) male prawns and seasonal length converted catch curves for (b) female (d) male, created from the 2013/14 data.

2.4.5 Time and size at maturity

Gravid female *M. dalli* and those carrying a spermatophore first appeared in the hand and otter trawls during November 2013 and disappeared in April 2014 (Fig. 2.13). These females ranged in size from 15 to 28.5 mm CL. The peak percentage of gravid females (34%) and the percentage carrying spermatophores (45%) were recorded in January 2014.
Figure 2.13 The percentage of female *Metapenaeus dalli* caught in hand and otter trawls that were gravid (red) or carrying a spermatophore (blue) between October 2013 and September 2014.

The estimated carapace length at 50% maturity (CL$_{50}$) for females, based on the CL of gravid females between December 2013 and April 2014, was 16.9 mm CL (95% CI = 16.7 to 17.1 mm CL), equivalent to an age of about 11 or 12 months (Fig. 2.14). The estimated CL$_{90}$ for females was 18.5 mm CL (95% CI = 18.2 to 18.7 mm CL), at about 12 to 13 months of age.

Figure 2.14 The logistic regression fitted to the proportion of gravid females in each 1 mm CL during the spawning season from December 2013 to April 2014 (red line) to estimate the size at maturity for female *Metapenaeus dalli*. The blue dotted line represents the carapace length where 50% (CL$_{50}$) of females are mature.
2.5 Discussion

This study uses data collected during a recent comprehensive field study of the western school prawn *Metapenaeus dalli* in both shallow and deep water of the Swan-Canning Estuary to provide the first quantitative estimates of growth and mortality and female size at maturity. In general, the results of this study and estimated times of reproduction are similar to those found by Potter et al. (1986) in a study conducted over 30 years ago when this species was much more abundant and supported a valued recreational fishery.

2.5.1 Reproduction

This study found that *M. dalli* has a strong seasonal spawning cycle in the Swan-Canning Estuary. Gravid female prawns (i.e. those with large green ovaries, as described by Tuma, 1967) and those carrying a spermatophore, first appeared in November and were last seen in April. During this period the average bottom water temperatures in the five study regions, varied from 20 to 27 °C. A strong seasonal cycle of reproduction was also found by Potter et al. (1986), who observed spawning between the same months as our study i.e., November to April. The 30-year gap in time between sampling in this study and Potter et al. (1986) indicates that the seasonal cycle established for *M. dalli* is unlikely related to a short-term event (e.g. a single weather event or response to an unusual estuarine condition). It is more than likely an evolutionary adaption of *M. dalli* to a seasonally oscillating reproductive cycle in response to longer-term hydrologic and climatic influences, in effect tuning itself with the estuarine environment. A similar conclusion was made by García (1988) while studying environmental effects on *Penaeus notialis* along the Ivory Coast. The evolutionary adaption of *M. dalli* to a solely estuarine life history was documented by Potter et al. (1986), where it was suggested the micro tidal nature of
the estuary and low flushing activity from freshwater discharge applied a positive selection pressure on *M. dalli* to remain in the estuary.

The start of reproductive activity in female *M. dalli* appears to be synchronised with an increase in surface and bottom water temperatures in the Swan-Canning Estuary to about 20 °C (cf. Figs 2.14 and 2.7a,b). The activity and emergence of invertebrates is strongly influence by temperature and the emergence time of several species of prawns increase greatly above 20 °C (Wassenberg and Hill, 1994). It is well documented that temperature strongly influences biological processes including the reproduction of ectothermic organisms (e.g. Atkinson, 1994; Clarke, 2003; Angilletta et al., 2004; Lloyd-Jones et al., 2012; Rodgers et al., 2013). The months where spawning activity was greatest (December, January, February and March) occur when the surface and bottom temperatures are between 22 and 27 °C (Fig. 2.7a,b). This peak reproductive output (i.e. December to March) indicates a life history strategy by *M. dalli* to optimise larval survival between 22 and 27 °C.

The majority of *M. dalli* active in this spawning cycle are from the recently maturing 0+ cohort (i.e. the prominent cohort in terms of abundance) that are close to 12 months of age, with only small numbers of the 1+ cohort found during this time (Fig. 2.9). This indicates that a large percentage of the female population will spawn during one season only, towards the end of their first year of life. Although, it is possible they spawn more than once in that season. Biotic and abiotic stressors such as: a) an increase in metabolic rate due to rising water temperature, b) expending energy for maturation and subsequent spawning, c) a significant growth phase (i.e. ≈ 2 mm CL month\(^{-1}\)) could stretch the physiological capability of female *M. dalli* (i.e.
reduction in fitness as energy is diverted from the immune function or reduced ability to escape and evade predators), and may become exacerbated or potentially lethal if food supply is limited (Rolff et al., 2004). This hypothesis could explain the significant decline in the catch of female, and to a lesser extent male, *M. dalli* after the spawning period (i.e. after April).

2.5.2 Growth

Similar to the strong seasonal reproductive cycle, *M. dalli* exhibited an equally strong seasonal growth pattern. Most growth occurred during the warmer months from October to March. In their first 5 to 6 months, female and male *M. dalli*, on average, grow to approximately 10 to 12 mm CL, respectively, and an estimated 0.88 to 1.47 g, respectively, in weight. Growth remained very slow over the colder winter months until the following October when female *M. dalli* grew at a similar rate in length to that during their first six months (≈ 2 mm CL month\(^{-1}\)) but much faster in terms of weight (≈ 1 g month\(^{-1}\)). Following winter, males grew much more slowly than females, at about half the rate of females and reached 18.8 mm CL by March.

The von Bertalanffy instantaneous growth parameter, *k*, calculated from the Somers (1988) seasonal model were similar for females (1.06) and males (0.97). However, the asymptotic carapace length (*L_\infty*) was much larger for females (33.7 mm CL, ≈ 23.8 g wet weight) than males (24.2 mm CL, ≈ 11.0 g wet weight). It should be noted that because of the high mortality rates for *M. dalli*, few individuals are likely to reach the asymptotic size. The estimated *L_\infty* is therefore likely to be an artefact of the model with little biological meaning and the 95 percentile for length may be a better estimate of maximum size for this species (see Hordyk et al., 2014). The difference in growth patterns between females and males has been recorded in many
species of penaeids that exhibit a wide variety of life history tactics in response to selective pressures (e.g. Primavera et al., 1998; Correa and Thiel, 2003; Callaghan et al., 2010; Mehanna et al., 2012; Accioly et al., 2013). The gender dimorphism, recorded in the current study and that of Potter et al. (1986) where female *M. dalli* grow much larger than males is more than likely a life history strategy to maximise fecundity and optimise reproductive capacity of the population (Ramirez Llodra, 2002). Similar observations were made by (Dall, 1958) in a study of the greentail prawn *Metapenaeus mastersii* (now *Metapenaeus ensis*) in the Brisbane river, although the size difference between genders was greater in the current study (the largest female = 30.5 mm CL and male = 24.1 mm CL).

The estimates for the asymptotic length appeared to differ quite significantly between the current study and the historical data summarised in Potter et al. (1986), with the historical estimate of $L_\infty$ for female and male prawns being 16 to 18% smaller (27.99 mm CL and ≈ 14.44 g for females and 19.97 mm CL and 6.34 g for males). These differences in growth could be related to a range of biotic and abiotic factors, as well as differences in fishing pressure and the sampling regimes in the thirty years between the studies. For example, the results of this study suggest that the ratio of natural mortality to the growth rate ($M/k$) is relatively high for *M. dalli* (females ≈ 2.9, males ≈ 2.6), which means that, even in unfished conditions, large individuals (at or near asymptotic size) are rare (Hordyk et al., 2014; Prince et al., 2015). Given the high level of fishing pressure during the historical study (Potter et al., 1986) the size distribution of *M. dalli* may have been truncated when the original study was conducted over 30 years ago, which could result in an under-estimate of the asymptotic length of the species (see also below).
The main differences between the studies in sampling regimes were the greater frequency of sampling and number of sites sampled during the current study, particularly by otter trawls in deeper water. Although the mesh size of the otter trawls was similar during both studies the hand trawl in the current study used smaller mesh (12 mm) than the historical study (19 mm). The difference in mesh size between the hand trawl nets should not adversely affect the catch of larger prawns (i.e., > 10 mm CL) and because of the presence of material such as macroalgae and jellyfish giving a smaller effective net mesh, may not have affected the capture of smaller prawns. An alternative explanation for the difference in $L_{\infty}$, alluded to by Potter et al. (1986), is an increase in recreational fishing during the 1970’s, leading to increased fishing pressure and potential selection of larger prawns by fishers. Thus, over $\approx 48$ nights of sampling in the shallows in the current study, only two groups of recreational prawn fishers were seen, whereas this was a common place activity in the 1970s and 1980s (Potter et al., 1986).

2.5.3 Mortality

Total instantaneous mortality was estimated using non-seasonal and seasonal versions of the length-converted catch curve (SLCC and LCC) methods (e.g. Pauly, 1983a, b; 1984, 1990). The difference between these methods is discussed in some detail in Pauly (1990) who concluded that the non-seasonal LCC methods can significantly overestimate instantaneous mortality ($Z$) and that this bias is amplified for small, short-lived species with strong seasonal cycles (e.g. $M. dallii$). The non-seasonal LCC in this study produced higher estimates of $Z$ when compared to the seasonal LCC, consistent with the conclusions of Pauly (1990). The difference
between the two methods was far more pronounced in females (0.98 year\(^{-1}\)) than males (0.24 year\(^{-1}\)).

A recent study by Hufnagl et al. (2013) tested the sensitivity of 8 methods for estimating Z, including the LCC and SLCC. They found that Z was underestimated in all 8 methods when seasonal growth and recruitment were present. In fact, Hufnagl et al. (2013) found the non-seasonal LCC method was consistently rated among the most accurate of methods across seasonal and non-seasonal scenarios. Furthermore, when mortality is low (i.e. < 5 year\(^{-1}\)) the non-seasonal LCC is generally a suitable choice.

When applying the above discussion to this study it seems the non-seasonal LCC result is more plausible. That is a female Z of 3.46 year\(^{-1}\) and a lower male Z of 2.69 year\(^{-1}\). This lower mortality of males than females is consistent with the virtual disappearance of larger female *M. dalli* in the 1+ cohort after the spawning season has completed in April. Combined with a greater abundance of female than male prawns in the 0+ cohort, this provides strong evidence to suggest a higher rate of mortality in the female population.

2.5.4 Assumptions in the study

The estimated rates of growth and mortality in this study have assumed that the catchability of the hand trawl and small otter trawls were the same over the size distribution of *M. dalli* caught during the current study. No estimates of catchability are available for either of these nets. Studies of other methods indicate that it is likely to vary from about 0.3 to 0.7 of the available population, depending on the size
and species of prawn. For example, Loneragan et al. (1995) used depletion studies to estimate the catchability of a small beam trawl (1 x 0.5 m wide mouth, with 2 mm mesh body, 1 mm cod end) for postlarval and juvenile *Penaeus semisulcatus* and *P. esculentus* in a range of sea grass habitats. The average catchability was higher for postlarvae (1-2 mm CL prawns, 0.66) than for juveniles (> 2 mm CL, 0.47). Joll and Penn (1990) used similar studies with a large, commercial otter trawl and estimated the catchability of *P. latisculatus* varied from 0.3 to 0.5. Clearly this is an important assumption that needs to be investigated, either through depletion studies or sensitivity analyses.

2.5.5 Conclusion

This study has reasonably established that the *M. dalli* population exhibits strong seasonal growth and reproductive cycles. These findings are similar to those discovered by Potter et al. (1986) over 30 years ago when this species was more abundant and supported a valued recreational fishery. Most of the growth is occurring during the warmer months between October and April, with little to no growth in the colder months from May to September. Reproductively active females were only found from November to April, when surface and bottom water temperatures were between 20 and 27 °C. Patterns of growth and reproduction were compared with those from Potter et al. (1986), who found a similar seasonal pattern, but the maximum size of *M. dalli* were about 10% smaller between 1977 to 1982.

The high mortality of the population and absence of larger *M. dalli*, particularly females is concerning as it appears that the majority of females are only spawning for 1 season, although possibly multiple times. Consequently, there is evidence to
suggest that the *M. dalli* population relies on a strong recruitment cycle and as such has developed some interesting life history strategies and evolutionary adaptations to an estuarine lifecycle.
3. Bioeconomic evaluation of restocking the western school prawn (*Metapenaeus dalli*) in the Swan-Canning Estuary

3.1 Abstract
The evaluation of the potential benefits of releasing cultured individuals has been identified as an important component of restocking and stock enhancement programs. Bioeconomic models provide a mechanism for integrating biological information with fisheries data and economic information to better assess the costs and benefits of release programs. The biological parameters estimated for the western school prawn *Metapenaeus dalli* in Chapter 2 were used as inputs in a bioeconomic model developed using the *EnhanceFish* software to evaluate the effectiveness of different stocking levels and sizes of *M. dalli* on the estimated population biomass in the Swan-Canning Estuary. The release sizes varied from 650,000 to 5 million prawns (i.e. 650,000, 1 million, 2 million, 5 million) and the size at release from 1 mm carapace length (CL) to 10 mm CL. The results from these simulations were compared with those from a non-restocking scenario. The greatest potential returns were obtained when the 5 million prawns were released at a size of 10 mm CL, however, such an aquaculture effort would require substantial capital expenditure to produce the required number of juveniles. The model results also highlighted that density-dependent processes are likely to be important and decrease the relative effectiveness of releases. At the current low population level, without any restocking, the population biomass was projected to remain virtually unchanged over a five-year period. The model results also highlight the need for empirical information on natural morality, the influence of density on growth and survival and how time-at-release might influence survival.
3.2 Introduction

3.2.1 Fishery management and bioeconomics

There is little doubt that understanding the population dynamics of a fishery is a critical component of successful restocking programs (e.g. Caddy and Defeo, 2003; Lorenzen, 2005, 2008; Ye et al., 2005; Leber, 2013). Fishery managers need to make decisions under challenging circumstances, such as the increasing uncertainty around policy, economics and environmental degradation brought about by climate change or other environmental factors (FAO, 2012). It is under these conditions that managers require tools to evaluate the trade-offs in the cost and benefits of different strategies, while also assessing the risks to existing fish stocks and associated ecosystem services.

Although not a panacea, bioeconomic theory provides a quantitative framework that allows fishery managers to evaluate the trade-offs of various fishery system dynamics (i.e. effort, harvest and stock size) against equilibrium reference points (Anderson and Seijo, 2010; Larkin et al., 2011). Bioeconomic tools such as EnhanceFish or the Excel model produced by Ye et al., 2005, provide a cost effective and powerful means to evaluate the viability of a portfolio of fishery management scenarios, such as changes to vessel and gear restrictions, spatial and temporal closures, setting harvest limits and implementing release programs for stock enhancement or restocking (Caddy and Defeo, 2003; Grafton, 2006; Cochrane and Garcia, 2009). For example, a bioeconomic assessment undertaken by Hart et al. (2013a) using the greenlip abalone fishery in south-western Australia to develop a base case from which various enhancement scenarios were applied. This included assessing fishing mortality (at various length classes), density and size of release on
their effect on spawning biomass, profit, gross value product and net present value. The model parameters were validated by previous experiments and surveys conducted in south-western Australia (e.g. Hart et al., 2013b, 2013c). Enhancement scenarios were then extended to the entire Australian greenlip abalone fishery, which demonstrated a significant increase in profitability (from $12 to $26 million) could be achieved from annual releases of 6.1 million juveniles at 4 cm in length.

3.2.2 Restocking bioeconomics

The biological effectiveness of a restocking program is based on the assumption that additional recruits will increase stock production by bypassing the recruitment bottleneck that occurs during the high mortality stage of larvae through to juveniles (Caddy and Defeo, 2003; Lorenzen, 2005; Bell et al., 2008). However, this assumption is constrained by the biological and socio-economic realities surrounding the target species (Johnston et al., 2010). Therefore, significant trade-offs exist between the cost and benefits of the various biological and socio-economic restocking scenarios (e.g. Caddy and Defeo, 2003; Lorenzen, 2005, 2008; Johnston et al., 2010; Larkin et al., 2011; Leber, 2013).

3.2.3 Objectives of chapter 3

The primary aim of this Chapter was to use the biological parameters (Chapter 2) estimated from the comprehensive sampling program to evaluate the potential success of restocking *M. dalli* in the Swan Canning Estuary through the development of a bioeconomic model. The biomass of the current population was estimated and the *EnhanceFish* software was used to evaluate the potential increase in population biomass from restocking with different numbers of released prawns and different sizes-at-release.
3.3 Material and methods

In order to develop an understanding of the potential for restocking to increase the biomass of the *M. dalli* population and start rebuilding its stocks, the data collected from the intensive sampling of this species (Chapter 2) need to be scaled up to the total Swan-Canning Estuary to estimate the biomass of the population (see 3.3.1 below). This involved estimating the extent of habitat that the population is likely to occupy and the densities of prawns within this distribution. The population parameters estimated in Chapter 2 were then used to examine the population dynamics of the wild population and how restocking of different magnitudes and size classes of prawns might increase the population in the estuary.

3.3.1 Estimation of the population biomass

The area that the population of *M. dalli* is likely to inhabit spans the sampling sites for this species selected as part of a major program on restocking this species in the Swan-Canning Estuary. These sites cover an area of approximately 35.1 km\(^2\) in the Swan-Canning Estuary. The area has been divided into two depth strata based on the bathymetry of the estuary: shallow waters < 2 m, cover an area of 15.5 km\(^2\) and were sampled using a hand trawl net; deeper waters, i.e. 2 to 17 m, cover an area of 19.6 km\(^2\) and were fished using a small otter trawl net.

The current biomass of the *M. dalli* population was calculated in *R* (R Core Team, 2014) using the swept area method. The total weight of the catch was calculated in each month for the otter and hand trawls separately using:

\[
C_m = \sum_{i=1}^{n} (w_i \times f_i),
\]
where $C_m$ is the total weight in kg for each monthly sample, $n$ is the number of 1 mm length classes, $w_i$ is the weight in kg of length class $i$ (i.e. calculated using the length-weight relationship for females and males separately, see Chapter 2, Fig. 2.), and $f_i$ is the number of prawns in length class $i$.

The mean catch $C_m$ for the sampling period was used to estimate the biomass for both the otter and hand trawls using:

$$B = \frac{\bar{C}_m}{v} \left(\frac{A}{a}\right),$$

where $\bar{C}_m$ is the mean monthly sample weight in kg, $v$ is the proportion of prawns caught in the net’s area of influence (i.e. catchability or efficiency of the net), $A$ is the area occupied by the stock, and $a$ is the monthly trawl area. The biomass, $B$, for the hand and otter trawls was combined and converted to tonnes in $B_0$, the estimated total biomass for the $M. dalli$ population in the Swan-Canning Estuary.

The vulnerability of prawns ($v$) to the hand and otter trawl nets has not been determined. However, Joll and Penn (1990) estimated the efficiency of a larger otter trawl net as between 0.30 to 0.51 for the western king prawn $Penaeus latisulcatus$. The efficiency of a small beam trawl Loneragan et al. (1995) was estimated at 0.47 for juvenile $P. semisulcatus$. These values were used to provide a guide to set the vulnerability ($v$) of $M. dalli$ to hand and otter trawls; a value of 0.4 was chosen for both methods. Empirical estimates of vulnerability would be valuable for refining the estimates of biomass.
3.3.2 Bioeconomic modelling

The restocking scenarios were evaluated using the *EnhanceFish* software developed by Lorenzen and Medley (2006). *EnhanceFish* is primarily designed for teleost fish populations that typically live longer than 1 to 2 years, e.g. the North Sea Sole (*Solea solea*). However, Lorenzen and Medley (2006) suggest the underlying population dynamics functionality implemented in *EnhanceFish* can be applied to crustacean and mollusc populations. It should be noted that the *EnhanceFish* model does not use a seasonal growth model or have functionality to model the intra-annual timing of release for stocked prawns. The timing of release in *EnhanceFish* is based on yearly time steps, where the numbers released and the size-at-release are the main determinants.

*EnhanceFish* uses an extended dynamic pool model that has been designed specifically for evaluating stock enhancement and restocking programs. It makes the following four key extensions to conventional dynamic pool models: i) includes parameters for size-dependent mortality, ii) includes density-dependent processes of the wild and hatchery fish (i.e. growth, mortality and reproduction), iii) defines the stock-recruitment relationship (SRR) so survival in the pre-recruit phase is known (i.e. prawns < 12 mm CL) and iv) splits the population structure into three components, see Figure 3.1 (i.e. wild phenotype, hatchery phenotype - naturally recruited, hatchery phenotype - stocked) (Lorenzen and Medley, 2006).
3.3.3 Recruitment

Currently, the SRR for the *M. dalli* population is unknown. The 2013/14 study described in Chapter 2 has accumulated a substantial set of data for *M. dalli* over 12 months, which covers a single recruitment cycle. These data are not extensive enough through time to estimate the SRR using either the Beverton and Holt (1957) or Ricker (1958) models. *EnhanceFish* offers an alternative method to estimate the SRR using Myers et al. (1999) steepness parameter. Myers et al. (1999) developed a standardised annual maximum reproductive rate (i.e. between 1 and 7) from a study of 700 spawner-recruit data sets. In data poor situations, *EnhanceFish* employs Myers et al. (1999) standardised slope along with an estimate of the current catch, relative effort and length at recruitment to calculate the approximate values for the
maximum recruits per unit of spawning stock biomass (SSB) and maximum average recruitment (i.e. the parameters a* and b* in the SRR).

The term ‘recruitment’ used throughout this chapter refers to when prawns become vulnerable to recreational fishing. This occurs from the 1st November each year when the recreational prawn fishery is open. At this time the main cohort of new recruits (i.e. the 0+ cohort) reaches a size of between 12 and 18 mm CL after approximately 10 to 12 months of growth (Fig 2.10, Chapter 2).

3.3.4 Natural mortality

A key extension to the dynamic pool model used in EnhanceFish is size dependent mortality by defining a function where natural mortality is inversely proportional to length for wild and hatchery prawns (Lorenzen and Medley, 2006). This relationship provides the basis to estimate survival at a given release size. The model outputs in this evaluation are conditional on the following assumptions of natural and fishing mortality: a) the total instantaneous mortality (Z) is estimated at 3.49 year\(^{-1}\) for female and 2.69 year\(^{-1}\) for male prawns, using the estimates made by the length converted catch curves in Chapter 2 (see Chapter 2), b) an estimated catch of 100 kg for 100 recreational fishers, based on anecdotal evidence of historically low levels of fishing for the last 15 years, c) fishing mortality (F) was estimated at 0.042 year\(^{-1}\) by configuring EnhanceFish with a natural mortality of 3.0 year\(^{-1}\), a catch of 100 kg and adjusting F until the population biomass was equal to 2.37 tonnes, the estimated population biomass.

Given fishing mortality is very low (\(\approx 0.014 \ M\)), natural mortality (\(M\)) approximates \(Z\). In this model \(M\) has been set to a value of 3.0 year\(^{-1}\) with a range of 2.69 to 3.49
year\(^1\) for wild and hatchery prawns. The size dependent mortality model in *EnhanceFish* has two parameters \((M_{1w} \text{ and } M_{1s})\) representing natural mortality in the wild and stocked prawns. Size dependent mortality was set the same value for both wild and stocked prawns.

### 3.3.5 Economics of restocking and prawn fishing

This study did not conduct a full assessment of the current socio-economics of the recreational prawn fishery. Furthermore, a literature search into similar studies of small recreational prawn fisheries did not find any information on the economic returns of recreational prawn fishing. Thus, the valuation of recreational utility, including fisher effort dynamics, was not included in the economic evaluation.

The operational costs associated with using the existing aquaculture facilities at the Challenger Institute have been incorporated into the model (Table 3.1). It is assumed these facilities can produce up to 2 million 15-day post larvae (PL15) prawns. However, stocking above 2 million PL15 will require additional infrastructure (i.e. tanks, pipes and heating) and therefore require extra capital investment. Any capital costs associated with the project have not been included in the economic assessment.

A base case scenario of stocking 650,000 PL15 (Table 3.1) was used to calculate the operational cost of 12 cents per PL15, approximately 1 mm CL. These costs were derived from a trial release of 650,000 PL15 conducted in January 2014. Improvements in techniques to increase larval survival from 11% to 30% will enable the production of 2 million PL15 using a similar sized seed stock of 600 adults that was used to produce 650,000 PL15. It has also been assumed that to culture 5 million PL15, the operational costs will be 250% greater than the cost of producing 2 million
PL15, i.e. a direct linear increase in the operational costs. This assumption is based on additional labour, prawn feed, power, heating and broodstock collection.

The cost of prawns grown in culture to sizes greater than 1 mm CL (i.e. > PL15) was calculated using the growth function created in Chapter 2. A growth rate of 1 mm every 12 days up to a size 10 mm CL was estimated. This rate was reduced by 25%, to 9 days, to allow for optimised growth conditions in culture, e.g. temperature, salinity and oxygen (e.g. Staples and Heales, 1991). The operational costs for the additional 9 days growth per 1 mm CL were calculated using the aquaculture, consumables, heat and power items only (Table 3.1), the broodstock collection was excluded. An operational cost of around 3 cents per prawn for each 1 mm CL in growth post PL15 was calculated, e.g. a prawn grown to 2 mm CL would cost 15 cents, this includes the 12 cents to PL15 plus 3 cents to 2 mm CL.

Table 3.1 An aggregated summary of the main operational costs of culturing 650,000 prawns to a length of 1 mm CL (i.e. PL15). Greg Jenkins, Director of the Centre for Australian Aquaculture Research and Development and Challenger Institute, Fremantle, provided information on the estimates.

<table>
<thead>
<tr>
<th>Item Description</th>
<th>Cost (AUD$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquaculture</td>
<td>64,060</td>
</tr>
<tr>
<td>Consumables (vehicles, food for prawns)</td>
<td>2,000</td>
</tr>
<tr>
<td>Heat and power</td>
<td>1,500</td>
</tr>
<tr>
<td>Broodstock collection</td>
<td>12,000</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>79,560</strong></td>
</tr>
</tbody>
</table>

3.3.6 Model parameters

The model parameters defined in Table 3.2 describe the initial configuration of the model used to evaluate various enhancement scenarios (see section 3.3.6). Parameter
values for growth, length-weight relationship and maturity were taken from the estimates made in Chapter 2.

*EnhanceFish* does not model the population growth and mortality of females and males separately. The estimates of growth and mortality for the separate sexes in Chapter 2 were averaged to provide the input to *EnhanceFish* and the range of values from this Chapter used to provide the range (Table 3.2).

**Table 3.2** Model parameters used in the *EnhanceFish* bioeconomic model to evaluate different restocking scenarios of *Metapenaeus dalli* in the Swan-Canning Estuary.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Range</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Life history</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>L₀</td>
<td>0.1</td>
<td></td>
<td>Length at larval/juvenile transition (mm)</td>
</tr>
<tr>
<td>A₀</td>
<td>0.04</td>
<td></td>
<td>Age at larval/juvenile transition (years)</td>
</tr>
<tr>
<td>Lᵣ</td>
<td>13</td>
<td>12–14</td>
<td>Length at recruitment (mm)</td>
</tr>
<tr>
<td>Aᵣ</td>
<td>1</td>
<td>0.9–1.1</td>
<td>Age at recruitment (years)</td>
</tr>
<tr>
<td>h</td>
<td>0.2</td>
<td></td>
<td>Heritability of life-history traits</td>
</tr>
<tr>
<td><strong>Growth</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>L∞</td>
<td>26</td>
<td>24–33</td>
<td>Asymptotic length at biomass → 0</td>
</tr>
<tr>
<td>K</td>
<td>1</td>
<td>0.97–1.06</td>
<td>von Bertalanffy growth rate</td>
</tr>
<tr>
<td>g</td>
<td>0</td>
<td></td>
<td>Competition coefficient</td>
</tr>
<tr>
<td>α</td>
<td>1.5x10⁻⁹</td>
<td>1.1x10⁻⁹–1.9x10⁻⁹</td>
<td>Coefficient of the length-weight relationship (converted from grams to tonnes)</td>
</tr>
<tr>
<td>β</td>
<td>2.79</td>
<td>2.68–2.89</td>
<td>Exponent of the length-weight relationship</td>
</tr>
<tr>
<td><strong>Natural Mortality</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M₁W</td>
<td>3</td>
<td>2.69–3.46</td>
<td>Mortality of wild phenotype</td>
</tr>
<tr>
<td>M₁S</td>
<td>3</td>
<td>2.69–3.46</td>
<td>Mortality of hatchery phenotype</td>
</tr>
<tr>
<td><strong>Reproduction</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lₘ</td>
<td>21</td>
<td></td>
<td>Length at maturity (mm)</td>
</tr>
<tr>
<td>p</td>
<td>-1.4</td>
<td></td>
<td>Steepness of maturity curve</td>
</tr>
<tr>
<td>r</td>
<td>1</td>
<td></td>
<td>Relative reproductive performance of stocked prawns</td>
</tr>
<tr>
<td><strong>Recruitment</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a*</td>
<td>330159</td>
<td></td>
<td>SRR estimates based on growth, mortality and maturity parameters, a catch of 0.01 tonnes and Myers et al. (1999) steepness parameter</td>
</tr>
<tr>
<td>b*</td>
<td>35437</td>
<td></td>
<td>Myers et al. (1999) steepness parameter</td>
</tr>
<tr>
<td>rₘ</td>
<td>5</td>
<td></td>
<td>Myers et al. (1999) steepness parameter</td>
</tr>
<tr>
<td><strong>Fishing</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>F</td>
<td>0.042</td>
<td></td>
<td>Fishing mortality</td>
</tr>
<tr>
<td>Lₙ</td>
<td>16</td>
<td></td>
<td>Gear selection length (mm)</td>
</tr>
<tr>
<td>q</td>
<td>-1.5</td>
<td></td>
<td>Steepness of gear selectivity curve</td>
</tr>
<tr>
<td><strong>Economics</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>γ₁</td>
<td>0.12</td>
<td></td>
<td>Cost of hatchery fish at PL15 (AUD$)</td>
</tr>
</tbody>
</table>
3.3.7 Model scenarios

Generally, commercial fishery bioeconomic assessments are used to estimate the numbers required and economic net present value (NPV) to achieve a target biomass with or without a release program. They do this by looking at various options to reduce or maintain current levels of exploitation (i.e. $F$) or consider closing fisheries until the target biomass is reached with or without stocking.

Model scenarios were developed to evaluate the change in overall biomass of the *M. dalli* population, knowing that there is currently very little fishing pressure and a relatively high natural mortality. The trial release of 650,000 PL15 in January 2014 was used as the lowest level of restocking. Three main options are considered, i) the stocking density (i.e. the number of prawns released), ii) the size-at-release of prawns and iii) a no restocking scenario. This last scenario was evaluated to determine how the wild population’s biomass is likely to respond without restocking. Various combinations of numbers released (i.e. 650,000, 2 million and 5 million) and size-at-release (i.e. 1, 3, 5 and 10 mm CL) were evaluated. This gave 12 scenarios based on different combinations of the number released and size-at-release. All scenarios are run over a 5-year period with a constant starting biomass ($B_0$).

3.3.8 Sensitivity analyses

Sensitivity analyses were conducted to explore the effects of variation in mortality and SRR on the model results. Probability density functions (PDF) were configured for each of these parameter sets. Input PDFs were defined as normal distributions with the mean set to the values shown in Table 3.2. The variances were manually adjusted to extend the range of values by $\pm 10\%$ of the mean. Monte Carlo
Simulations were used to sample the input PDFs and generate output PDFs that were displayed graphically and checked visually in *EnhanceFish*.

### 3.4 Results

#### 3.4.1 Estimation of the 2013/14 population biomass

A total of 4,110 *M. dalli* were caught in the otter trawl and 588 prawns in the hand trawl during the sampling period. The 15 mm CL size class contained the largest number of prawns (69) caught in the hand trawl (Fig. 3.2b), while it was the 11 mm CL size class (607) for the otter trawl (Fig. 3.2a). The length distribution of prawns in the hand trawl was bimodal around the 15 and 16 mm CL size classes. In contrast, the length distribution of prawns in the otter trawls was relatively evenly distributed with a small peak at 11 to 17 mm CL.

![Figure 3.2](image)

*Figure 3.2* The number of female (■) and male (□) *Metapenaeus dalli* caught in each 1 mm carapace length class size for (a) hand and (b) otter trawls over the 12-month sampling period between October 2013 and September 2014.
In terms of the total weight of prawns caught over the 12 months, the total hand trawl catch weighed 2 kg and the total otter trawl catch weighed 11.6 kg. When converted to weight, the 16 mm CL size class was the heaviest in the hand trawls, with a total weight of 0.24 kg and the 18 mm CL class was the heaviest in the otter trawls (0.97 kg) (Fig. 3.3a, b). The bimodal distribution in weights of the otter trawl shows an increase in weight of the female population (Fig. 3.3b) in size classes 17 to 30, although the numbers in the larger CL classes had decreased greatly (Fig. 3.2b).

**Figure 3.3** The weight (kg) of female (■) and male (□) *Metapenaeus dalli* caught in each 1 mm carapace length class size for (a) hand and (b) otter trawls over the 12-month sampling period between October 2013 and September 2014.
The mean weight of prawns caught in hand trawls during the sampling period was $0.17 \pm 0.08$ kg (1± SE) and $0.97 \pm 0.15$ kg for the otter trawls. The largest weight of catches for the hand trawl (i.e. 1 kg) was recorded in December (Fig. 3.4a), while the heaviest weight for the otter trawls (i.e. > 1 kg) was caught from December to March (Fig. 3.4b). The mean density km$^{-2}$ (± 1 SE) of $M. \ dalli$ caught using the hand trawl was $19.02 \pm 9.02$ kg km$^{-2}$ and $105.93 \pm 16.06$ kg/km$^{-2}$ for the otter trawl (Fig. 3.4c, d).

![Figure 3.4](image)

**Figure 3.4** Total weight (kg) and density (kg km$^{-2}$) of female (■) and male (■) *Metapenaeus dalli* in each month for hand trawls (a and b, respectively) and otter trawls (b and d, respectively) over the 12-month sampling period between October 2013 and September 2014.

The estimated biomass for the hand trawl area of 15.5 km$^2$ was calculated at 0.29 tonnes and the otter trawl area covering the shallower water (19.6 km$^2$) was 2.09
tonnes. The total biomass \( B_0 \) estimated for the combined area of 35.1 km\(^2\) was therefore 2.37 tonnes.

### 3.4.2 Bioeconomic modelling

A release of 650,000 1 mm CL \( M. \ dalli \) was estimated to produce the lowest increase in biomass (0.1 tonnes) over a 5-year period (Table 3.3). The largest predicted increase in biomass was 4.1 tonnes from the initial population biomass, achieved by stocking 5 million, 10 mm CL \( M. \ dalli \). However, the most notable difference between scenarios is the relatively low increment in biomass increase between releases of 2 to 5 million (i.e. a mean of 0.08 ± 0.05 tonnes) in all stocking sizes. A greater relative increase in population biomass was achieved by moving from a release 650,000 to 2 million prawns (i.e. 0.44 ± 0.36 tonnes).

**Table 3.3** An estimate of the total biomass, in tonnes, over a 5-year period for restocking scenarios of \( Metapenaeus \ dalli \) in the Swan-Canning Estuary for increasing stocking density and stocking size. Initial estimate of the population biomass was 2.37 tonnes.

<table>
<thead>
<tr>
<th>Stocking Density</th>
<th>Stocking Size (mm CL)</th>
<th>1</th>
<th>3</th>
<th>5</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>650,000</td>
<td></td>
<td>2.47</td>
<td>2.70</td>
<td>3.06</td>
<td>4.80</td>
</tr>
<tr>
<td>2,000,000</td>
<td></td>
<td>2.51</td>
<td>2.78</td>
<td>3.18</td>
<td>6.33</td>
</tr>
<tr>
<td>5,000,000</td>
<td></td>
<td>2.53</td>
<td>2.81</td>
<td>3.22</td>
<td>6.57</td>
</tr>
</tbody>
</table>

The estimated operating costs (Table 3.4) have been made by scaling the operational cost per single prawn (i.e. 0.12 cents) from the operational costs of producing 650,000 prawns by Challenger Institute in 2013/14. These costs do not include capital or additional costs required to increase the production capacity of the aquaculture facilities needed to generate a stocking density of 5 million prawns. It is therefore no surprise the lowest operational cost (AUD $79,560) is a stocking density
of 650,000 1 mm CL prawns. In contrast, the highest cost (AUD $682,500) is associated with a stocking density of 5 million 10 mm CL prawns.

**Table 3.4** An estimate of total operational costs (AUD$) for restocking scenarios of increasing stocking density and stocking size of *Metapenaeus dalli* in the Swan-Canning Estuary.

<table>
<thead>
<tr>
<th>Stocking Density</th>
<th>Stocking Size (mm)</th>
<th>1</th>
<th>3</th>
<th>5</th>
<th>10</th>
</tr>
</thead>
<tbody>
<tr>
<td>650,000</td>
<td></td>
<td>79,560</td>
<td>117,000</td>
<td>162,500</td>
<td>273,000</td>
</tr>
<tr>
<td>2,000,000</td>
<td></td>
<td>79,560</td>
<td>117,000</td>
<td>163,500</td>
<td>273,000</td>
</tr>
<tr>
<td>5,000,000</td>
<td></td>
<td>198,900</td>
<td>292,500</td>
<td>408,750</td>
<td>682,500</td>
</tr>
</tbody>
</table>

The *EnhanceFish* model also predicts how the wild population will respond to different release strategies. In all cases, it predicts that biomass in the wild population of *M. dalli* will decline over the five years of the simulation. For a release of 1 million prawns, the biomass of the wild population was predicted to decrease to 1.65 tonnes, 1.36 tonnes for 2 million prawns and 1.24 tonnes for a release of 5 million prawns (Fig. 3.5).

Changes to the size-at-release (i.e. 1, 3, 5 or 10 mm CL) for each of the stocking density scenarios did not influence the rate of decline in the wild population. The greatest increase in biomass, relative to each size-at-release, was found at a density of around 1 million prawns (Fig. 3.5). Varying the size-at-release had relatively little impact on the predicted biomass as more than 1 million prawns were released (Fig. 3.5).

The sensitivity analysis revealed that changing the mortality and recruitment parameters by ±10% did not have a significant effect on the stability of the model.
This analysis was extended to study the effect of changing natural mortality on biomass over a 5-year period (Fig. 3.6). At the current level of natural mortality (i.e. 3.0 year\(^{-1}\)), without restocking, the population biomass will change very little over the next 5 years (Fig. 3.6). In contrast, reducing natural mortality to half the current rate (i.e., 1.5 year\(^{-1}\)) results in a predicted increase in population biomass from 2.37 to 4.2 tonnes, i.e. an approximate 180% increase in biomass.

![Figure 3.5](image1)

**Figure 3.5** Change in the biomass of the *Metapenaeus dalli* population in the Swan-Canning Estuary with the number of prawns released and different sizes-at-release over a 5-year period. Wild population = ■ (dashed line), 1 mm = ■, 3 mm = ■, 5 mm = ■, 10 mm = ■.

![Figure 3.6](image2)

**Figure 3.6** The population biomass of *Metapenaeus dalli* in the Swan-Canning Estuary projected forward 5 years without restocking for (a) a natural mortality rate of 3.0 year\(^{-1}\) (■) and (b) a natural mortality rate of 1.5 year\(^{-1}\) (■).
3.5 Discussion
The primary aim of this preliminary bioeconomic assessment was to explore the population dynamics and costs associated with restocking *Metapenaeus dalli* in the Swan-Canning Estuary. In general, the model results indicated that the small to moderate potential increases in biomass of the *M. dalli* population would be achieved for releases of 650,000 to 5 million prawns and at a range of sizes (1 to 10 mm CL). The estimated biomass increase for different numbers and sizes-at-release ranged from a 0.1 tonnes (650,000 at 1 mm CL) to 4.2 tonne (5 million at 10 mm CL) over a 5 year period. These preliminary results require further exploration to investigate the behaviour of the *EnhanceFish* model for a short-lived species and compare these predictions with those of a bioeconomic model developed specifically for prawns (e.g. Ye et al., 2005). Some of the important outcomes from this exploratory modelling exercise and the limitations and assumptions of *EnhanceFish* for short-lived species are discussed below.

3.5.1 Population biomass
The significant decline in *M. dalli* in recent times is supported by anecdotal evidence from recreational prawn fishers who have witnessed a substantial decline in their catches. Unfortunately, there is very little scientific evidence and literature available to quantitatively estimate the decline in catch or create an index of relative abundance. Historical records indicate, however, that the largest commercial prawn catch in the Swan-Canning Estuary was 14 tonnes in 1959 and that the last significant catch of 3 tonnes, was recorded in 1975 (Smith et al., 2007). It should be noted that *M. dalli* and *Penaeus latisulcatus* are not distinguished in the commercial catch records so these may be over-estimates of the historical range of *M. dalli* catches. A large recreational fishery continued to target *M. dalli* until the 1990’s
(Smith et al., 2007). Although the annual catch of the recreational fishery is not known, it was very significant given its popularity as a regular summer activity (i.e. December to March) until about 20 years ago.

The current population biomass estimate of approximately 2.37 tonnes, derived from the average catches over the 12 months in the nearshore and offshore waters, is low in terms of historical commercial catches in the Swan, e.g. the model predicts that the record catch of 14 tonnes in 1959 would have required a population biomass of approximately 34 tonnes with a fishing mortality of 0.6 year\(^{-1}\) and natural mortality of 2.4 year\(^{-1}\) (i.e. assuming the same growth, reproduction and life history model parameters, and a total instantaneous mortality of 3 year\(^{-1}\)). The current low population size could potentially affect the rate of population increase. This depensatory behavior, where at low abundance mortality is relatively high, even with very low fishing pressure, suppresses key population processes (e.g. growth, mortality and reproduction) and is characterised by a population shifting from one equilibrium to another (Myers et al., 1995; Stephens et al., 1999; Frank and Brickman, 2000). The *M. dalli* population appears to be exhibiting depensatory behavior; it has a relatively high natural mortality, low biomass and is predicted to grow very little over the next 5 years.

Fortunately, if these model predictions of a very low relative *M. dalli* population size are realistic, restocking has the potential to have a marked positive impact and facilitate rebuilding the population size (e.g. Liermann and Hilborn, 1997; Lorenzen, 2005; Cabral et al., 2013). One of the most successful restocking programs was carried out in the Northern Japanese Scallop (*Patinopsecten yessoensis*) fishery in
Hokkaido (Bell et al., 2008). In the early half of the twentieth century this fishery had annual harvest of up to 80,000 tonnes per annum. However, in 1945 the fishery collapsed to an annual harvest of only 6,000 tonnes per annum for the next 25 years. During this period, local fishery cooperatives released large numbers of cultured juveniles to habitat selected for its low potential of predation. Since then, the total scallop harvest has increased to around 300,000 tonnes per annum with a value of around ¥37.6 billion (Uki, 2006).

3.5.2 Number released and size-at-release

The results indicate that the population biomass growth over the five-year model period was at a faster rate with releases of up to 1 million prawns and that the population growth rate declined when the numbers released exceeded 1 million. This effect is amplified by also increasing the size of prawns at release, e.g. EnhanceFish predicted higher rates of increasing population growth when the size-at-release was larger and approached the population’s natural recruitment size. This phenomenon is not unusual in restocking or stock enhancement programs, e.g. in the North Sea sole (Solea solea) stock enhancement program, Lorenzen (2008b) found an 81% increase in yield by increasing the same proportional abundance of stocked fish in the recruit life stage rather than at earlier stages in the life-cycle. When releases of larvae or juveniles were simulated, the predicted yield increased by only 4% and 29%, respectively.

As density increases in populations, density-dependence can lead to significant changes in growth, survival and reproductive output (e.g. Miller et al., 1988; Rose et al., 2001; Lorenzen, 2005; Lewin et al., 2006). The results from the EnhanceFish model show that stocking prawns at densities exceeding 1 million produces the
greatest compensatory response in the population, shown by a decrease in population growth rate with an increasing stocking density, possibly caused by density-dependent mortality or reduced growth. Similarly, prawns released at small juvenile sizes (1, 3 and 5 mm CL) elicit strong compensatory responses (e.g. juvenile density-dependent mortality). This is evidenced by the relatively low increase in total biomass at these release sizes, e.g. a density of 650,000 prawns yields an increase in biomass of 0.1 and 0.69 tonnes in the 1 and 5 mm CL sizes respectively. In contrast to the smaller sizes-at-release, the 10 mm CL prawns (i.e. closer to the size of recruitment into the recreational prawn fishery) have a strong positive population growth response, e.g. a 2.43 tonne increase at a stocking density of 650,000. These larger prawns appear to circumvent the high level of density-dependent mortality suffered by releases of smaller juveniles (i.e. 1 to 5 mm CL). Ye et al. (2005) also estimated that the optimal size-at-release for brown tiger prawns *Penaeus esculentus* was 10 mm CL (≈ 1 g wet weight), similar to the release size for karuma prawns *Penaeus japonicas* in Japan (Hamasaki and Kitada, 2006).

In a study of six demersal marine fish species in 17 populations (i.e. across the North Sea, Irish Sea, Barents Sea, Baltic Sea and Northwest Atlantic) Myers and Cadigan (1993) found the greatest source of variability in stock sizes was related to density-dependent mortality in the early juvenile life stage. For example, a 100% increase in abundance of whiting (*Merlangius merlangus*) at age 0 resulted in a 26% increase at age 1. Myers and Cadigan (1993) suggest the primary reasons for this variability are competition for food, habitat availability and predator-prey interactions. Similarly, Ye et al. (2005) found in their bioeconomic model, that the greatest source of risk and uncertainty to a large commercial Penaeid prawn (*Penaeus esculentus*)
enhancement program was post-release mortality and density-dependent mortality caused by the release of 21 million prawns.

3.5.3 Timing of release

It was not possible to vary the time of release using the EnhanceFish model. However, the time of release can have a significant impact on the survival of released individuals (e.g. Caddy and Defeo, 2003; Bell et al., 2005; Leber et al., 2008; Gardner et al., 2013). For example, Hervas et al. (2010) found that the timing of release affected the post-release mortality of stocking white sea bass (Atractoscion nobilis). White sea bass released in spring had the highest chance of survival, while those released in winter had the lowest.

Releases of *M. dalli* close to, or during winter, are likely to be less successful than those during warmer months because of the cessation of growth when water temperatures are less than 21 ºC. It is possible that releases earlier than the normal time of recruitment (i.e. around October to November, depending on seasonal variability), when water temperatures are starting to increase, may be more successful than those during the peak of the natural recruitment cycle. The primary benefit of an earlier release time is the potential reduction in competition with the wild population. This strategy could also reduce density-dependent effects within the wild population. However, the earlier releases would only be possible if the prawns could be spawned and cultured outside their breeding season.
3.5.4 Wild population’s response to restocking

The mixing of wild and hatchery reared fish can have detrimental effects on both populations if not managed carefully. The scale of the hatchery release and the relative abundance and fitness of the wild population will significantly influence productivity and overall success of restocking (Lorenzen et al., 2010, 2012). The EnhanceFish model shows that the wild population of *M. dalli* declines by approximately 0.1 tonnes per million prawns released for all restocking scenarios. However, the net population biomass (i.e. wild and hatchery prawns combined) increases despite this decline in the wild population.

Restocking genetically and phenotypically fit hatchery prawns carries the risk of displacing the wild genotype. Safeguards to preserve the wild genotype are particularly important for the *M. dalli* population as this population is at historically low levels. Selection of quality wild seed stock is fundamental to the evolutionary adaption of the broodstock and, as a consequence, the fitness of the released individuals. Careful husbandry and management during aquaculture production is critical to post-release survival, growth, recruitment and reproduction of the hatchery prawns (Lorenzen, 2008a, 2012).

3.5.5 Costs and benefits of restocking

Bypassing the density-dependent early juvenile life stage with larger juveniles, at least 10 mm CL, appears to be an effective strategy for restocking prawns. However, there are clear trade-offs between the size-at-release, the increasing costs of aquaculture production, and negative effects on the wild population. Releasing
prawns at larger sizes incurs additional operational costs (i.e. around 3 cents per 1 mm increase in CL) associated with labour, prawn feed (e.g. algae and rotofor), heating and power. It may also require additional capital costs for the production facilities to grow prawns to the larger size. Indeed, the additional capital costs of investment required to produce the 24 million 1 g *Penaeus esculentus* for 100 tonne enhancement in Exmouth Gulf were one reason this program was discontinued (Loneragan et al., 2004, 2013). Furthermore, there is evidence to suggest that the longer prawns remain in culture the higher, the risk of maladaptation to the wild environment (e.g. they will be more exposed to predation and out competed for food) (e.g. Masuda and Tsukamoto, 1998; Tomiyama et al., 2011; Ochwada-Doyle et al., 2012). Thus, this increases the risk and potential costs associated with higher post release mortality.

Currently, very little is known on the benefits of restocking small recreational fisheries, which is particularly apparent for prawns where no obvious information is available, such as *M. dalli* in Western Australia or the eastern king prawn (*Penaeus plebejus*) in the east coast of Australia (M. Taylor, NSW Fisheries, pers. comm.). Moreover, recreational fisheries need to include the social benefits from recreational fishing and the economic benefits can be difficult to estimate precisely. This contrasts with commercial fisheries, where there is a substantial amount of information available, in terms of catch, effort, operational costs and revenue. Evaluating the costs and benefits of restocking *M. dalli* requires a complete assessment of the current socio-economics of the recreational prawn fishery. Such a study would provide information on the benefits of restocking by valuing the recreational utility of the fishery. This information could then be used to quantify the
benefits in monetary terms, thus enabling a comprehensive analysis of the benefits and costs of restocking *M. dalli*.

3.5.6 Future investigation

These preliminary results indicate the predictions of the potential success of restocking of *M. dalli* from the *EnhanceFish* model require further investigation. A future investigation could explore this potential by: a) focusing on refining the estimates of mortality used in the model from empirical studies to estimate density-dependent mortality, release mortality and variation in mortality with size of prawns; b) investigating the importance of time-at-release on the population biomass, and c) comparing the predictions and behaviour of the model with those from a model designed specifically for short-lived penaeid prawns (Ye et al., 2005).

Ye et al. (2005) developed a tool to evaluate the economic viability, biological effectiveness and risk of a potential stock enhancement program for the Brown Tiger Prawn (*Penaeus esculentus*), which is fished commercially in the Exmouth Gulf Prawn Trawl Fishery (Loneragan et al., 2013b; Sporer et al., 2013). This bioeconomic model was created to include all components of a release program; from the production of prawns in the hatchery, through their culture and release in the Exmouth Gulf to their eventual capture in the fishery. The model uses weekly estimates of the model parameters for growth and mortality and works in weekly time steps. Furthermore, the model design allowed the researchers to explore and quantify the effect of uncertainty for various enhancement scenarios and investigate the influence of time-at-release on prawn survival (Loneragan et al., 2004; Ye et al., 2005).
In contrast, *EnhanceFish* is a generalised model, originally designed for teleost fish populations, that can also be configured for a range of release programs, such as small community based programs; e.g. Thmorda reservoir in Cambodia, or large commerical operations e.g. North Sea Sole (*Solea solea*), by changing key parameters and data associated with a particular enhancement or restocking program (Lorenzen and Medley, 2006). It was created to provide fishery managers with a tool to quantitatively optimise enhancement scenarios by assessing, impact on yield, wild stock abundance and structure, economic performance, uncertainty and changes in fishing effort (Lorenzen and Medley, 2006).

Both models provide a valuable framework for researchers, managers, and fishers to consider all aspects of a release program and identify important knowledge gaps that are likely to influence the reliability of the model predictions.

3.5.7 Conclusions

Restocking the *M. dalli* population requires stocking densities and sizes-at-release that effectively and economically boosts its recruitment capacity (Walters and Kitchell, 2001; Lorenzen, 2005, 2008). However, it should be noted that it is impossible to avoid a compensatory response in the *M. dalli* population with increasing stocking densities (e.g. Myers and Cadigan, 1993, 1995; Liermann and Hilborn, 1997; Lorenzen, 2005). Furthermore, this preliminary quantitative analysis is the first step in developing an understanding of the *M. dalli* population’s compensatory responses. Additional experiments and field studies that test this
behaviour will further develop scientific knowledge and optimise the stocking density and size-at-release strategies.
4. Conclusions and recommendations

This study investigated the potential of restocking the Swan-Canning Estuary with *Metapenaeus dalli* and is part of a larger integrated research and development program conducting a trial to improve the depleted status of the stock. The primary objectives of this study were:

1) To use the data from a spatially and temporally comprehensive sampling program for *M. dalli* in the Swan-Canning Estuary to quantitatively estimate the biological parameters for growth, mortality and size at maturity; and

2) Develop a bioeconomic model to evaluate the potential increase in population biomass from restocking with different numbers of released prawns and different sizes-at-release.

Prawns were collected from 20 nearshore, shallow sites (i.e. < 2 m deep) using hand trawls and from 16 offshore, deeper sites (i.e. 2 to 17 m deep) using otter trawls in each month between October 2013 and September 2014. Length frequency analysis was used to estimate the growth, mortality and size at maturity of the *M. dalli* population and examine the fit to the data provided by a standard von Bertalanffy growth curves and ones with terms included to model seasonal growth. These quantitative estimates of growth are the first to be made for this penaeid. The results demonstrated that *M. dalli* exhibits a strong seasonal pattern of growth, with most growth occurring in the warmer months (i.e. October to March) and little to no growth during the colder months (i.e. April to August). The asymptotic carapace length (*L_∞*), estimated by the Somers’ (1988) seasonal growth model was much longer for females (33.7 mm CL, ≈ 23.8 g) than males (24.2 mm CL, ≈ 11.0 g).
However, the instantaneous growth parameter $k$ was similar for both sexes (females $= 1.06$ and males $= 0.97$).

The information from the length frequency distributions of the $M.\ dalli$ population in the Swan-Canning Estuary between 1977 and 1982 (i.e. Potter et al., 1986), was used to quantitatively estimate growth parameters and these parameters were compared with those from the current study. The estimates for the asymptotic length from the historical data were almost 16 to 18% smaller than those obtained from the current data for both females (16% smaller, $L_\infty = 28.0$ mm CL and $\approx 14.4$ g wet weight) and males (18% smaller, $L_\infty = 20.0$ mm CL and $\approx 6.3$ g wet weight). The estimates of $k$, however, were similar in magnitude between studies ($2013/14 \approx 1.02$; historical $\approx 1.03$). The difference in $L_\infty$ between the studies is possibly due to changes in fishing levels between the time periods, with recreational fishing pressure being far greater in the 1970s and 1980s than the current levels, which is likely to have removed the larger prawns in the population.

The data on the presence of gravid females and females carrying a spermatophore demonstrate that $M.\ dalli$ has a strong seasonal reproductive cycle with reproductive activity recorded between November and April, when water temperatures in the estuary varied from 20 to 27 °C. This timing of reproduction is similar to that reported by Potter et al. (1986) for $M.\ dalli$ 30 years ago. These authors concluded that this seasonal pattern of reproduction was an adaptation to the hydrologic and climatic conditions in the estuary. Thus, this timing of reproduction coincides with the warm, dry summer and early autumn months, when water movement due to tidal action and freshwater discharge is limited and salinities and temperatures remain
relatively stable. These environmental and hydrological conditions are ideal for spawning, retention of larvae within the estuary and good recruitment of many local fish species, such as atherinids, the gobbeguts *Ororhinchus rueppelli* and the black bream *Acanthopagrus butcheri* (e.g. Prince and Potter, 1983; Chrystal et al., 1985; Potter and Hyndes, 1999; Sarre and Potter, 1999).

The length frequency data and estimates of growth from the 2013/14 data were used to estimate the mortality of *M. dalli* for the first time using seasonal length converted catch curves and non-seasonal length converted catch curves described by Pauly (1990). The relatively high mortality of the population (females = 3.46 year$^{-1}$ and males = 2.69 year$^{-1}$) leads to the virtual absence of larger prawns after 12 months, particularly females. These findings, combined with the seasonal pattern of reproduction, imply that the majority of female prawns probably live for only one spawning season, although they may spawn multiple times during this period. Consequently, recruitment to the *M. dalli* population is determined largely by the spawning success in one summer/autumn period. Since this species is confined to estuaries in south-western Australia, no recruitment is likely to come from sources outside the Swan-Canning Estuary, which highlights the need to develop strategies for population recovery.

A bioeconomic model was developed in *EnhanceFish* to evaluate the potential contribution of different scales of restocking and different sizes-at-release to the population biomass of *M. dalli* in the Swan-Canning Estuary. Data from the current sampling program and estimates of the area of habitat available for *M. dalli* were used to calculate a current population biomass of approximately 2.4 tonnes. This
estimate, combined with the biological parameters for growth, mortality and reproduction provide the input parameters for the EnhanceFish model. In general, the results from the bioeconomic model indicated that small to moderate potential increases in biomass of the *M. dalli* population would be achieved for releases of 650,000 to 5 million prawns and at a range of sizes (1 to 10 mm CL). The estimated increases in biomass for different numbers (650,000, 1 million, 2 million and 5 million prawns) and sizes-at-release (1 mm to 10 mm CL) ranged from 0.1 tonnes (650,000 at 1 mm CL) to 4.2 tonne (5 million at 10 mm CL) over a five year period. The rate of increase in population biomass decreased as the numbers released increased above 1 million prawns, indicating that density-dependent effects are increasing.

It should be noted, however, that EnhanceFish has some important constraints when applied to short-lived species, such as *M. dalli*, that need to be considered. For example, it does not use a seasonal growth model nor have the functionality to model the intra-annual timing of release for stocked prawns. Nonetheless, as a preliminary exploration into the potential of restocking the Swan-Canning Estuary, the results are encouraging for releases of up to 1 million prawns with a size-at-release of 10 mm CL. The model also included only those operational costs for producing the prawns and not any capital costs. These will be particularly important if production is scaled up to a level that would require additional infrastructure and attempts are made to produce larger sizes of prawns for release. This may require a different grow-out environment such as the experimental raceways that were used to grow tiger prawns from older postlarvae to a size of 10 mm CL (≈ 1 g) (Ye et al., 2005).
4.1 Recommendations for future research

Despite an extensive search for literature, no information was found on the socio-economic benefits of recreational prawn fisheries for any species of *Metapenaeus* or *Penaeus*. Neither, were studies of the economic returns or value of recreational utility found for recreational prawn fisheries, including fisher effort dynamics. Studies like these would be a useful addition to our understanding and evaluation of the potential benefits of restocking *M. dalli* in the Swan-Canning Estuary.

The preliminary results from this bioeconomic evaluation require further exploration to investigate the behaviour of the *EnhanceFish* model for a short-lived species, particularly through comparing the performance of this model with other models. Further research in the following areas would achieve this:

i) focusing on refining the estimates of mortality used in the model from empirical studies to estimate density-dependent mortality, release mortality and variation in mortality with size of prawns;

ii) investigating the importance of time-at-release on the population biomass, and

iii) comparing the predictions and behaviour of the model with those from a model designed specifically for short-lived Penaeid prawns (i.e. that developed by Ye et al., 2005).
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Publication List

The contents of Chapter 2 were presented at an international Conference organised by the Fisheries Society of the British Isles entitled ‘Integrated perspectives on fish stock enhancement: integrating biological, socio-economic and managerial dimensions of fish stocking and introductions’

Estimation of biological parameters for evaluating the success of *Metapenaeus dalli* releases in a temperate Australian estuary

Andrew Broadley, James R. Tweedley, Brian Poh & Neil R. Loneragan
Centre for Fish and Fisheries Research, Murdoch University, Australia
j.tweedley@murdoch.edu.au

**INTRODUCTION**

- The Western School Prawn (*Metapenaeus dalli*) was an important recreational and commercial species in the Swan-Canning Estuary. However, over the last fifty years, their numbers have declined significantly.
- The aim of this study is to assess the feasibility of restocking by evaluating the biological and economic parameters of this species.

![Fig. 1](image)

**RESULTS**

- Western School Prawns complete their life cycles within estuaries, reaching maturity within 12 months and some live for up to 24 months.
- Protracted mating and spawning period between December and April.
- Time of reproduction coincides with water temperatures above 20 °C.

**METHODS**

- Sampling was conducted at 20 shallow sites (i.e., ≤1.5 m deep) and 16 deep sites (i.e., 1.5 to 7 m deep) in the Swan-Canning Estuary every 28 days on the new moon between October 2013 and June 2014.
- Prawns were collected using a 4 m wide hand trawl (9 mm mesh) in shallow waters and a 2.6 m wide otter trawl (26 mm mesh) in deep water.
- Length frequency analysis undertaken in R using the package Mixsats to fit finite mixture distribution models to histograms.
- Growth models of current (2013/14) and historical (i.e., 1980-1982 from Potter et al., 1986) data created in R, with parameters estimated using the FDA package implementation of Simen’s seasonal adaptation of the von Bertalanffy growth model.

**RESULTS CONT.**

- Strong seasonal pattern of growth, with virtually all growth occurring in the austral spring and summer (October to February) and very little growth in winter (June to August).
- Seasonal pattern of growth more pronounced in 2013/14 than in 1980-82.

**CONCLUSION**

- These data, once complete, combined with information on the costs of aquaculture production will provide the basis for a bio-economic assessment for restocking this species.

**REFERENCES**


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