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IMAS
INSTITUTE FOR MARINE AND
ANTARCTIC STUDIES

**Exploring biases in estimating survival probability of marine
resources from mark-recapture analyses:**

a case study with the southern rock lobster *Jasus edwardsii*

Ziya Kordjazi

Submitted in fulfilment of the requirements for the degree of Doctor of Philosophy,

University of Tasmania, June 2015

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STATEMENT OF Co-AUTHORSHIP

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Abstract

Capture-mark-recapture (CMR) modelling is commonly used for direct estimation of demographic parameters in the wild. Whilst it is well known that the precision of estimated parameters in CMR studies increases with sample size, duration of study and the number of recapture events there are few studies that explore the design of CMR projects to achieve optimal outcomes. Research funding and time allocation are two constraints that place restrictions on the research design. This study addresses this challenge and seeks to demonstrate issues that need to be addressed and ways to design CMR that minimise costs while still providing the precision and accuracy required for assessment of marine resources .

A twelve year tagging data set for *Jasus edwardsii* at a non-fishing site (Crayfish Point Scientific Reserve, Tasmania, Australia) was used in the study. Cormack-Jolly-Seber (CJS) modelling was used to compare estimates of survival probability with increasing numbers of surveys from 3 to 12 for both equal (annual) and unequal survey intervals.

The study found that more surveys were required to obtain precise estimates of survival probability in females than in males, and the number of surveys required increased when intervals between surveys were unequal. For annual surveys, stable estimates of survival probability generally required at least five years, which is often beyond the three-year duration commonly used in scientific studies.

Five annual surveys were used to determine the effect of sample size and fishing effort (the number of traps set per day and days fished) on the precision of estimated survival probability. Larger sample sizes (≥ 500) and more sampling effort were required to determine sex specific survival probability estimates.

Tag induced mortality (TIM) and tag loss reduce the number of tagged lobsters in the population and result in an under-estimation of survival probability. TIM is often investigated in aquaria studies with small sample sizes and over short time periods. In this study, a model based on three

annual-surveys was used to provide *in-situ* estimates of TIM proportion. Estimates of TIM proportion were high and varied being on average between 25% and 40%. Capturing sufficient lobsters during the third survey that were tagged in the first survey and also seen in the second survey was a major limitation in using this method.

Tag loss is usually estimated by double tagging in the wild. This study investigated the fate of new tags and existing (old) tags applied to lobsters. Estimated tag loss rate was different between males and females and the annual rate of tag loss was similar between years during the two year study indicating that tag loss is chronic and needs to be accounted for throughout the entire period of mark-recapture programs. In general, older tags were lost at a slightly higher rate than new tags.

The design of mark-recapture programs will be dependent on the species under study and although these results are specific for *J. edwardsii*, the issues considered and methods used in this thesis should assist researchers in the design of tagging surveys. Importantly, both tag loss and tag induced mortality can be of a magnitude that impacts estimates from mark recapture studies and the thesis demonstrates issues to be considered in designing a tagging program that meets the objectives of the study and also enables estimation of issues such as tag loss rate and TIM proportion which can bias survival probability and population estimates.

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Chapter 1

General Introduction

Chapter 1 General Introduction

Capture-mark-recapture (CMR) modelling is a tool that can be used for investigating the dynamics of fished populations where it is not practical to capture all members of a population or to make direct observations as in aquatic populations (Bell *et al.*, 2003). CMR provides an opportunity to track individually marked animals over time (Lebreton *et al.*, 1992) and to directly estimate demographic parameters in the wild.

Whilst, a wide range of CMR models associated with the objective of the studies (Yoccoz *et al.*, 2001) has been used with closed, open, and combined (robust) population models (Otis *et al.*, 1978; Pollock *et al.*, 1990), some challenges, including research funding associated with study design (sample size and study period) and the effect of tagging on animals (tag loss and tag induced mortality) put restrictions on CMR studies. This affects the desired precision and accuracy in the estimation of demographic parameters including those used in assessing exploitable fish stocks and thus the setting of sustainable catch limits.

Collecting data in CMR studies on marine resources can be an expensive process as it often involves the charter of vessels and gear, scientific and technical labour, and tagging equipment. Whilst, it is well known that the accuracy and precision of estimates can improve with increasing sample size and survey period, research funding and time can affect the final CMR design. Minimising costs can have substantial benefits but needs to be balanced against the objectives to ensure that sustainable management of the resource can be achieved. This is mainly through ensuring the appropriate precision and accuracy in parameter estimates.

This study has focused on the design of southern rock lobster tagging experiments to determine design issues to meet the objectives of minimal costs while ensuring appropriate outcomes. In particular, the study focused on issues that impact the precision and accuracy of parameter estimates including survey design (i.e. number of surveys, samples, and sampling effort) and tagging effects that could bias estimates.

1.1 Study area

This study was conducted in the Crayfish Point Scientific Reserve (CPSR), Tasmania, Australia, where southern rock lobster has been protected from fishing since 1971. Tagging surveys have been undertaken approximately annually in the reserve since 2000 as well as several ad-hoc surveys undertaken within years associated with other research activity. However, the objectives for many of the surveys varied resulting in considerable variation in the number of days that sampling was undertaken in each survey, the number of traps used each day and the number of lobsters captured and tagged. While these inconsistencies in sampling design can cause issues in the reliability of estimates, they also provide a basis from which to understand how these changes influence the precision and accuracy of results.

The population of lobsters in the CPSR is only affected by natural mortality, as removal of tagged lobsters from the resource from fishing was not an issue, and emigration from the area was also considered minimal as southern rock lobster has high site fidelity (Barrett *et al.*, 2009). Gardner *et al.* (2003) also found no evidence of migration of *Jasus edwardsii* in Tasmania. As such, the CPSR provides an appropriate study site to determine survival or natural mortality (1-survival) rate. However, caution does need to be used when applying these estimates to fished populations as the size composition and density of lobsters is not similar to exploited regions of the resource.

Another benefit of the CPSR is that it is bounded by large regions of sand habitat making the reef study region a bounded and constant area between surveys. However, a major perturbation in the lobster population within the CPSR occurred in November 2005 when 1998 tagged lobsters were introduced to the population as part of a separate study (Green and Gardner, 2009). The effect of this perturbation on the CJS mark recapture models was assessed. Other principal changes occurred in the annual surveys with a declining number of survey days associated with minimising research costs. The major change was a reduction from a two week (12 days) to a single working week (Monday to Friday) sampling regime.

1.2 Cormack -Jolly-Seber (CJS) model

The Cormack-Jolly-Seber (CJS) model was developed to study apparent survival probability from capture-recapture data (Cormack, 1964; Jolly, 1965; Seber, 1965), where it is not possible to separate natural mortality (death) and permanent migration. Capture probability is defined as the probability that animals alive and present at one capture occasion are captured. The survival probability is defined as the probability that animals alive at one capture occasion are still alive at the next. The main CJS model assumption is that all animals in the population have the same capture and survival probabilities at a single capture occasion.

Assumptions of CMR methods are outlined below and violation of these may lead to bias in parameter estimation (Pollock *et al.*, 1990; Pollock *et al.*, 2007). These include:

- (1) An equal probability of capture for every individual in the population at each sampling event. Biases in population estimates will occur if tagged and non-tagged animals do not mix freely and uniformly after tagging. Biases in estimates of survival probability will also occur if different tagging cohorts do not mix freely with each other.
- (2) An equal probability of survival for every marked animal from one survey event to the next, irrespective of when it was tagged.
- (3) Tag loss is zero or negligible. By decreasing the number of animals available to be recaptured in the population, tag loss can result in serious under-estimation (negative bias) of survival rates (Pollock *et al.*, 1990) and over-estimation (positive bias) of population size (McDonald *et al.*, 2003; Pollock and Alpizar Jara, 2010)
- (4) After sampling all animals are released immediately back to the location of capture and the sampling period is of short duration compared to the periods between successive tagging events. Recapture rate may be affected if the tagged animals are held for longer periods before release (Courtney *et al.*, 2001).

(5) Emigration is permanent. Temporary emigration may lead to biased survival probability and population size estimates.

Pollock *et al.* (1990) showed that violations of the first assumption related to inherent heterogeneity in recapture and survival probabilities, which could cause small positive or negative bias in estimates of survival probability. Similarly, heterogeneity of survival rates in combination with heterogeneous capture rates could cause positive or negative biases in survival rates (Pollock *et al.*, 1990). Both tag loss and tag induced mortality impact assumption 3. Frusher *et al.* (2009) found that there was little difference in tag loss between small and medium sized lobsters that reflected the size composition in fished regions although larger lobsters (well above the legal size limit and rare in the commercial fishery) tended to have a higher rate of tag loss. Although sample sizes were small, Frusher *et al.* (2009) also reported that tag loss in smaller lobsters tended to be greater than the average during the first year after tagging. Higher tag loss for smaller lobsters was also found for the European spiny lobster *Palinurus elephas* (Gonzalez-Vicente *et al.*, 2012). In addition, sex and the stage of moulting have an impact on tag loss (Melville-Smith and Chubb, 1997; Frusher *et al.*, 2009; Gonzalez-Vicente *et al.*, 2012). Each reported a lower rate of tag loss in females than in males.

Previous studies of tag induced mortality (TIM) have been undertaken in artificial environments (aquarium and sea-cage) and have shown high estimates (30-50 %) for lobsters. Some of these high estimates have been considered to be biased due to additional stress associated with bottom contamination in artificial environments (Montgomery and Brett, 1996), low sample size (Dubula *et al.*, 2005; Claverie and Smith, 2007) or short experimental duration (Claverie and Smith, 2007).

1.3 Approach and thesis structure

A long term T-bar tagging dataset conducted on *Jasus edwardsii* population from 2000 to 2012 in the Crayfish Scientific Point Reserve, near Hobart, Tasmania, Australia provided an ideal

opportunity to investigate how accuracy and precision of survival probability of lobster would change with sample size, survey number and interval. Chapter 2 was designed to find whether a three-year survey period which is common in scientific investigation due to standard funding durations, could provide reliable and precise estimations of survival probability.

Sample size and sampling duration have been shown to impact the estimates of growth (Fiske *et al.*, 2008) and the precision of other estimated population parameters (Lindberg and Walker, 2007) in both matrix and CMR modelling. The precise and the most cost-effective survey number derived in this study (five surveys - Chapter 2) was used to investigate the effect of sample size and fishing effort on the precision of the survival probability estimates and to determine the minimum sample size and effort required to obtain precise estimates (Chapter 3).

As the estimates of survival probability are conditional on the number of tagged lobsters alive in the population, it is also important to understand what may impact the number of lobsters that are available for recapture, that is, if there are changes between the number tagged at time “ t ” and those available to be captured at time “ $t+1$ ”. In addition to a direct bias induced on survival rate (i.e. 1 - natural mortality), tagged induced mortality (where the lobster dies from the tagging process and is no longer available to be captured in the following survey) and tag loss (where the lobster is still alive but no longer carries the tag and cannot be identified as a tagged lobster during the next survey) change the number of tagged lobsters available to be recaptured. In the second part of this thesis I investigated tag induced mortality proportion (Chapter 4) and tag loss rate (Chapter 5).

Tag induced mortality (TIM) rate and/or proportion is usually estimated in an artificial environment where it is restricted by small sample size and short study duration. Biased estimates of TIM rate and/or proportion are considered to be usually obtained by these methods, as the effect of predators on tagged animals and other environmental circumstances cannot be replicated by laboratory and cage studies. In Chapter 4, a three-survey model was used to provide an estimate of TIM. In addition to providing an *in-situ* estimate of TIM rate and/or proportion, Chapter 4 also explores

sampling design issues which may impact the use of this method and the reliability of the estimate, especially the sample size required to obtain suitable estimates.

Tag loss is normally estimated by the use of double tagging methods in the field, where two tags are synchronously applied to each lobster. In Chapter 5, annual tag loss was estimated for lobsters over two survey years. In addition, the rate of tag loss for new tags and existing tags was compared to evaluate the effect of the timing of tagging on tag loss.

In Chapter 6, the results from chapters 2 to 5 are discussed and a design outlined that should address the key concerns highlighted in this thesis.

Chapter 2

Estimating survival probability of rock lobsters from long-term tagging programmes – how survey number and interval influence estimates

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Chapter 2 Estimating survival probability of rock lobsters from long-term tagging programmes – how survey number and interval influence estimates

Abstract

A long-term tagging data set on southern rock lobster *Jasus edwardsii* conducted at the Crayfish Point Scientific Reserve (CPSR) Taroona near Hobart, Tasmania, Australia was used to determine how survey number and survey duration affected the precision of survival probability estimates for male and female lobsters in order to ensure sustainable exploitation of the population. Tagging surveys were undertaken twice a year during 2000-2003 with unequal time-intervals between surveys and then once per year with equal time intervals during 2004-2012 during the January-February period. The most parsimonious Cormack-Jolly-Seber (CJS) model for estimating survival probability of lobsters was dependent on (i) timing of the tagging and recapture surveys, (ii) time between consecutive recapture surveys, and (iii) sex. The number of surveys required to provide a precise survival probability varied with sex and time between recapture surveys. For surveys where there was unequal time between recapture surveys, seven and five surveys were required for female and male lobsters, respectively, whereas only five surveys were required when annual surveys were undertaken. Thus, lobster tagging projects using annual surveys should ideally extend to at least five years, which is beyond the three year project duration common in marine science.

Key words: Capture-mark-recapture (CMR), Cormack-Jolly-Seber (CJS) model, survival probability, lobster.

2.1 Introduction

Capture-mark-recapture (CMR) modelling is widely used for estimating demographic parameters including survival rate and population size (Pollock *et al.*, 1990). Estimating survival probability is especially important for harvested species where one is attempting to sustainably exploit the population. CMR studies provide an opportunity to follow an individually marked animal through

37time (Lebreton *et al.*, 1992), and can provide insights into biotic and abiotic environmental factors that impact survival and migration and thus the productivity of the population.

CMR designs, including marking and data collection methods are varied and determined by the objective(s) of the study (Yoccoz *et al.*, 2001). Different applications of CMR models require examination of the assumptions made in each case to minimise violations (Pollock *et al.*, 1990) whilst attaining the most precise and accurate estimates of demographic parameters. Otis *et al.* (1978) reported that factors affecting CMR studies in trap-based fisheries can be categorised into two groups: 1) statistical design and data recording, including the number of capture surveys, interval between surveys, size and shape of traps, spacing of traps and number of traps at a specific location; and 2) field procedures, including the method of marking or tagging, method of recapture, use of bait, time of day when traps are checked and handling of the animals.

Sampling design or data collection methods may also be limited by the costs of surveying. While the precision of estimated parameters increases with the number of sampling events and tagged animals (Beasom *et al.*, 1986), there is a trade-off with the cost of the program. For example, Rozas and Minello (1997) noted that resources and time may be wasted by conducting excessive sampling events while the precision of the results may be affected by too few samples. In addition, designing a mark recapture plan beyond the required precision wastes funds (Frusher and Hoenig, 2001b; Frusher *et al.*, 2009). However, there are few cases where these have been empirically tested. For example, the impact of the number of recapture events on the estimation of survival probability in open populations has not been examined.

The aim of this study was to determine how the number of surveys and different time intervals between surveys affect the precision of estimates of survival probability for male and female rock lobsters. This was achieved using a long-term, mark-recapture data set of southern rock lobster *Jasus edwardsii*, collected over 13 years (2000-2012) from the Crayfish Point Reserve on the east coast of Tasmania, Australia.

2.2 Methods

2.2.1 Study site

Tagging surveys were conducted at the Crayfish Point Scientific Reserve (CPSR), Taroona, near Hobart, Tasmania, Australia (42° 57' 08"S 142° 21' 20"E), which has been closed to fishing for *Jasus edwardsii* since November 1971. The reserve covers around 1.24 km² and has a mixture of temperate rocky reef and sandy substrate to a maximum depth of 18 m. A major change to the population of *J. edwardsii* occurred in November 2005 when 1998 tagged lobsters (565 males and 1433 females) were translocated to the CPSR from deeper water regions off southern Tasmania. All these lobsters were below the legal minimum lengths (LML) of 105 mm CL (females) and 110 mm CL (males) (Green and Gardner, 2009).

2.2.2 Field methods

A long term tagging program of *J. edwardsii* in the CPSR has been ongoing since 2000. This work has been undertaken to address several projects, which resulted in variation between surveys in terms of effort, sample size and survey interval. Seventeen major tagging surveys were undertaken comprising eight surveys of unequal time intervals from 2000 to 2003 and nine surveys at approximately equal (annual) time intervals from 2004 to 2012 (Table 2-1). Collectively, this provided a long-term (13-year) capture-recapture experiment.

As the number of traps and the number of trapping days varied between surveys, the number of lobsters captured per trap was used to standardise these variations to enable comparisons between surveys. Trapping occurred during consecutive days during all surveys. On the first day of each survey, traps were baited and set in the early afternoon. On subsequent days the traps were checked in the morning, re-baited and re-set. All untagged lobsters were tagged by uniquely coded T-bar tags (Hallprint T-bar anchor tag; TBA1, Hallprint Pty Ltd, 27 Jacobsen Crescent, Holden Hill,

South Australia 5088, Australia) on the ventral side of the first or second abdominal segment. The tag number, length of carapace and sex of all lobsters were recorded and lobsters were released back immediately to the wild after tagging.

Lobsters captured in recapture surveys following the initial tagging event were in one of three groups:

- (i) untagged lobsters that were subsequently tagged;
- (ii) lobsters that had been tagged in a previous survey of this study; and
- (iii) lobsters tagged in other studies, such as prior to the first survey of this study (i.e. prior to 2000) or been translocated to the reserve, when group (iii) lobsters were first seen in a survey used in this study, they were considered “untagged” and treated as newly tagged lobster with the original tag number being treated as a new tag number. For these lobsters we assume that they had no difference in tag induced mortality proportion and that their probability of recapture was equal to newly tagged lobsters.

2.2.3 Mark-recapture modelling (statistical methods)

Model selection and data analyses were based on Cormack-Jolly-Seber (CJS) models (Cormack, 1964; Jolly, 1965; Seber, 1965). Program Mark was used for fitting the models (White and Burnham, 1999).

Survival probability (ϕ) and recapture probability (p) were estimated from the survey data set of time (t) and sex (g) by maximum likelihood. Sixteen parameter combinations of survival and capture probability were estimated with constant (time and sex-independent) survival probability ($\phi(.)$), time specific survival probability ($\phi(t)$), sex specific survival probability ($\phi(g)$) or time and sex specific survival probability ($\phi(g*t)$) with constant (time and sex-independent) capture

probability ($p(.)$), time specific capture probability ($p(t)$), sex specific recapture probability ($p(g)$), or time and sex specific recapture probability ($p(g*t)$). The fully parameterised (saturated) model was $\phi(g*t)p(g*t)$.

Models were run for a set of surveys that had at least three survey events. The first survey was the tagging survey and was followed by a series of mark-recapture events to estimate the change in the accuracy and precision of survival probability.

Two designs were chosen to determine the impacts of study design on the accuracy and precision of survival probability estimates.

1. Effect of increasing the number of subsequent surveys. Apparent survival probability was estimated and compared for models that used 2, 3, 4, etc. subsequent surveys after the initial tagging survey (Table 2-1). This was repeated using every survey (e.g. Nov 00, Nov 01 etc.) as the initial tagging event.
2. Effect of different tagging times. This design kept the number of surveys constant and varied the initial tagging event. For example, survival probabilities estimated from three survey events with a different initial tagging event were compared (Table 2-2). This was repeated for models using 4, 5, 6 etc. surveys.

Table 2 - 1 Example of Design 1 using Jan-Feb 2000 as the initial tagging survey and survival probability estimated from models with increasing numbers of subsequent surveys. Time intervals were unequal.

Tagging Events (S1)	Recapture events for Tagging events (Jan-Feb 2000). The interval between recapture events was unequal.															
	Nov 00	Nov 01	Dec 01	Jan-Feb 02	Nov 02	Jan-Feb 03	Nov 03	Jan-Feb 04	Jan 05	Jan-Feb 06	Jan 07	Jan 08	Feb 09	Jan 10	Jan 11	Feb 12
JF 00 (S1)	S2	S3														
JF 00 (S1)	S2	S3	S4													
...	...															
JF 00 (S1)	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	
JF 00 (S1)	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17

Table 2 - 2 Example of Design 2 involved comparison of the three survey models run over different time periods.

Tagging Events	Tagging event (S1) and two Recapture events (S2 and S3) associated with each initial tagging event											
	Nov 00	Dec 01	Jan-Feb 03	Jan-Feb 04	Jan 05	Jan-Feb 06	Jan 07	Jan 08	Feb 09	Jan 10	Jan 11	Feb 12
Nov 2000	S1	S2	S3									
Dec 2001		S1	S2	S3								
...												
Jan 2010										S1	S2	S3

Akaike Information Criteria (AICc), that accounts for small sample sizes (Hurvich and Tsai, 1991) , was used to select the best model(s). Delta AICc, the difference in AICc of each candidate model relative to the model found to have the lowest AICc, was used to determine the most parsimonious model from the candidate set of models (Burnham and Anderson, 1998). In general, any models with the delta AICc score of <2 was considered to have substantial support in CMR studies (Boulanger *et al.*, 2006).

Sixteen candidate models were available for each analysis. Based on the delta AICc scores each model was coded relative to the Delta AICc value. The codes of 1, 2, 3, 4 and 5 were applied to the models with the value of $0 \leq \Delta AICc < 2$, $2 \leq \Delta AICc < 4$, $4 \leq \Delta AICc < 7$, $7 \leq \Delta AICc < 10$, and $\Delta AICc \geq 10$, respectively. For each initial starting survey (e.g. Jan-Feb 00 in Table 2-1) CJS analyses were undertaken for each combination of surveys by incrementally adding a survey and repeating the analysis until the last survey in February 2012 (i.e. 3 surveys, 4 surveys, 5 surveys etc). For each subsequent initial starting survey (e.g. Nov 00, Nov 01 etc) the number of possible CJS analyses declined (e.g. 9 possible analyses for Nov 00, 8 possible analyses for Nov 01 etc). To determine the preferred model for each starting date, the analyses for each model (e.g. $\phi(g)p(g)$) were summed across all analyses (e.g. Jan-Feb00:3 surveys + Jan-Feb00:4 surveys + Jan-Feb00:5 surveys etc) and divided by the total number of analyses. As it is possible to have a range of models that are equally likely, all models that were within 0.2 units of the lowest value were considered to be likely.

Relative standard error (RSE) was used to determine the precision of the estimated survival probability with low relative standard error, indicating high precision of estimates (Lindberg and Walker, 2007).

Relative standard error was calculated as $= \frac{\overline{SE}}{\bar{X}}$, where \overline{SE} and \bar{X} are the standard error and mean of the parameter estimates, respectively.

2.3 Results

A total of 12,317 *J.edwardsii* were tagged in 17 separate surveys from 2000 to 2012. The number of captured lobsters (capture per trap) was variable between traps within surveys, between surveys and between sexes. The mean number and standard deviation of males captured per trap (1.62 ± 0.47) was greater than the mean number of females (1.12 ± 0.40) (Figure 2-1, Figure 2-2 and Table 2-3).

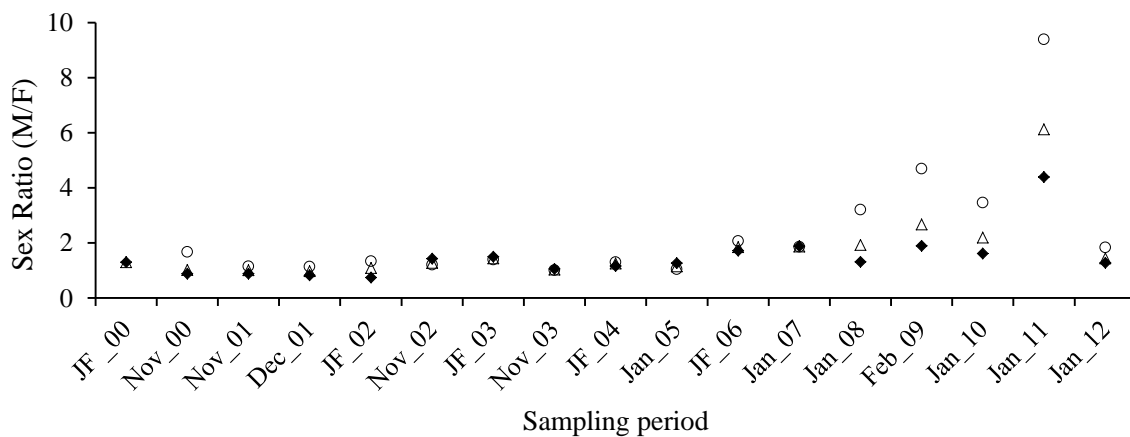


Figure 2 - 1 Sex ratio (M/F) of newly tagged (◆), recaptured (○) and total captured (Δ) lobsters for each sample period. A value of more than one indicates that the sample was skewed towards males.

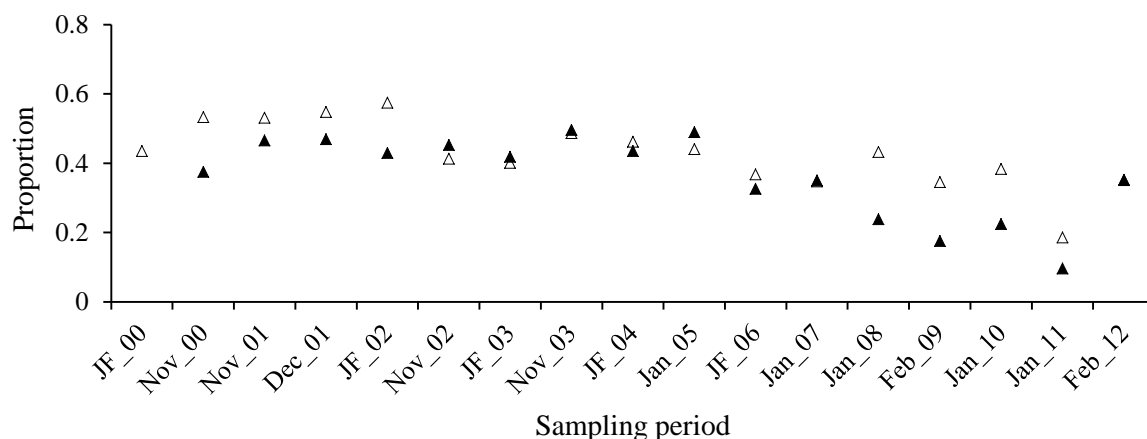


Figure 2 - 2 Proportion of newly tagged females (Δ) to total newly tagged lobsters and proportion of recaptured females (▲) to total recaptured lobsters.

Although the sex ratio (and standard deviation) of males and females captured from 2000 to 2005 was similar (1.14 ± 0.15), the sex ratio became skewed towards males from 2006 onwards (2.57 ± 0.16). This peaked in the 2011 survey when the sex ratio was 6.02 (Table 2-3).

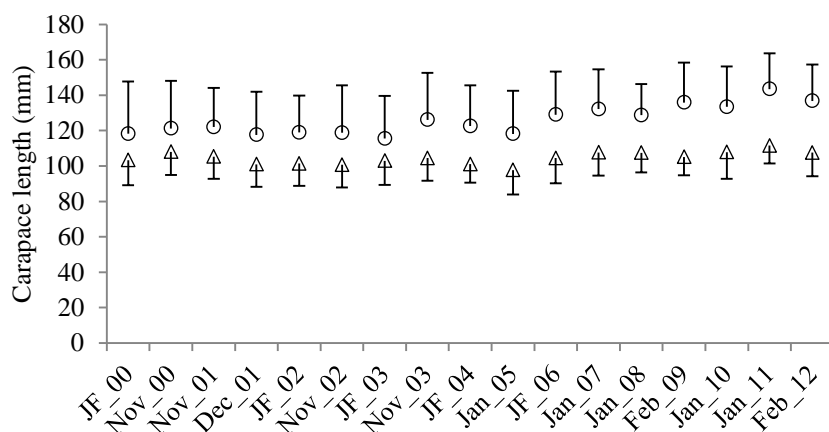


Figure 2 - 3 Comparison of mean carapace length (mm) for male (○) and female (Δ) newly tagged lobsters for surveys from 2000 to 2012. Error bars are standard deviation (+sd for males and -sd for females).

The mean size of newly tagged lobsters showed only minor changes between surveys prior to translocation in January –February 2006 (Figure 2-3).

2.3.1 Model selection

The effect of sex and time on estimated survival probability varied both within and between models for the equal and unequal survey periods (Table 2-4). The preferred models for the equal time period shifted from sex dependent in the early survey period to either constant (sex/time-independent) and/or sex dependent models towards the end of the survey period before time became a factor in the latter surveys. The preferred model for the unequal time period surveys tended to be the sex-dependent (g) or sex and time-dependent ($g*t$) during the period when greater than one survey was undertaken each year (2000 – 2003). In all models the recapture probability was sex and time dependent.

For models using a different number of surveys and equal recapture intervals, the most parsimonious model was usually sex dependent for five or more surveys. In contrast, where there was an unequal recapture interval, the most parsimonious model was primarily sex and time dependent irrespective of the number of surveys (Table 2-5). The following results on the effect of the number of sampling events, the start data and the precision of survival probability estimates are reported for each sex and based on outputs from the $\phi(g)p(g*t)$ as the most consistent model across all analyses.

2.3.2 Survival probability – effect of number of sampling events

The apparent survival probability estimates of females were higher than males although the precision (RSE) was more variable for females for any given model (Figure 2-3).

In general, at least five surveys were required to obtain stable and thus reliable estimates of survival probability for males whereas estimates for females required approximately seven surveys before they became stable.

Table 2 - 3 Summary of sampling statistics for each survey event (TC=total capture, F=female, M=male).

Parameters	Jan- Feb 2000	Nov 00	Nov 01	Dec 01	Jan- Feb 02	Nov 02	Jan- Feb 03	Nov 03	Jan- Feb 04	Jan 05	Jan- Feb 06	Jan 07	Jan 08	Feb 09	Jan 10	Jan 11	Feb 12
Trap lifts (days)	16	9	8	7	10	4	11	9	7	10	12	7	4	4	4	4	4
TC Female	697	1170	672	1057	660	358	704	487	455	946	548	638	246	129	242	75	373
TC Male	904	1192	675	1028	723	453	1010	502	567	1086	1017	1191	474	345	530	459	547
Total Captures	1601	2362	1347	2085	1383	811	1714	899	1022	2032	1565	1829	721	474	772	534	920
Traps	833	891	444	602	495	310	801	399	403	708	885	419	232	239	240	240	240
Capture/trap	1.92	2.65	3.03	3.46	2.79	2.61	2.14	2.25	2.53	2.87	1.77	4.36	3.11	1.98	3.22	2.22	3.83
Capture /trap (F)	0.84	1.31	1.51	1.75	1.33	1.15	0.88	1.22	1.13	1.34	0.62	1.52	1.06	0.54	1.01	0.31	1.55
Capture /trap (M)	1.08	1.34	1.52	1.71	1.46	1.46	1.26	1.26	1.41	1.53	1.15	2.84	2.04	1.44	2.21	1.91	2.28
Sex Ratio New Tag (M/F)	1.30	0.88	0.88	0.83	0.74	1.42	1.49	1.05	1.16	1.27	1.72	1.88	1.31	1.89	1.61	4.39	1.28
Sex Ratio Recapture (M/F)		1.66	1.15	1.13	1.33	1.20	1.40	1.01	1.30	1.04	2.07	1.85	3.2	4.7	3.46	9.38	1.84
Sex Ratio (Total)																	
Capture (M/F)	1.30	1.02	1.00	0.97	1.09	1.27	1.44	1.03	1.25	1.14	1.85	1.87	1.93	2.67	2.19	6.02	1.44
Translocated Lobster Total Recap / trap											0.23	0.36	0.20	.015	0.27	0.17	0.18
Translocated Lobster (M) Recap/trap											0.12	0.22	0.14	0.12	0.19	0.15	0.14
Translocated Lobster (F) Recap/trap											0.11	0.14	0.06	0.02	0.08	0.02	0.04

Table 2 - 4 Dependency of survival probability (ϕ) on sex (g), time (t) and/or sex and time (g^*t) in the range of models that were equally likely to be within 0.2 units of the lowest value of coded Delta AICc run for different tagging events with equal and unequal survey intervals. The general model is $\phi(x)p(g^*t)$ where “x” within the mentioned threshold is shown in the table and (.) denotes ϕ in the model was independent of sex and/or time.

Survey intervals	Tagging (or recapture) events												
	Jan-Feb 2000	Nov 00	Nov 01	Dec 01	Jan-Feb 02	Nov 02	Jan-Feb 03	Nov 03	Jan-Feb 04	Jan 05	Jan-Feb 06	Jan 07	Jan 08
Equal (Annual)	-	(g)	-	(g)	-	-	(g);(.)	-	(.);(g)	(.);(g)	(.);(g)	(t)	(.);(t)
Unequal	(g)	(g*t)	(g*t)	(g*t)	(g)	(g)	(g)	(g*t);(t)	(.);(g)	(.);(g)	(.);(g)	(t)	(.);(t)

Table 2 - 5 Dependency of survival probability (ϕ) on sex (g), time (t) and/or sex and time (g^*t) in the range of models that were equally likely to be within 0.2 units of the lowest value of coded Delta AICc run with the same number of surveys in equal and unequal survey intervals. The general model is $\phi(x)p(g^*t)$ where “x” within the mentioned threshold is shown in the table and (.) denotes ϕ in the model was independent of sex and/or time.

Survey intervals	number of surveys used in the models												
	3	4	5	6	7	8	9	10	11	12	13	14	15
Equal (Annual)	(g*t); (g);(t);(.)	(g);(t)	(g)	(g)	(g)	(g)	(g)	(g)	-	-	-	-	-
Unequal	(g*t)	(t)	(t)	(g*t)	(g*t);(g)	(g);(g*t)	(g)	(g)	(g)	(g*t);(g)	(g*t)	(g*t)	(g*t)

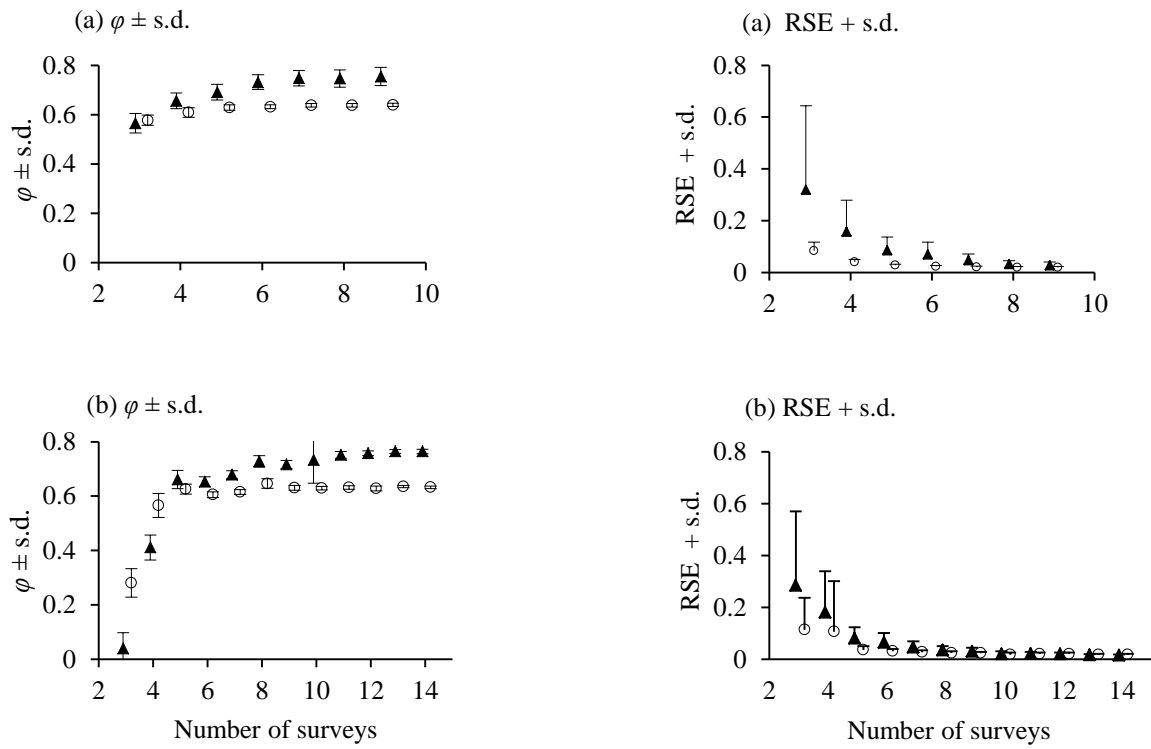


Figure 2 - 4 Average survival probability estimates ($\phi \pm \text{s.d.}$) and average relative standard error (RSE+s.d.), combined from several studies with the same number of surveys for males (○) and females (▲) with (a) equal (annual) and (b) unequal intervals between surveys.

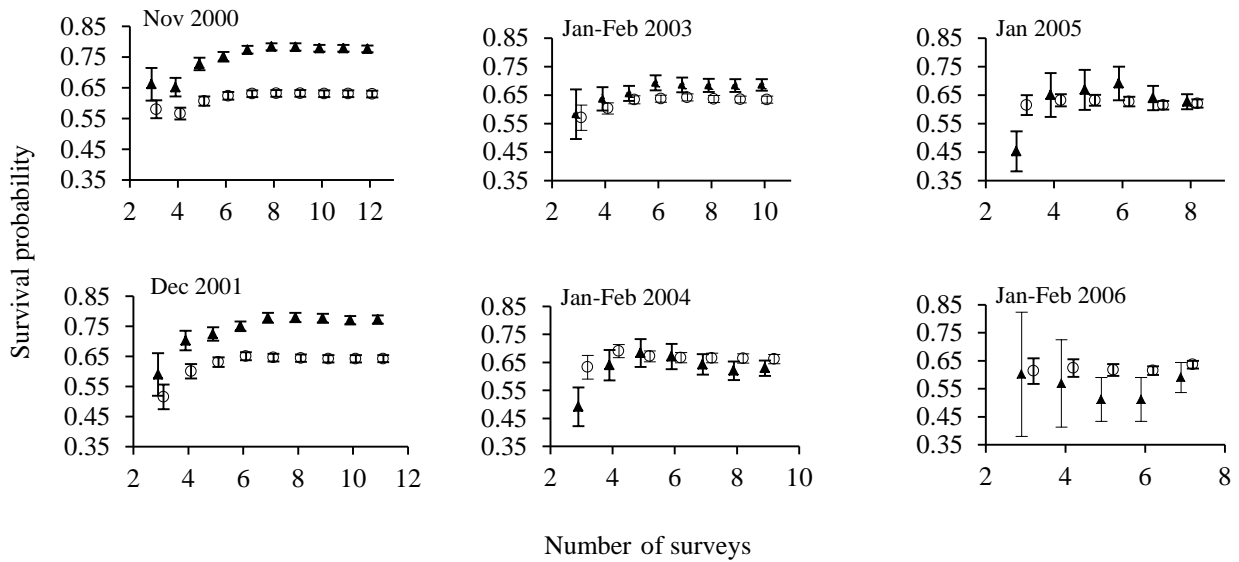


Figure 2 - 5 Annual survival probability estimates ($\phi \pm \text{SE}$) with different number of surveys and equal (annual) intervals for every study based on start times (Design 1); Males (○), Females (▲).

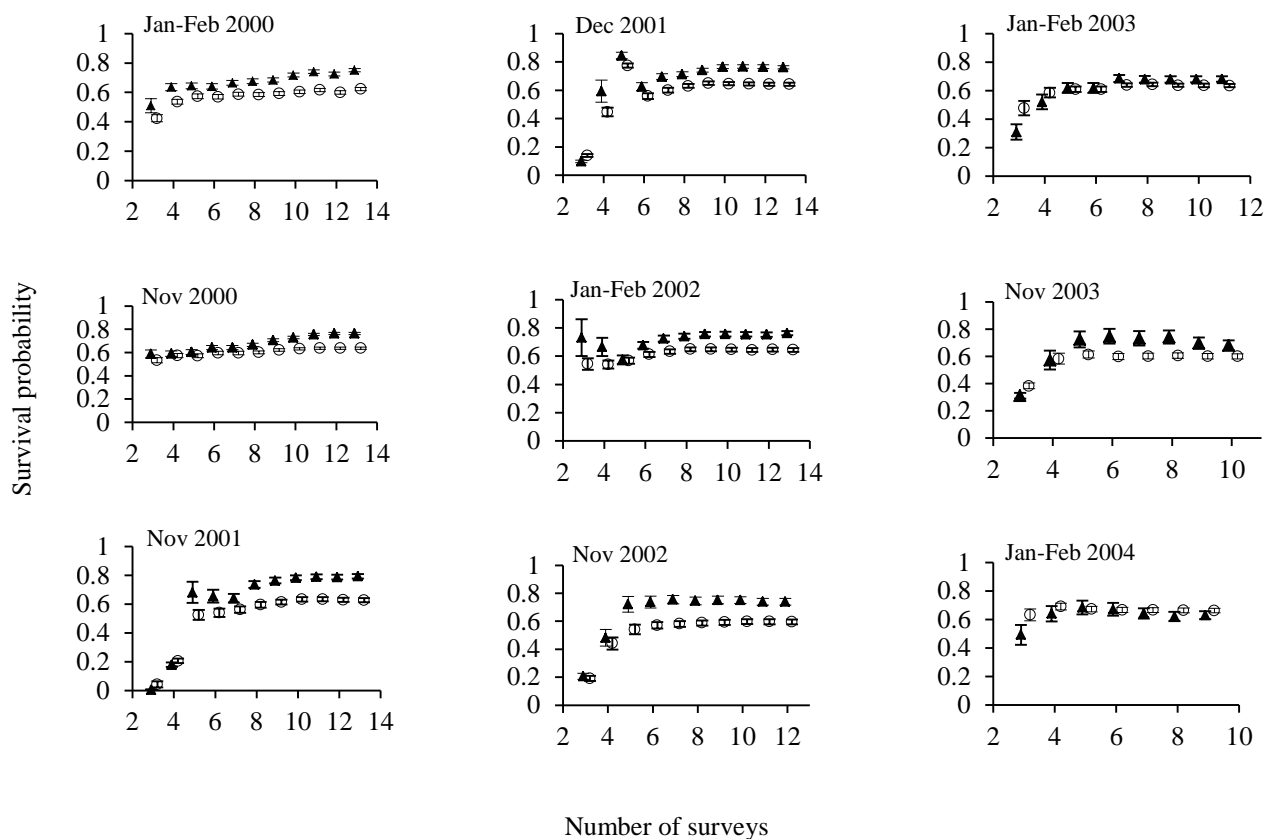


Figure 2 - 6 Annual survival probability estimates ($\phi \pm \text{SE}$) with different number of surveys and unequal intervals for every study based on start times (Design 1); Males (○), Females (▲). Only annual surveys were undertaken after Jan-Feb 2004.

2.3.3 Annual Survival probability_– effect of start date

Estimated annual survival probability of males was relatively stable at around 0.63 yr^{-1} irrespective of the time of the initial survey period whereas apparent survival probability of females varied depending on the period selected (Figures 2-4 and 2-5). There was a trend of decline in estimates of apparent survival probability of females from approximately 0.77 yr^{-1} at the start of the time series to 0.60 yr^{-1} for the later surveys. The pattern was similar for both males and females irrespective of whether there were equal or unequal periods between surveys.

In all but two model runs, at least 5 surveys were required before male and female apparent survival probability estimates stabilised.

2.3.4 Precision of survival probability estimates

The desired level of precision of survival probability estimates varies with the application. In these analyses we have arbitrarily chosen $\pm 5\%$ as the level of precision. At the 5% level, males required fewer surveys to provide both precise estimates of survival probability than females, with models unable to establish precise estimates for females in the latter period of the time series. Estimates from unequal surveys were more variable in the number of surveys required to provide precise estimates (Table 2-6).

Table 2 - 6 Number of surveys required to obtain precise estimate of survival probability that has a 95% probability of being within 5 % of the overall mean survival probability estimate. Models were classified as not acceptable (NA) where the precision of survival probability estimation was above this threshold.

Survey	Sex	(Independent) Tagging events												The range of the required surveys
		Jan-Feb 00	Nov 00	Nov 01	Dec 01	Jan-Feb 02	Nov 02	Jan-Feb 03	Nov 03	Jan-Feb 04	Jan 05	Jan-Feb 06	Jan 07	
Equal (Annual)	M	-	4	-	4	-	-	4	-	4	4	5	5	4-5
	F	-	4	-	4	-	-	5	-	9	8	NA	NA	4-NA
Unequal	M	4	3	7	5	5	6	5	5	4	4	5	5	3-7
	F	4	4	7	5	6	7	7	NA	NA	8	NA	NA	4-NA

2.4 Discussion

In this study, models were developed to understand the precision of survival probability estimates for the southern rock lobster *Jasus edwardsii* relating to the number of surveys and the interval between surveys. Although, the site used for collection of data was a scientific reserve closed to fishing, a major perturbation in the reserve occurred when a large number of lobsters were translocated into the reserve in 2005 (Green and Gardner, 2009), resulting in an increase in density in the reserve. This translocation appears to have had a greater impact on female lobsters where sex ratio changed in favour of males and estimated survival probability of females decreased. In contrast, translocation of lobsters into the site did not coincide with any change in the survival probability of males where it was consistent over time. The cause of the decline in apparent annual

survival probability of female lobsters is uncertain, as CJS models cannot separate emigration from the study site and mortality, although previous research has shown that this species has high site fidelity (Gardner *et al.*, 2003; Barrett *et al.*, 2009). The Crayfish Point Reserve used in this study has been protected for 40 years and the habitat and or food for lobsters may have reached its carrying capacity, thus resulting in emigration of females from the reserve after density was increased by translocation of additional lobsters into the site. Emigration has been found to increase capture heterogeneity and reduce capture probability resulting in an increasing bias and decreasing precision of survival probability estimates (Bailey *et al.*, 2010).

Females had greater survival probability than males but the precision of estimates were higher for males than for females. Apparent survival probability of males and females were estimated to be between 0.60-0.63 and 0.75-0.78 yr⁻¹, respectively. Sex specific differences in survival probability estimates was also reported by (Frusher *et al.*, 2009) in an analysis of a similar dataset using a lower number of annual survey periods. Sex differences have been reported in other studies of crustaceans although there is inconsistency in the patterns as higher survival probability estimates were obtained for female western king prawn (*Penaeus latisulcatus*) from fishing and natural mortality (Xiao and McShane, 2000), whereas Bell *et al.* (2003) found the opposite pattern with higher apparent survival probability for male edible crabs *Cancer pagurus*.

The most parsimonious model for estimating apparent survival probability was affected by whether the design involved equal or unequal survey intervals. For the design with unequal survey intervals, both the timing of tagging and the time interval between recapture surveys decreased the precision of survival probability estimates for both males and females.

Although Program MARK allows for different time intervals between surveys, the variation in this study was extreme (from 1-13 months) and useful for testing differences in interval. It resulted in a lack of precision in survival probability estimates for periods of unequal recapture intervals. When compared to equal surveys, the parsimonious model for estimating survival probability of lobsters

in unequal surveys was sex and time-dependent when the first tagging survey occurred in November and December. There also appeared to be a synergistic effect between the timing of the initial tagging event and the time interval between subsequent recapture events. In surveys where tagging events commenced in November or December, and where the time interval between the subsequent surveys was small (e.g. one or two months), survival probability of both males and females estimated by 3-5 surveys declined substantially in precision. Crustaceans demonstrate substantial variability in catchability due to mating, moulting and compensatory periods (Ziegler *et al.*, 2004). The increased bias and decreased precision of survival probability estimates obtained in this study when models used unequal time periods between surveys, suggests that behaviour affecting catchability was not constant during these periods. By comparison, Nowicki *et al.* (2008) found that population size of freshwater crayfish was not biased where time interval between surveys was constant and also short enough (two weeks) to avoid trap-shy behaviour. Furthermore, unbiased annual survival probability was obtained using an annual (constant) survey interval (in this study). Therefore, it seems that a large difference in survey intervals may affect the estimate of survival probability in lobsters.

The number of surveys required to provide adequate precision in the estimation of survival probability in lobsters was dependent on sex and time between recapture intervals. Prior to the impact of translocation a minimum of five surveys were required to obtain stable estimates of survival probability with the most parsimonious model being sex-dependent irrespective of whether equal (annual) or unequal survey time intervals were involved.

In general, females required more surveys (five annual surveys or seven unequal interval surveys) than males (four annual surveys or five unequal interval surveys). Similarly, Nowicki *et al.* (2008) found that estimation of population size for a freshwater crayfish was not biased with five surveys when short (fortnightly) and equal intervals between surveys was applied. Thus, for an annual model, the length of a tagging project should be longer than the standard three year project duration that is common in science. Managers and funding agencies need to be aware of the benefits of

longer-term projects and the loss in the precision of important assessment parameters that can occur if project duration is reduced.

Chapter 3

**Sample size and fishing effort influence estimates of survival
probability in a capture-mark-recapture study on southern rock
lobster (*Jasus edwardsii*)**

Chapter 3 Sample size and fishing effort influence estimates of survival probability in a capture-mark-recapture study on southern rock lobster (*Jasus edwardsii*)

Abstract

Five annual capture-mark-recapture surveys on *Jasus edwardsii* commencing in November 2000 at Crayfish Point Scientific Reserve (CPSR), Hobart, Tasmania, Australia, were used to evaluate the effect of sample size and fishing effort on the precision of estimated survival probability. Random subsampling was conducted from every annual survey with groups of 200, 250, 300, 400, 500, 750 and 1000 individuals; and 12 combinations of effort subsampled at three levels of the number of traps (15, 30 and 50) over four levels of the number of sampling-days (2, 4, 6 and 7). The most parsimonious Cormack-Jolly-Seber (CJS) model for estimating survival probability shifted from a constant model towards sex-dependent models with increasing sample size and effort. Whilst, a sample of 500 lobsters or setting 50 traps over four consecutive sampling-days provided enough support for obtaining precise survival probability estimations for males and females separately, 30 traps over four sampling days was adequate if an estimate of survival probability for combined sexes was sufficient for management of the fishery. The choice of effort in designing mark-recapture studies is dependent on the objectives of studies, the budget and the precision of information required by management for assessment of the fishery.

Key words: capture-mark-recapture (CMR), survival_probability, sample size, effort, Cormack-Jolly-Seber (CJS), lobster

3.1 Introduction

Capture-mark-recapture (CMR) modelling is a common method of estimating demographic parameters in field studies where live recapture or resighting is possible (Pollock *et al.*, 1990; Lebreton *et al.*, 1992; Nichols, 1992). Although CMR methods are widely applied, there is often limited attention given to sampling design. Planning the appropriate intensity and duration of sampling to obtain a specified level of precision is a challenging issue and can bias or affect estimates if there is insufficient data. For example, McKelvey and Pearson (2001) reported that 98% of samples collected in small mammal population studies were too small for estimating population density.

Precision and accuracy of estimated survival probabilities are known to increase with sample size and duration of sampling (Robson and Regier, 1964; Pollock *et al.*, 1990). However, sampling intensity and effectiveness may be constrained by environmental and biological factors as well as the pragmatic constraints of research funding and time. Tagging studies often rely on data collected by trapping and in these studies the sampling efficiency is affected by the catchability of animals, which in turn is affected by environmental and physiological factors. For example, catchability of animals by baited-traps such as lobster (Ziegler *et al.*, 2003), crab (Williams and Hill, 1982) and crayfish (Somers and Stechey, 1986) consistently decline in winter. In addition, several biological characteristics of crustaceans including moulting and mating can occur at different times of the year and have been shown to influence the precision and accuracy of survival rate estimates (Kelly *et al.*, 1999; Ziegler *et al.*, 2004).

Perhaps most importantly, research funds and time often place the greatest limitation on CMR studies commonly manifesting in restrictions on sample size and duration of sampling. Other resourcing constraints include the cost of tagging equipment such as transmitters in

satellite telemetry (Lindberg and Walker, 2007) or the cost of boat charter, crews and scientists.

It is commonly assumed that CMR methods are robust to restrictions in sampling effort as long as effort remains consistent within the study. However, a number of studies have shown that this assumption may be invalid (Robson and Regier, 1964). For example, sample size and sample duration have been shown to influence estimates of growth rate (Fiske *et al.*, 2008), meta-population estimates of interchange among populations (Burke *et al.*, 1995) as well as the precision of estimated parameters (Lindberg and Walker, 2007) in both matrix and CMR modelling.

This study examined the effect of sample size and fishing effort on the precision of parameter estimates in five annual mark-recapture surveys of the southern rock lobster *Jasus edwardsii*. Previous work (Chapter 2) showed that precision of survival probability estimates increased with the number of recapture surveys although beyond five surveys the improvements were not significant.

This chapter used five annual sampling events to:

1. evaluate the effects of sample size and fishing effort on the precision of survival probability estimates; and to
2. determine the minimum sample size and effort needed to obtain precise survival probability estimates.

3.2 Methods

3.2.1 Study area

In CMR studies, precision of survival probability estimates can be dependent on the size of the population (Robson and Regier, 1964) and the size of the sampling area (Pennington and Vølstad, 1991; 1994). The effect of varied population size and sampling area was limited in this study by conducting sampling within the Crayfish Point Scientific Reserve (CPSR), Tarooma, Tasmania, Australia (42° 57' 08"S 142° 21' 20"E). This reserve contains rocky reef habitat and is surrounded by sand, which limits rock lobster movement (Barrett *et al.*, 2009). The reserve contains a temperate rocky reef of 1.24 km² with maximum depth of 15 m. Fishing in the reserve has been banned since its declaration in November 1971 and the reserve now holds a dense population of southern rock lobsters (Green and Gardner, 2009).

3.2.2 Field method

Mark-recapture data were collected for five separate research projects on *Jasus edwardsii* from 2000 to 2005. As the surveys were originally undertaken to address different objectives, there was some variation between surveys in effort including the number of traps used, the number of sampling days and number of lobsters tagged. These factors, in addition to likely changes in catchability and abundance, resulted in differences in the number of lobsters tagged and/or recaptured between surveys (Table 3-1). However all surveys used consistent sampling protocols and similar equipment. On the first sampling day, traps were baited and set in the afternoon. On subsequent days, traps were checked in the morning, rebaited and reset. We refer to days when traps are hauled and the contents of the trap sampled as "sampling days". All untagged lobsters were tagged by a uniquely coded T-bar tag (Hallprint

T-bar anchor tag; TBA1, Hallprint Pty Ltd, 27 Jacobson Crescent, Holden Hill, South Australia 5088, Australia) on the ventral side of the first or second abdominal segment. The tag number, length of carapace and sex of each lobster were recorded.

3.2.3 Varying sample size

Previous research found that five surveys was the most economical number of surveys for estimating survival probability at a 5% level of precision (Chapter 2, Kordjazi *et al.*, 2015). The surveys used here were restricted to those that provided an opportunity to subsample up to 1000 tagged lobsters (Table 3-1). Random subsampling of both (newly) tagged and recaptured lobsters was undertaken with the statistical package R (R core team 2013) from every annual survey, with the number of individuals subsampled without replacement set at 200, 250, 300, 400, 500, 750 or 1000. Approximately equal numbers of males and females were selected as the average sex ratio across all surveys was 1.16 (M/F). Recapture histories were established based on the first survey undertaken in November 2000 and four successive recapture events. This process was repeated ten times by selecting ten independent subsamples for each level of sample size.

3.2.4 Varying sampling effort

The effect of sampling effort on estimates of survival probability was examined by subsampling a combination of the number of traps deployed each sampling day and the number of days sampled during each of five annual surveys (Table 3-1).

To quantify the effect of the number of traps on the precision of lobster survival probabilities, a total of 50, 30 and 15 traps were randomly subsampled from each day of sampling in each

survey. A random design was used to ensure that tagged lobsters that would be distributed across the entire reef region had equal probability of being captured in each survey.

Table 3 - 1 Summary of sampling data

	Annual surveys				
	Nov 2000	Dec 01	Jan-Feb 03	Jan-Feb 04	Jan 05
Sampling-days	10	7	11	7	10
Total traps deployed	891	602	761	403	708
Traps/day	89-90	86	69-78	49-63*	68-73
Lobsters captured	2362	2085	1714	1022	2032

*The lowest number of traps was 49 which occurred on two days in January-February 2004 survey.

To determine the effect of the number of days of sampling, 7, 6, 4 and 2 sampling days were randomly selected out of the total number of days sampled each survey. As such, 3*4 (=12) combinations of both the number of traps and days of sampling were applied in each survey, with each of these combinations replicated 10 times. For each subsample, 5-year capture histories were constructed based on lobsters that were tagged in the initial year's survey (November 2000), and seen or not seen in the four subsequent recapture surveys in the program R (R core team 2013).

Model selection and data analyses were based on Cormack-Jolly-Seber (CJS) models (Cormack, 1964; Jolly, 1965; Seber, 1965). The best model and survival probability of males and females was determined using the capture-recapture analysis in program MARK (White and Burnham, 1999), accessed through R via the RMark package (Laake and Rexstad, 2012).

Survival probability (ϕ) and recapture probability (p) were estimated from each survey data set. The likelihoods of survival probability and recapture probability were estimated incorporating factors of time (t) and sex (g). Sixteen parameter combinations of survival and capture probability were estimated with constant survival probability ($\phi(.)$), time specific survival probability ($\phi(t)$), sex specific survival probability ($\phi(g)$) or time and sex specific survival probability ($\phi(g*t)$) with constant capture probability ($p(.)$), time specific capture

probability ($p(t)$), sex specific recapture probability ($p(g)$) or time and sex specific recapture probability ($p(g*t)$). The fully parameterised (saturated) model was $\phi(g*t)p(g*t)$.

Delta AICc (AIC corrected for low sample sizes) , the difference in AICc of each candidate model relative to the model found to have the lowest AICc, was used to determine the most parsimonious model for each analysis from sixteen candidate set of models (Burnham and Anderson, 1998). Whilst, the model with the lowest delta AICc value was considered the most parsimonious, all models within 2 units of the lowest delta AICc value (lowest delta AICc value +2) also had substantial support and needed to be considered (Boulanger *et al.*, 2006).

In this study, the process of data analysis was repeated 10 times by 10 independent subsamples without replacement for each level of sample size and effort. To find the best model based on the delta AICc scores among these 10 analyses, each model was coded relative to the Delta AICc score. The codes of 1, 2, 3, 4 and 5 were applied to the models with the value of $0 \leq \Delta AICc < 2$, $2 \leq \Delta AICc < 4$, $4 \leq \Delta AICc < 7$, $7 \leq \Delta AICc < 10$, and $\Delta AICc \geq 10$, respectively. Whilst the model with the lowest average coded delta AICc ($\Delta AICc$) value among these 10 analyses was evaluated as a parsimonious model, all models that were within 0.2 units of the lowest average coded delta AICc ($\Delta AICc$) value were considered to be likely models.

The best model among the 10 repetitions of each level of subsample was then used to examine the effect of sex and time on survival probability of lobster with varying sample size. Relative standard error (RSE) was used to determine the precision of the estimated survival probability with low relative standard error, indicating high precision of estimates (Lindberg and Walker, 2007). Relative standard error was calculated as $RSE = \frac{\overline{SE}}{\bar{X}}$

where \overline{SE} and \bar{X} are average standard error and mean of the parameter estimates, respectively.

3.3 Results

3.3.1 Sample size

The most parsimonious model for estimating survival probability (ϕ) and recapture probability (p) changed from constant (sex and/or time-independent) and/or sex-dependent to only sex-dependent with increase in the number of tagged lobsters in the sample (Table 3-2). The number of equally supported models (i.e. within 0.2 units of the most parsimonious model) also decreased from 3 when 250 lobsters were tagged to one when 750 or more lobsters were tagged. Thus sex differences were significant when sufficient lobsters were tagged.

Table 3 - 2 Comparison of model performance with increasing sample size. (*) indicates models that were considered equally parsimonious (i.e. within 0.2 units of the most parsimonious model)

Model	Coded	Model	Coded	Model	Coded	Model	Coded
Sample size	$\Delta AICc$	Sample size	$\Delta AICc$	Sample size	$\Delta AICc$	Sample size	$\Delta AICc$
250	250	500	500	750	750	1000	1000
$\phi(.) p(g)$	1.3*	$\phi(g) p(.)$	1.5*	$\phi(g) p(g)$	1.6*	$\phi(g) p(g)$	2.1*
$\phi(.) p(.)$	1.4*	$\phi(.) p(g)$	1.6*	$\phi(.) p(t)$	2.1	$\phi(g) p(g^*t)$	3
$\phi(g) p(.)$	1.4*	$\phi(.) p(.)$	1.8	$\phi(g) p(t)$	2.1	$\phi(t) p(.)$	3.2
$\phi(g) p(g)$	2	$\phi(g) p(g)$	1.9	$\phi(g) p(.)$	2.2	$\phi(t) p(g)$	3.3
$\phi(.) p(t)$	2.3	$\phi(.) p(t)$	2.5	$\phi(.) p(.)$	2.4	$\phi(g^*t) p(g)$	3.3
$\phi(t) p(.)$	3	$\phi(g) p(t)$	2.6	$\phi(t) p(.)$	2.4	$\phi(.) p(g)$	3.4
$\phi(g) p(t)$	2.5	$\phi(t) p(.)$	2.7	$\phi(.) p(g)$	2.4	$\phi(.) p(.)$	3.5
$\phi(t) p(g)$	3.1	$\phi(t) p(g)$	2.8	$\phi(t) p(g)$	2.7	$\phi(g) p(.)$	3.5
$\phi(t) p(t)$	3.7	$\phi(t) p(t)$	3.1	$\phi(t) p(t)$	2.8	$\phi(.) p(t)$	3.5
$\phi(.) p(g^*t)$	3.4	$\phi(g) p(g^*t)$	3.1	$\phi(g) p(g^*t)$	3	$\phi(g) p(t)$	3.5
$\phi(g) p(g^*t)$	4.1	$\phi(g^*t) p(.)$	3.1	$\phi(g^*t) p(g)$	3.1	$\phi(g^*t) p(.)$	3.6
$\phi(t) p(g^*t)$	4.3	$\phi(.) p(g^*t)$	3.3	$\phi(.) p(g^*t)$	3.3	$\phi(.) p(g^*t)$	3.7
$\phi(g^*t) p(.)$	4.6	$\phi(g^*t) p(g)$	3.6	$\phi(g^*t) p(.)$	3.3	$\phi(t) p(t)$	4
$\phi(g^*t) p(g)$	4.8	$\phi(g^*t) p(t)$	3.9	$\phi(t) p(g^*t)$	3.8	$\phi(g^*t) p(t)$	4
$\phi(g^*t) p(t)$	4.6	$\phi(t) p(g^*t)$	4.2	$\phi(g^*t) p(t)$	3.9	$\phi(t) p(g^*t)$	4.1
$\phi(g^*t) p(g^*t)$	5	$\phi(g^*t) p(g^*t)$	4.6	$\phi(g^*t) p(g^*t)$	4.3	$\phi(g^*t) p(g^*t)$	4.2

3.3.1.1 Survival probability and precision of survival probability

To determine the survival probability of males and females separated, the most parsimonious model which occurred across all sample sizes ($\phi(g)p(g)$) was selected. Both estimated average survival (Figure 3-1a) and average recapture probability (Figure 3-1c) increased with sample size. Females had a higher average survival probability and lower average recapture probability than males. The precision of survival probability estimates and recapture probability improved with sample size for both sexes as average standard errors of estimates decreased (Figure 3-1b). Although the degree of precision would be determined by managers or assessors of the fishery, in this study a relative standard error (RSE) of 0.05 was chosen to compare the relative precision across different models.

A sample of at least 500 lobsters was required to obtain a survival probability estimate with a $RSE < 0.05$ for both males and females (Figure 3-1b). Reliable survival probability estimates for females alone could, however, be obtained from a sample of 400 female lobsters. This resulted in a 5-10% probability of resighting each lobster.

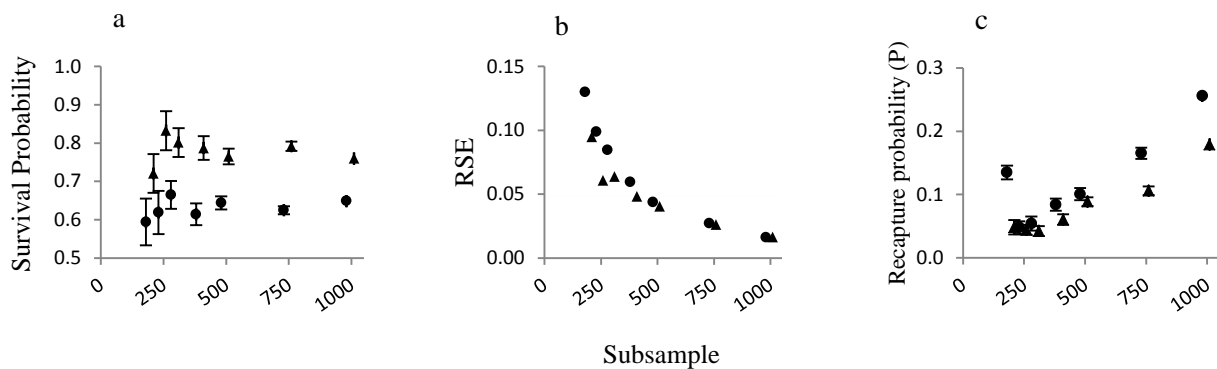


Figure 3 - 1 Average estimated survival probability (a), relative standard error (RSE) of survival probability (b) and average recapture (resighting) probability (c) of males (●) and females (▲) at different levels of subsample size (number of individuals). The horizontal line in (b) indicates a 5% level of precision.

3.3.2 Sampling effort

As effort, both in number of sampling days and number of traps increased (i.e., from bottom right quadrat in Table 3-3 to top left quadrat) the survival and recapture probability estimates for the most parsimonious models tended to shift from sex and time independent to sex dependent and time and sex dependent, respectively.

3.3.2.1 Survival probability and precision of survival probability

The most common sex-time independent and sex dependent models found across the combinations of the number of traps and sampling days was $\phi(.)p(g)$ and $\phi(g)p(g)$ respectively (Table 3-3). To determine the appropriate sampling design a precision level (RSE) of 5% was used. If a combined sex model is considered appropriate then 15 traps at 6 days or 30 traps at 4 days would be sufficient. Although 50 traps at 4 days also produced precise estimates, this has not been considered as the required precision, <5% RSE, was not obtained with fewer trap (Figure 3-2a). If the survival probability between the different sexes was considered sufficient to warrant different survival estimates then at least 30 traps for 6 days is required or 50 traps at 4 days (Figure 3-2b). If less precision is required (e.g. 10%), 15 traps for 4 days or 30 traps for 2 days would be the minimum required for estimating annual survival probability for combined sexes and 30 traps for 4 days for separate sexes.

Table 3 - 3 Dependency of survival probability (ϕ) and recapture probability (p) on sex (g), time (t) and/or sex and time (g^*t) in the five top models for combinations of sampling days and number of traps. General model is $\phi(x)p(y)$ where “ x ” and “ y ” have been shown in the following table as $(x)/(y)$. (.) denotes where ϕ and/or p in the model were independent of sex and/or time. (*) indicates the most parsimonious models in the range of $Z+0.2$, where Z is the average of the coded $\Delta AICc$ value among ten repetitions. The values of $0 \leq \Delta AICc < 2$, $2 \leq \Delta AICc < 4$, $4 \leq \Delta AICc < 7$, $7 \leq \Delta AICc < 10$, and $\Delta AICc \geq 10$ were coded from 1 to 5, respectively.

Number of traps	Parsimonious models	Number of sampling days							
		7		6		4		2	
50	First	(g)/(g*t)	1.0*	(.)/(g*t)	1.3*	(.)/(g)	1.8*	(.)/(.)	1.3*
	Second	(.)/(g*t)	1.1*	(g)/(g*t)	1.6	(.)/(t)	2.0*	(.)/(t)	1.5*
	Third	(t)/(g*t)	2.5	(t)/(g)	1.8	(g)/(g)	2.1	(g)/(.)	1.6
	Forth	(g*t)/(g)	3.0	(t)/(g*t)	2.0	(.)/(.)	2.1	(.)/(t)	1.7
	Fifth	(g*t)/(g*t)	3.9	(g*t)/(g)	2.8	(t)/(.)	2.1	(g)/(t)	1.7
30	First	(.)/(g*t)	1.7*	(.)/(g)	2.0*	(.)/(t)	1.7*	(.)/(.)	1.0*
	Second	(g)/(g*t)	2.0	(.)/(t)	2.2*	(.)/(.)	1.7*	(.)/(g)	1.4
	Third	(g)/(g)	2.6	(g)/(t)	2.3	(.)/(g)	1.8*	(g)/(.)	1.7
	Forth	(.)/(g)	2.8	(g)/(g)	2.5	(t)/(.)	1.8*	(g)/(g)	1.9
	Fifth	(.)/(t)	2.9	(.)/(.)	2.5	(g)/(.)	2.0	(.)/(t)	2.1
15	First	(g)/(g)	1.5*	(.)/(g)	1.2*	(.)/(.)	1.0*	(.)/(.)	1.0*
	Second	(.)/(g)	1.6*	(g)/(g)	1.4*	(.)/(g)	1.0*	(.)/(g)	1.2*
	Third	(.)/(.)	1.8	(.)/(.)	1.6	(g)/(.)	1.3	(g)/(.)	1.3
	Forth	(g)/(.)	2.1	(g)/(.)	1.7	(g)/(g)	1.8	(g)/(g)	2.2
	Fifth	(.)/(t)	2.7	(t)/(g)	2.8	(.)/(g*t)	2.3	(.)/(t)	3.4

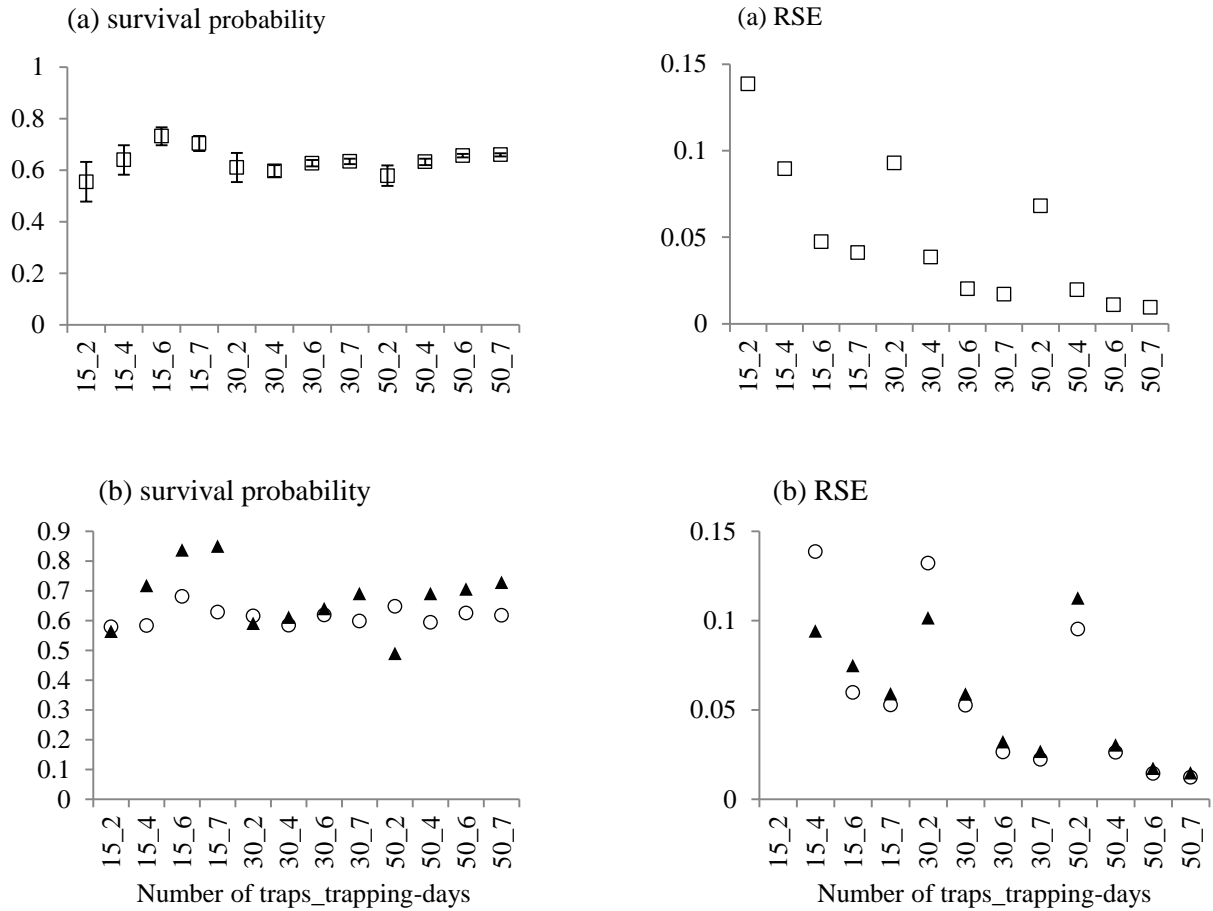


Figure 3 - 2 Survival probability (ϕ) and relative standard error (RSE) of lobsters - combined sexes (a) and different sexes (b) (males (\circ) and females (\blacktriangle)), estimated by sex-time independent model ($\phi(\cdot)p(g)$) and sex dependent model $\phi(g)p(g)$ respectively, associated with each combination of fishing effort.

3.4 Discussion

The trade-off between the desired precision of survival probability and cost-effectiveness of sampling was evaluated with southern rock lobster data by subsampling from existing data with variation in the number of tagged lobster (sample size) and fishing effort (the number of traps set and number of days sampled). As expected, both the sample size and sampling effort affected the precision and accuracy of survival probability estimates. Decreasing effort, both in terms of numbers of individuals sampled and the allocation of effort (days sampled and traps set) led to decreased precision in estimates of both survival probabilities (ϕ) and

recapture probabilities (p). Importantly, under-sampling led to biased estimates of survival probability and also minimised the ability to determine differences between sexes. Finally, the minimal increase in precision at higher levels of sampling effort, suggests that an optimal ratio of sampling to precision can be achieved which can be used to find the balance in sampling effort that delivers quality data without excessive effort and cost.

Although sex specific ($\phi(g)$) was an equally likely model for each of the sample sizes, it was only when 750 or more lobsters were tagged that it had substantially greater support than the combined sexes ($\phi(.)$) models. However, at the pre-determined RSE of 0.05, 500 tagged lobsters was sufficient to provide precise estimates of survival probability indicating the importance of both model selection (at 500 tagged lobsters, the sex specific model was not the only parsimonious model) and interrogating the model outputs (i.e. plotting means, standard deviations and RSE).

While catch rates could be used to determine the effort needed to achieve a targeted sample size of recaptured lobsters, catch rates can vary considerably and we selected the number of traps set and consecutive sampling days as alternative metrics for determining sampling design.

The number of captured lobsters increased with more sampling effort, that is, with more traps, more trapping days, or both. Not surprisingly, it was possible to obtain a satisfactory estimate of survival probability by trading off the amount of traps versus days of sampling. Understanding the trade-off between these two aspects of sampling can assist in optimizing sampling strategies (Comeau *et al.*, 2009).

Daily catchability of rock lobsters is known to vary significantly due to factors such as swell and moon phase (Srisurichan *et al.*, 2005) which have not been included as explanatory variables in this study. Because of this we speculated that there was potential for different

outcomes from short sampling periods with lots of traps each day versus longer sampling periods with fewer traps each day. However, our results suggest that sampling design is free to be driven by the optimal, pragmatic balance between the number of traps and sampling-days that result in the most cost effective and operationally feasible sampling program.

Increasing the number of traps from 15 to 50 enabled more precise survival probability estimates. Whilst, four days of sampling for 50 traps subsampled was required to obtain precise survival probability estimates for males and females separately, 30 traps set for four sampling-days was sufficient for obtaining precise survival probability estimates for combined sexes. Although, fewer traps in a fixed sampling area can impact the effectiveness of bait plumes and thus reduce the probability of capture of animals (Acosta and Perry, 2000), sampling plans based on setting fewer traps over more days may, however, be advantageous in some situations. Whilst more accurate and precise parameter estimates are often pursued in mark recapture studies with the use of more traps (Zimmerman and Palo, 2011), data collected with 15 traps provided precise parameter estimates for combined sexes, where 6 sampling-days were required to obtain precise survival probability estimates in the level of 0.05. This can also be impacted by the logistics of the sampling methodology being undertaken as the costs of each additional trap needs to be compared to the cost of labour and vessel charter, including scientist's time with adding additional days to the survey. Of particular importance in Australia is the need to minimise costs by not sampling over a weekend (Saturday and Sunday). Increasing the traps used across 4 days of sampling is preferred.

A priori determination of the desired level of precision and accuracy of estimated parameters, population size and sampling area can assist in planning sampling design in a CMR study (Robson and Regier, 1964; Lindberg *et al.*, 2002). This process is necessary for many research projects because of the constraint of research funding that limits the extent of data

collection. While this study may provide a guide, we recommend that prior to planning large scale tagging programs that pilot programs are undertaken and analyses such as those undertaken in this chapter be used to determine a cost effective sampling design.

Chapter 4

Estimating tag induced mortality of southern rock lobster in the wild

How does variation in the size and the number of tagged lobsters affect the tag induced mortality estimates?

Chapter 4 Estimating tag induced mortality of southern rock lobster in the wild - How does variation in the size and the number of tagged lobsters affect the tag induced mortality estimations?

Abstract

A long-term capture-mark-recapture study conducted since 2000 at the Crayfish Point Scientific Reserve (CPSR), Hobart, Tasmania, Australia, was used to estimate *in situ* tag induced mortality (TIM) proportion. A three-survey model was used that assumed the impact of tagging was most important immediately after the animal was tagged and thus occurs before the next survey. The average of TIM proportion estimates for lobsters (combined sexes) was 35%. Estimates of TIM proportion for lobsters were varied and dependent on the number of recaptured lobsters. To provide reliable estimates of TIM proportion in this study at least 15 lobsters had to be captured in two subsequent surveys after the initial tagging survey. As tagging normally only marks a small proportion of a population, tag recovery rates are relatively low and the need to recover sufficient lobsters on two separate surveys is a major constraint of the method. High TIM proportion estimates need to be factored into the analysis of mark recapture studies that are used to estimate population parameters otherwise overly optimistic projections would result which would potentially lead to over-estimation of the resource and inappropriate catch allocations.

Key words: capture-mark-recapture (CMR), tag induced mortality (TIM), proportion, lobster.

4.1 Introduction

Capture-mark-recapture (CMR) methods are commonly used for estimating demographic characteristics of crustaceans in the wild, including survival probability, population size and migration, provided the tags can be retained over a series of moults (Pollock *et al.*, 1990). In

any tagging study, the effect of tag loss and tag induced mortality (TIM) should be investigated as both result in fewer tagged animals being available for recapture and will therefore bias estimation of population parameters.

TIM is defined as the death of a tagged animal due to factors either directly or indirectly related to the tagging process including stress caused by recapture, handling by the tagger, tag wounds, infection of the tagging area, and predation as a result of capture and tagging.

Additionally, TIM may be influenced by traits or biological aspects of tagged animals such as moult stage or size. In lobsters, various studies have demonstrated that the most appropriate period of the moult cycle for tagging to minimize tag loss and tag induced mortality is during the inter-moult stage (Montgomery *et al.*, 1995; Moriyasu *et al.*, 1995; Frusher *et al.*, 2009).

Determination of TIM proportion, tag loss rate and tag related injuries has traditionally been estimated in aquaria facilities or with caging studies conducted in the field (Montgomery and Brett, 1996; Davis *et al.*, 2004; Dubula *et al.*, 2005; Mattei *et al.*, 2011; Gonzalez-Vicente *et al.*, 2012). Aquaria based studies of TIM can provide biased values as the artificial environment can place additional stress on captured animals and these environments also fail to replicate interactions with predators, prey and habitat. In addition, aquaria studies usually comprise small sample sizes (Mattei *et al.*, 2011) and/or short time periods (Montgomery and Brett, 1996; Claverie and Smith, 2007).

In-situ measurements of TIM, where tagged animals are rapidly released into the wild to mix and behave with the population from which they were captured, can overcome several of the biases associated with aquaria or caged studies. Frusher *et al.* (2009) developed a model based on three survey events that compared the survival of lobsters tagged in a second survey compared to lobsters previously tagged in the first survey. The method assumes that TIM has already impacted the previously tagged lobsters and thus provides an estimate of the initial

impact of tagging. The timing of surveys dictates the period over which the impact of tagging is considered. However, the method does require animals to be recaptured on two subsequent occasions which can be an impediment when tag return rates are low. Recognising CMR studies are expensive and there is a trade-off between effort and the precision of estimates, it is desirable to determine what sample size and sampling effort is required to gain reasonable estimates of TIM proportion in the wild. In this study, capture-mark-recapture data collected over 13 years (2000-2012) was used to estimate TIM proportion in the wild for the southern rock lobster *Jasus edwardsii*.

The aim of this study was to:

- 1) Determine tag induced mortality proportion for *J. edwardsii*.
- 2) Evaluate the effect of the number and size range of initially tagged lobsters on estimates of tag induced mortality proportion.

4.2 Method

4.2.1 Study area

A long term annual tagging and recapture survey was conducted from 2000 to 2012 at the Crayfish Point Scientific Reserve (CPSR), near Hobart, Tasmania, Australia (42° 57' 08"S 142° 21' 20"E). This scientific reserve contains rocky reef habitat and is surrounded by sand, which limits rock lobster movement (Barrett *et al.*, 2009). The reserve contains a temperate rocky reef of 1.24 km² with maximum depth of 15 m. Fishing for lobsters in the reserve has been banned since November 1971 and the reserve now holds a dense population of southern rock lobsters (Green *et al.*, 2009).

4.2.2 Field methods

A tagging program of southern rock lobster in the CPSR has been conducted since 2000 to address a number of research projects. In this study we used the surveys undertaken approximately annually at the start of each year as this is the inter-moult period for both males and females and catchability is higher. Catchability declines as waters cool from February to August (Ziegler *et al.*, 2003). On the first day, traps were baited and set in the early afternoon. On subsequent days the traps were checked in the morning, re-baited and re-set. All untagged lobsters were tagged by a uniquely coded T-bar tag (Hallprint T-bar anchor tag; TBA1, Hallprint Pty Ltd, 27 Jacobsen Crescent, Holden Hill, South Australia 5088, Australia) on the ventral side of the first or second abdominal segment. The tag number, length of carapace and sex of all lobsters were recorded and lobsters were released back immediately to the wild after tagging.

4.2.3 Estimating tag induced mortality (TIM) proportion

A three-year tagging model (Figure 4-1) was used to estimate tag induced mortality (TIM) proportion. This three-year tagging model developed by Frusher *et al.* (2009) is described in Appendix 4-1. As tagging surveys were undertaken annually, the TIM estimates are the annual proportion of lobsters that died from tagging during a year. Untagged lobsters were tagged on the first and second surveys (N_1 and N_2 , respectively). During the second survey lobsters that were tagged in the first survey are also captured ($R_{[21]}$) and returned to the sea with newly tagged lobsters. It was assumed that TIM had already occurred for this group of tagged lobsters between the first and second survey. In the third survey, lobsters were caught from both the first ($R_{[31]}$) and second surveys ($R_{[32]}$). It was assumed that over the second year

of study, TIM (I_2) only impacts lobsters that were tagged in the second survey. Whilst, lobsters tagged in the first survey experienced the impact of tagging over the first year of study (I_1), it was assumed that tagged lobsters recaptured in the second survey had already survived the impact of tagging and thus no impact of tagging between surveys 2 and 3 occurs. For estimating TIM proportion only a component of the $R_{[31]}$ was required. These were the lobsters that were caught in both the second and third surveys. It was assumed that no TIM occurs for these lobsters between the second and third surveys. These are denoted as $R_{3[21]}$.

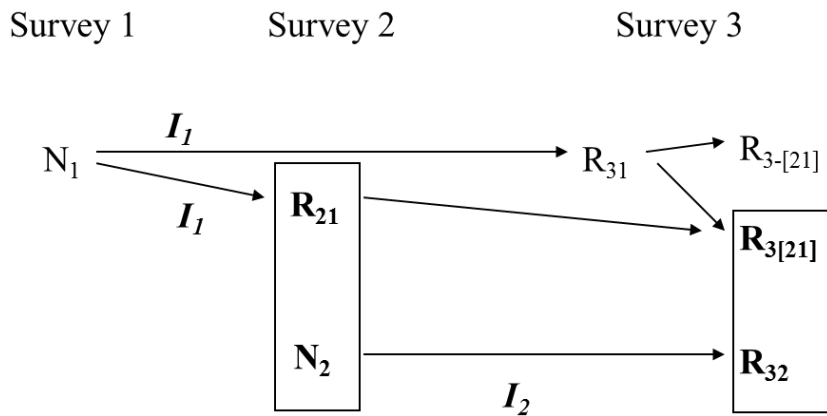


Figure 4 - 1 Three-year tagging survey method for estimating TIM proportion. The bold and boxed texts indicate the data used (from Frusher *et al.*, 2009). I_1 and I_2 denote the impact of tagging on mortality of lobsters tagged in the first and second tagging events, respectively.

Tag induced mortality (TIM) proportion for the period between survey 2 and 3 (I_2) is calculated as:

$$I_2 = 1 - \left(\frac{R_{[32]} * R_{[21]}}{R_{3[21]} * N_2} \right) \quad (\text{Equation 1})$$

where

I_2 : Tag induced mortality (TIM) proportion for the period between survey 2 and 3.

$R_{[21]}$: Number of tagged lobsters captured in the second survey that were tagged in the first survey

$R_{[32]}$: Number of tagged lobsters captured in the third survey that were tagged in the second survey

$R_{3[21]}$: Number of tagged lobsters captured in the third survey that were tagged in the first survey and also captured in the second survey.

N_2 : Number of lobsters tagged in the second survey.

This model assumes that no tagged lobsters are lost to migration and that tagged lobsters from each tagging event mix evenly with each other and that each tagged lobster, irrespective of when it was tagged, has an equal probability of being captured.

4.2.4 Criteria for determining TIM

To ensure that sufficient recaptures are obtained to estimate TIM, the largest sample size should be used. This would be the entire tagged population in each survey. However, to minimise any size dependent tag induced mortality impacts, estimates of TIM were calculated for increased size ranges. The most frequent 10 mm CL size bin of male and female lobsters capture in the CPSR from 2000 to 2012 was 120-130 mm CL and 100-110 mm CL for males and females, respectively (Figure 4-2). This size bin was used as the smallest bin and increasing samples were obtained by expanding the size bin width incrementally by 5mm CL either side of these bins.

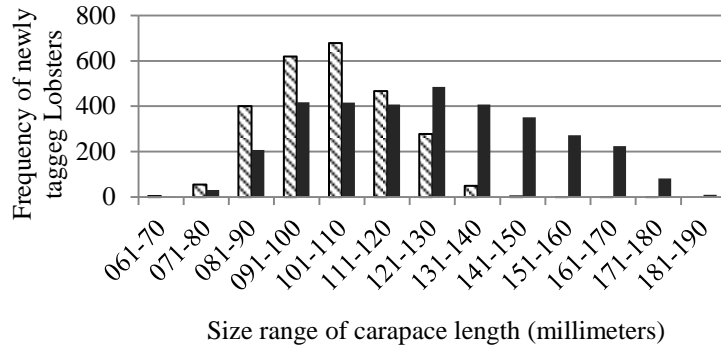


Figure 4 - 2 Size frequency of male (black column) and female (dashed column) lobsters tagged from 2000 to 2012.

For example, male TIM was estimated for size bins 120-130; 115-135; 110-140 etc. TIM was estimated for lobsters in each size group for male and female lobsters, separately.

4.2.4.1 Determination of size bin

A major restriction in the use of the *in-situ* method is the low sample size of recaptured lobsters. From Figure 4-1 the lowest sample size would be expected with the $R_{3[21]}$ group as this requires lobsters to be recaptured on two separate occasions.

To select the size bin with the highest numbers of tagged and recaptured lobsters ($R_{3[21]}$ and $R_{[32]}$), the relationships between size bin width and $R_{3[21]}$ and $R_{[32]}$ were separately evaluated.

4.2.4.2 Determination of the minimum number of recaptured lobsters

Delta TIM (ΔTIM) defined as the change in the estimated TIM between successive (overlapping) size bin widths in every survey was used (i) to determine the accuracy of the TIM estimates, and (ii) to find the minimum number of recaptured lobsters in the third survey ($R_{3[21]}$ and $R_{[32]}$) which resulted in accurate estimates. Accuracy of TIM proportion estimate was considered high, when estimates were within $\pm 10\%$.

$$\Delta TIM = TIM_{(n)} - TIM_{(n-1)} \quad (\text{Equation 2})$$

4.2.4.3 Determination of sample size and effort

To determine the amount of effort required to obtain reliable estimates of TIM proportion in the CPSR, the number of lobsters that needed to be tagged was initially estimated using the following equation:

$$\widehat{N}_i = R_{3[21]} / (\beta_{[21]} * \beta_{3[21]}) \quad (\text{Equation 3})$$

where

\widehat{N}_i : Estimated number of lobsters to be tagged in the first survey

$R_{3[21]}$: Number of tagged lobsters captured in the third survey that were tagged in the first survey and also captured in the second survey.

$\beta_{[21]}$: Proportion of lobsters tagged in the first survey that were recaptured in the second survey ($R_{[21]}/N_i$).

$\beta_{3[21]}$: Proportion of lobsters recaptured in the third survey that had also been recaptured in the second survey divided by the number recaptured in the second survey that had been tagged in the first tagging survey ($R_{3[21]}/R_{[21]}$).

To determine the effort required to catch the estimated number of lobsters (N_i), the capture rate (per trap lift) was estimated for the first survey in each estimate of TIM proportion.

Capture rate per trap is estimated by the following formula:

$$C_{ri} = N_i / T_i \quad (\text{Equation 4})$$

Where

Cri : Capture and/or recapture rate in survey i

Ni : Number of lobsters tagged and/or recaptured in survey i

Ti : Number of traps applied in survey i

For the first survey, Ni is the total number of the tagged lobsters, whilst in the second and third surveys, Ni is the number of recaptured lobsters ($=R_{[21]}$) and the number of lobsters recaptured over two consecutive surveys ($=R_{3[21]}$), respectively.

The number of traps was estimated by:

$$\hat{T}_i = \hat{N}_i / Cri$$

Where \hat{N}_i is the estimated number of lobsters required to generate TIM proportion estimates (equation 3) and Cri is the observed capture rate for the same survey (equation 4).

4.3 Results

4.3.1 Determination of criteria for estimating TIM proportion

4.3.1.1 Determination of size bin

Generally, there was an almost linear increase in the number of recaptured lobsters versus size bin width until approximately 95-155 mm CL and 80 –130 mm CL for males and females respectively after which an increase in the size bin resulted in fewer or no additional returns (Figures 4-3 and 4-4).

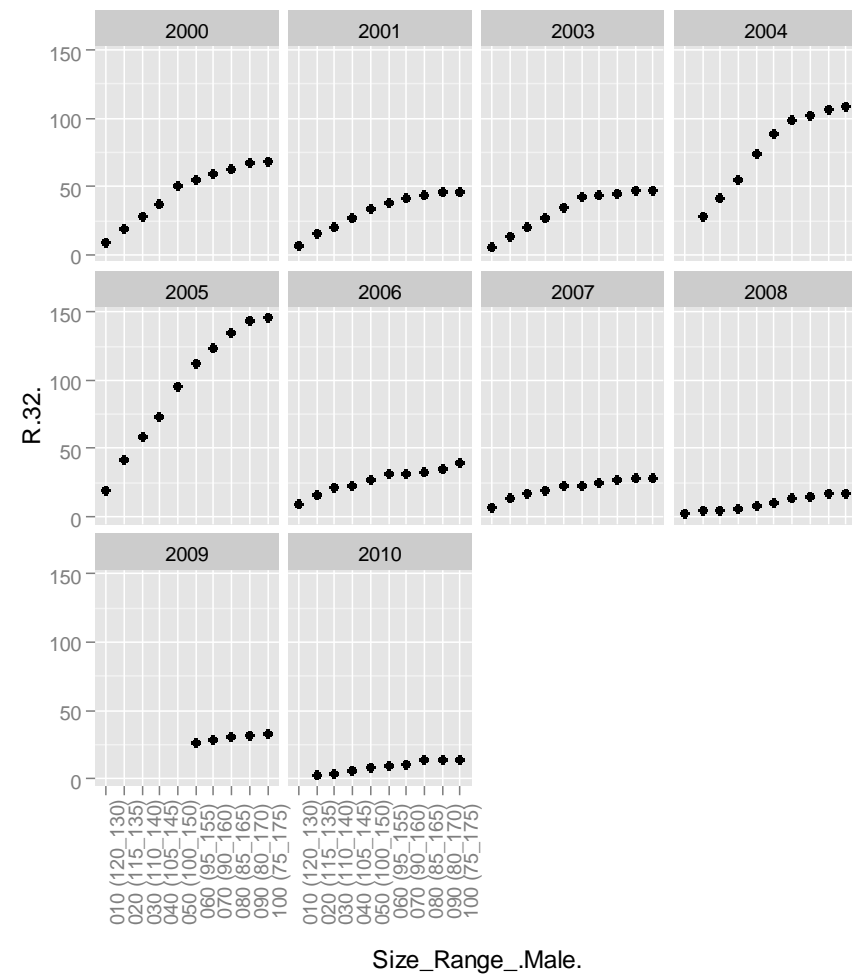
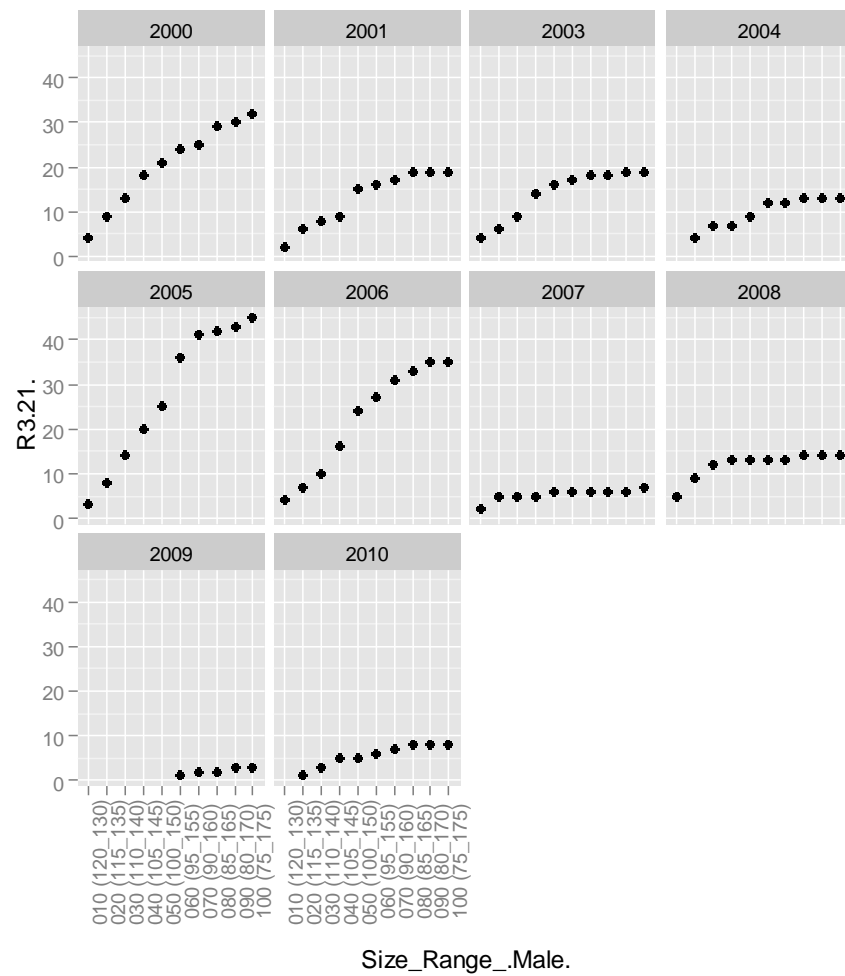


Figure 4 - 3 The number of recaptured lobsters at $R_{3[21]}$ and $R_{3[2]}$ for different size bin widths.

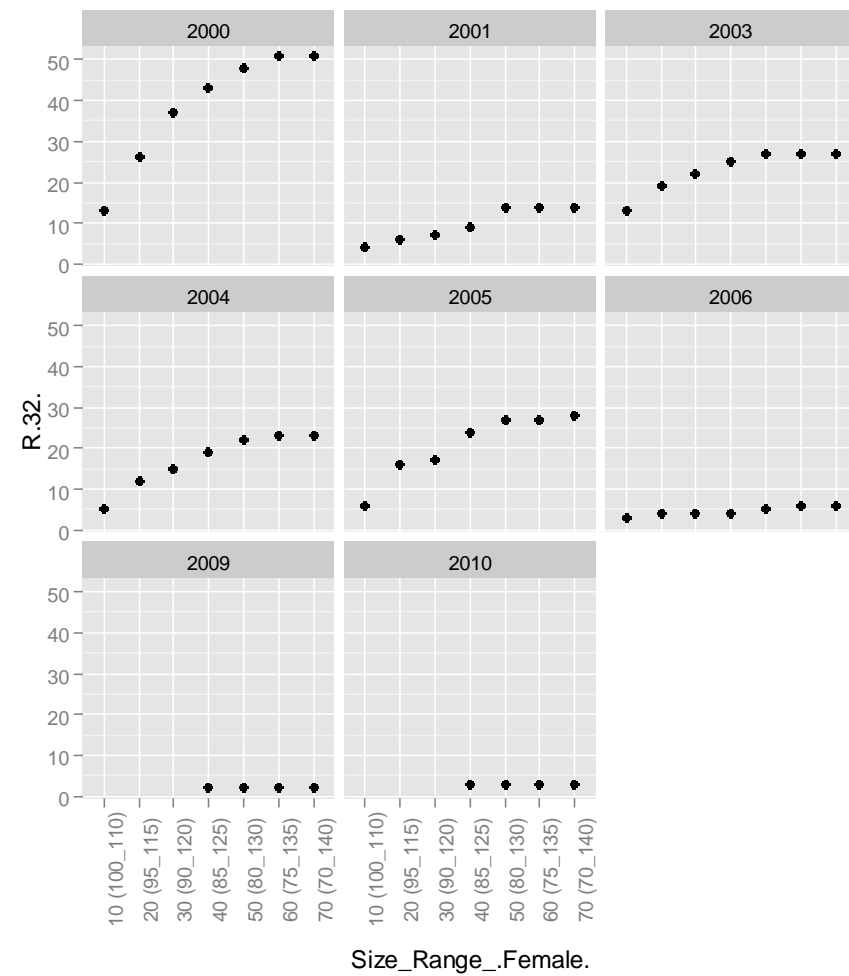
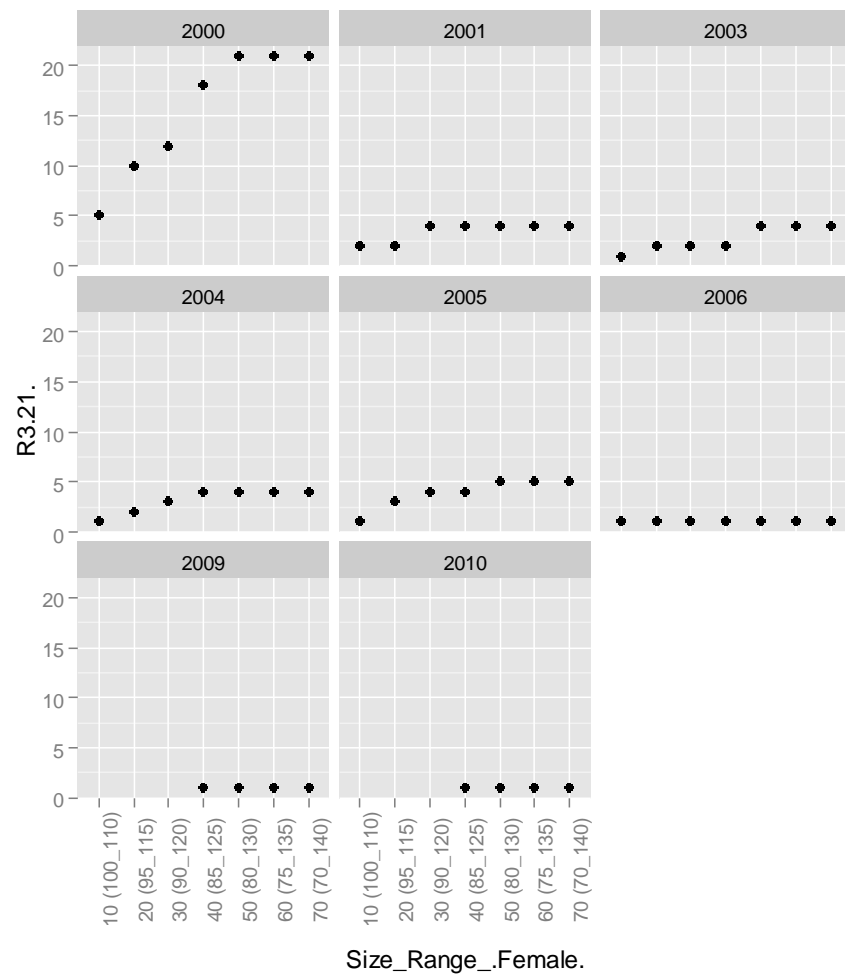


Figure 4 - 4 The number of recaptured lobsters at $R_{3[21]}$ and $R_{3[32]}$ for different size bin widths.

4.3.1.2 The effect of the number of $R_{3[21]}$ and $R_{[32]}$ on the estimated TIM proportion

The accuracy of the estimated TIM proportion increased with the number of recaptured lobsters in the third survey (i.e. $R_{3[21]}$ and $R_{[32]}$). The number of $R_{3[21]}$ and $R_{[32]}$ required to obtain accurate estimates was at least 15 and 40 recaptured lobsters, respectively (Figure 4-5). Unfortunately fewer female lobsters were recaptured in these larger size bins making an estimate for females difficult although results do appear similar.

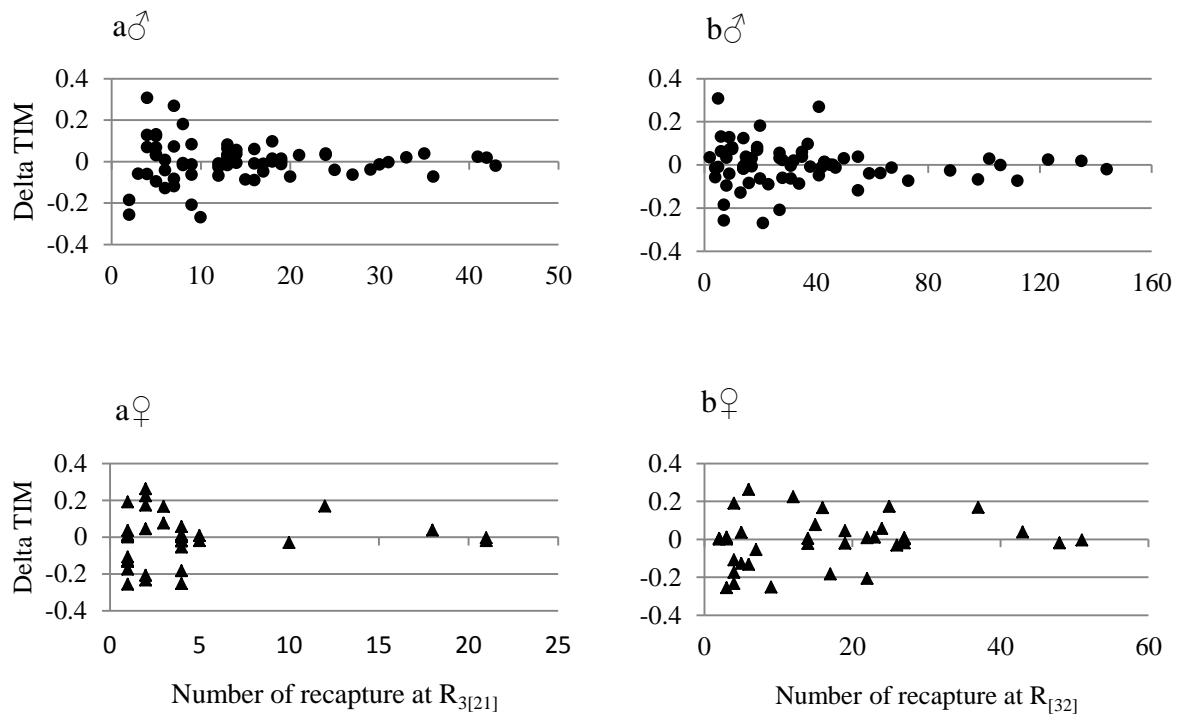


Figure 4 - 5 Accuracy of estimated TIM proportion against the number of recaptured lobsters at $R_{3[21]}$ (a) and $R_{[32]}$ (b) for males (♂) and females (♀), separately.

4.3.1.3 Determination of effort

As 1998 lobsters were translocated to the study area in November 2005, capture rate after this perturbation may not reflect the normal situation. As such, only three “three-survey” studies (i.e. include the initial tagging event and two subsequent recapture events) were undertaken before this perturbation (i.e. studies that started in Nov 2000, Dec 2001 and Jan-Feb 2003), could be analysed to determine the effort required to estimate TIM proportion.

The estimated sample size and effort is based on 15 lobsters being recaptured in survey 3 ($R_{3[21]}$) as determined in section 4.3.1.2. Only surveys in Nov 2000, Jan-Feb 2003 and Jan 2005 met the criteria of >15 lobsters in $R_{3[21]}$ and >40 lobsters in $R_{3[32]}$. For females, Nov 2000 was the only set of surveys where sufficient female lobsters were recaptured in the third survey (Table 4-1). However, unlike males where the number captured in the third survey for analyses starting in Nov 2000, Dec 2001 and Jan-Feb 2003 was more than 15, females had substantially lower recapture rates, particular between the second and third surveys. This resulted in the estimated number of traps required in each of the surveys to be large for females.

Across all surveys, using the mean proportion of lobsters recaptured and capture rates, the maximum number traps required to capture sufficient lobsters to produce an accurate estimate of TIM for males was 572 (~600) and 1173 (~1200) traps in the first survey for males and females, respectively (Table 4-2), whilst the number of traps required to capture sufficient lobsters to produce an accurate estimate of TIM proportion for combined sexes was 371 (~400).

Table 4 - 1 Estimated and observed parameters in every three survey study started in 2000, 2001 and 2003. Bold indicates when the observed value was greater than the estimated. T1, T2, and T3 denote the number of traps set in the first, second and third surveys, respectively.

Sex/ Studies		Males			Females			Combined Sexes		
		Nov	Dec	JF	Nov	Dec	JF	Nov	Dec	JF
Parameters		2000	2001	2003	2000	2001	2003	2000	2001	2003
N_1	Observed	267	255	246	448	393	228	715	648	474
	Estimated	167	239	217	320	1474	855	238	486	338
$R_{[21]}$	Observed	73	55	38	109	48	14	182	103	52
	Estimated	46	52	33	78	180	52	61	77	37
$R_{3[21]}$	Observed	24	16	17	21	4	4	45	20	21
	Fixed	15	15	15	15	15	15	15	15	15
T_1	Observed	891	602	801	891	602	801	891	602	801
	Estimated	557	564	707	636	2257	3004	297	451	572
T_2	Observed	602	801	403	602	801	403	602	801	403
	Estimated	376	751	356	430	3004	1511	201	601	288
T_3	Observed	801	403	708	801	403	708	801	403	708
	Estimated	501	378	625	572	1511	2655	267	302	506

Table 4 - 2 Summary of the mean observed and estimated parameters. (Cr= Capture rate)

	Males			Females			Combined sexes		
	Observed	Estimated	Mean Cr	Observed	Estimated	Mean Cr	Observed	Estimated	Mean Cr
N1	256	197	0.34	356	563	0.48	612	305	0.82
$R_{[21]}$	55	42	0.09	57	80	0.09	112	53	0.19
$R_{3[21]}$	19	15	0.03	10	15	0.01	29	15	0.05
Trap S1	765	572	-	765	1173	-	765	371	-
Trap S2	602	445	-	602	872	-	602	285	-
Trap S3	637	480	-	637	1077	-	637	332	-

4.3.2 Estimation of TIM proportion

4.3.2.1 Estimation of TIM proportion with observed data

From 2000 – 2012 there were only four occasions for males, one occasion for females and five occasions for combined sexes where there was greater than 15 and 40 recaptures of lobsters in both categories $R_{3[21]}$ and $R_{[32]}$, respectively. Estimates of TIM proportion varied in the reserve (Table 4-3). In 2005 TIM proportion was as low as 12% whereas in 2000 it was almost 34%. The average of TIM proportion estimates for combined sexes was 35%, whilst TIM proportion was 25% and 37% across all male surveys and the only estimates for females, respectively.

Table 4 - 3 Estimates of TIM proportion with observed data in the size bin of 95-155 (males) and 80-130 (females).

	Sex	survey										Overall TIM proportion estimate $\bar{X} \pm \text{s.d.}$
		Nov 2000	Dec 2001	Jan- Feb 2003	Jan- Feb 2004	Jan 2005	Jan- Feb 2006	Jan 2007	Jan 2008	Feb 2009	Jan 2010	
Number of Sampling- days	M&F	10	7	11	7	10	12	7	4	4	4	
Number of Tagged Lobsters*	M	267	255	246	132	359	312	255	116	69	116	
	F	488	393	228	143	363	191	156	91	56	90	
N_1	M&F	755	648	474	275	722	503	411	207	125	206	
Number of Tagged Lobsters*	M	255	246	132	359	312	255	116	69	116	85	
	F	393	228	143	363	191	156	91	56	90	34	
N_2	M&F	648	474	275	722	503	411	207	125	206	119	
Capture rate	M	0.33	0.29	0.45	0.29	0.41	0.24	0.19	0.57	0.10	0.23	
$\text{Beta}_{3[21]} =$	F	0.19	0.08	0.29	0.15	0.23	0.04	0.00	0.00	0.33	0.50	
$R_{3[21]}/R_{[21]}$	M&F	0.25	0.19	0.40	0.23	0.37	0.20	0.17	0.45	0.15	0.25	
Capture rate	M	0.27	0.22	0.15	0.32	0.25	0.36	0.12	0.20	0.14	0.22	
$\text{Beta}_{[21]} =$	F	0.22	0.12	0.06	0.19	0.06	0.14	0.03	0.07	0.05	0.02	
$R_{[21]}/N_1$	M&F	0.25	0.16	0.11	0.25	0.15	0.28	0.09	0.14	0.10	0.14	
Capture rate	M	0.22	0.15	0.32	0.25	0.36	0.12	0.20	0.14	0.22	0.11	
$\text{Beta}_{[32]} =$	F	0.12	0.06	0.19	0.06	0.14	0.03	0.07	0.05	0.02	0.09	
$R_{[32]}/N_2$	M&F	0.16	0.11	0.25	0.15	0.28	0.09	0.14	0.10	0.14	0.10	
Number of Recaptured Lobsters	M	73	55	38	42	88	112	31	23	10	26	
	F	109	48	14	27	22	27	5	6	3	2	
$R_{[21]}$	M&F	182	103	52	69	110	139	36	29	13	28	
Number of Recaptured Lobsters	M	24	16	17	12	36	27	6	13	1	6	
	F	21	4	4	4	5	1	0	0	1	1	
$R_{3[21]} > 15$	M&F	45	20	21	16	41	28	6	13	2	7	
Number of Recaptured Lobsters	M	55	38	42	88	112	31	23	10	26	9	
	F	48	14	27	22	27	5	6	3	2	3	
$R_{[32]} > 40$	M&F	103	52	69	110	139	36	29	13	28	12	
TIM proportion Estimates**	M	0.34	0.47	0.29	0.14	0.12	0.49	-0.02	0.74	-1.2	0.54	0.25±0.12
	F	0.37	0.26	0.34	0.59	0.38	0.13	Inf	Inf	0.93	0.83	0.37
	M&F	0.36	0.44	0.38	0.34	0.26	0.57	0.16	0.77	0.12	0.59	0.35±0.06

* Number of tagged lobsters in the first tagging event for every three-year survey.

**TIM proportion was only estimated in four occasions for males, one occasion for females and five occasion for combined sexes when $R_{3[21]} > 15$ and $R_{[32]} > 40$.

4.4 Discussion

In this study we applied a three survey model to estimate tag induced mortality (TIM) proportion of lobsters tagged annually in the wild. Previous attempts to estimate annual TIM proportion have been reliant on aquaria studies or caging experiments. A major benefit of this method is that it estimates TIM proportion *in-situ* by comparing tag returns from multiple tagging years. However, a weakness of the method is the need to recapture animals on two occasions. For our dataset we found that at least 15 lobsters had to be recaptured from two occasions $R_{3[21]}$ and 40 on one occasion $R_{[32]}$. Over the three “three-survey” periods that we could estimate the number of traps required to capture 15 lobsters in the third survey, there was considerable variability in the proportion of lobsters captured between surveys for both sexes, although the proportion was substantially lower for females resulting in estimates that are probably beyond logistical capabilities. On average 600 traps would be required for males, 1200 for females and 400 for combined sexes.

The average proportion of annual TIM for males was estimated at 0.25 and varied from 0.12 in 2005 to 0.34 in 2000. The only estimate for females was 0.37. There are a number of reasons for such stark contrasts in TIM proportion including the quality of handling and tagging process. In this study, tagging was conducted by different people ranging from those with no prior experience to staff with many years’ experience, which may have led to differences in stress and injury during tagging. For example, inappropriate handling can result in the loss of some appendages including antenna or legs, which can affect foraging and defence against predators and thus increased mortality rates (Herrnkind *et al.*, 2001; Parsons and Eggleston, 2005; Frisch and Hobbs, 2011). Parsons and Eggleston (2005) mentioned three reasons for the higher predation rate on injured lobsters all of which could contribute to the estimate of TIM proportion in this study: 1) releasing a mixture of organic compounds (e.g., blood) that may attract predators; 2) reduction in the defensive capacity of injured lobsters; and 3) losing cooperative group defence by injured lobsters.

However, other studies using aquaria and cage methods have also reported high estimates of TIM proportion. Montgomery and Brett (1996) indicated that TIM was occurring after 4 weeks for *Jasus verreauxi* tagged with T-anchor tags and held in aquaria although environmental conditions within the aquaria made it difficult to separate out TIM from other stressors as mortality occurred in both tagged and untagged lobsters after week 15. Dubula *et al.* (2005) found that the rate of TIM in *Jasus lalandii* was 27% and 46% in aquarium and in sea-cage studies, respectively. They reported that the higher rate of TIM estimates in sea-cages was due to cannibalism as they were unable to separate lobsters in sea cages as they moulted. Claverie and Smith (2007) suggested that the high rate of TIM (48%) over a 60 day-study for the galatheid, *Munida rugose* (Fabricius, 1775) was associated with infection. Both Dubula *et al.* (2005) and Claverie and Smith (2007) observed black necrotic tissue around the wound associated with tagging.

Our study also found high TIM proportion similar to these studies. While there are potential explanations for such high proportion and/or rate estimates for each of the methods, the consistency in high values across the different methods would suggest that TIM proportion and/or rate is high for crustaceans. If these estimates are correct, then caution needs to be applied in the use of tagging data to estimate survival probability or population size. Tagging studies used to estimate population parameters assume that all animals tagged in a survey are equally available for recapture at the next survey with the difference being survival rate. With substantially fewer lobsters available for recapture due to TIM, survival probability and population estimates that do not account for TIM would be over-estimated.

Although only one estimate of TIM proportion was available for females, it is encouraging to see that it is relatively similar to the estimate for males and thus an estimate of TIM proportion for combined sexes may be appropriate given the consistency across species. In this study, TIM proportion could be estimated for combined sexes on five occasions. There was considerable consistency between these estimates with a mean of 0.35 and standard deviation of 0.06 indicating

that TIM proportion may be consistent between years and could be estimated on a small number of occasions and applied for longer time periods. An alternative estimate of TIM proportion can be achieved by aggregating data across all surveys. This provided a TIM estimate of 0.30, 0.42 and 0.39 from males, females and combined sexes respectively. Although the data is limited, both the single estimate (Nov 2000) and the aggregated data indicate that TIM proportion is slightly higher for females than males. However, this difference is far less than the difference in recapture probability between males and females suggesting that there are other biological issues influences recaptures rates such as dominance hierarchies (Frusher and Hoenig, 2001).

From 2008 onwards there was a reduction in the effort to 4 days in response to cost savings. Unfortunately for calculating TIM proportion the effort was insufficient to recapture enough tagged lobsters in the $R_{3[21]}$ and $R_{[32]}$ categories. Of particular importance is the need to maintain consistency in sampling effort across all the surveys used. Although appropriate numbers of lobsters were tagged in 2007 during a 7 day survey, the subsequent survey days were reduced to 4 in 2008 and 2009 resulting in insufficient lobsters to estimate TIM proportion.

The extremely low estimate of TIM in 2005 may reflect a perturbation to the population as over 1998 tagged lobsters (28% male, 72% female) were translocated into the study area three months before the first annual recapture event of the three-year survey which started in 2005.

This study highlights the need for TIM proportion and/or rate to be estimated as an essential part of tagging programs and provides a method to do so that can be incorporated into a standard tagging program. Tagging programs can provide valuable information on population parameters for assessing fish stocks but large values of TIM proportion, as found in this and other surveys, would result in biased estimates that are overly optimistic and lead to over exploitation of stocks. TIM proportion needs to be estimated in tagging programs and incorporated into analyses for improved estimates of survival rates and population size.

Appendix 4-1 Development of *in situ* tag-induced mortality model (From Frusher *et al.*, 2009)

In a tagging program the first survey captures untagged animals and the released lobsters include lobsters tagged during that survey. On subsequent surveys the catch will include lobsters tagged on previous surveys as well as untagged lobsters that are subsequently tagged during this event. Released lobsters include a mix of previously tagged and newly tagged lobsters. If the initial impact of tagging occurs between surveys then tagged lobsters recaptured in any subsequent survey would have survived the impact of tagging. By comparing the fate of previously tagged lobsters against the fate of newly tagged lobsters on subsequent surveys, the difference should be the initial impact of tagging. A minimum of three surveys are required to estimate tag induced mortality and the following example illustrates the method:

During each of two surveys we tag a number of animals. During the second survey (S2) we would recapture animals tagged during the first survey (S1) and during the third survey (S3) we would recapture animals that were tagged during both the first (S1) and second (S2) surveys.

If we assume that initial tag loss and tag-induced mortality occurs shortly after tagging and before the following survey then recaptures will be a function of the total number tagged, the initial impact of tagging (which includes both initial tag loss and initial tag-induced mortality), and the recapture rate.

$$r_{i+1,i} \propto N_i(1 - \theta_{I_i})\theta_{R_{i+1}} \quad (1)$$

where $r_{i+1,i}$ = expected recaptures during survey $i+1$ from animals tagged during survey i

N_i = number of animals tagged during survey i

θ_{I_i} = number of animals that die from the initial impact of tagging during survey i

θ_{R_i} = probability of recapture during survey i (recapture rate)

For three surveys a tagging/recapture matrix can be establish as follows:

Survey 1 (S_1)	Survey 2 (S_2)	Survey 3 (S_3)
Tagging (N_1)	Recapture ($r_{21} = N_1(1 - \theta_{I_1})\theta_{R_2}$)	Recapture ($r_{31} = N_1(1 - \theta_{I_1})\theta_{R_3}$)
	Tagging (N_2)	($r_{32} = N_2(1 - \theta_{I_2})\theta_{R_3}$)

Recaptures during S3 will also include recaptures of animals that were recaptured and released during S2 from tagging event 1. This subset of recaptures can be stated as follows:

$$r_{3[21]} = r_{21}\theta_{R_3} \quad (2)$$

Note that equation (2) is similar to the other equations in the tagging/recapture matrix with the number recaptured and returned during survey 2 now becoming the number of tagged lobsters released (i.e. r_{21} replaces N_1 in the equation estimating r_{31}) and as these animals were previously tagged, the initially impact of tagging for this cohort has already occurred (i.e. $\theta_{I_1} = 0$ and therefore $(1 - \theta_{I_1}) = 1$).

If we divide r_{32} by $r_{3[21]}$ it is possible to estimate the initial impact of tagging associated with tagging event 2.

$$\frac{r_{32}}{r_{3[21]}} = \frac{N_2(1 - \theta_{I_2})\theta_{R_3}}{r_{21}\theta_{R_3}} \quad (3)$$

If we assume that the probability of capturing a tagged animal in the population is equal for animals tagged during surveys 1 and 2 then θ_{R_3} cancels out and θ_{I_2} is the only unknown.

$$1 - \theta_{I_2} = \frac{r_{32}r_{21}}{r_{3[21]}N_2} \quad (4)$$

and

$$\theta_{I_2} = 1 - \frac{r_{32}r_{21}}{r_{3[21]}N_2} \quad (5)$$

The following is a worked example.

Let the number of animals tagged be 100 and 200 for S1 and S2 respectively. The initial impact of tagging is 10% for S1 and 20% for S2 and the tag recovery rate is 30% in S2 and 40% in S3.

Thus, $N_1 = 100$, $N_2 = 200$, $\theta_{I_1} = 0.1$, $\theta_{I_2} = 0.2$, $\theta_{R_2} = 0.3$ and $\theta_{R_3} = 0.4$

Using the tagging/recapture matrix above, we get:

S ₁	S ₂	S ₃
Tagging N ₁ = 100	Recapture $r_{21} = 100*(1-0.1)*0.3 = 27$	Recapture $r_{31} = 100*(1-0.1)*0.4 = 36$ and $r_{3[21]} = 27*0.4 = 10.8$
	Tagging N ₂ = 200	$r_{32} = 200*(1-0.2)*0.4 = 64$

From equation 5 we can estimate θ_{I_2}

$$\theta_{I_2} = 1 - \frac{64 \times 27}{10.8 \times 200} = 0.2$$

It is worthwhile noting that neither chronic tag loss nor natural mortality has been incorporated into the methodology. It is assumed that natural mortality would be equivalent for both groups between surveys and thus cancel out. If tag loss is chronic and linear, then tag loss would be equivalent for both groups and θ would be a measure of tag-induced mortality only.

Chapter 5

Estimating tag loss rate for the southern rock lobster *Jasus edwardsii*

Chapter 5 Estimating tag loss rate for the southern rock lobster *Jasus edwardsii*

Abstract

A three-year double tagging program was conducted from 2012 to 2014 in a no-fishing area at Crayfish Point Scientific Reserve (CPSR), near Hobart, Tasmania, Australia, where the population contains lobsters tagged over the last two decades, to determine tag loss rate and evaluate the effect of synchronous and non-synchronous double tagging of *Jasus edwardsii*. The rate of tag loss depended on the time interval between applying the first and second tags on lobsters and sex. Males had a higher rate of tag loss than females over the first year of the study. Lobsters that were double-tagged but at two different times (non-synchronous) had a slightly higher rate of tag loss than those double tagged synchronously, which was associated with the higher rate of tag loss for the older tag. A higher rate of tag loss for older tags suggests that tag loss is both chronic and non-linear. If tag loss rate is substantially non-linear, multi-year tagging programs would need to account for tag loss in the analysis.

Key words: capture-mark-recapture (CMR), tag loss rate, lobster

5.1 Introduction

Tagging is widely used to estimate demographic parameters in aquatic animals (Bell *et al.*, 2003) including those harvested in fisheries. The method involves tracking the fate of tagged animals over time, although, tag loss decreases the number of tagged individuals in the population (Montgomery and Brett, 1996), which can bias both survival probability and population size estimates. Tagged individuals will also be lost due to tag induced mortality (TIM), natural mortality, fishing mortality, or when tagged animals emigrate from the study area. Tags may be lost due to tissue infection or insufficient anchoring (“type I tag loss”) or may be shed during moulting in crustaceans where the tag is attached to the exoskeleton (“type II tag loss”) (Beverton and Holt, 1957). Research reported

here was on crustaceans where tags were inserted into the muscle rather than attached to the exoskeleton so were not technically exposed to type II tag loss. Where tag loss affects the estimates obtained from tag-recapture models, the fate of tags needs to be investigated to minimise bias in interpretation of model estimates.

Two broad approaches are used for estimating tag loss rate. The first is direct observation where tagged animals are monitored in a captive environment such as aquaria or sea-cages. However, captivity may reduce environmental factors that would otherwise lead to tag loss, including infection (Courtney *et al.*, 2001), entanglement (Frisch and Hobbs, 2006) and tag removal by other animals (Melville-Smith and Chubb, 1997). The second method is indirect and involves double-tagging individuals in the field and estimating the rate of tag loss from the relative proportion of recaptured individuals with both tags or only one tag still attached (Frisch and Hobbs, 2006; Frusher *et al.*, 2009). Although the direct method provides an absolute measurement of tag loss rate, the indirect method is considered to be more realistic of what might occur in nature, provided sufficient individuals are released and recaptured (Frisch and Hobbs, 2006).

Several studies have been conducted to assess the effect of factors such as size-at-tagging, sex, moulting stage and insertion position of the tags on the rate of loss of T-bar tags in lobsters where two tags were inserted synchronously (i.e. two tags applied at the same time). They include studies under artificial conditions such as cage and aquarium trials (Montgomery and Brett, 1996) and in the field (Melville-Smith and Chubb, 1997; Frusher *et al.*, 2009; Gonzalez-Vicente *et al.*, 2012). Double-tagging studies have also been conducted on the same species examined here, *Jasus edwardsii*, with Frusher *et al.* (2009) reporting that size had no significant impact on tag loss rate although tag loss rate was almost double in male lobsters. A higher rate of tag loss rate in males has also been reported in other lobster species (Gonzalez-Vicente *et al.*, 2012; Xu *et al.*, 2014). The probability of losing a tag also varies with moult cycle and increased during the late pre-moult (Moriyasu *et al.*, 1995). Both Melville-Smith and Chubb (1997) and Frusher *et al.* (2009) reported

that dorsal tags were more likely to be lost or damaged compared to tags inserted into the ventral parts of lobsters. Frusher *et al.* (2009) found tag loss to be chronic with an approximately constant rate of loss over time. In this study we explored in more detail the fate of tags by comparing newly tagged lobsters with previously tagged lobsters.

5.2 Methods

5.2.1 Study area

This study was conducted at the Crayfish Point Scientific Reserve (CPSR), in Tasmania, Australia (42° 57' 08"S 142° 21' 20"E). This scientific reserve contains rocky reef habitat and is surrounded by sand, which limits rock lobster movement (Barrett *et al.*, 2009). The reserve contains a temperate rocky reef of 1.24 km² with maximum depth of 15 m. Fishing in the reserve has been prohibited since its declaration in November 1971 and the reserve now holds a dense population of *Jasus edwardsii* (Green *et al.*, 2009) that have been tagged for various projects over the last two decades.

5.2.2 Field method

A three-year double tagging program was conducted from 2012 to 2014. On the first day of each annual survey baited traps were set in the early afternoon. On subsequent days the traps were checked in the morning, all lobsters removed, re-baited and re-set. In 2012 and 2013 untagged lobsters were double tagged and lobsters that had been single tagged in previous surveys had an additional tag inserted. All tags are uniquely coded T-bar tags (Hallprint T-bar anchor tag; TBA1, Hallprint Pty Ltd, 27 Jacobsen Crescent, Holden Hill, South Australia 5088, Australia) and were applied on both sides of the ventral surface of the first or second abdominal segment. The tag number, length of carapace and sex of all lobsters were recorded and lobsters were immediately released after tagging.

In 2012, during the first survey there were two groups of lobsters released: 1) lobsters bearing no tag on which two new tags (2NT) were applied synchronously, 2) lobsters carrying a single old tag on which a new tag (ONT) was applied as the second tag (non-synchronously double tagged lobsters). In 2013, in addition to these groups, a third group of lobsters was released: 3) lobsters carrying two old tags (2OT) where no new tag was applied. This included both group 1 and group 2 lobsters tagged in 2012.

Tag loss was estimated as the proportion of double tagged lobsters tagged in survey S_t that had lost a tag in a future survey (S_{t+1}):

$$Tag\ loss = \frac{N_{1T(S_{t+1})}}{N_{DT(S_{t+1})}}$$

Where tag loss is the proportion of tags lost between survey (t) and the following survey ($t+1$)

N_{1T} = Number of double tagged lobsters in S_t that had lost one tag in survey S_{t+1}

N_{DT} = Number of double tagged lobsters in S_t that had lost no tags in survey S_{t+1}

5.3 Results

In 2012 and 2013, 340 and 235 double-tagged lobsters were released back to the wild, respectively. The number of recaptures is shown in Table 5-1. The annual rate of tag loss was slightly higher during the second year (17.9%) compared to the first year (15.7%) after tagging (Figure 5-1).

Table 5 - 1 Fate of tags on newly tagged and previously tagged lobsters.

Tagging Year	Recapture Year	Number of Double Tagged Lobsters	Double tagged lobsters including lobsters with	Survey period Year	Recaptured lobsters			1 tag loss		
					Originally double tagged Total	Male	Female	Total	Male	Female
2012+2013	2013+2014	575	2 new tags and 1 old and 1 new tag	1	102 (48+54)	72	30	16 (8+8)	13	3
2012+2013	2013+2014	285	2 new tags	1	43 (19+24)	33	10	6 (2+4)	4	2
2012+2013	2013+2014	290	1 old and 1 new tag	1	59	39	20	10*	9	1
2012	2013 and 2014	340	2 old tags	1	18**	13	5	4	2	2
2012	2014	240	2 new tags and 1 old and 1 new tag	2	70	43	27	25	12	13
2012	2014	160	2 new Tags	2	28	19	9	9	5	4
2012	2014	180	1 old and 1 new tag	2	42	24	18	16***	7	9

*including 6 old tags and 4 new tags lost

**Double tagged lobsters recaptured over two consecutive recapture events in 2013 and 2014

*** including 4 old tags and 12 new tags lost

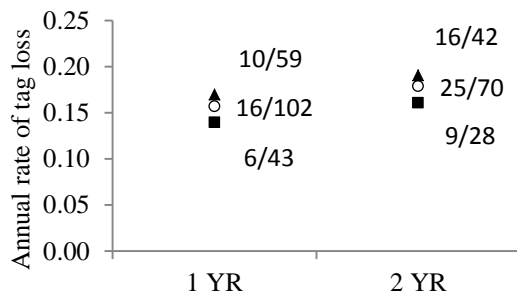


Figure 5 - 1 Annual rate of tag loss for lobsters double tagged synchronously (■), non-synchronously (▲) and combined (○) for one and two years since tagging. The number of single tagged and double tagged lobsters captured for non-synchronous, combined and synchronous are provided from top to bottom respectively.

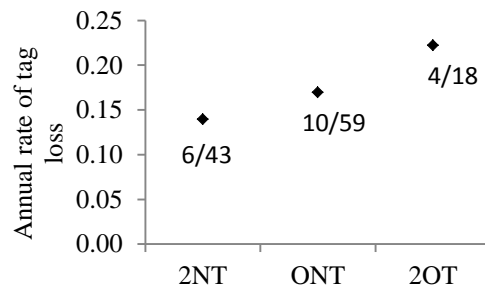


Figure 5 - 2 Annual rate of tag loss for lobsters captured in 2014 which had either two new tag (2NT), an old and a new tag (ONT) or two old tags (2OT) when released in 2012 and/or 2013. The number of single tagged and double tagged lobsters captured are provided below each data point.

Lobsters tagged non-synchronously had a slightly higher rate of annual tag loss than those tagged synchronously indicating that older tags are lost at a greater rate than new tags (Figures 5-1 and 5-2). This resulted in slightly higher overall annual tag loss rates of lobsters recaptured after two years as these included a greater proportion of older tagged lobsters (Figure 5-1).

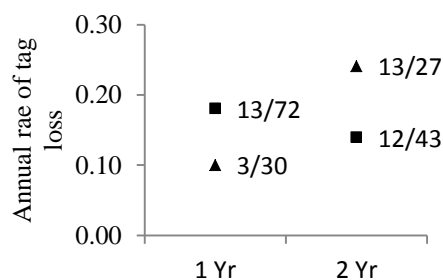


Figure 5 - 3 Annual rate of tag loss for males (■) and females (▲) lobsters double tagged (synchronous and non-synchronous combined) over the first and second year of study. The number of single tagged and double tagged lobsters captured is adjacent to each data point.

Males and female lobsters showed both different rates and different trends in tag loss. Males had a higher rate of annual tag loss during the first year and then only a minor decrease in the rate of annual tag loss over the second year, whereas females had a lower initial rate of tag loss in the first year followed by a substantial increase in annual tag loss rate in the second year (Figure 5-3).

5.4 Discussion

A double tagging program conducted over a two year period showed that the rate of tag loss was relatively constant over time, with non-synchronous double tagged lobsters having a slightly higher rate of tag loss than synchronous double tagged lobsters. The rate of tag loss found in this study was higher than reported in other studies for both this species and for other lobster species (Table 5-2). Several factors may influence the difference between

studies including the proportion of synchronous or non-synchronous tagged lobsters in the populations, species, sex, experience level of taggers, experimental conditions (e.g. field vs laboratory) and tag insertion position (dorsal or ventral).

Table 5 - 2 Summary of the tag loss rate estimated for different species of lobsters.

Species	Experiment	Tag position	Sex	% TL Annually		Annual TL %		References
						Year 1	Year 2	
<i>Jasus edwardsii</i>	Field	ventrally	M & F	15.7	Male Female	18.0 10.0	14.0 24.0	Current study
<i>Jasus varreauxi</i>	Laboratory	dorsally	M & F	8.0				Montgomery and Brett (1996)
<i>Jasus edwardsii</i>	Field	ventrally	M F	13.4 6.1				Frusher <i>et al.</i> (2009)
<i>Palinurus elephas</i>	Field	dorsally	M F	6.8 5.0				Gonzalez-Vicente <i>et al.</i> (2012)

In this study many of the lobsters captured already had a single tag and the extra (double) tag inserted was a new tag. The increased rate of tag loss of lobsters that had either one or two old tags (i.e. tags inserted a least one year prior to the year of capture) indicates that, over time, tags are able to work their way out of the lobster or are weakened such that they are more easily removed. Although the difference was relatively small, further studies into the longevity of tags inserted into lobsters are required as chronic and non-linear tag loss rate is problematic in many mark recapture models such as multi-year mark recapture models.

Similar to Frusher *et al.*, 2009, this study also found annual tag loss rate to be chronic and similar from year to year although at a higher rate. The continued loss of tags needs to be factored into mark recapture experiments as it reduces the number of tagged lobsters in the population available for recapture during the recapture surveys and would result in an under-estimation (negative bias) of survival rates (Pollock *et al.*, 1990) and over-estimation (positive bias) of population size (Pollock and Alpizar Jara, 2010).

The rate of tag loss varied between sexes. Females had a lower rate of tag loss than males over a 12-month period between tagging and recapture event which was similar to previous studies undertaken on *Jasus edwardsii* and *Palinurus elephas* in the field (Frusher *et al.*, 2009; Gonzalez-Vicente *et al.*, 2012). Although the rate of tag loss for both sexes was higher than Frusher *et al.* (2009), females were estimated to have approximately half the tag loss rate of males. In contrast, Gonzalez-Vicente *et al.* (2012) estimated substantially lower tag loss rates for both males and females of *P. elephas* with only a minor difference between sexes. The lower tag loss rate in females may be due to the ventral positioning of the tag which would be protected by the larger pleopods and the developing eggs in females which occur for 5-6 months of the year.

However, in contrast to the findings of Frusher *et al.* (2009), the annual tag rate declined for males between the first and second year and rapidly increased for females. This may have been associated with lower sample sizes in the current study, as Frusher *et al.* (2009) and Gonzalez-Vicente *et al.* (2012) tagged 700 lobsters and 2486 lobsters (64.2 % females) respectively, whereas only 575 lobsters were double tagged in this study. Although an accurate number could not be determined, greater than 575 lobsters need to be double tagged and approximately 1000 lobsters is likely to be a better sample of tagged individuals. Given the magnitude of tag loss rate and the contrasting results for males and females and between different studies on the same species, further research conducted over longer periods of time (i.e. > 3 years) and with larger sample sizes is warranted.

Although the number of lobsters tagged in this study was low and this made it difficult to estimate the rate of tag loss for males and females over two years, results from this study confirm that tag loss rate is relatively high (between 5-20%) and thus important if population size and survival or mortality rate are to be estimated from tagging studies. The history of the tag (i.e. whether synchronously or non-synchronously tagged) and the sex of the species

being tagged will also bias population parameter estimates from tagging studies. *In-situ* double tagging experiments should be an important component in the initial design of tagging studies.

Chapter 6

General Discussion

Chapter 6 General Discussion

The development of future management strategies is dependent on the quality and quantity of data available for assessment of natural resources. In the marine environment the provision of such data is even more problematic as it is not possible to directly observe aquatic animals. For fisheries, the sources of data are either fisheries dependent, where the industry supplies information (e.g. catch and effort data in mandatory log books) or independent where scientists control the design and collection of information or a combination of both where fishers participate in scientifically designed data collection projects or programs. Fisheries dependent data collected by routine fishing operations is limited by (i) regulatory factors including minimum size limit, seasonal closures, fishing effort and landings and (ii) fishers' decisions associated with economic yield as fishermen chase profits (e.g. they may change the locations of fishing to obtain the best catch rate). While fisheries dependent data comes from all regions of the fishery and during times when the fishery is operating, fishery independent data is normally associated with short time intervals and few locations due to funding constraints.

Annual estimates of population size are required to determine the amount of fish available for harvesting. Population size is affected by birth, immigration, emigration and mortality (natural and fishing). In fishing areas, the density of legal size animals can be substantially reduced by fishing. Survival or natural mortality ($1 - \text{natural survival}$) estimates have proven to be difficult to estimate in fished regions where attribution between fishing and natural causes of mortality are confounded. Studies in regions where fishing is prohibited can provide valuable insights into natural mortality as it is not complicated by fishing. However, the size structure and density in fish protected areas is different to that within fished areas and extrapolation to fished areas needs to be undertaken with caution. However, given the

difficulty in estimating survival probability, these estimates do provide valuable insights into the potential magnitude of natural mortality.

Fisheries dependent data relies on catch and effort data to obtain population size and there are concerns that this data will become less reliable as fisheries adopt output controls such as quota systems. Under a quota system fishers aim to maximise return per fish (\$/kg) rather than maximising catch. This can result in fishers changing behaviour to target different locations and/or times of the year when fish prices are higher but catch rates are lower or they may prefer to discard less valuable catches and allocate their quota to more valuable fish (i.e. high grading). Under quota management systems, fishery independent methods for estimating population size may become increasingly required if the relationship between catch rates and abundance dissolves.

Tagging is one of the most widely used tools in fisheries science and capture-mark-recapture models are able to estimate both survival probability and population size. Many of these models require multiple tagging events and a large number of animals to be tagged to provide sufficient precision and accuracy in parameter estimates. However, like most fishery independent studies, surveys can be expensive and thus it is important to minimise costs while ensuring the appropriate outcomes are achieved. While there is a range of software available for analysing CMR data, including statistical methods for selecting the best fitting model from a range of possible models (Lebreton *et al.*, 1992), limited attention has been given to some of the pragmatic issues associated with sampling design including the costs of research and the trade-off between costs and accuracy and precision of estimated parameters. For example, small sample size rendered 98% of small mammal studies (McKelvey and Pearson, 2001) ineffective highlighting the need for careful consideration of the number of surveys, sampling days per survey and sampling units per sampling day.

CMR studies are also likely to be restricted by funding and time. Although it is well known that the accuracy and precision of estimates increase with sampling design, there is a trade-off between research funding and the precision of estimates. A cost-effective CMR design can minimize the cost of the study, whilst ensuring that the required precision and accuracy of estimates are provided for sustainable fisheries management. The aim of this study was to examine a cost-effective sampling design for estimating survival probability of *Jasus edwardsii* in the Crayfish Point Scientific Reserve (CPSR) where fishing has been banned since 1971.

Although, this study was conducted in a no-fishing area, the outcomes of this study can be used as a guideline for applying CMR studies in fished and non-fished areas. Whilst, the most cost efficient outcomes for sampling design are discussed in this study, some issues including density and size structure of target animals need to be taken into account for applying the outcome of this study for the same species in fished areas. For estimating survival probability, no-fishing areas can be beneficial as there are no confounding effects of fishing and the probability of recapture of each tagged lobster is higher. For example, Olsen and Moland (2011) found that 50% of tagged animals can be removed by fishing. However, the size structure of the target animal is often wider in the no-fishing area where super legal size individuals can be a considerable proportion of the population. Survival rates may therefore be lower than a fished population as older animals are expected to have higher mortality rates.

The current study was conducted on *Jasus edwardsii*, which has high site fidelity (Gardner *et al.*, 2003; Barrett *et al.*, 2009). The CPSR, which is surrounded by non-reef habitat, is an ideal place to undertake the study as apparent survival probability estimates are unlikely to include effects of migration. In studies where animals have low site fidelity or high rates of migration mark-recapture survival probability estimates will be underestimates of actual

survival probability as emigration of tagged lobsters will have the same impact as natural mortality, tag loss or tag induced mortality.

In multiyear CMR studies, any specific pressure on the target population may affect the outcome of the study. The result of this study showed that translocation of lobsters to the no-fishing area affected sex ratio of tagged and/or recaptured lobsters in favour of males. Whilst, 72% of translocated lobsters were females, survival probability of females reduced from 0.78 yr⁻¹ to 0.60 yr⁻¹ since translocating in November 2005. It appears that translocated lobsters may have been in excess of the carrying capacity of the CPSR resulting in either emigration or death of tagged females.

6.1 Developing a capture-mark-recapture design for estimating survival probability

In this study, a long term tagging data set from a population of *Jasus edwardsii* at Taroona, Tasmania, Australia was used to investigate the impact of the number of surveys and interval size between surveys (Chapter 2), and sample size and effort (Chapter 3), on the precision of survival probability estimates.

Whilst three-annual surveys are common in CMR studies, the result of this study showed that at least five annual surveys were required to obtain precise survival probability estimates at the level of 5% RSE for males and females, separately. Similarly Nowicki *et al.* (2008) also required five surveys to obtain a precise estimate of population size for white-clawed crayfish (*Austropotamobius pallipes*). In comparing between surveys undertaken at either equal or unequal time intervals between surveys, unequal survey intervals required more surveys to obtain similar precision to surveys with approximately equal intervals between surveys. Thus, equal interval survey designs were more cost-effective than unequal interval designs.

Examination of the amount of sampling effort required to achieve acceptable estimates of survival probability for males and females found that with increasing sampling effort (number of surveys and sample size) the most parsimonious model for estimating survival probability shifted from sex-independent towards sex-dependent. Although there was a 0.2 unit difference in survival probability between males and females, at least four sampling days using 50 traps were required for the most parsimonious model to be sex specific. However, depending on the requirements of the users of the data, sex differences of this magnitude may not warrant the extra costs associated with sampling.

A minimum of 500 lobsters (of approximately equal number of each sex) were needed to be tagged to obtain precise survival probability estimates. Depending on the costs of the survey, this could be achieved in the CPSR with either 30 traps over 4 sampling-days or with fewer traps (e.g. 15 traps) over longer sampling-days (e.g. six sampling-days) to estimate survival probability of combined sexes. Although the CPSR is in relatively protected waters and close to the research laboratories resulting in the requirement of a small vessel for a short period of time, the cost of vessel use and labour have resulted in many surveys being restricted to four days to match the Australian working week (Monday to Friday). Lower effort would also result in lowering the precision of the survival probability estimates. This study used a relative standard error of 5% and fisheries managers and/or assessment scientists may consider less precision is required, which would result in fewer traps or fewer trapping days. Thus it is important for the experimental design to be matched to the expected level of precision required by management.

6.2 Tag induced mortality (TIM) and tag loss (TL)

Tag induced mortality (TIM) and tag loss (TL) both reduce the initial number of tagged animals in the population and can bias parameter estimates. If the rate of tag loss or tag

induced mortality is substantial, survival probability would be under-estimated and population size overestimated (Pollock *et al.*, 1990; McDonald *et al.*, 2003).

In this study, a three-survey model was used to obtain *in situ* estimates of TIM proportion (Chapter 3).

Estimates of TIM proportion for this study were relatively high being on average between 25% and 40%. Although high, these estimates are similar to other studies across different species of lobster irrespective of the sampling design both *in situ* (current study) and in aquaria and/or in sea-cage studies (Montgomery and Brett, 1996; Dubula *et al.*, 2005). For example, Dubula *et al.* (2005) found that the rate of TIM in *Jasus lalandii* was 27% and 46% in aquarium and in sea-cage studies, respectively. When TIM rate is high, and not accounted in mark-recapture designs, estimates of population size will be substantially overestimated and would lead to inappropriate management decisions of the amount of allocated catch.

The precision of TIM proportion estimates is dependent on the number of tagged lobsters recaptured in two successive surveys from the initial tagging event. For lobsters in the CPSR a minimum of 15 lobsters were required to be recaptured in the third survey that were tagged in the first survey and also recaptured in the second survey. To meet this criterion for separate estimates of TIM proportion for males and females separately, trapping effort was greater than required to estimate survival probability. Approximately, 600 trap lifts were required for males and 1200 for females. The higher effort required for females reflected the lower recapture probability. While there was limited data available to compare differences between sexes, the available data did suggest that differences may be small and thus a combined estimate might be sufficient. A combined estimate required approximately 400 trap lifts which is almost double the estimated number of traps lifts required to estimate survival probability. Whereas survival probability estimates could be achieved with 50 traps over four days, to be able to estimate TIM proportion over four days (and thus fit into a

standard working week) twice as many traps would be required and consideration needs to be given to the extent of habitat that can be surveyed as well as the logistics of servicing this many traps.

In addition to TIM, tag loss also impacts the outcomes from survival probability estimates and in lobsters has been assumed to be low. Tag loss can be caused by either the method (poor tag insertion) or the environment including the tagged animal rejecting or removing the tag. The design of double tagging experiments including the timing of double tagging, sample size and sex differences can also affect the precision and accuracy of tag loss rate estimates.

Estimates of tag loss rate from this study were higher than previously reported for *J. edwardsii* (Frusher *et al.*, 2009) and other species (Montgomery and Brett, 1996; Gonzalez-Vicente *et al.*, 2012). A possible explanation for the higher rate of tag loss was asynchronous application of both tags. Old tags have a slightly higher rate of tag loss and many lobsters in this project already had an existing tag and thus the second tag was added to an existing single tagged lobster (Chapter 5). Other studies have also reported that tag durability decreased with the time after tagging both in lobsters and fish (Frusher *et al.*, 2009; Rude *et al.*, 2011; Bodine and Fleming, 2014).

This study was unable to define the minimum sample size required to obtain a precise estimate of tag loss rate due to the low number of lobsters double tagged. However, the number of double tagged lobsters is expected to be in excess of the number required to be tagged to estimate survival probability or tag induced mortality.

6.3 Combined estimates

Estimates of tag induced mortality and tag loss have not been incorporated into the survival

estimates. Results from this study found that at least 5 surveys were required to obtain precise estimates of survival based on Cormack-Jolly-Seber (CJS) mark-recapture models (Chapter 2). These survival estimates are based on recaptures across a number of surveys where tag loss and TIM would have different impacts. For example, TIM only impacts between the first and second annual survey and not between subsequent surveys. While these estimates are not additive they can provide approximate estimates of the mortality schedule. For example, in a study commencing in 2000, male mean annual survival from a 5 survey design was estimated at 61% (Figure 2-4), TIM at 34% (Table 4-3) and tag loss was 16% (Table 5-2). Combined these add to 111%. In contrast, for the study commencing in 2005, these estimates were 63%, 12% and 16% respectively and combined these totalled 91%. Although survival estimates will alter when TIM and tag loss are incorporated into the CJS models, these would suggest that natural mortality is low which is consistent with the stock assessment models used for this species. Further research incorporating TIM and tag loss into the estimation of survival is required given the magnitude of these estimates found in this study.

6.4 Survey design

In this study I have demonstrated the need to consider sample design in developing tagging studies as well as the need to consider factors influencing the fate of tagged lobsters (tag induced mortality and tag loss). Ideally it would be appropriate to design a tagging survey to achieve all of these objectives (survival probability, TIM, tag loss). The number of lobsters required to provide precise and accurate estimates of the required values varies for each of the objectives and studies need to focus on the maximum number needed. For example, using the results of this study for southern rock lobster a five year study is required. In the initial survey at least 1000 lobsters should be double tagged to estimate tag loss rate followed by

two other tagging surveys where at least 400 trap lifts are undertaken to estimate tag induced mortality proportion and a further two recapture events (fourth and fifth survey) is required to estimate survival probability. Such a design should optimise outcomes while also being cost-effective.

Combined studies that target the objectives of the study and provide for additional information necessary in interpreting parameter estimates should be considered as normal practice in designing tagging projects for marine animals where estimates are required for the management of marine resources. This thesis brings together a series of methods that can be used to develop an optimal design providing prior knowledge on catch rates is available. While estimation of the proportion of lobsters dying from tagging (TIM) and the rate of tag loss require greater effort and number of animals to be tagged, these estimates are likely to be relatively consistent over time and may only need to be estimated at less regular intervals if longer-term tagging studies are envisaged.

6.5 Future Work

Although, a 12-year tagging data-set was used for estimating survival probability, there were some restrictions to use the whole data-set in this study. As individual surveys were originally designed for different projects, number of traps and trapping-days were varied among the surveys. This issue placed some restrictions on using the whole data-set. For example, only one and three out of 10 three-survey designs provided enough data to estimate tag induced mortality proportion for females and males, separately.

Further research that should be undertaken include:

- 1) The results of this study were conducted on lobsters in a no-fishing area and while I suggest that they can be applied as a cautionary guideline for studies in a fished area, I believe that greater effort is required to examine cost-effective CMR designs on the species in fished areas where fishing reduces the size structure and density of target species and also removes tagged animals in the population.

- 2) Behaviour of species can affect CMR study design. Unfortunately, there were limited sample sizes to look at the effect of different size groups of lobsters. For example, it has been shown that there is a size hierarchy in catchability of lobsters (Frusher and Hoenig, 2001a; Ihde *et al.*, 2006). Further work on understanding the impacts of this behavioural trait on survival and how it might bias survival probability estimates would be worthwhile. Developing a model using size as a co-variate based on increased numbers of small and medium sized lobsters (males) and small and large lobsters (females) should be undertaken.
- 3) There are a range of environmental factors that potentially impact catchability and thus CMR outcomes. These can include water temperature, lunar cycle (moon phase) and swell. Recent research showed that the East Australian Current (EAC) is expanding southwards which is expected to impact recapture probability of lobsters. It has been shown that moon phase impacts daily catch rates of western rock lobster, but it is unclear how lunar cycle (moon phase) affects the catch rate of lobster in the Crayfish Point Scientific Reserve where the maximum depth is 15 m.

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