Toward Adaptive Management of Parera (*Anas superciliosa*) and Mallard (*A. platyrhynchos*) Duck in New Zealand.

A Thesis presented in fulfilment of the requirements for the degree
of

Masters of Science
In
Conservation Biology

At Massey University, Manawatu,
New Zealand

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2012

Abstract

Wildlife exploitation is encumbered with uncertainty. To ensure sustainability of wildlife populations managers must understand the consequences of, and account for, uncertainty in their decisions. This is most pertinent if the goal is to optimise or maximise the harvest or take.

Uncertainty can be separated into four main categories: environmental variation, partial management control, structural uncertainty (e.g., density dependence) and partial observability. This thesis examines the first three categories in the context of mallard (*Anas platyrhynchos*) and parera (grey duck, *A. superciliosa*) harvest in New Zealand, and specifically addresses sustainable and maximum annual mallard harvest.

A simple heuristic harvest model is proposed to represent a population subject to a seasonal annual harvest. The heuristic model is then converted into a series of quantitative models that can be used to predict the effect of regulations on hunter behaviour (partial management control). Specifically, how regulations may affect hunter effort (hours hunted) and the consequences of hunter effort on, harvest rates, survival, and productivity. Survival and productivity were further evaluated as a function of post-harvest population size (structural uncertainty). Harvest rate, survival, and productivity data were derived from 22,500 (1,024 recaptures; 3100 recoveries) mallard and parera banded from 1997 to 2009 in the Eastern and Hawke's Bay Fish and Game Regions and a telemetry study of 46 mallard in the Eastern Region. Harvest data and reporting rate estimates were derived from a randomised hunter survey over the study period.

In the Eastern Region hunter effort explained changes in survival better than any of the other candidate models ($w_i = 0.851$). In the Hawke's Bay changes in survival was explained by changes in season length ($w_i = 0.334$), hunter effort ($\Delta QAIC_c = 0.739$; $w_i = 0.231$), and spring temperature in the year of banding (*SpcT*) ($\Delta QAIC_c = 0.1.53$; $w_i = 0.155$). Correlation of harvest rates and effort approached significance (P=0.053) in the Eastern Region for adults only while in the Hawke's Bay data there was no relationship. This was assumed a consequence of reporting rate confounding harvest rate estimates as correlation between hunter effort and harvest was good in both Eastern (R=0.85, $t_{(10)}$=5.3193, P<0.001) and Hawke's Bay (R=0.76, $t_{(8)} = 3.3878$, $P = 0.0095$).

A deterministic model was developed (from the quantitative models), to maximise annual harvest subject to the criteria that harvest should not compromise the ability to maximise the following season's harvest. The performance of the quantitative models was validated using a partially stochastic model to simulate harvest. Harvest simulations were used to predict 2010 (outside of the study period) harvest (41,549 mallard and parera; SE=3,552) in the Eastern Fish and Game Region. Simulations predicted harvest accurately (42,045; SE=1,992). Simulations indicated that mallard harvest was not sustainable over a 10 year period when juvenile female: adult female ratios ≤ 0.8 when constrained by Eastern Regions regulation set (season length 30 to 71 days). When productivity increased (≥ 0.95 juvenile female: adult female) long term harvest was viable under the most relaxed season constraint (71 days). This has important implications when managing breeding habitat.

It was proposed that managing populations within similar climate zones would reduce environmental uncertainty. Survival of mallard and parera were analyzed using a set of linear climate covariate models fitted to data from 91,500 mallard and parera banded throughout New Zealand (1969–2009). Climate explained changes in survival better than or was comparable to the alternate candidate models in 11 of 17 data sets.

The quantitative models in this thesis provide a platform for Fish and Game managers to initiate an adaptive management approach to mallard and parera harvest management in New Zealand.

Should Fish and Game wish to review current mallard and parera management areas, establishing management units on homogenous climate zones would contribute to creating a good management system.

Acknowledgements

I would like to acknowledge and thank my supervisor, Doug Armstrong who had to read multiple drafts of this thesis from beginning to end! New Zealand Fish and Game for their contribution towards my study fees. Eastern Fish and Game Council who provided the opportunity, and Eastern staff who assisted with the banding and unselfishly covered for me in my quest to get this thesis finished. To David Klee from Auckland Waikato Fish and Game who collaborated with me on the telemetry work. I would like to thank Wildlife Service, Acclimatisation Societies, and Staff at Hawke's Bay Fish and Game who helped with, and undertook, banding in other regions. To the late Rod Cossie who facilitated access to the Department of Conservation banding records. Richard Barker who took time out to tutor me in Program MARK, and assist with many of my inane questions. I would also like to thank Richard for his support and assistance with statistical problems over the years but most importantly for his humour. I would like to make special mention of Murray Williams for his encouragement to write this thesis but more importantly for his inspiration and mentoring over the years. To my parents who have always shown support and encouragement.

Most importantly I would like to thank my wife Karen for her eternal optimism, unwavering support, and encouragement, and to Joshua and Emma who missed out on weekends with their dad while I wrote this thesis.

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Chapter 1

Mallard and Parera (Grey Duck) Management in New Zealand

Introduction

The relationship between harvest regulation and sustainable, or optimal harvest, is fraught with uncertainty. In the interest of their clients and the exploited wildlife population, managers need to develop robust (statistically and politically defensible) methods of measuring the effects of their management on the status of the population of interest. Such measurement is necessary in order that regulation strategies maximise harvest while ensuring the long term wellbeing of the exploited wildlife population.

Unfortunately statistically ideal study designs are not usually practicable because they require politically unacceptable manipulation of harvest levels. To deal with this impracticability the traditional control and treatment experimental design has undergone a paradigm shift in some areas of resource management to an adaptive management (AM) protocol. Surprisingly there are few examples of AM in population management (Armstrong et al., 2007) despite its obvious benefits such as transparency of process, assistance to management, and progression of knowledge.

Adaptive management is designed to assist managers learn about complex ecological systems by monitoring the results of a suite of management initiatives (Gregory et al., 2006) while accounting for uncertainty. The key features of an adaptive approach are: (1) the recognition of competing hypothesis about biological process; (2) the measurement of uncertainty for each hypothesis; (3) incorporation of these uncertainty measures into a decision-making process, and; (4) the updating of the uncertainty measures through time (Williams, 1997).

Williams et al. (1996) nicely categorise uncertainty associated with wildlife exploitation into four areas:

1. Environmental variation (precipitation for instance);
2. Structural uncertainty (lack of knowledge of the underlying biological mechanisms for example competing theories on the effects of harvest on survival; (Anderson and Burnham, 1976, Burham and Anderson, 1984)).
3. Partial observability of population status and population change (e.g. (Barker and Sauer, 1992, Francis et al., 1998, Nichols et al., 2000, Pagano and Arnold, 2009a, Pollock et al., 2002);
4. Partial controllability (the effect of harvest regulations on harvest rates; (Johnson et al., 1997, Williams and Nichols, 2001).

In the harvest scenario it is imperative that managers develop an understanding of how harvest regulations affect state variables[1] (partial controllability) and how the population responds (structural uncertainty) at different population sizes (e.g. density dependence). Models can be developed to explain competing hypotheses, with their relative support adjusted based on how accurately they predict outcomes (Conn and Kendall, 2004). Where theory is contentious the AM approach allows individuals or organisations to submit competing testable hypotheses within a transparent process. Perhaps one of the biggest advantages of AM is that it provides a framework for thinking rigorously about the system, the benefits of management and, what needs to be monitored (McCarthy et al., 2012).

AM however, is not without its problems. One of the key ingredients of AM is the allocation of different treatments (in the case of harvest; regulations) to different locations. Sutherland (2001) highlights a problem with this approach in that exploiters are sensitive to different rules (regulations) in different areas particularly when regulation constraint appears unnecessary. Nevertheless the alternatives are less favourable such as zero harvest or, decision making in a vacuum of knowledge.

Waterfowl harvest in North America is a case in point, where the harvest regulation process has seen the adoption of Adaptive Harvest Management (Nichols et al., 1995a, Williams et al., 1996, Johnson and Case, 2000, Nichols, 2000, Williams, 2000).

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[1] State variables include population size, survival rate, and productivity.

In 1995 North American wildlife managers introduced an adaptive management approach to setting harvest regulations of the mid-continent mallard (*Anas platyrhynchos*) (Nichols, 2000, Nichols et al., 2007). This was due to a number of factors: (1) recognition that the effect of environmental influences and socio-economic factors were difficult to predict (Williams, 2000); (2) poor relationship between regulations and harvest rates (Williams, 2000, Johnson and Case, 2000); (3) an inability to agree about resource status and proposed regulations (Humburg et al., 2000) and; (4) a re-evaluation of historic data by Anderson and Burnham (1976) that indicated little relationship between harvest mortality and survival (Nichols, 2000).

Game Bird Management in New Zealand

New Zealand game birds include a number of introduced *Galliformes* and a mix of introduced and native *Anseriformes* excluding teal (see Schedule 3 Wildlife Act 1953). Mallard (*Anas platyrhynchos*) are the most numerous and widespread waterfowl in New Zealand (Heather and Robertson, 1996) and in combination with the grey or parera (*A. superciliosa*) make up the majority of the hunters bag (Barker, 2009) (see Box 1.1) and consequently provide the focus for this Thesis.

Johnson and Williams (1999) wrote in respect to waterfowl harvest prior to 1995 that *"North America, for all its success, historically has had several shortcomings, including a lack of well-defined objectives, a failure to account for uncertain management outcomes, and inefficient use of harvest regulations to understand the effects of management."* New Zealand has similar shortcomings and is generally in an inferior state of affairs. Management goals have been articulated in New Zealand Sports Fish and Game Management Plans but there is generally no consistent periodic information about national game bird populations or goal related variables and no framework for understanding the effects of management and harvest regulations on game bird populations. Nevertheless, New Zealand game bird management has the statutory direction, the Fish and Game organisation has political flexibility, geographically the country is small, and there is a lack of confounding migratory issues (such as in Europe and North America), all of which provide opportunities to overcome the current shortcomings. In addition, and most importantly, there is a desire amongst waterfowl managers to improve the way game birds are monitored and managed (Kaiteriteri Fish and Game workshop 2009).

Box 1.1. Mallard and Parera in NZ; A story of changing fortunes

Parera were once the primary game bird for both Maori (Best, 1942, Turbott, 1967) and European (Caithness, 1982a, McDowall, 1994) but are now considered critically endangered (Miskelly et al., 2008). In many parts of the country they could be considered an unavoidable by-catch of the mallard harvest. Ironically limiting the mallard harvest to reduce this by-catch may in fact expedite the pareras demise through increasing the prevalence of hybridisation and introgression with the Mallard.

The mallard was first introduced into New Zealand from Britain in 1867 (Williams, 1981, Marchant and Higgins, 1991, McDowall, 1994). The initial releases were not particularly successful and it wasn't until the 1930s, when mallard from the USA were imported, that they multiplied and spread widely (Williams, 1981, McDowall, 1994)[1].

Caithness (1982b) reports a steady ascendency of the mallard over parera in the hunters bag 1968-1980. Currently parera comprise about 5% of the total waterfowl harvest (R. J. Barker University of Otago, unpublished data.).

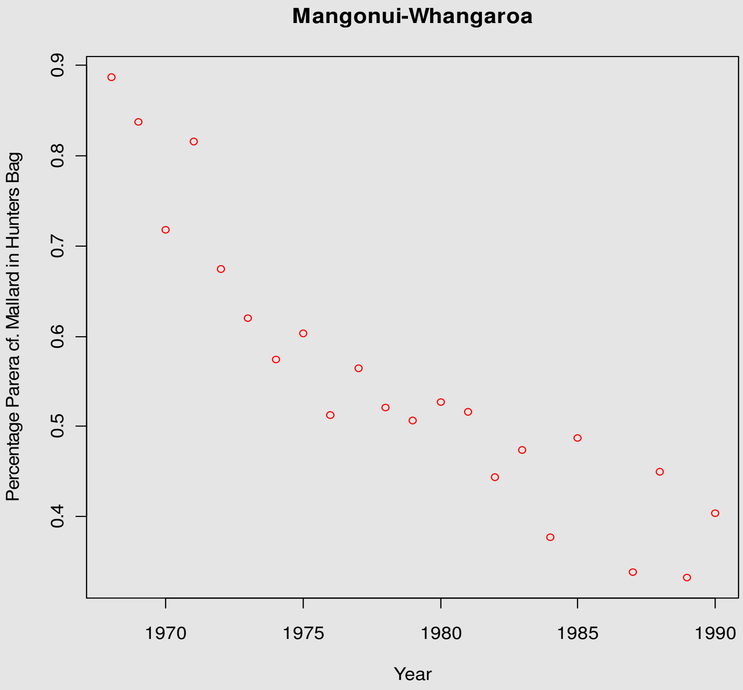

Figure 1.1: Proportion of Parera out of the total bag of parera, mallard and hybrids in the Mangonui – Whangaroa Acclimatisation District (Northland Fish and Game), one of the last strongholds of the parera (extrapolated from Caithness Waterfowl Diaries; unpubl. Wildlife Service reports).

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[1] A recent genetic study suggests the US sourced birds may have been originally of UK-source (Guay et al., In Press).

An examination of average parera and mallard harvest from historic hunter diaries (Figure 1.1) exemplifies the changing status of mallard and parera at a regional level.

It wasn't until relatively recently that hybridisation of the parera and mallard was recognised as an issue (Gillespie, 1985). To explain the demise of the parera, Williams and Basse (2006) tendered three hypotheses:

1. Demographic and competitive ascendancy (with the mallard);
2. Genetic assimilation (with the mallard) and;
3. Habitat destruction and disturbance.

They conclude that all factors have played a role; the mallard is more fecund, bigger, and has a higher survival rate. The contribution of hybridisation to their demise was not so clear but the genetic integrity has been compromised. Loss of wild wetlands has had an impact on the species but does not explain the displacement by the mallard.

In New Zealand there are two subspecies of Parera *A. superciliosa superciliosa* and *A superciliosa rogersi* (Marchant and Higgins, 1991, Rhymer et al., 2002, Rhymer et al., 1994) the New Zealand grey or parera and the Australian black duck respectively. The problem of genetic assimilation is further complicated as the sub-speciation *Anas superciliosa superciliosa* and *A. superciliosa rogersi* may not reflect the actual intraspecific diversity (Rhymer et al., 2002). In an examination of Mitochondrial DNA (mDNA) Rhymer et al. (2002) discovered a distinct haplotype, primarily confined to parera in the North Island of New Zealand. The other haplotype was found throughout New Zealand and Australia. Analysis of the genetic rate of mDNA divergence between the two distinct haplotype groups suggests that they separated 830,000 years ago, which is about the same as the estimated separation between mallard and parera (*A. superciliosa)* about 1Mya (Rhymer et al., 2002).

The fourth possibility of disease to explain the demise of the Parera appears to have received little consideration in the literature. McDowall (1969) makes reference to introduced birds with their diseases probably initiating the decline and extinction of New Zealand Birds. Both Tennyson and Martinson (2006) and Worthy and Holdaway (2002) however, hold a contrary view and suggest the evidence is against disease as a cause of extinctions. The current lack of evidence one-way or the other makes this issue un-resolvable.

Nevertheless Conroy et al. (2002) discuss the role of disease in the ascendancy of the mallard over the North American black duck (*Anas rupribes*). They report a mortality of 86% black duck vs. 0% for mallards in an outbreak of duck virus enteritis (herpes virus) in two captive game farm flocks, and note that disease has resulted in severe morbidity and mortality at times but believe that it is not limiting the population.

Stanislawek et al. (2002) have looked at paramyxoviruses and influenza viruses in wild mallard in New Zealand as part of the national H1N1 (Bird Flu) monitoring program. Isolates, however, were reported only coming from mallard (i.e. the study does not appear to differentiate between the mallard and the parera, despite the sample including parera and their hybrids). An interesting observation is the number of reported pathogenic isolates decreases from north to south, coinciding with parera distribution (sample sites do not include the West Coast). This is, however, believed to be a function of the timing of the adult moult being later (most of the isolates are from adults) in the South Island compared with the northern regions (Woldek Stanislawek pers. com.).

The distinct genetic divergence within the species, hybridisation with the mallard duck, and its rapid demise has put Fish and Game managers in a quandary. The statutory definition further exasperates the situation.

The First Schedule of the Wildlife Act (1953) defines grey (parera) and mallard duck as:

- Grey duck (*Anas superciliosa*) and any cross of that species with any other species, variety, or kind of duck (except on Chatham Islands).
- Mallard duck (*Anas platyrhynchos*) and any cross of that species with any other species, variety, or kind of duck (except on Chatham Islands).

It is not clear at which point a parera becomes a mallard or a cross thereof and vice versa.

Uncertainty aside, sustainable management of parera looks dubious. Banning hunting may not be feasible while mallard continue to be harvested due to the difficulty in differentiating parera and mallard and their hybrids in the field. As previously mentioned the option of banning all dabbling duck harvest apart from being politically unacceptable is also not a solution as it will probably expedite introgression with the mallard.

New Zealand game bird management is vested with Fish and Game Councils. Fish and Game Councils are crown entities comprising a council of elected hunters and anglers. There are 12 independent regions and one National Council funded solely from the sale of licence holders. Regional boundaries are politically and catchment based. Functions and responsibilities are dictated by statute and annual management guided by 10 year Sports Fish and Game Management Plans.

Statutory direction includes a requirement to monitor (Section 26Q Conservation Law Reform Act 1990 Act) and for the National Council to coordinate management between regions. Even so there is little or no coordination of management or monitoring or any scientific advisory capacity. Many regions have no monitoring of the primary game bird (the mallard duck) nor are there processes or policy for setting regulations based on mallard population status. That is in most instances regulation setting is *ad hoc*.

Many of the Sports Fish and Game Management Plans do include goals such as "*To manage sustainable populations of sports fish and game bird species for recreational* *harvest."* (e.g. Eastern Region Sports Fish and Game Management Plan 2000) but there is no process to determine sustainability. What's more it is likely that sustainability under represents the intent of Councils compared with some goal of maximum sustainable yield (MSY) or maximum cumulative harvest. The coordinating body, New Zealand Fish and Game have gone some way towards meeting their responsibilities with the development of a Research Strategy that includes objectives to monitor the primary game bird the mallard but there is no documented method on how this is to be done or how the information should be utilised in harvest strategies.

Conservation Biology and Harvest: The introduction to this Conservation Biology thesis requires some comment about developing a harvest strategy around the parera, listed as critically endangered (Miskelly et al., 2008) and mallard duck an introduced species. More recently within Fish and Game circles, mallard and parera are collectively referred to as greylard due to the preponderance of hybrids. Differentiating them at a harvest or management level is now very difficult. To date the debate about their continued harvest has been minimal due to many of the reasons I discuss in Box 1.1. Consequently reference to mallards in this text usually includes mallard hybrids and parera unless specifically differentiated.

The development of an AM protocol for grey or parera duck (a native species) and mallard duck, introduced wildlife, under the guise of a Conservation Biology thesis may appear at odds with the modern day ethos associated with endangered and threatened native wildlife (for example see Soulé (1985)). I would argue however, that Conservation Biology is about sustainable or wise use of the resource and the concepts are as applicable to introduced as native species. Aldo Leopold (1930) wrote that the prevalent notion of early conservation efforts was to enact laws to restrict hunting and thereby string out the remnant wildlife supplies and make them last longer. Conservation Biology includes developing strategies to harvest or take species where the take is intentional or otherwise. As Briskie (2006) comments there are lessons to be learned from the study of introduced species. In a forum "Conservation of Wild Living Resources" Levin (1996) writes "it is imperative to develop principles that synthesize what has been learned from individual case studies, and allow the application of those principles to new situations." Many of the concepts and lessons learned in mallard harvest crossover to harvest management in overt take situations such as recreational and commercial fisheries and possibly management of accidental by-catch. Furthermore a better understanding of mallard duck in New Zealand may also identify opportunities to improve or build on existing native waterfowl management.

Thesis Objectives and Format

Objectives

1. To attain a better understanding of the effects of mallard and parera management on goal related variables, primarily, the effects of harvest regulations on harvest (partial management control).
2. To ascertain evidence of density dependence on mallard recruitment and survival (structural uncertainty).
3. To formulate harvest regulation strategies (using results from 1 & 2) that will optimise harvest over the long term in the Eastern Region (Hawke's Bay Fish and Game Council have implemented a fixed 8 week season policy).
4. To establish if environmental stochasticity explains temporal and spatial variations in survival rates of mallard and parera across New Zealand (in part environmental uncertainty). If so there is a sound argument to manage populations within similar climate zones.
5. To use the results of 1, 2, and 4 as a platform to launch an adaptive management approach to mallard and parera management in New Zealand.

Format

Chapter 2: A simple heuristic harvest model is constructed which is then deconstructed into two areas of uncertainty, - structural and partial management control. These two areas of uncertainty are investigated using empirical data from 22,500 mallard and parera banded from 1997 – 2009 in the Eastern and Hawke's Bay Fish and Game regions, results from a preliminary telemetry study conducted in the Eastern Region and; fortnightly random surveys of hunters during the waterfowl season over the same period. The findings were then amalgamated to develop a simple simulation model to determine the optimal harvest strategy incorporating structural and partial control uncertainties.

The format for this chapter follows the decision theory approach suggested by Possingham et al. (2001) Harvest objectives are established, management options evaluated, and then this large ecologically complex problem is broken down into smaller manageable problems that can be conceptualised through simple empirical models. These simple models provide the foundation to an adaptive management approach.

Chapter 3: Monitoring the consequences of management and ecological processes is very expensive. Rather than 12 independent Fish & Game Councils conducting their own independent monitoring programs it is better to rationalise this effort. If it can be shown that mallard and parera population dynamics can be explained by climatic covariates then there is a reasonable argument to manage these species within similar climate zones. This will mitigate environmental uncertainty and provide additional statistical power (by reducing environmental uncertainty) within an adaptive management framework.

Spatial and temporal changes in survival rates is explored using historic and contemporary data from 91,500 mallard and parera banded in discrete geological areas throughout New Zealand.

Chapter 2

Optimal harvest strategies utilising an adaptive management framework.

Introduction

Fish and Game New Zealand has as an objective, "sustainable harvest of game birds", which include mallard duck, but no strategy to implement sustainability or structured nationwide measure of state variables associated with achieving their objective.

In this chapter I suggest an objective of sustainability is understated. Some form of maximum cumulative harvest is probably more appropriate than just that of sustainable management. I look at different harvesting options and identify the most appropriate method of achieving the objective of optimal harvest.

In a stepwise fashion harvest management processes are deconstructed into smaller components. Retrospective analysis is used to evaluate the effects of management on harvest in Eastern and Hawke's Bay Fish and Game Regions. The results of this analysis are then used to run a simulation to determine a harvest strategy that will optimise harvest in the Eastern Region.

Management Goals and Objectives

Fish and Game New Zealand have articulated mallard and parera harvest goals ("sustainable management of sports fish and game birds for recreational harvest") and objectives ("adequate information relevant to the region's needs for effective management of the sports fish and game bird resource" and "develop techniques to determine the size of late summer/autumn mallard population") but as Johnson et al. (1997) point out many managers fail to recognise that a goal to provide sustainable hunting opportunities is in itself, not sufficient for defining a unique harvest strategy. The organisation has not identified methods to achieve their objectives nor clearly articulated how the objectives relate to the goal of sustainable management of game birds for recreational harvest. Nor has the organisation identified procedures and processes to incorporate this knowledge into the regulatory process of harvest management.

It is important that goals and objectives are explicit (Pollock et al., 2002, Possingham et al., 2001, Yoccoz et al., 2001). The Sports Fish and Game Management Plan goal of "sustainable management of game birds for recreational harvest" is relatively non-specific and loose. Providing a harvest of 10 mallard a year would undoubtedly be sustainable, and perhaps could be considered a limited harvest but probably doesn't reflect the intent of Fish and Game Councils. A goal that incorporates the objective to maximise or optimise recreational harvest over time would be more consistent with Councils intention and the interests of hunters. A more specific goal might be: "To maximise the cumulative mallard harvest over time." A key presumption of this goal is that harvest is sustainable otherwise it is not achievable in the long term. Secondly maximisation of the cumulative harvest suggests some form of annual maximisation. A competing objective to the maximum cumulative harvest may be longer season length[1] which could compromise the goal of maximum cumulative harvest, therefore developing an understanding of the consequences of extending season length is important.

Management Options

Newton (1998, Table 14.2) lists 6 different harvesting options:

- Free-for-all (no controls);
- Fixed quota (same number harvested each year);
- Fixed effort (same total effort each year);
- Variable quota (harvest level is dependent on the population size);
- Fixed percentage (same proportion of the population is harvested each year);
- Fixed escapement (same proportion of the population are left to breed at the end of each year).

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[1] Many hunters only hunt the opening weekend so they would like the opportunity to shoot as many ducks on those two days whereas the keen hunters want a long season and may forgo the opportunity to shoot some maximum harvest in order that they may continue hunting over an extended period.

Pros and cons of the options include: Free-for-all risks population collapse or extinction. Fixed quota has inefficiencies (over and under harvest) and are unstable (Sutherland, 2001). Total effort is difficult to control and can also lead to population decline (Newton 1998). Variable quota is sensitive to change in densities, but requires intensive annual monitoring to implement. Fixed percentage has the same disadvantage as variable quota but risks over harvest. Fixed escapement can result in large fluctuations in harvest with no harvest in some years and also involves intensive monitoring (Newton, 1998).

Variable quota is the only option that affords the opportunity to attain the maximum sustainable yield (MSY). The MSY is a single maximum constant yield that can be taken from a population at equilibrium (Hunter and Runge, 2004). In many instances however uncertainty in wildlife management (population stochasticity, incomplete control of harvest rates and, effects of management) (Johnson et al., 1997), precludes attaining the MSY (Strickland et al., 1996) or risks overharvest (Clark, 1996). History shows there are many examples in resource management where harvest has exceeded some sustainable threshold (Ludwig, 2001) resulting in population collapse (Reynolds et al., 2001, Wade, 2001, Newton, 1998) or extinctions (Mace and Reynolds, 2001). Furthermore Hunter and Runge (2004) argue that the MSY policy is inefficient away from the equilibrium[1], which is usually the case in a stochastic environment. They suggest a state dependent strategy (SDS) performs better when the population is not at equilibrium and is on par with MSY when it is. SDS is a variable quota approach with a set of management decisions conditional on the state of the population (e.g. population size).

Mallard harvest in New Zealand is distinctly different to nearly any other harvest or take regime. There is no commercial value in the size of the harvest (it is illegal to sell any game birds) hence no desire or foundation to maximise the economic yield. There are limited benefits in a constant effort paradigm as there is no debt servicing or ongoing maintenance of plant. Constant effort would be difficult to achieve as the

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[1] Under a set carrying capacity the population will increase until the carrying capacity is reached. Density dependence above this level results in mortality exceeding births whence the population returns to carrying capacity or equilibrium. If the population falls below equilibrium, density dependant mortality decreases and density dependant productivity increases and the population grows until equilibrium is reached.

number of hunters that partake in any one year is not known prior to the season. Finally, with the mallard, there are no conservation values (minimum population size) or concerns that might be associated with the harvest of native or endemic species. In fact there is a segment of society that would happily see them exploited to extinction. Nevertheless grey duck harvest which is inherent in any mallard harvest is questionable (see Box 1.1).

For the majority of the 12 Fish and Game Regions the *status quo* is one of fixed regulations (the same set of regulations every year). Anderson (1975a) points out fixed regulations entail inefficiencies associated with under and over harvest, and ultimately runs the risk of over exploitation (nevertheless this approach has stood the test of time). For the remaining regions regulations are based on population size or trends however there is no stratagem on how the regulations relate to sustainability or cumulative harvest policy. A SDS involves the least risk and is consistent with the goal of maximising cumulative harvest.

State Dependent Strategy

Requirements of an optimal harvest strategy are:

1. An objective function describing goals of management;
2. A set of regulatory options and;
3. A mathematical description of the management system (Johnson et al., 1997, Williams and Nichols, 2001).

Objective Function

Management objectives expressed as an objective function (a mathematical expression of the objective) (Williams, 1982, Williams, 1997) help determine monitoring requirements (Yoccoz et al., 2001) and allow optimisation theory (Intriligator, 1971, Williams, 1982, Williams et al., 2002). The set of regulatory options are the control variables, and a mathematical description of the system can be derived as a set of predictive models incorporating state variables.

A point of difference between New Zealand and North American harvest management strategies is that the objective function in North America includes a conservation component in the maximum cumulative harvest goal. If the breeding population falls below the goal expressed in the North American Waterfowl Management Plan a devaluing factor reduces the value of the harvest (Nichols et al., 2007). In New Zealand there is no equivalent minimum population requirement.

Without constraint therefore the goal of maximum cumulative harvest or optimal yield, simply becomes:

$$\max \sum_{t=1}^{T} H_t \qquad [2.1]$$

Where H_t is harvest in year t and T is some time in the future. In other words the objective is to maximise harvest over some period t to T years. Alternatively hunters may determine that they would prefer to hunt a minimum (and maybe a maximum) of some season length at the expense of cumulative harvest. The goal may then be expressed subject to the constraint that season length is of some minimum and maximum period for example $30 \leq SL \leq 71$.

Maximising the cumulative harvest over time requires an ability to manipulate population levels through harvest management. In the short term this goal may be too ambitious, a more achievable goal may be to maximise annual harvest ($H_{(t)\max}$) while ensuring sustainability.

Regulation Option Set

Williams and Nichols (2001) represent the vector of population states as:

$$\underline{x}_t = \underline{G}(\underline{x}_{t-1}\underline{a}_t z_{t-1}) \qquad [2.2]$$

Where $\underline{G}$ is the vector of models predicting outcomes $\underline{x}_t$, regulatory options A_t, and random variation z_{t-1}. They attribute a value $V(A_t|\underline{x}_t)$ (the value of the regulatory option given the state of the population) as accumulated utilities over some time frame:

$$V(A_t|\underline{x}_t) = E\{\sum_{\mathrm{T}=t}^{T}(R(\underline{a}_{\mathrm{T}}|x_{\mathrm{T}})|x_t)\} \qquad [2.3]$$

Where the expectation is a function of environmental uncertainty and partial controllability and R represents the return or harvest utility. With prudent utilisation of the regulatory option set $V(A_t|\underline{x}_t)$ can be maximised using a backwards iteration algorithm (Williams and Nichols, 2001).

Currently in the Eastern Region, a set of regulatory options have been implemented depending on the estimated size of the population. If the population is above the top threshold population size a relaxed set of season regulations are applied. When the population is below this level but above the bottom population size threshold an intermediate set of regulations apply and below this a restricted set. The threshold levels were set in a relatively arbitrary manner with little understanding of the consequences on long term harvest rates and population sizes. This threshold approach does not maximise long term harvest nor does it incorporate the constraint of long term mallard sustainability. Nevertheless, the threshold approach is easily understood, and easy to implement, should Eastern Region governors wish to persevere with this approach I investigate the harvest value of different threshold strategies given different population states and determine a set of sustainable strategies.

To maximise harvest I formulate an alternative approach to Williams and Nichols (2001) whereby I determine the optimal amount of hunter effort required to maximise harvest subject to sustainability constraints and then determine the corresponding regulatory options set that will achieve the optimal effort.

The next step is to devise a mathematical description of the management system and formulate models that adequately explain population dynamics under different harvest strategies.

Management System

The management system can be portrayed as a simple heuristic model of the harvested population (Figure 2.1). The heuristic harvest model can be decomposed into two parts, structural uncertainty and, partial management control. Partial management control comprises the effects of regulations on, hunter effort, harvest rates, and survival. While structural uncertainty consists of density dependent, survival, and recruitment.

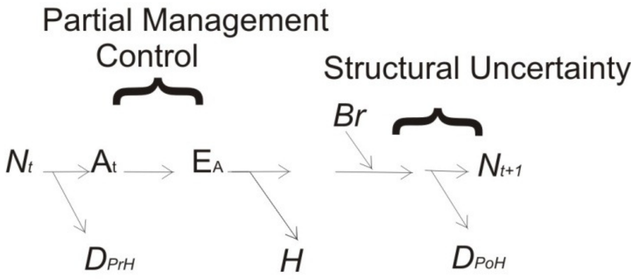

Figure 2.1. A simple heuristic harvest model where N_t is the assessed population in year t, A_t season regulations, E hunter effort (hours hunted waterfowl given the season regulations A), H harvest, Br births, D_{PrH} deaths pre-harvest and, D_{PoH} deaths post-harvest.

The key components of this model to be understood are:

1. What is the effect of regulations on hunter behaviour, specifically the consequence of altering season length on hunter effort (hours hunted)?
2. What influence does effort have on harvest rates?
3. What is the effect of harvest rates on annual survival?
4. Do long hunting seasons interfere with breeding?
5. Does density dependence apply to survival and productivity at normal harvest population levels?
6. What is the size and composition of the population?

The following investigates solutions to these questions through assessing the performance of quantitative models and cumulates in utilising the best models to derive a stochastic representation of the heuristic model which is tested on data (harvest survey) outside of the study period.

Partial Management Control

Regulations and Harvest Rates: A set of season regulations (harvest constraints; A_t) are implemented dependent on the state of the population (x_t). For example when the population (N_t) is low or declining, restrictive regulations (a_r) may be introduced. Season regulations in waterfowl harvest are usually confined to daily bag limits (BL) and season length (SL), but harvest may also be constrained through restrictions on permitted gear (e.g. only two shells in the shotgun) and techniques (e.g. only birds in flight may be shot).

Season length constrains the total available hunting period. The assumption is that hunters will hunt for fewer hours in shorter seasons; however anecdotal evidence suggests hunters may be more inclined to hunt more frequently in shorter seasons. Effort (E) may also be a function of weather conditions and economic conditions (e.g. in years of high unemployment hunters may be less inclined to buy a licence and go hunting, alternatively they might just be less compliant i.e. not buy a licence but still hunt). Success may also be important. If hunters are successful they may be more inclined to continue hunting. Partial control of season regulations over total effort could be expressed as the function:

$$E \cong f(A, WC, EC, \frac{d}{hr}, Hu) \qquad [2.4]$$

Where E= total effort, A= season regulations, WC =weather conditions, EC =economic conditions, $\frac{d}{hr}$ = ducks per hour and Hu= the number of hunters.

Reducing total effort (E) through regulations should reduce harvest rate (h):

$$h = f(E) \qquad [2.5]$$

and in turn will affect total harvest (H):

$$H_t = N_t h_t \qquad [2.6]$$

An increased effort will increase harvest rate but this relationship may not be linear. In wet years for example, ducks may be spread more widely and require more effort for every kill. The evidence however of a relation between mallard harvest regulations and harvest rates (Conroy et al., 2005, Giudice, 2003, Johnson and Clinton, 1996, Otis, 2004, Sedinger and Rexstad, 1994) and harvest rates and survival (Nichols and Hines, 1983) is not clear.

Despite the importance of mallard harvest regulations and their influence on hunter effort I was unable to find any research on this aspect of harvest control, nor in relation to hunter effort and mallard survival.

Regulations and Recruitment: A common complaint amongst New Zealand hunters is that the shooting season is too long as reputedly the ducks have eggs in them by the end of June (Caithness, 1968-1991). The hunting season that commences in the first weekend in May overlaps with mallard pairing which is initiated in mallards as early as March (Balham, 1952). Laying commences late July early August (Heather and Robertson, 1996, Marchant and Higgins, 1991, Williams, 1981) (see Figure 3.1). In some years broods have been observed as early as April (pers. obs.).

If longer seasons are impacting on productivity in the following breeding season it is important to include this in the harvest model.

Structural Uncertainty

Sustained harvesting depends on depleted populations restoring their size through density dependent increase in per capita birth rate and decreased non-harvest mortality (Newton, 1998). Uncertainty stems from; the relationship between harvest and mortality subsequent to harvest, and secondly post harvest population size and productivity.

Density dependence in waterfowl is not well understood. For large mammals the effects of density dependence tend to be most pronounced near carrying capacity while the opposite holds for animals with life strategies similar to insects and some fish stocks (Fowler, 1981). In the US mallard recruitment rate may be inversely related to the size of the breeding population suggesting recruitment rates may be density dependent (Kaminski and Gluesing, 1987). Pospahala et al. (1974) suggest US mallard recruitment may be related to density dependent factors (availability of nesting habitat) as wells as factors independent of density. Hill (1984) examined mallard density dependence in Kent UK. He/she reports evidence of weak density dependence in duckling mortality and the proportion of nest predation increased as nest density increased. They also reported that overwinter loss was higher in years following a good breeding season. The extent of the overwinter loss buffered fluctuations in juvenile mortality, leading to little change in the size of the breeding population.

The nature of the relationship between waterfowl harvest and post harvest mortality has attracted a reasonable amount of debate and investigation (Burham and Anderson, 1984, Burnham et al., 1984, Nichols et al., 1984, Barker et al., 1991, Smith and Reynolds, 1992, Sedinger and Rexstad, 1994, Johnson et al., 1997). Post-harvest mortality may be additive to harvest mortality or, due to less competition for resources, reduced exposure to disease etc., post-harvest mortality may decrease. The population may completely compensate for the harvest mortality, or some variant where the true relationship may lie somewhere between completely additive and completely compensatory (Nichols et al., 1984).

Anderson and Burnham (1976) and Burnham and Anderson (1984) report the relationship between North American mallard harvest rates and survival may be density dependent whereby post-harvest mortality may decrease when subject to harvest mortality up to some threshold (compensatory mortality). Smith and Reynolds (1992) on more recent data rejected both the compensatory and additive models. They reported that their rejection of the compensatory model was stronger than the rejection of additive model. Sedinger and Rexstad (1994) didn’t agree with some of the findings of Smith and Reynolds (1992) and went on to suggest there is little evidence that restrictive harvest regulations improve survival. Barker et al. (1991) examined compensatory verse additive harvest of parera in New Zealand and rejected the completely compensatory model but were unable to reject the completely additive model.

To assist in resolving this debate with respect to structural uncertainty in the US mid - continent mallard harvest, managers have, within an adaptive harvest framework, formulated four competing models (Johnson et al., 1997, Nichols et al., 2007); two additive and two compensatory survival, with either strongly, or weakly, density dependent recruitment.

Methods

Fish and Game conduct an annual hunter survey. 120 hunters are randomly selected every fortnight during the season and questioned on, number and species shot, of those that are shot how many are not recovered, and hours hunted, for each Fish and Game Region (Eastern and Hawke's Bay; Figure 2.2). At the end of each season hunters were also asked an additional question to determine reporting rate (see Harvest and Harvest Rates, page 29). Harvest survey data was used to evaluate the effect of regulations on hunter effort (see below) form 1997 - 2009.

Recapture and recovery data from 22,500 mallard and parera banded in both regions over the same period was analysed to determine survival (see Survival, page 25) and harvest rates (page 29).

Effort

Total and average effort are analysed in relation to participation (number of licence sales), season regulations (season length and bag limit), hunter success and, weather.

Total annual hours spent hunting waterfowl was defined as a measure of effort. Effort was estimated from the game hunter survey (R. J. Barker University of Otago unpublished data). Not all hunters (land owners[1]) require a licence and not all hunters are compliant (hold a licence when required to). Furthermore, total hours spent waterfowl hunting may extend beyond the mallard season (Canada goose seasons are often longer than the mallard season) but this is generally presumed insignificant.

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[1] Hunters that live and hunt on their own land do not require a licence, this number has variously been estimated at 10% of total hunters, but this estimate is unreliable.

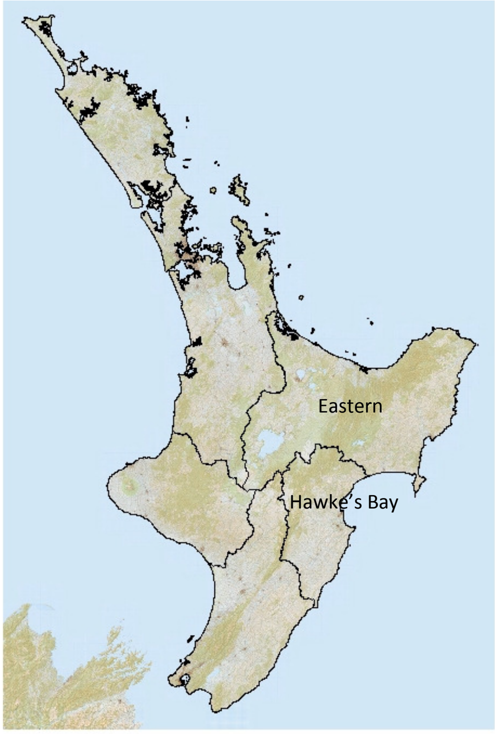

Figure 2.2. Eastern and Hawke's Bay Fish and Game Regions.

Two approaches were used to examine the premise that hunters would exert more effort if they were more successful (converse of the law of diminishing returns (Strickland et al., 1996)). (1) Opening weekend harvest and total days hunted during seasons of the same length; (2) catch rate (mallard per hour hunted waterfowl). It was assumed that a successful opening weekend would encourage hunters to hunt for more days in the remainder of the season and or that hunters would hunt for more days if catch-rate (over the whole season) was higher (success begets success).

Annual variability of licence sales was predicted to be, in part, a function of economic buoyancy. A profile of New Zealand hunter demographics suggests a predominance of blue collar occupations (Brocklesby et al., 1995). It was predicted blue collar workers are more vulnerable to unemployment than other sections of the community and that in times of high unemployment (Statistics NZ; http://www.stats.govt.nz/) hunters would be less likely to purchase a licence or there would be an increase in non-compliance (less licensed hunters). The latter assumption was not tested.

Anecdotal evidence suggests hunters are more successful when the weather is poor and therefore may be tempted to hunt for longer. Conversely hunters may be more incline to hunt when the weather is good.

To examine the above relationships package stats in R (2.11.0) (R Development Core Team, 2005) was used. Pearson' product-moment for correlation; lm for linear models, and nls for non-linear models. Normality of the residuals was tested using Shapiro-Wilk normality test in R.

Multiple models are compared under Akaike's Information Criterion adjusted for low sample size AICc (Burham and Anderson, 2002). If the models were similar the linear model was utilised.

10 linear models were used to investigate average effort. The likelihoods of the normal linear models were calculated:

$$L_i = \left[\frac{1}{(2\pi)^{0.5}}\right]^n \left[\frac{1}{\sigma}\right]^n \exp^{\left[-\frac{1}{2}\sum_{i=1}^{n}\frac{(y_i-\mu_i)^2}{\sigma^2}\right]} \tag*{[2.7]}$$

Where:

$$\mu = E(y)$$

$$\sigma^2 = \frac{(y_i-\mu_i)^2}{n}$$

And log models:

$$L_i = \left[\prod_{i=1}^{n}\frac{1}{y_i}\right]\left[\frac{1}{(2\pi)^{0.5}}\right]^n \left[\frac{1}{\sigma}\right]^n \exp^{\left[-\frac{1}{2}\sum_{i=1}^{n}\frac{(\log(y_{i)}-\theta_i)^2}{\sigma^2}\right]} \tag*{[2.8]}$$

Where:

$$\theta_i = \left[\log(\mu_i) - \frac{\sigma^2}{2}\right]$$

(Burnham and Anderson, 2002).

Model parameters were:- year (*Yr*), total licence sales (Hunters – *Hu*), season length (*SL*), and average climate conditions(winter precipitation – *WR*) and, temperature (winter temperature – *WT*). Climate data were obtained from NIWA's web site (http://cliflo.niwa.co.nz/) for Whakatane (Whakatane AWS - B76995), and Napier (Napier AWS -D96484). The number of parameters in each model included 1 for the intercept, 1 for each of the regression coefficients and 1 for variance (Burnham and Anderson, 2002).

Model estimates and 95% credible intervals[1] for the top ranked model were derived in Program WINBUGs (Spiegelhalter et al., 2003) with non-informative normal priors, 1000 iteration burn-in and 100000 iterations for model parameter estimates.

Survival

Band recapture (*n*=1,024) and recovery data (*n*=3,100) from 22,500 mallard and parera that were banded in the Eastern Fish and Game Region from 1997-2009 and Hawke's Bay Fish and Game Region from 2000-2009. Trapping and banding methodology is described in Appendix A. Birds were banded annually mid - late January (post fledging but prior to the onset of maturation characteristics) during the study period with individually numbered stainless-steel bands. Birds were recorded as either mallard or parera, male or female, and adult or juvenile (hatch year; refer page 153). Recoveries were predominately from harvested birds during the waterfowl season which commences in the first week of May. Recaptures were recorded at subsequent banding periods.

A wide selection of models was investigated to help interpret temporal changes in survival. Interpretation of the candidate models is given in Table 2.1 and includes:

- 8 climate models (refer to Chapter 3 for rational) where survival was constrained to be a linear function of either precipitation or temperature data;
- 3 models where survival was constrained to a linear function of season regulations;
- 1 model where survival was constrained to a linear function of total harvest and;

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[1] With the 95% credible interval there is a 95% chance that the true estimate lies within the interval, whereas 95% confidence limit is a random variable (Link and Barker, 2010) whereby if the data were repeated numerous times 95% of these confidence intervals would encompass the true mean (McCarthy, 2007).

- 1 model survival was constrained to a linear function of either total or average effort;
- The remainder were nested versions of the most general (global) model.

All the starting linear models assumed a sex-effect with the primary covariate (e.g. sex*precipitation). Post the initial run, the equivalent additive model of the top ranked model was run to check the legitimacy of including the sex effect.

Table 2.1: Candidate model set and Explanation

Model	Explanation
$\{S_{(g*t)}P_{(g*t)}r_{(g*t)}F_{(g*t)}\}$	Global Model: Survival (S), recapture (P), conditional reporting (r) and, fidelity (F) are group (cohort; adult female, adult male, juvenile female, juvenile male banded in the same year) and time dependent.
$\{S_{(g)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Group Model: Survival (S) is group dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile (1)) and time dependent.
$\{S_{(sex)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Sex Model: Survival (S) is sex dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile at first capture(1)) and time dependent.
$\{S_{(sex+age)*t}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Sex and age model: Survival (S) is sex, age (adult or juvenile at first capture), and time dependent
$\{S_{(g*t)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Time dependent group Model: Survival (S) is group and time dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile).
$\{S_{(.)}P_{(.)}r_{(.)}F_{(.)}\}$	Fully constrained dot Model: Survival (S), recapture (P), conditional reporting (r) and, fidelity (F) are constant over time (dot model).
$\{S_{(.)}P_{(g*t)}r_{(g*t)}F_{(g*t)}\}$	Survival dot Model: Survival is constant over time. The rest of the parameters are group and time dependent.
$\{S_{(sex*cc+age})P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Climate Models: This model represents 8 different models where survival (S) is sex and age dependent and is constrained by one of the climate covariates (cc); winter rain (*WR*); spring rain in the preceding year (*SppR*); spring rain in the current year (*SpcR*); summer rain (*SuR*); spring temperature in the preceding year (*SppT*); spring temperature in the current banding year (*SpcT*); winter temperature (*WT*) and summer temperature (*SuT*).
$\{S_{(sex*A+age})P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Season Regulation Model: Season regulations (*A*) are either season length (*SL*), or Bag limit (*BL*).
$\{S_{(sex*H+age})P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Harvest Model. *H* represents annual mallard and parera harvest
$\{S_{(sex*E+age})P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Effort model: Effort is expressed as total effort (*E*) or average effort (*Eave*)

1). Birds trapped as juveniles at first encounter are modelled as juveniles for the first year and then adults in subsequent years.

Other models were contemplated for inclusion in the model set but were excluded in favour of those that were considered most likely to represent the true situation (Anderson and Burnham, 2002, Anderson et al., 2001) and have sufficient foundation for reliable inference (Steidl et al., 1997).

Data were analysed in programme MARK (White and Burham, 1999) using Burnham Live-Dead format. Parameter estimates are computed using maximum likelihood and were based on model structure originally proposed by Burnham (1993) – “Both live and dead recoveries” but subsequently reparameterized[1] (Cooch and White, 2009). Encounter probabilities are a function of the bird surviving (S_t) from year t to t_{+1}, if they are recaptured (P_t) in year t, if they are killed they are found and reported (the conditional reporting rate, r_t (Barker et al., 2005) and, that the bird shows fidelity to the study area and are available for recapture (F_t). Encounter histories are recorded as a 1 if the bird is encountered at a recapture (trapping) or recovery (shot or found dead) occasion otherwise 0 (Cooch and White, 2009). Encounter histories were generated from Microsoft Excel pivot tables.

Linear models were transformed on the logit scale (Cooch, 2001). Because the logit transformation has difficulty estimating parameters that are near the boundary (0, 1) the parameter count was determined from the closest approximating Sine transformed model. Conditional reporting rate was largely constrained to age and time specific in accordance with findings of Nichols et al., (1990) and Caithness et al. (1991). Recapture and fidelity parameters were predominately left general (cohort and time dependent) to limit the size of the candidate model set (Anderson et al., 2001).

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[1] Program MARK uses different parameterization than the format presented by Burnham (1993). The terminology used by Burnham is based on the Brownie (1985) M1 model where the underlying model is a function of survival (S) and band recovery probability “f”.
In MARK f has been reparameterized such that:

$$f = r(S-1)$$

$$r = \frac{f}{(S-1)}$$

Where r is the probability of the band being recovered and reported (the conditional reporting rate). The advantage is that we can elicit all the survival information from the model (under the f parameterization there is some survival and some reporting rate probability).

Data fit was determined on the most paramaterised model (*Global Model*), with 100 bootstrap simulations (White et al., 2001). Models were ranked on Akaike's Information Criterion adjusted for low sample size (AICc) or Quasi likelihood AICc (QAICc) when over-dispersion was suspected (Burnham and Anderson, 2002, White et al., 2001). Over-dispersion was incorporated where the data did not fit the model ($P \leq 0.05$) using a variance inflation factor ($\hat{C}$). $\hat{C}$ was calculated using two techniques; deviance of the data were divided by the mean deviance of the simulated data, or, the global model estimate of $\hat{C}$ was divided by the mean $\hat{C}$ of the simulated data. Whichever of the techniques produced the largest $\hat{C}$ estimation was used to adjust for over inflation of variance (Cooch and White, 2009). If QAICc was used the number of parameters in the model were increased by 1 for $\hat{C}$ (Burnham and Anderson, 2002).

Over inflation of the variance was suspected due to a compromise of the assumptions (see Brownie et al., 1985, Nichols, 2005). Personal observation indicates the probability of encountering a banded bird is not an independent event. Recovery of a second banded bird was usually from the same trap sample suggesting that the birds had stayed together following banding. Anderson et al (1994) indicate that this (and heterogeneity) are probably a common issue in capture-recapture data. If QAICc was used the number of parameters in the model was increased by one for $\hat{C}$ (Burham and Anderson, 2002).

Additive models (parallel on the logit scale) are denoted "+" and multiplicative variables "*" such as time (every estimate is different).

Model averaging (White et al., 2001) was used to obtain estimates of survival and conditional reporting rate unless the top ranked model received overwhelming support (ΔAIC>2 from the next best model; (Burham and Anderson, 2002)[1]).

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[1] The larger the Δ_i the less plausible that the fitted model is the best model given the data (Burnham and Anderson, 2002). As a rough rule of thumb they suggest:

Δ_i	Level of Empirical Support
0-2	Substantial
4-7	Considerably less
>10	Essentially None

Mean and process variance were derived from a random effects model (Burnham, 2001, Burnham and White, 2002) (not reported in the model set).

Harvest and Harvest Rates

Harvest (H) was determined from hunter survey data (R. J. Barker University of Otago unpublished data) (see above). Harvest rate (h_t) is defined:

$$H_t = N_t h_t \qquad [2.9]$$

Where N_t is the population at time t.

Most literature reporting waterfowl harvest studies use the Brownie et al. (1978, 1985) parameterization, for example:

$$E(R_{ii}) = N_i f_i \qquad i=1,\ldots\ldots,k \qquad [2.10]$$

$$E(R_{ij}) = N_i S_i \ldots. S_{j-1} f_j \quad i=1,\ldots.,k \qquad j=i+1,\ldots\ldots,l \qquad [2.11]$$

Where $E(R_{ij})$ is the expected number of band returns by hunting season (banded in the i^{th} year and recovered in the j^{th} year), N_i the number of banded birds in the i^{th} year, f_i the birds recovered in the year of banding, f_j the recovery rate in subsequent years, and, S_{j-i} the survival rate (k and l are the final banding and recovery years respectively).

To avoid confusion later on in this paper where N is used to denote population size notation is changed: $R_{ii} = m_{ii}$ and $N_i = M_i$

The probability of a band being recovered (f) is dependent on the probability that the bird is killed (k) and retrieved (γ) and the band reported (λ) to Fish & Game or Department of Conservation:

$$f = k\gamma\lambda \tag{2.12}$$

Harvest rate is the probability that the bird is killed and retrieved:

$$h = k\gamma \tag{2.13}$$

$\therefore$

$$h\lambda = f \tag{2.14}$$

And:

$$h = \frac{f}{\lambda} \tag{2.15}$$

Where (k) is the probability that the bird is killed, the retrieval rate (γ), and, the probability that once the bird has been killed and retrieved it is reported (λ) to Fish and Game or the Department of Conservation.

Because not all bands recovered are reported (Nichols et al., 1995b, Royle and Garrettson, 2005) 120 randomly selected hunters were surveyed each year during the study period in order to attain an estimate of reporting rate. Hunters were asked if they had shot a banded bird in the last 3 years had they reported it. I was mindful of recall bias (Barker, 1991) over the 3 year period but considered that shooting a banded bird was an out of the ordinary event for many hunters. A possible decrease in reporting rate over time (Guillemain et al., 2011) was considered as hunters become acclimatised (fatigued) to shooting a banded bird, therefore temporal changes in reporting rate were tested.

The probability of reporting a bird is an independent Bernoulli trial with two possible random outcomes; a reported bird is recorded as a success and, an unreported bird a failure.

By invariance the MLE for h is given:

$$\hat{h} = \frac{\hat{f}}{\hat{\lambda}} \qquad [2.16]$$

and has large sample variance (R. J. Barker pers.com.):

$$\sigma^2_{\hat{h}} = \frac{\sigma^2_{\hat{f}}}{\lambda^2} + \frac{h^2\sigma^2_{\hat{\lambda}}}{\lambda^2} \qquad [2.17]$$

$$= h^2\left\{\frac{\sigma^2_{\hat{f}}}{f^2} + \frac{\sigma^2_{\hat{\lambda}}}{\lambda^2}\right\} \qquad [2.18]$$

Harvest rate ($\hat{h}_A$) under a set of common season regulations (A) were considered normally distributed[1] with variance $\sigma^2_{\hat{h}_A}$.

From 2001-2006 Eastern Region fixed season length to 57 days. This provided an opportunity to examine the effect of restrictive vs. relaxed bag limits on harvest rates. Bag limits under 10 per day were considered restrictive[2].

Program R (2.11.0) package stats (R Development Core Team, 2005) was used to examine Pearson's product-moment correlation between harvest and harvest rate, harvest and survival, harvest and season regulations. No comparative analysis of harvest rates and survival was made due to confounding of covariance.

Population size

The approximate population size ($\hat{N}_t$) was estimated from mark recovery and harvest data using the Lincoln-Petersen Estimate. A sample of the population was trapped and marked (banded). A second sample of the population was obtained, but rather than trapped they were harvested as part of the normal hunting season. Harvest was

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[1] A beta distribution may have been more appropriate as it is the natural prior conjugate of a binomial distribution (Bolstad, 2007).

[2] This was the opinion of the Council setting the season regulations.

determined through the hunter survey (R. J. Barker University of Otago unpublished data). The number of marked birds in the harvest was estimated from banded birds retrieved and reported by hunters in the year of initial capture. Because not all banded birds that are retrieved are reported the reporting rate was estimated through random surveys of hunters (see harvest rates above).

Assumptions:

- the population was closed (N_t is constant; no net immigration or emigration, and survival between banding and harvest, February and May is 1)[1];
- That the trap sites were representative of the population across the study area;
- That the period February – May provided ample opportunity for banded birds to mix with birds without bands and the harvest is a random sample from population N;
- Non-recovery of shot banded birds was constant from year to year;
- That there was no band loss and;
- Reporting rate was not 1 but could be accounted for through a survey of hunters (see above).

N is estimated:

$$\hat{N}_i = \frac{M_i H_i}{m_{ii}}$$ (Seber, 1982)[2] [2.19]

$$\cong \frac{M_i \hat{H}_i}{\hat{m}_{ii}}$$ [2.20]

where:

$$\hat{m}_{ii} = \frac{m_{ii}}{\hat{\lambda}}$$ [2.21]

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[1] The population can be considered closed to immigration and emigration as recoveries of banded birds (m_{ii}) are confined within New Zealand. Recoveries from outside the immediate study area are no less likely than from within it. Emigration from the immediate study area, the population of interest, is assumed to be off-set by an equal number of birds immigrating into the study area.

[2] Seber (1982) uses a different notation than I present here.

M_i is the number of trapped and banded (marked) birds in year i; H_i is the total mallard harvest in the i^{th} year; m_{ii} is number of birds banded in the i^{th} year and recovered in the i^{th} year; $\hat{\lambda}$ is the estimated reporting rate.

And estimated variance:

$$\hat{v}_{\hat{N}_i} = \frac{(M_i+1)(H_i+1)(M_i-m_{ii})(H_i-m_{ii})}{(m_{ii}+1)^2(m_{ii}+2)} \quad \text{(Seber, 1982)} \qquad [2.22]$$

$$\cong \frac{(M_i+1)(\hat{H}_i+1)(M_i-m_{ii})(\hat{H}_i-m_{ii})}{(m_{ii}+1)^2(m_{ii}+2)} \qquad [2.23]$$

The Lincoln-Petersen Estimate of $\hat{N}_i$ is equivalent to:

$$\hat{N}_i = \frac{\hat{H}_i}{\hat{h}_i} \qquad [2.24]$$

There are a number of issues with this approach. In estimating both population size (N_i) and the variance ($\hat{v}_{\hat{N}_i}$), harvest in year t ($\hat{H}_i$) is an estimate and further, the expected number of recovered bands ($\hat{m}_{ii}$) for that harvest is a function of reporting rate. For the variance estimate m_{ii} was not adjusted by the reporting rate (effectively inflating the variance), but it was for N_i. Secondly survival between banding and the beginning of the banding season is probably not 1, particularly for the juveniles. Therefore the population size of the respective cohorts should be reduced down (if survival is less than 1 the number of marked birds in the harvest would be smaller than if it were 1; increasing the marked birds in the harvest reduces the estimated population size) by the appropriate survival estimate for this period.

Density Dependence

Post- harvest survival and productivity will be dependent on post- harvest population size if density dependence is apparent.

Annual survival was recorded from banding (end of February) until the following February. The heuristic model (Figure 2.1) identifies three periods of mortality; banding until harvest, harvest, and post harvest. Consistent with these three periods of mortality, survival is defined akin to that presented by Johnson et al. (1997) except that slightly different time periods are used, and the model include the post banding pre-harvest period in the juvenile survival model:

$$\phi_{jt} = \alpha_{jt}\beta_{jt}\delta_{jt} \tag{2.25}$$

where ϕ_{jt} is the annual survival of the j^{th} cohort in year t, β_{jt}, hunting season survival and, α_{jt}, and δ_{jt}, are the post-banding pre-harvest, and post-harvest survival respectively. α_{jt} was estimated from the telemetry study of 46 mallard duck in the Eastern Region. VHF transmitters fixed either as a harness backpack or tail mount to 16 juvenile female, 10 juvenile male, 8 adult female, and 12 adult male mallard randomly selected from the trap sample. Either 6 gram Sirtrack single-stage tail mount transmitters (5 month life) using glue and cable ties based on protocols outlined in Giroux et al. (1990), or 16 gm Sirtrack two-stage transmitters (10 month life) attached via harnesses around the with the aerial down the back of the bird. The two-stage transmitters were attached to juvenile females as we were particularly interested in their survival over the study period (it was anticipated that these transmitters would provide a better signal). The two-stage transmitters have a mortality switch that allows time since death to be calculated. A three element handheld Yagi receiving antenna and automatic scanning receiver (Samuel and Fuller, 1996) were used from a vehicle or by foot. Two aerial surveys were conducted prior to and post the dabbling duck hunting season as per procedures detailed in Gilmer et al., (1981).

Known Fate procedure (Cooch and White, 2009) (based on Kaplan-Meier methodology (Kaplan and Meier, 1958) and stagger entry design (Pollock et al., 1989a) in Program MARK (White and Burham, 1999) was used to assess survival over the study periods. Covariates of interest were age, weight and, transmitter type (single-stage tail mount, single-stage backpack and, two-stage backpack). Survival periods were monthly intervals January 2011- October 2011.

Initial results suggest $\alpha = 1$ for adults and 0.91 for juveniles and:

β_{jt} is defined:

$$\beta_{jt} = (1 - k_{jt}) \quad [2.26]$$

$$= 1 - \left(\frac{h_{jt}}{\gamma\lambda} \right) \quad [2.27]$$

Where k is the cohort and time dependent kill rate and, γ the recovery rate ($\gamma = 1 - c$; c is the crippling loss determined from the Eastern Fish and Game hunter survey for the period at 5%).

Post-harvest survival (δ_{jt}) as a function of post-harvest population size ($N_{PoH,t}$) inferred density dependence:

$$\delta_{jt} = f(N_{PoH,t}) \quad [2.28]$$

Population Size and Survival: Post-harvest adult population size was defined:

$$\hat{N}_{jtPoH} = N_{jt}\alpha_{jt}\beta_{jt} \quad [2.29]$$

for the j^{th} cohort where the respective proportion of N_t attributed to each cohort was determined from the trap sample.

lm and nls (R package stats; (R Development Core Team, 2005)) were used to investigate the functional form (linear, exponential, or quadratic polynomial) of the relationship between post harvest population size and survival, and post harvest population size and productivity. The most appropriate (parsimonious) model was selected using Akaike’s information criteria for low sample size (AICc) (Burnham and Anderson, 2002).

Density Dependent Recruitment: Recruitment (*B*) was determined as the ratio of juveniles to adults in the trap sample[1]. Concerns were that adult females would be underrepresented in the trap sample due to trap timing coinciding with adult female wing moult[2] and would therefore not be available for trapping at some sites. Trap bias was tested using a simple model of population change proposed by Cowardin and Johnson (1979):

$$\Delta N = \frac{N_{t+1}}{N_t} = \frac{N_t S_{AFt} + N_t B_t S_{JFt}}{N_t} = S_{AFt} B_t S_{JFt} \qquad [2.30]$$

Where ΔN = population change; N_t = the spring population in year *t*; S_{AFt} = the adult female survival rate and; S_{JFt} is juvenile survival from autumn to the following spring. B_t = the number of juvenile females in the autumn population produced per adult female in the previous spring population.

It was anticipated that population change (ΔN) would be approximately 1 over the 13 and 9 years study period for the Eastern and Hawke's Bay Regions respectively. Average survival was determined under a Random effects model (Burnham, 2001, Burnham and White, 2002) and evaluated against average recruitment over the period.

Recruitment and Regulations

Evidence of a causal relationship between season length and recruitment (ratio of juveniles to adults in the trap sample) was investigated by dividing game seasons into two treatments; ≥57 days (long season) and <57 days (short season) and analysed using a two-tailed Student t-test.

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[1] Trapping occurred late January early February when the majority of young had fledged but still retained their juvenile sex characteristics (see Appendix A).

[2] Timing of moult is not well documented in New Zealand (Marchant and Higgins, 1991). Males commence moult after females have finished nesting (Heather and Robertson, 1996, Williams, 1981) and females commence about 1 month after males (Heather and Robertson, 1996). Habitat conditions and weather influence timing of the prebasic moult in mallard (Baldassarre and Bolen, 2006) and perhaps timing of breeding? I therefore have some misgivings about using trap sample as an index of productivity. Other methods embarked on such as brood counts in the Eastern Region did not cover the full study period.

Partial Management Control- Results

Season Regulations and Effort – Eastern Region

Participation: Economic indices are not a particularly useful indicator of participation in the Eastern Region. The correlation, between licence sales and the unemployment rate (Figure 2.3) was negative but not significant at the 0.05 level (R=-0.447, $t_{(11)} = -1.6559$, $P = 0.126$). Nevertheless unemployment rate appears to perform better than the method used for budgeting purposes. Licence sales are currently predicted on the previous two year sales trend (R=0.189, $t_{(8)} = 0.5455$, $P = 0.6$). An alternative approach examined was average licence sales for the last two years but this is no better than the two year trend (R=-0.153, $t_{(8)} = -0.4384$, $P = 0.67$).

Effort: Total effort declined over the study period (R=-0.830.69, $t_{(10)} = -4.7419$, $P = 0.0008$, Figure 2.7[1]) which appears to be due to a decline in average effort (R=-0.83, $t_{(10)} = -4.7334$, P= 0.0008, Figure 2.6) which is probably in part, a function of fewer days on average hunted over the study (R= -0.64, $t_{(10)} = -2.6345$, $P = 0.02$, Figure 2.4)..

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[1] The 2001 data have been removed as the surveyor made up a number of results.

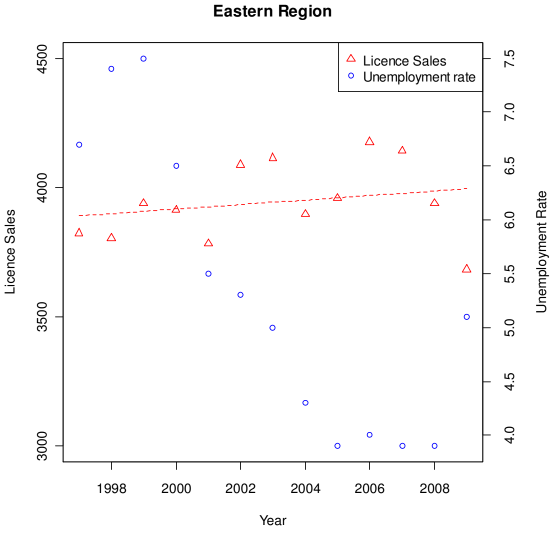

Figure 2.3. Eastern Region Licence Sales 1997-2009 and unemployment rate. There is no evidence of a linear change in participation (red dashed line) through this period (R^2 = 0.035, $F_{(1)(10)}$= 0.359, P=0.56)

Season regulations: The error associated with estimated average effort (Figure 2.6) makes comparison between the season regulations dubious. The variances, however are reasonable and to be expected considering the difference in effort exerted by keen hunters vs. the more casual hunter[1].

Some of the decline in effort may have been a consequence of increasing restriction in season length over the period (R=-0.49, $t_{(11)}$ = -1.8834, P = 0.086), but if this was so we might expect a strong relationship between average days hunted and season length but this was not the case (R= 0.32, $t_{(10)}$ = 1.0745, P = 0.31, Figure 2.10).

Correlation between season length and total hours hunted was positive but not significant (R = 0.4183, $t_{(10)}$ = 1.4565, P = 0.18; Figure 2.8) as was average hours hunted (R = 0.2825, $t_{(10)}$ = 0.9316, P = 0.37 Figure 2.9). In explanation Eastern Region hunters hunted on average for more hours in the restricted (30 and 43 day) seasons than the majority of the intermediate (57 day) and 1 of the relaxed seasons (71 day) (Figure 2.8). The relationship appears non-linear (exponential); as season

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[1] 120 fortnightly surveys of hunters during the season picks up limited active hunters after opening weekend and their activity will vary from those that hunt all day to those that hunt for an hour after work.

length increases average hours per day hunted decreases (Figure 2.12). The average number of days hunted in the 30 day season was less than any of the other season but not the average number of hours. Hunters therefore must have hunted for more hours per day.

The correlation between bag limits and hours hunted was relatively strong (R= 0.7024, P= 0.011 Figure 2.11).

Effect of success on effort: To examine the relationship between opening weekend success and effort in the Eastern Region years were divided into 57 (n= 6) and 71 (n=4) day seasons (there were only one 30 and 43 day seasons). The results were mixed; there was no correlation between opening weekend harvest and days hunted in the 57 day season (R=0.242, $t_{(4)} = 0.4994$, $P = 0.64$), while the correlation in the 71 day season was reasonable (R=0.975, $t_{(2)} = 6.1957$, $P = 0.025$) but this is likely to be a function of the limited data points.

In the second analysis where ducks per hour were proposed as an index of a good season, hunters did not increase days hunted in high ducks per hour years (R=0.112, $t_{(11)} = 0.3745$, $P = 0.72$).

Correlation between total effort and total harvest in the Eastern Region was good (R=0.86, $t_{(10)} = 5.3193$, $P = 0.0003$). Total harvest may explain the incongruous result between bag limits and effort. Bag limits are strongly correlated with total harvest (R =0.63, $t_{(10)} = 2.5758$, $P = 0.028$). this was anticipated as Eastern Region used bag limits to try and manage harvest (2001-2006). So in years when game was assessed as plentiful liberal bag limits were used therefore causation between bag limits and effort is questionable.

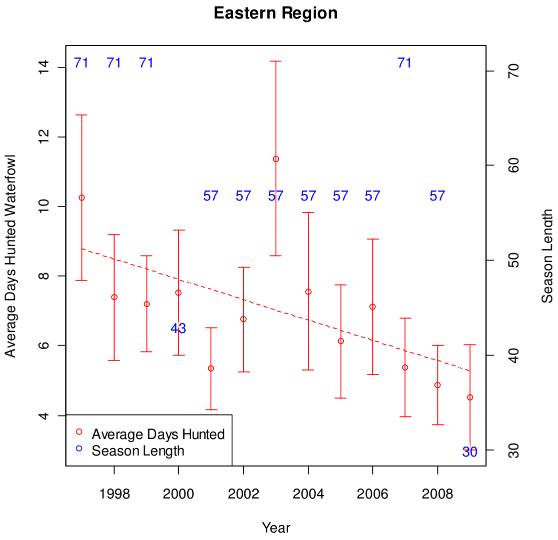

Figure 2.4. Average days ($\overline{D} \pm 95\%\ CI$) spent waterfowl hunting per licence holder in the Eastern Region 1997-2009. Season length superimposed. The 2001 effort estimate was compromised and is probably wrong. Average days over the study period have decreased (R= -0.57, $t_{(11)}$ = -2.3131, P = 0.04).

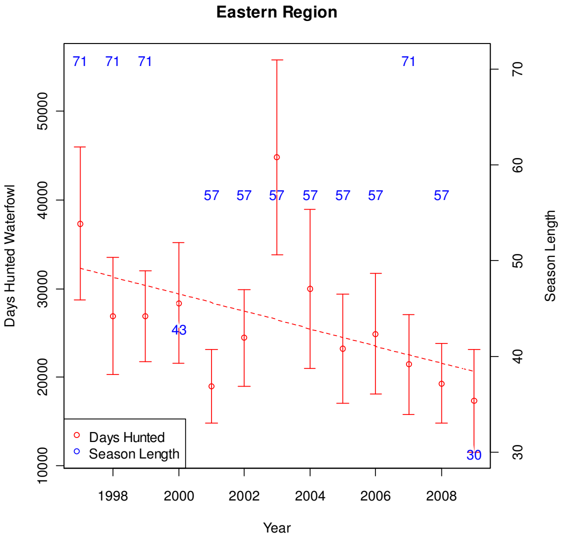

Figure 2.5. Total days ($\widehat{D} \pm 95\%\ CI$) hunted waterfowl 1997-2009, Eastern Region showing season length. Regression line on days hunted (red dashed line). Total days may be decreasing over the study (R =0.24, $t_{(11)}$ = -1.8148, P = 0.09). The 2001 days estimate was compromised and is probably wrong.

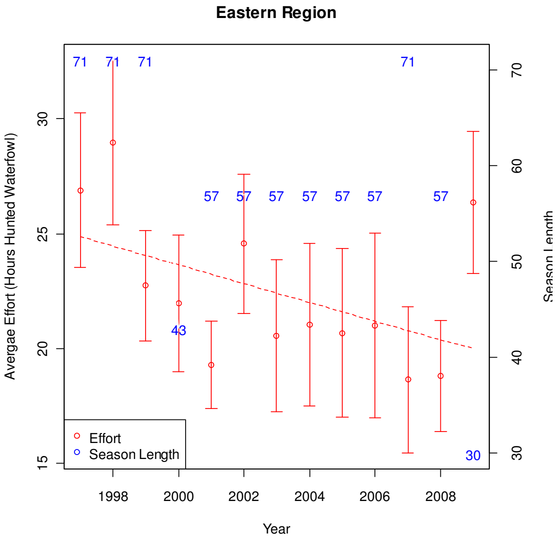

Figure 2.6. Average effort ($\overline{E} \pm 95\%\ CI$) spent waterfowl hunting in the Eastern Region 1997-2009. Season length superimposed. The 2001 effort estimate was compromised and is probably wrong. Average hours hunting waterfowl may be decreasing (R= -0.48, $t_{(11)}$=-1.8148, P = 0.097)?

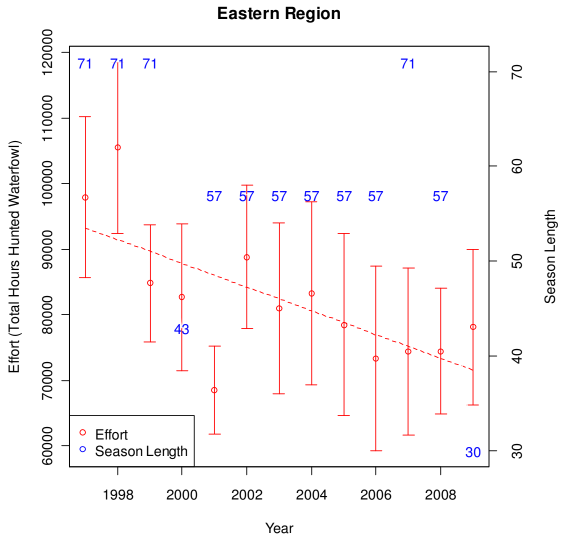

Figure 2.7. Total effort ($\widehat{E} \pm 95\%\ CI$) spent waterfowl hunting in the Eastern Region 1997-2009. Season length superimposed. The 2001 effort estimate was compromised and is probably wrong and has been removed from the correlation analysis (see below). Total hours hunting waterfowl has decreased (R= -0.83, $t_{(10)}$=-4.7419, P <0.001). The difference between average hours hunted and total hours hunted for 2009 is a function of the low number of licensed participants in this year.

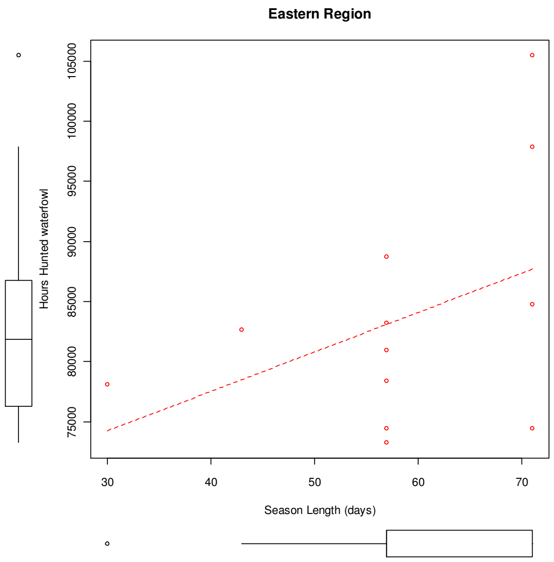

Figure 2.8: Scatterplot of waterfowl season length in the Eastern Region against total hours hunting waterfowl. (R = 0.42, $t_{(10)}$ = 1.4565, P = 0.18).

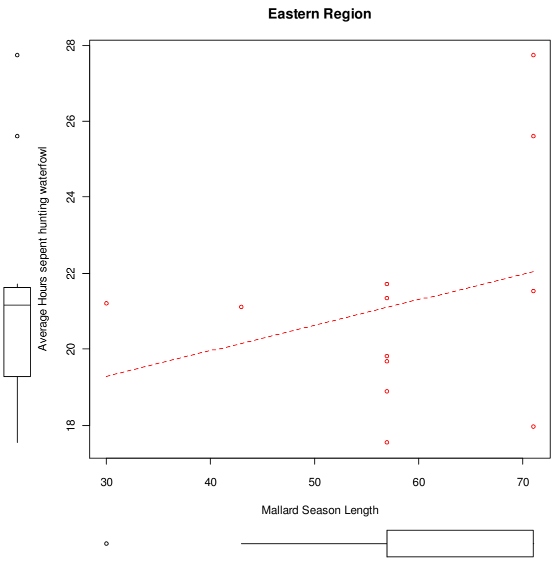

Figure 2.9: Scatterplot of mallard season length against average hours hunting waterfowl in the Eastern Region (R = 0.2825, $t_{(10)}$ = 0.9316, P = 0.374).

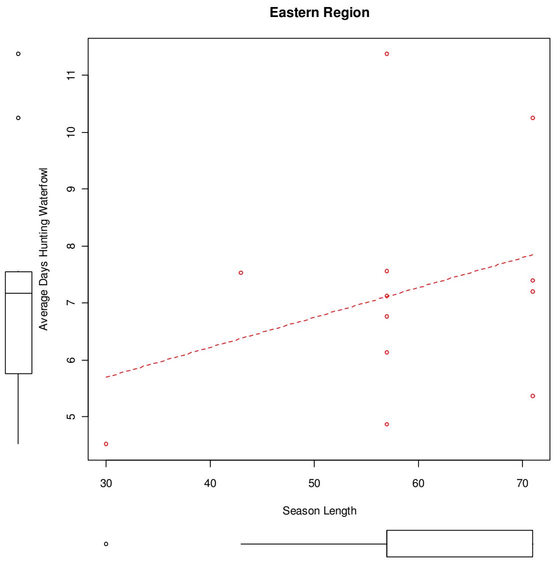

Figure 2.10. Scatterplot of average days Eastern hunters spent hunting waterfowl as a function of season length. (R=0.32, $t_{(10)}$ = 1.0745, P = 0.31).

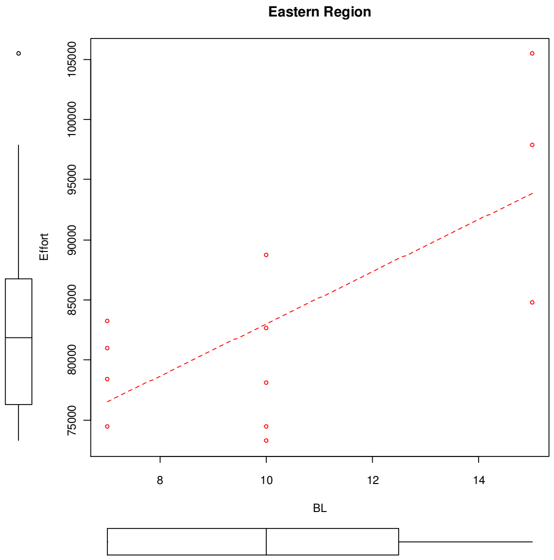

Figure 2.11. The relationship between effort and bag limits (*R* = 0.702, *P* = 0.011) in the Eastern Region.

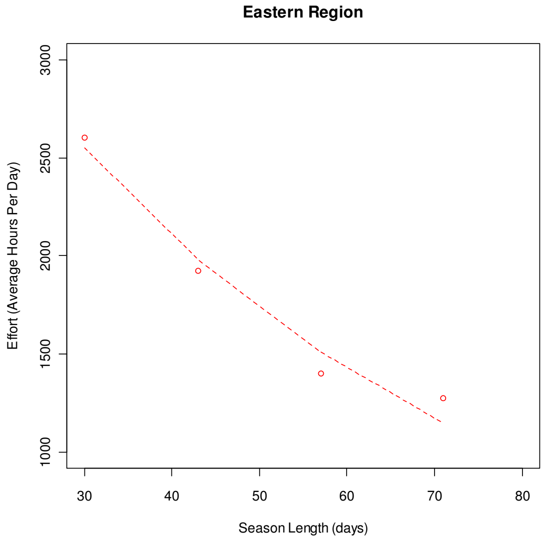

Figure 2.12. Average hours hunted per day as a function of season length. Red dashed line is the exponential model.

Predicting Effort: To help understand what influences average effort spent hunting waterfowl in the Eastern Region 13 models (Table 2.2) were developed. Covariates were mallard season length (*SL;* days), winter temperature (*WT; C^o*), winter precipitation (*WR*; mm), number of licence holders (*Hu*) and year (*Yr*).

Table 2.2: Models of average effort spent hunting waterfowl per licence holder in the Eastern Region. *Yr*=Year; *Hu* =Number of Licence Holders*; SL* = Season Length*; WT* = Winter Temperature (Whakatane AWS)*; WR* = Winter Rain (Whakatane AWS).

Model	AICc	ΔAICc	AICc Weights	Model Likelihood	Num. Param.
$\overline{E} = 1326.7845 - 0.6513Yr$	55.4222	0.0000	0.870339	1	3
$\overline{E} = 1348.79297 - 0.66205Yr - 0.00693SL$	60.1149	4.692676	0.083308	0.095719	4
$\overline{E} = 1332.03168 - 0.65506Yr - 0.0149SL + 0.02399WR$	62.5864	7.1642	0.02421	0.027817	5
$\overline{E} = 1176 - 0.5658Yr + 0.02991SL - 0.005622Hu$	64.12882	8.70664	0.011196	0.012864	5
$\overline{E} = 1305 - 0.6424Yr - 0.003626SL + 0.4431WT$	66.1400	10.71782	0.004096	0.004706	5
$\overline{E} = 16.28138 + 0.10137SL$	67.4477	12.02553	0.00213	0.002447	3
$\overline{E} = 58.897615 + 0.144146SL - 0.011401Hu$	67.6212	12.19904	0.001953	0.002244	4
$\overline{E} = 52.583994 - 0.007683Hu$	67.8162	12.39397	0.001772	0.002036	3
$\overline{E} = 1277.43876 - 0.6306Yr - 0.011SL + 0.54719WT$ $+ 0.02453WR$	70.8360	15.41385	0.000391	0.00045	6
$Log_e(\overline{E}) = Log_e(SL) + Log_e(0.895695SL) - 0.01433SL$	71.6589	16.23673	0.000259	0.000298	4
$\overline{E} = 45.123208 + 0.145982SL - 0.01181Hu + 1.576129WT$	71.8839	16.46167	0.000232	0.000266	5
$\overline{E} = 78.157134 + 0.169204SL - 0.015915Hu - 0.021376WR$	73.3719	17.9497	0.00011	0.000127	5
$\overline{E} = 64.058629 + 0.170504SL - 0.016226Hu + 1.566976WT$ $- 0.020927WR$	80.0751	24.65289	3.86E-06	4.43E-06	6

The *Year* model was the top ranked model (*wi*=0.87) with the only other model to receive any degree of support was the *Year-Season Length* model (ΔAICc=4.69; *wi*=0.08). The year parameter in the top model was highly significant (*P*=0.0008) whereas the season length parameter in the second ranked model was not (*P*=0.90). Note also that the season length parameter is negative suggesting as season length increases average effort decreases.

The *Year-Season length-Winter Rain* model suffered the consequence of parsimony, with one less parameter it would have been the second ranked model. The positive effect of the Winter Rain parameter was reasonably significant (*P*=0.12) considering the climate data were from Whakatane and the effort data were from the whole of the Eastern Region. This suggests hunters hunt more on average in wet weather.

The negative prefix on the hunter parameter (*P*=0.23) suggests average hours hunting increases with fewer hunters. An explanation is that the more itinerate hunters hunt less on average than the keener more permanent hunter. Although the support for these models (*Hunter*) was poor it could have important implications when trying to establish a predictive model of total hours hunted. Total effort is calculated as the product of average effort and licence sales, therefore estimates of total effort is confounded by the number of licensed hunters. Because the number of hunters has a marked twofold influence on the total effort estimate I reran the analysis using total effort (Table 2.3).

Table 2.3: Models of total effort spent hunting waterfowl in the Eastern Region. *Hu* =Licence Holders*;* *SL* = Season Length*;* *WT* = Winter Temperature (Whakatane AWS*);* *WR* = Winter Rain (Whakatane AWS).

Model	AICc	ΔAICc	AICc Weights	Model Likelihood	Num. Par.
$E = 4134353 - 2022.2Yr$	248.3575	0.0000	0.9182	1.0000	3
$E = 3347132 - 1583.18Yr - 25.77Hu + 167.37SL$	253.6880	5.3305	0.0639	0.0696	5
$E = 221112.42 - 41.94Hu + 487.03SL$	257.7037	9.3462	0.0086	0.0093	4
$E = 199780.49 - 29.38Hu$	259.7860	11.4285	0.0030	0.0033	3
$E = 170075.01 - 43.45Hu + 493.83SL + 5839.93WT$	260.0271	11.6696	0.0027	0.0029	5
$E = 64341.5 + 329.7SL$	260.1874	11.8299	0.0025	0.0027	3
$Log_e(E) = Log_e(SL) + Log_e(3528.7SL) - 0.0151SL$	262.7883	14.4308	0.0007	0.0007	3
$E = 247648.95 - 4816Hu + 521.56SL - 29.45WR$	263.8576	15.5001	0.0004	0.0004	5
$E = 195214.87 - 49.32Hu + 526.39SL - 27.78WR + 5827.78WT$	268.6635	20.3060	0.0000	0.0000	6

The top ranked model (*Year*) of the total effort received 92% of the support, while the next ranked model (*Year-Hunter-Season Length*; ΔAICc=5.33; *wi*=0.06; parameter estimates: P=0.011, 0.059 and 0.328 respectively) was of considerably better fit (R^2 = 0.81 cf. the *Year* model 0.69).

The total effort models fit the data noticeably better than the average models (fit of average effort models *Year* and *Year-Season Length* were respectively, R^2= 0.49 and 0.49).

Given the better fit and utility of incorporating season length I used the second ranked total effort model to predict effort (Figure 2.13; Equation [2.31]):

$$E = 3347132 - 1583.18Yr - 25.77Hu + 167.37SL \qquad [2.31]$$

Using Program Winbugs the expected total effort and 95% credible intervals can be calculated for a given year, season length, and predicted number of hunters (Figure 2.14).

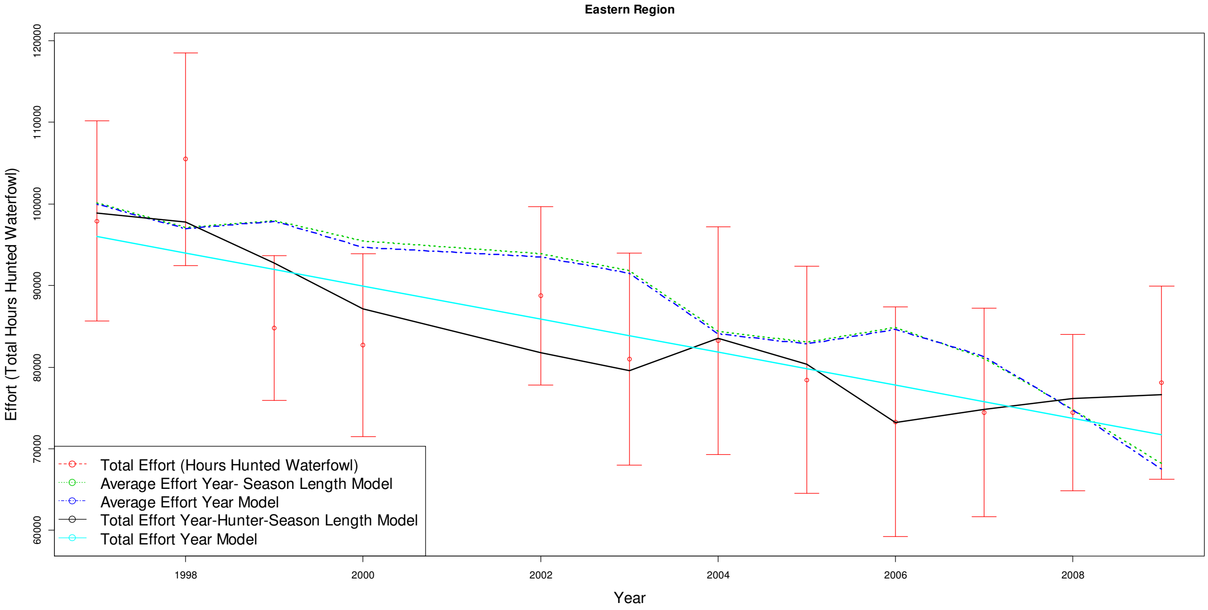

Figure 2.13 Total effort in the Eastern Region 1997-2009 (excluding 2001) against the top (*Year* - light blue solid line - R^2=0.69, $F_{(1)(10)}$= 22.49, P<0.001) and second top ranked total effort model *Year-Season Length* (Black solid line; R^2=0.808, $F_{(3)(8)}$= 11.23, P= 0.003), and average effort models, the dashed lines (*Year, Year-Season Length* R^2=0.49 both models). The average effort models have been fitted by multiplying their respective parameter estimates by the number of licences sold in each year.

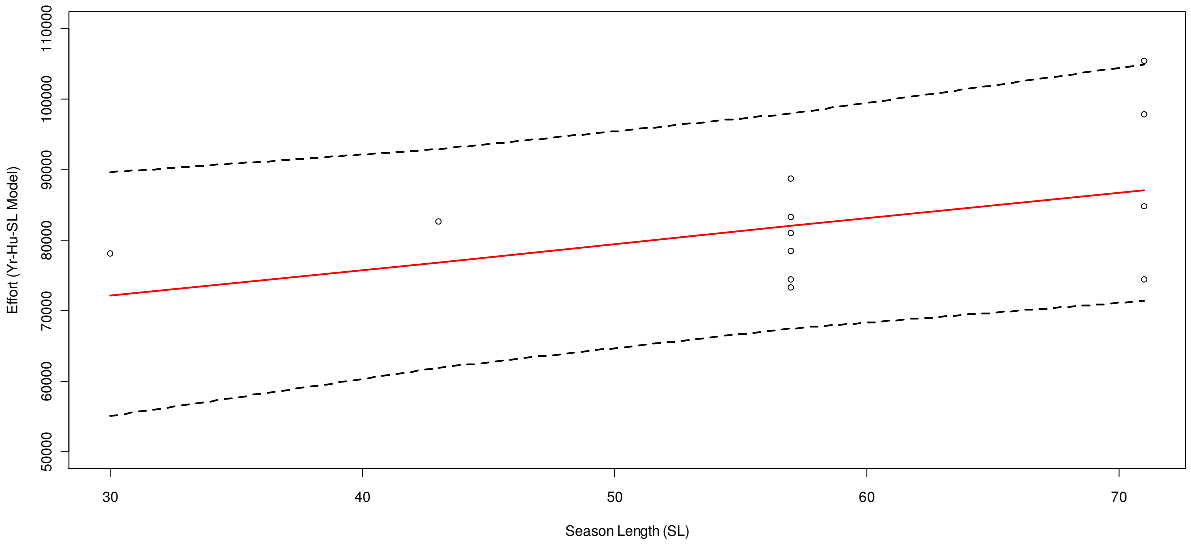

Figure 2.14. Eastern Region effort as a function of year (2009, in this example) the number of hunters (3900) and season length (red line) with 95% credible interval (dashed line) (1000 burn in, 100,000 simulations in WINBUGs). Points show estimated effort derived from the hunter survey.

Hawke's Bay Effort

Participation: From 2000-2009 licence sales have increased in the Hawke's Bay (R=0.89, $t_{(8)}$= 5.7385 P= 0.0004, Figure 2.15). Participation may not have increased to the same extent as licence sales because this period also coincided with an increase in law enforcement. At the beginning of the period a much greater percentage of unlicensed hunters were encountered compared with the Eastern Region. Over the study period compliance (proportion of licensed holders among encountered hunters) increased.

Unemployment appears to have a slightly more pronounced effect on Hawke's Bay licence holders than in the Eastern region (Figure 2.15). Nevertheless the relationship with licence sales is also not significant at the 5% level (R=-0.58, $t_{(8)}$ = -2.0277, P = 0.077). In the short term, the most appropriate predictor of licence sales a linear extrapolation of the last two years licence sales (R=0.69, $t_{(8)}$ = 2.3505, P = 0.057), alternatively the average of the previous two years also performs well (R=0.73, $t_{(8)}$ = 2.6141, P = 0.039).

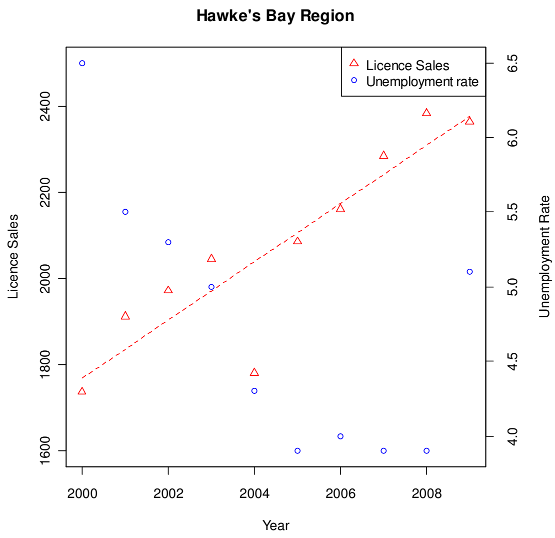

Figure 2.15. Hawke’s Bay Region Licence Sales 2000-2009 (linear model of licence sales red dashed line R^2=0.80, $F_{(1)(8)}$= 32.93 P=0.0004); and unemployment rate.

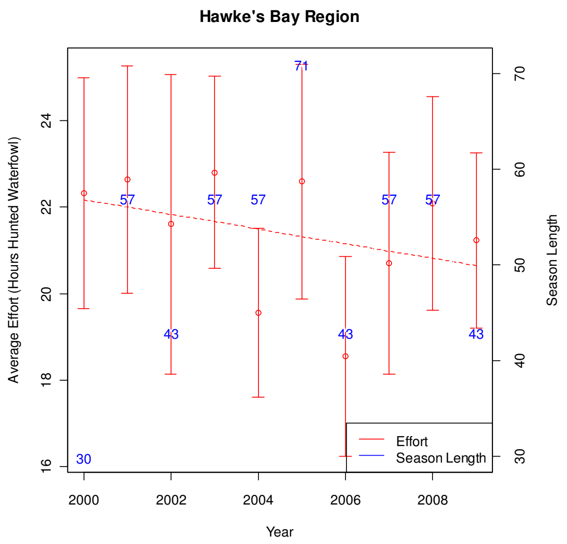

Figure 2.16. Average effort ($\bar{E} \pm 95\% \, CI$) hunting waterfowl in the Hawke’s Bay Region, season length (days) superimposed. Regression of average effort over study period is not significant (R^2=0.09, $F_{(1)(8)}$= 0.818, P= 0.392).

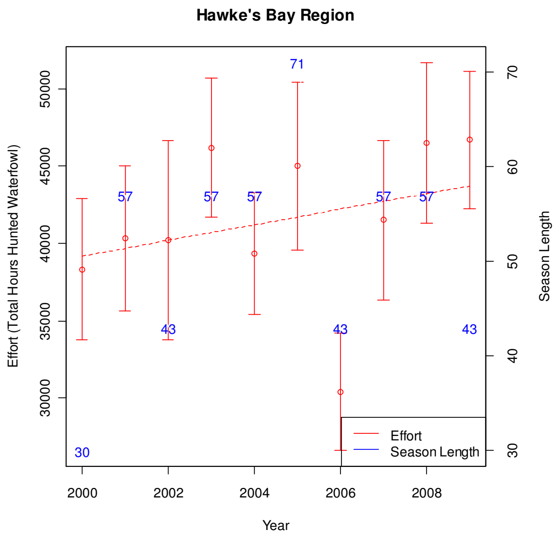

Figure 2.17. Total Hours spent hunting waterfowl($\hat{E} \pm 95\% \, CI$) in the Hawke’s Bay 2000-2009.

Effort: Consistent with the Eastern Region total effort did not increase with participation (R =0.39, $t_{(8)}$ = 1.218, P = 0.26). There was no evidence that total effort changed over the study period (R =0.30, $t_{(8)}$ = 0.9044, P = 0.39), nor average effort (R =-0.35, $t_{(8)}$ = 1.0882, P = 0.31).

Season Regulations: The confidence intervals associated with the average effort (Figure 2.16) estimate indicate no significant difference between years. Nor was there a significant correlation between season length and average effort (R=0.21, $t_{(8)}$ = 0.6139, P = 0.55; Figure 2.21).

On average there was very little difference in average hours hunted (18.5-22.8). This compares with the range in Eastern region 18.6 – 28.9. Behaviour between the two regions was similar; more hours were expended on average in the 30 day season than the all but two of the 57 day seasons, and average hours were greater in 2 of the 43 day seasons than 2 of the 57 day seasons. This appears in part, a result of more days hunted in some of the short seasons (Figure 2.18). Consistent with Eastern Region, season length appears to have an exponential relationship with average effort per day hunted (Figure 2.19).

There was no correlation between effort and bag limits (R=0.05, $t_{(8)}$ = 0.154, P = 0.88).

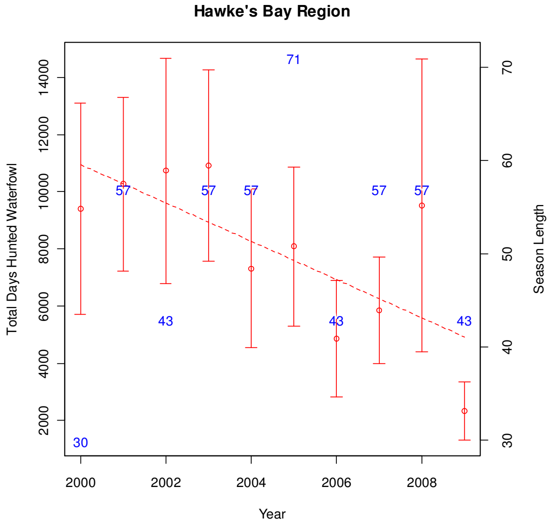

Figure 2.18. Total days ($\widehat{D} \pm 95\%\ CI$) spent waterfowl hunting in the Hawke's Bay Region 2000-2009. Season length (days) superimposed.

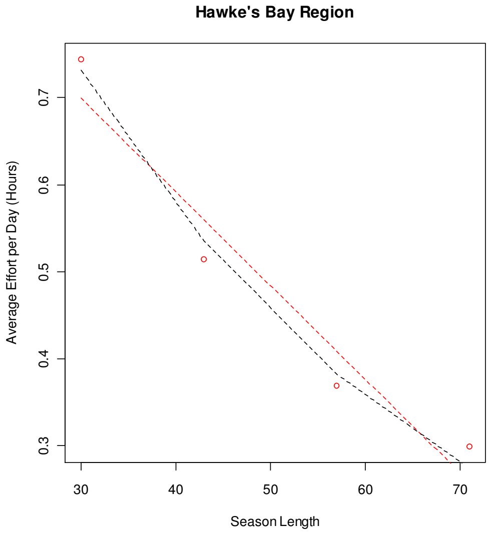

Figure 2.19. Average effort per day against season length. The red dashed line is the linear model and black dashed line the exponential model

Effect of success on effort: For the Hawke's Bay data I was only able to test the correlation between opening weekend harvest and total days hunted for one season length (57 days; n=5); all the other season lengths were for ≤2 years. A successful opening weekend does not appear to inspire hunters to hunt for more hours in the remainder of the season (R=0.046, $t_{(5)} = 0.1042$, $P = 0.92$). Neither did a good season (ducks per hour) result in hunters hunting for more days (R=0.058, $t_{(4)} = 0.1164$, $P = 0.91$).

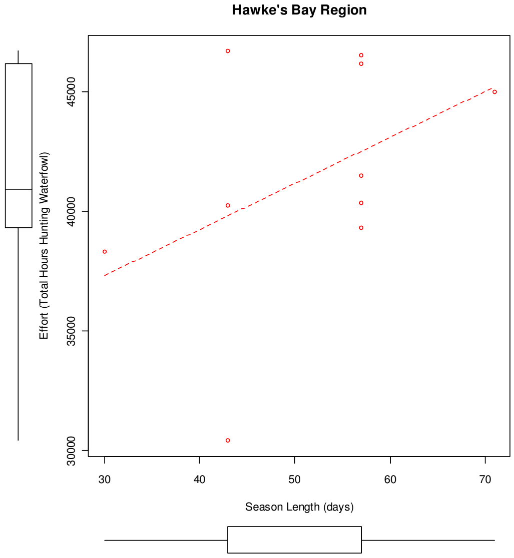

Figure 2.20. Scatterplot of waterfowl season length in the Hawke's Bay Region against total hours hunting waterfowl. ($R = 0.44$, $t_{(8)} = 1.3984$, $P = 0.20$).

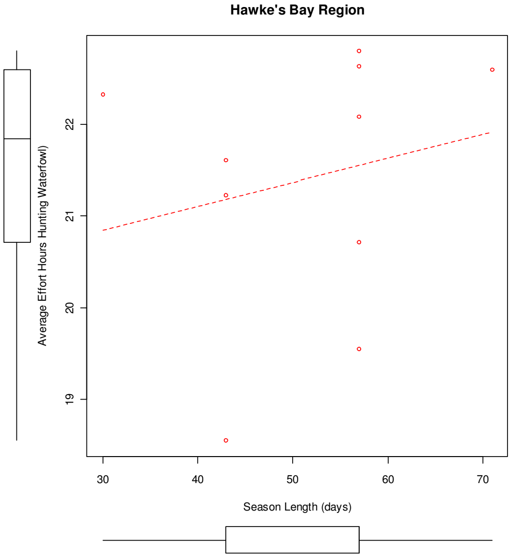

Figure 2.21: Scatterplot of mallard season length against average hours hunting waterfowl in the Hawke's Bay Region. (R = 0.21, $t_{(8)}$= 0.6139, $P = 0.55$)

Predicting Effort: 10 models were developed to predict average effort in the Hawke's Bay but three of these (*Season Length-Hunter, Season Length-Year* and *Season Length-Winter Rain*) were excluded as the residuals failed the Shapiro-Wilk normality test.

Of the remaining candidate models (Table 2.4) the *Year (wi*=0.37*), Season Length (wi*=0.24*)* and *Hunter (wi*=0.20*)* models received 81% of the support, however none of the parameters were significant (P>0.31). The *Season Length-Winter Rain-Winter Temperature* model had the lowest deviance and the climate covariates were significant (P=0.014 & P=0.025 respectively) suggesting hunters hunt less in the rain but more in warm weather.

Table 2.4: Models of average effort spent hunting waterfowl in the Hawke's Bay Region. *Yr*=Year, *Hu* =Licence Holders*; SL* = Season Length*; WT* = Winter Temperature (Napier AWS)*; WR* = Winter Rain (Napier AWS).

Model	AICc	ΔAICc	AICc Weights (*wi*)	Model Likelihood	Number of Parameters
$\overline{E} = 351.9629 - 0.1649Yr$	43.0045	0.0000	0.3706	0.0371	3
$\overline{E} = 20.05395 + 0.0266SL$	43.8418	0.8372	0.2438	0.0244	3
$\overline{E} = 22.3983 - 0.00047Hu$	44.2686	1.2640	0.1970	0.0197	3
$\overline{E} = 1449 - 0.7213Yr + 0.04026SL + 0.007665Hu$	44.9876	1.9831	0.1375	0.0138	4
$\overline{E} = -0.9241 + 0.05155SL - 0.05963WR + 2.56113WT$	47.5563	4.5517	0.0381	0.0038	5
$Log_e(\overline{E}) = Log_e(SL) + Log_e(1.153746SL) - 0.01936SL$	53.7818	10.7773	0.0017	0.0002	4
$\overline{E} = 13.63244 + 0.03234SL + 0.62742WT$	56.3327	13.3282	0.0005	0.0000	5

Given the poor performance of the average effort models I looked at total effort (not shown). The *Season Length* model was the top ranked total effort model but this is also a poor fit (R^2 =0.196) (probably due to the 2006 data point which is the only significantly different effort estimate; Figure 2.22). Nevertheless I report the *Season Length* model (Equation [2.32]; Figure 2.23) as the starting model to predict effort in the Hawke's Bay:

$$E = 31555.6 + 192.8SL \qquad [2.32]$$

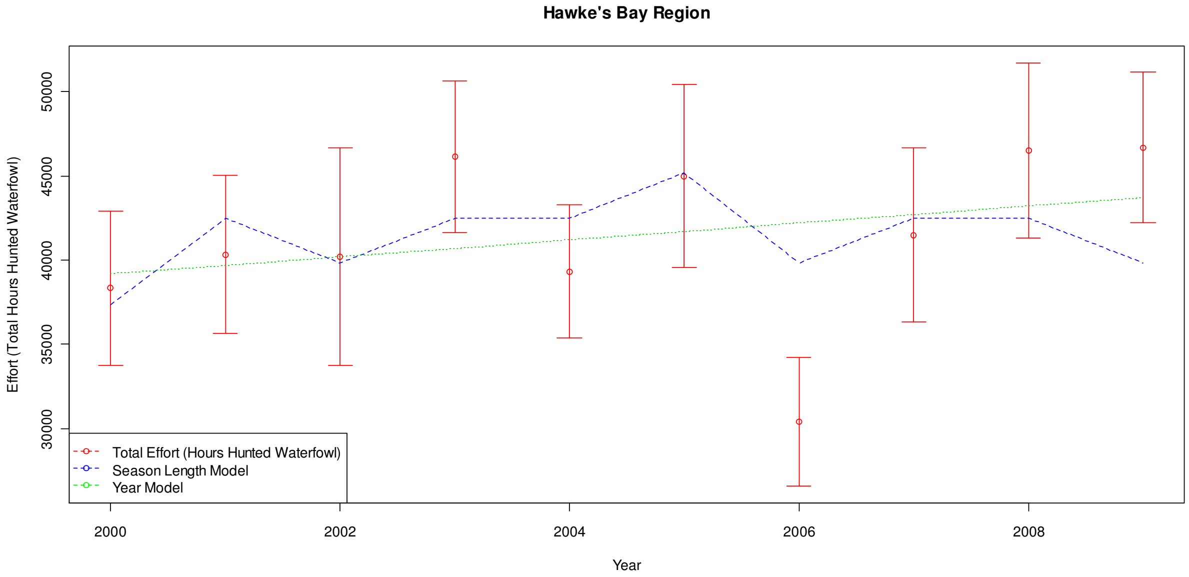

Figure 2.22: Comparisons of *Season Length* and *Year* models and effort (hours hunted waterfowl) in the Hawke's Bay estimated from the hunter survey.

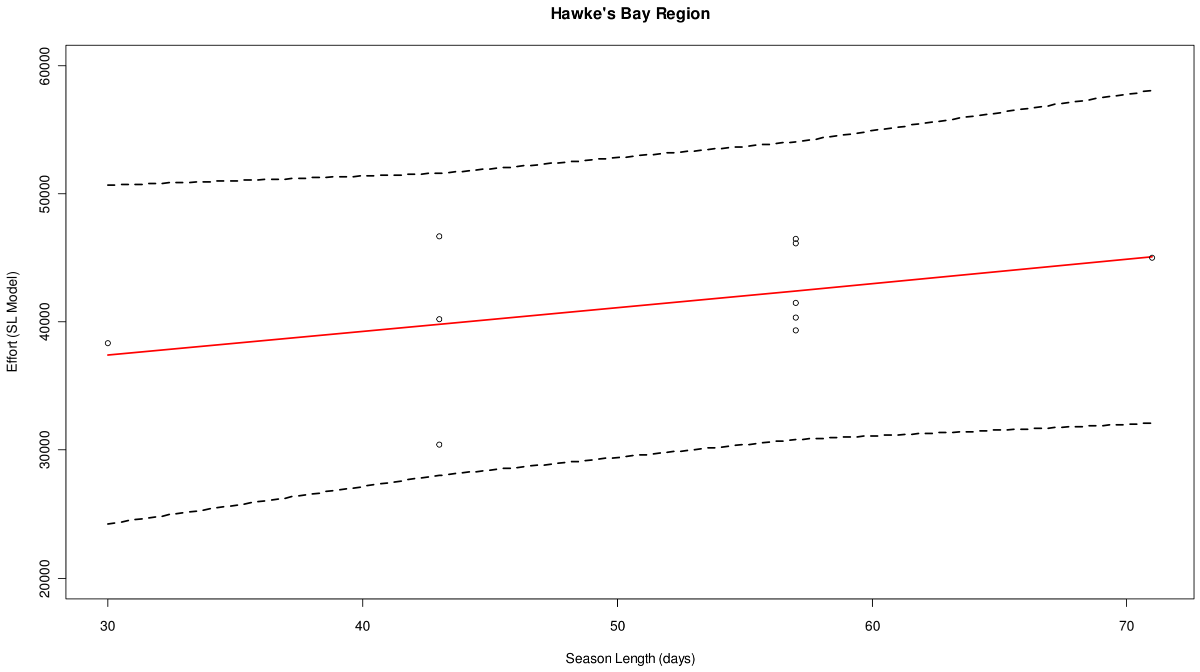

Figure 2.23. Hawke's Bay effort derived from the season length model (red line) with 95% credible interval (dashed line) (1000 burn in, 100,000 simulations in WINBUGs). Points show estimated effort from the hunter survey.

The flat shape of the graph (Figure 2.23) is indicative of the small influence that season length has on hunter effort.

Season Regulations and Survival

It was anticipated that survival rate would decrease with increasing effort, and harvest. Further that survival would decrease with season constraint.

Eastern Survival

Goodness of fit of the global modal suggested the data did not fit the assumptions (P <0.01); accordingly $\hat{C}$ was adjusted (1.1304) (Table 2.7).

The *Effort* model (survival is constrained as a linear function of total effort) received the greatest support (87%). Weight of evidence ($\frac{w_1}{w_2}$; the relative weight of the two models) over the second ranked model, *Season Length,* was clear (18).

The additive *Effort* model ($Logit(\underline{\hat{S}}) = \beta_1 + \beta_2 Sex + \beta_3 E + \beta_4 age$) did not improve QAICc ranking, that is, increasing effort affected the sexes differently (weight of evidence in favour of the model incorporating the sex-effect was in excess of 300).

Estimates of adult and juvenile female survival as a function of effort can be seen in Figure 2.24 and Figure 2.25.

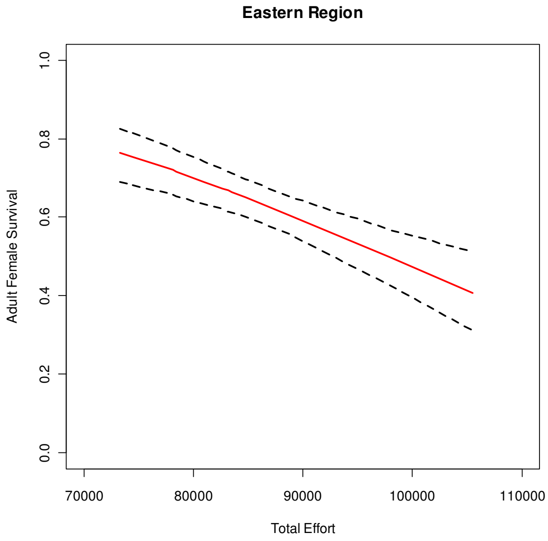

Figure 2.24. Adult female survival as a function of total effort (hours spent hunting waterfowl) in the Eastern Region extrapolated from the *Effort* model. Black dashed line 95% CI.

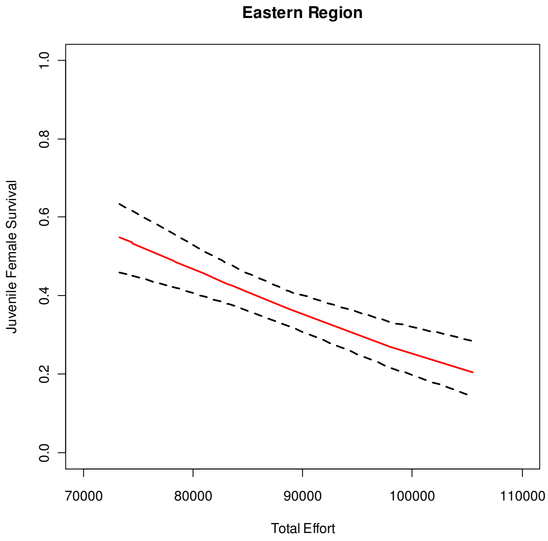

Figure 2.25. Juvenile female survival as a function of total effort (hours spent hunting waterfowl) in the Eastern Region extrapolated from the *Effort* model. Black dashed line 95% CI. .

The vector of beta estimates for the *Effort* model[1] is given in Table 2.5 and for the *Season Length* in Table 2.6. Interpolation of the Beta estimates using the logit function suggests that for an average total effort of 83500 hours hunting waterfowl we might expect annual adult female survival to be 66.5%. A 10% increase in effort would see a 14% decrease in survival and a 10% decrease in effort may result in a 12% increase in adult female survival[2].

---

[1] The beta (β) estimates are the parameter estimates for the logit model for example β_1 is the estimate of the intercept value for the model $\underline{\hat{S}} = \beta_1 + \beta_2 Sex + \beta_3 Age + \beta_4 Effort + \beta_5 Sex.Effort$

[2] Note the effect size (increase, decrease in survival) is inconsistent as the back transformation on the logit scale is not linear.

Table 2.5. Beta estimates of linear Effort model for the Eastern Region

Parameter	Beta	SE	95% Confidence Interval	
Survival Intercept (β_1)	1.1416462	0.5817415	0.0014329	2.2818595
Sex (β_2)	3.5544631	0.958988	1.6748467	5.4340795
Age (β_3)	-0.9796978	0.0753614	-1.1274062	-0.8319894
Effort (β_4)	-0.0000069	0.0000068	-0.0000203	0.0000064
Sex*Effort (β_5)	-0.0000411	0.000011	-0.0000627	-0.0000196

Although the relative level of support for the *Season Length* model was considerably less (ΔQAICc 5.7, *wi*=0.047, Table 2.7) than the *Effort* model it still is interesting to examine the respective survival rates attributed to the different season lengths. The effect of regulations on juvenile female survival appears more pronounced (Figure 2.27) cf. adult females (Figure 2.26). It is also apparent that the high effort reported for the 43 day season (Figure 2.8) (and corresponding high harvest rate, Figure 2.35), has influenced juvenile female survival rates, with survival rates lower than the 57 days seasons (P=0.03, $t_{(0.05)(2)(379)}$= 2.1669).

Table 2.6. Beta estimates of linear SL adult female mallard survival model for the Eastern Region.

Parameter	B	SE	Lower (95%CI)	Upper (95% CI)
Survival Intercept	1.3036585	0.4023911	0.5149719	2.0923451
Sex	1.0536488	0.3895007	0.2902275	1.8170701
Age	-0.9242892	0.0754635	-1.0721977	-0.7763807
SL	-0.0130448	0.0067121	-0.0262005	0.0001109
Sex*SL	-0.0180165	0.0064288	-0.030617	-0.005416

The scatterplot & density matrix graph of adult and juvenile female survival (model averaging) against total mallard harvest and season length (Figure 2.28) are also interesting, particularly the adult females where survival does not appear linear. The rate of decrease in adult female survival appears to increase once harvest exceeds about 50000. This may explain why the total effort explained changes in survival better than total mallard harvest. Also of note is the high degree of variability in survival for both cohorts relative to season length particularly for the 71 day season. The high degree of fit between effort and survival is to be expected based on the relative weight that this model received in the model averaging process.

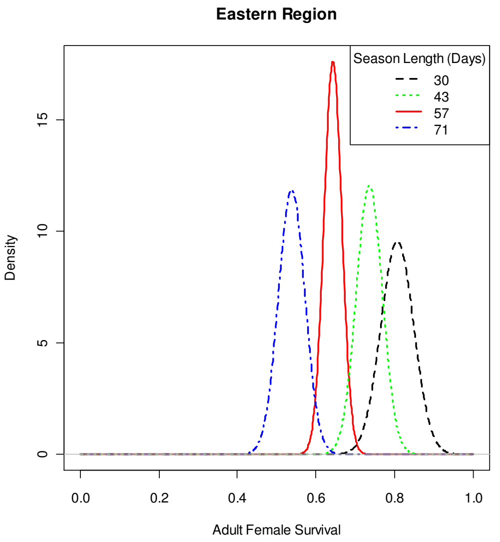

Figure 2.26. Probability distributions of annual adult female survival in Eastern Region mallard, against season lengths of 30, 43, 57 and, 71 days from second top ranked season length model (ΔQAICc 5.767).

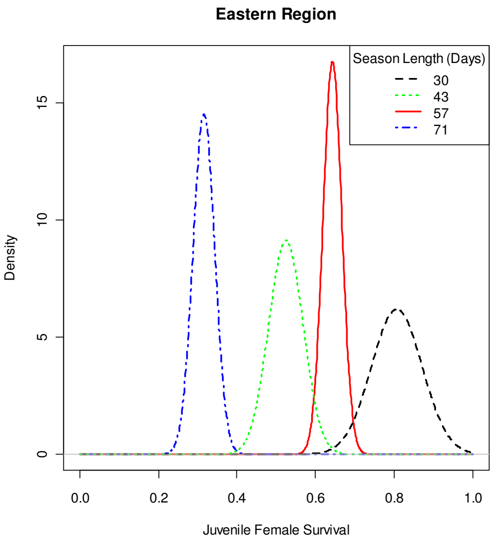

Figure 2.27. Probability distributions of annual juvenile female survival in Eastern Region mallard, against season lengths of 30, 43, 57 and, 71 days from second top ranked season length model (ΔQAICc 5.767)..

Table 2.7. Models of survival, recapture, recovery and fidelity parameters of 15,747 mallard banded 1997-2009 in the Eastern Region models ranked on QAICc (c-hat 1.1304). *S*= Survival, *p*=recapture, *r*= conditional recovery rate, *F*= fidelity, *SL*= Season length, *BL*= Bag Limit, *H*= total harvest; *E*= effort; *WR*= winter rain, WT= winter temperature SuR = summer rain, *SuT*= summer temperature, *SppT*= spring temperature in the year prior to survival year, *SpcT* = spring temperature in the current year, *SppR*= spring rain in year prior to survival year. Climate data extrapolated from Whakatane AWS.

Model	QAICc	Delta QAICc	AICc Weights	Model Likelihood	Num. Par	QDeviance
{S(sex*E+age) p(g*t) r(age*t) F(g*t)}	20531.63	0	0.85104	1	123	949.5843
{S(sex*SL+age) p(g*t) r(age*t) F(g*t)}	20537.39	5.7677	0.04759	0.0559	123	955.352
{S(sex*SL+E+age) p(g*t) r(age*t) F(g*t)}	20538.26	6.6392	0.03078	0.0362	124	954.1931
{S(sex*Eave+age) p(g*t) r(age*t) F(g*t)}	20539.32	7.6908	0.01819	0.0214	122	959.3053
{S(sex*H+age) p(g*t) r(age*t) F(g*t)}	20540.07	8.441	0.0125	0.0147	123	958.0252
{S(sex*(BL+SL)+age) p(g*t) r(age*t) F(g*t)}	20540.16	8.5373	0.01192	0.014	125	954.0611
{S(sex*BL+age) p(g*t) r(age*t) F(g*t)}	20540.82	9.1991	0.00856	0.0101	123	958.7839
{S(sex*SpcT+age) p(g*t) r(age*t) F(g*t)}	20542.66	11.0365	0.00342	0.004	123	960.6208
{S(sex*WR+age) p(g*t) r(age*t) F(g*t)}	20542.98	11.3576	0.00291	0.0034	123	960.9421
{S(sex+E+age) p(g*t) r(age*t) F(g*t)}	20543.07	11.4461	0.00278	0.0033	122	963.0608
{S(sex*SuR+age) p(g*t) r(age*t) F(g*t)}	20543.16	11.5346	0.00266	0.0031	123	961.1195
{S(sex+SL+E+age) p(g*t) r(age*t) F(g*t)}	20544.07	12.4492	0.00169	0.002	123	962.0335
{S(sex*WT+age) p(g*t) r(age*t) F(g*t)}	20544.29	12.6615	0.00152	0.0018	123	962.2465
{S(sex+Hrs+age) p(g*t) r(age*t) F(g*t)}	20544.57	12.949	0.00131	0.0015	122	964.5639
{S(sex*SpcR+age) p(g*t) r(age*t) F(g*t)}	20544.7	13.0765	0.00123	0.0014	123	962.6608
{S(sex*SppR+age) p(g*t) r(age*t) F(g*t)}	20548.58	16.9555	0.00018	0.0002	123	966.5397
{S(sex*SppT+age) p(g*t) r(age*t) F(g*t)}	20549.01	17.3836	0.00014	0.0002	123	966.9681
{S(sex*SuT+age) p(g*t) r(age*t) F(g*t)}	20549.43	17.8047	0.00012	0.0001	123	967.3889
{S((sex+age)*t) p(g*t) r(age*t) F(g*t)}	20573.51	41.8882	0	0	171	893.7298
{S(g) p(g*t) r(g*t) F(g*t)}	20641.77	110.1404	0	0	149	1006.852
{S(sex*E+age) p(g*t) r(age*E) F(g*t)}	20642.03	110.409	0	0	100	1106.626
{S(g*t) p(g*t) r(g*t) F(g*t) PIM}	20684.21	152.5858	0	0	193	959.4356
{S(sex*t) p(g*t) r(age*t) F(g*t)}	20705.97	174.3471	0	0	144	1081.24

Model	QAICc	Delta QAICc	AICc Weights	Model Likelihood	Num. Par	QDeviance
{S(sex) p(g*t) r(age*t) F(g*t)}	20709.73	178.1003	0	0	120	1133.775
{S(sex) p(g*t) r(age) F(g*t)}	20816.92	285.2943	0	0	96	1289.607
{S(.) p(.) r(.) F(.)}	20899.76	368.1363	0	0	5	1555.584

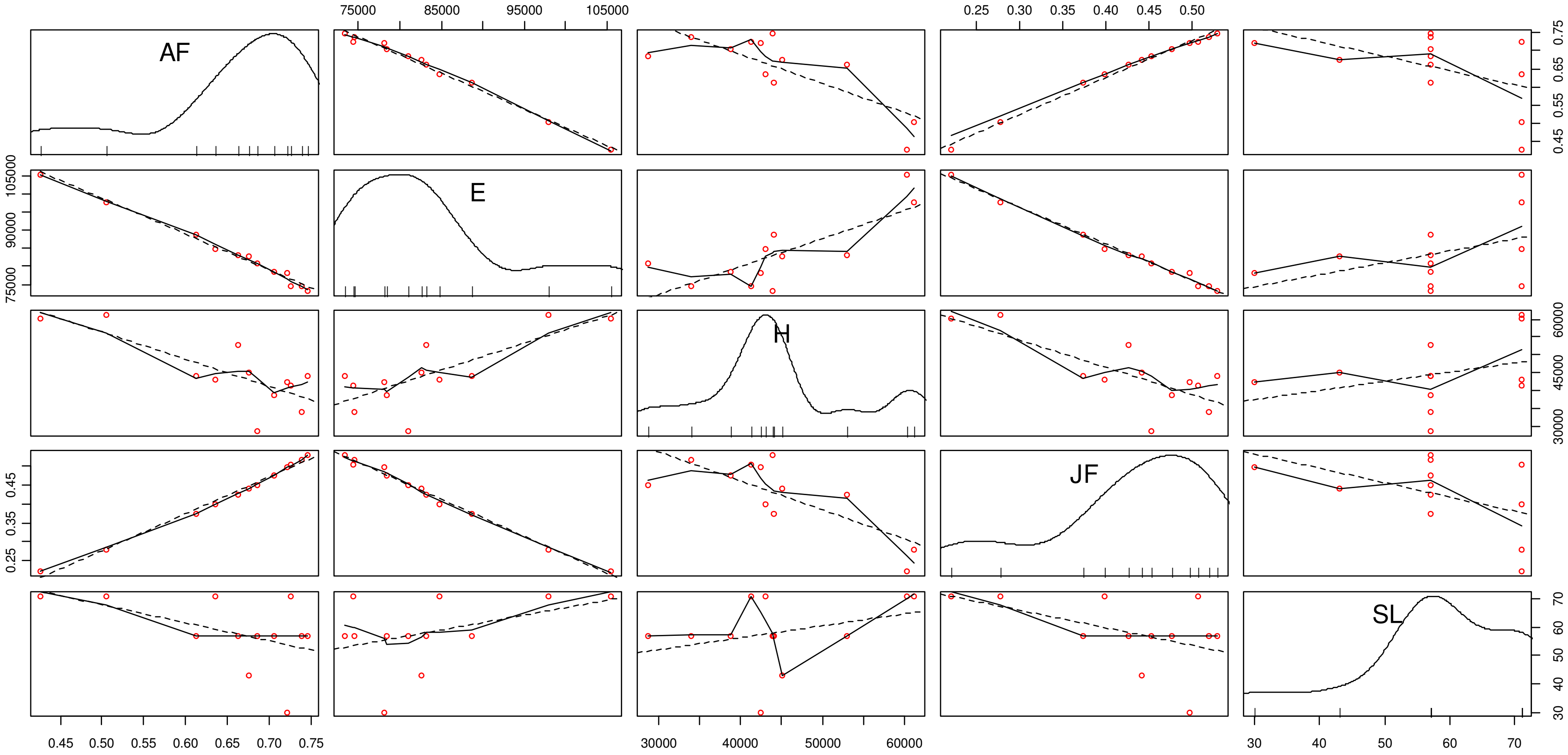

Figure 2.28. Scatter-plot and density matrix of Eastern Region adult female (AF) & juvenile female (JF) survival (model averaging), total mallard harvests (H), effort (E) and, season length (SL). To interpret this graph read the intersection of the parameters of interest. For example, for the affect harvest levels (H) has on adult female survival (AF), cross from the "AF" in the top left hand corner to intersection of the harvest density (H) column (i.e. the top row, middle column). This is a scatter-plot of harvest and adult female survival with the values of the harvest given at the base of the middle column and the values for the adult female survival at the right hand end of the top row. The black dashed line is the best fit line and the solid line the lowess (locally weighted scatter-plot smoothing).

Hawke's Bay Survival

Goodness of fit analysis of the Hawke's Bay global model suggested the data did not fit all of the assumptions (*P* <0.01), therefore $\hat{C}$ was adjusted (1.1586) accordingly.

The *Season Length* model was the top model (w_i=0.33) in the Hawke's Bay data set but the degree of support relative to the next top model (*Effort* ΔAICc=0.74, w_i=0.23; Figure 2.31 and Figure 2.32) was not strong (relative weight = 1.4). Survival distributions from the *Season Length* model are shown in Figure 2.29 for adult females and Figure 2.30 juvenile females.

The Beta estimates for the SL model are given in Table 2.8.

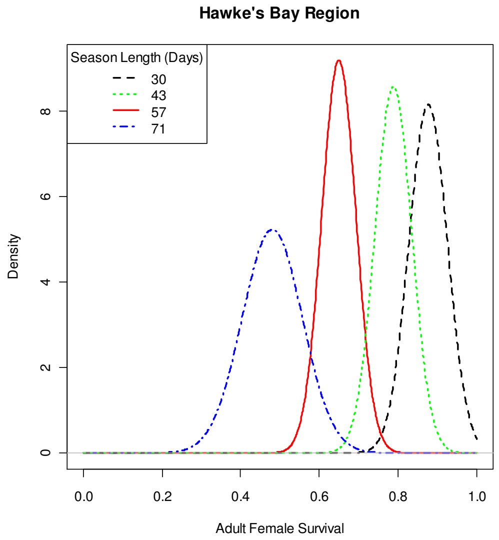

Figure 2.29. Distributions of adult female survival in Hawke's Bay Region mallard, against season lengths of 30, 43, 57 and ,71 days from top ranked *Season Length* model .

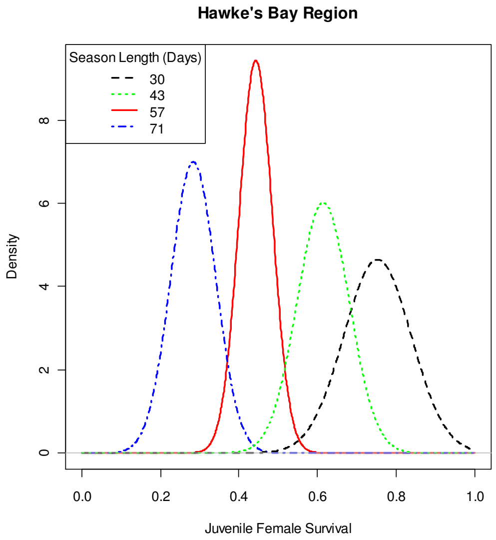

Figure 2.30. Distributions of juvenile female survival in Hawke's Bay Region mallard, against season lengths of 30, 43, 57 and,71 days from top ranked Season Length model .

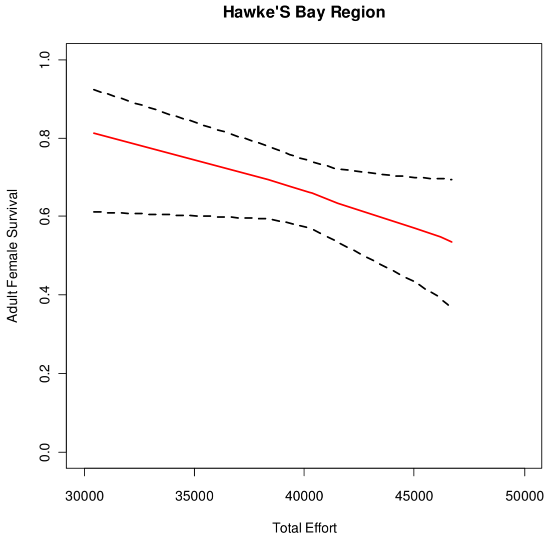

Figure 2.31. Annual adult female survival as a function of effort in the Hawke's Bay extrapolated from the Effort model (ΔQAICc=0.739). Black dashed line is 95% CI.

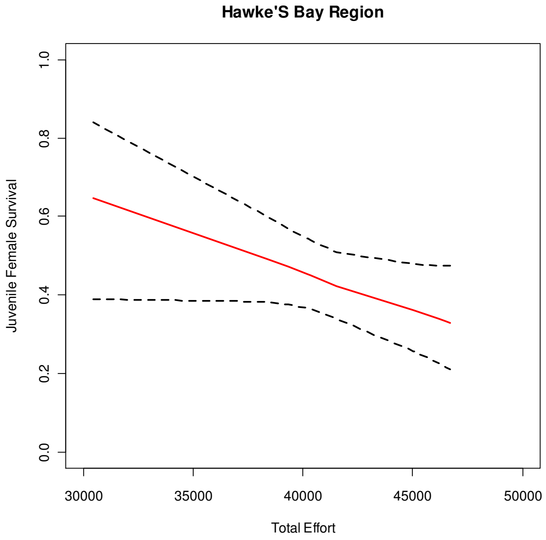

Figure 2.32. Annual juvenile female survival as a function of effort in the Hawke's Bay extrapolated from the *Effort* model (ΔQAICc=0.739). Black dashed line is 95% CI

Table 2.8. Beta estimates of linear SL adult female mallard survival model for the Hawke's Bay Region.

Parameter	B	SE	Lower (95%CI)	Upper (95% CI)
Survival Intercept	1.748243	0.593575	0.584837	2.91165
Sex	1.697045	0.639758	0.443119	2.950971
Age	-0.84655	0.121436	-1.08456	-0.60853
SL	-0.02201	0.010534	-0.04266	-0.00137
Sex*SL	-0.02763	0.011439	-0.05005	-0.00521

Table 2.9: Models of survival, recapture, recovery and fidelity parameters of 6777 mallard banded 2000-2009 in the Hawke's Bay Region models ranked on QAICc (c-hat 1.1586). *S*= Survival, *p*=recapture, *r*= conditional recovery rate, *F*= fidelity, *SL*= Season length*, BL*= Bag Limit, *WR*= winter rain, WT= winter temperature SuR = summer rain, *SuT*= summer temperature, *SppT*= spring temperature in the year prior to survival year, *SpcT* = spring temperature in the current year, *SppR*= spring rain in year prior to survival year. Climate data extrapolated from Napier AWS.

Model	QAICc	Δ QAICc	QAICc Weights	Model Likelihood	Num. Par	QDeviance
{S(sex*SL+age) p(g*t) r(age*t) F(g*t)}	8533.086	0	0.33446	1	91	517.9676
{S(sex*E+age) p(g*t) r(age*t) F(g*t)}	8533.825	0.739	0.23114	0.6911	91	518.7066
{S(sex*SpcT+age) p(g*t) r(age*t) F(g*t)}	8534.621	1.5355	0.15521	0.4641	91	519.5031
{S(sex*SuT+age) p(g*t) r(age*t) F(g*t)}	8535.438	2.3518	0.1032	0.3086	91	520.3194
{S(sex*Drake BL+age) p(g*t) r(age*t) F(g*t)}	8535.669	2.5831	0.09193	0.2749	91	520.5507
{S(sex*SppR+age) p(g*t) r(age*t) F(g*t)}	8537.599	4.5126	0.03503	0.1047	91	522.4802
{S(sex*WT+age) p(g*t) r(age*t) F(g*t)}	8539.596	6.5104	0.0129	0.0386	91	524.478
{S(sex*SpcR+age) p(g*t) r(age*t) F(g*t)}	8539.814	6.7281	0.01157	0.0346	91	524.6958
{S(sex*SuR+age) p(g*t) r(age*t) F(g*t)}	8539.974	6.888	0.01068	0.0319	91	524.8556
{S(sex*SppT+age) p(g*t) r(age*t) F(g*t)}	8540.836	7.7504	0.00694	0.0207	91	525.718
{S(sex*WR+age) p(g*t) r(age*t) F(g*t)}	8540.94	7.8542	0.00659	0.0197	91	525.8218
{S(g) p(g*t) r(g*t) F(g*t)}	8546.78	13.6943	0.00036	0.0011	110	492.5469
{S(sex+age)*t p(g*t) r(age*t) F(g*t)}	8565.918	32.8316	0	0	125	480.652
{S(g*t) p(g*t) r(g*t) F(g*t)}	8581.477	48.391	0	0	142	460.8782
{S(sex*Hen BL+age) p(g*t) r(age*t) F(g*t)}	8583.054	49.9676	0	0	91	567.9352
{S(sex) p(g*t) r(age*t) F(g*t)}	8590.802	57.7161	0	0	88	581.8402
{S(.) p(.) r(.) F(.)}	8655.989	122.903	0	0	5	815.2733

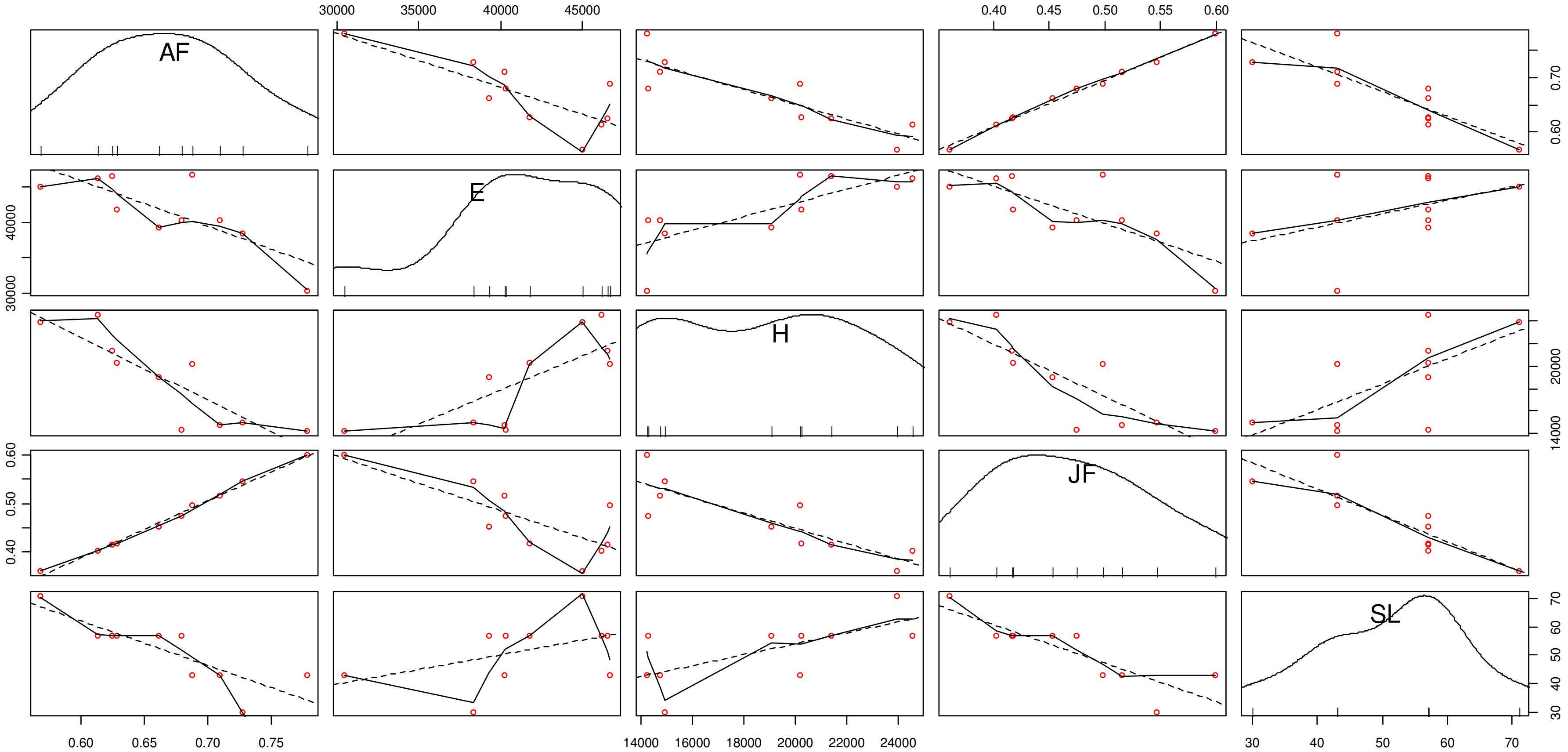

Figure 2.33. Scatter-plot and density matrix of Hawke's Bay Region adult female (AF) & juvenile female (JF) survival (model averaging), total mallard harvests (H), effort (E) and, season length (SL). To interpret this graph read the intersection of the parameters of interest. For example, for the affect harvest levels (H) has on adult female survival (AF), cross from the "AF" in the top left hand corner to intersection of the harvest density (H) column (i.e. the top row, middle column). This is a scatter-plot of harvest and adult female survival with the values of the harvest given at the base of the middle column and the values for the adult female survival at the right hand end of the top row. The black dashed line is the best fit line and the solid line the lowess (locally weighted scatter-plot smoothing).

Harvest and Harvest Rates

The heuristic harvest model proposes season regulations constrain effort which in turn influence harvest rates. An increase in effort should result in an increase in harvest rate. Harvest rate, however, will be influenced by amongst other things; environmental factors (for example flooding may disperse birds), topographical features (hill country will be different to lowland), and hunter density (low hunter density may result in birds not being disturbed and moved on reducing their vulnerability to harvest).

Eastern Region

Reporting Rate: There was no evidence of any trend (R=0.452, $t_{(11)}$ = 1.6787, P =0.12) in reporting rate (0.62, *SE*=0.0014) in the Eastern Region over the study period. Harvest rate (h) was therefore calculated using the average reporting rate for the period as:

$$\hat{h}_i = \frac{\hat{f}_i}{0.62} \qquad [2.33]$$

where f_i is the recovery rate in the year of banding.

Season Regulations: Juvenile harvest rates ($\widehat{h}_J$=0.15; *SE*=5.28x10^{-7}) were significantly (P<0.0001, $t_{(0.05)(1)(15746)}$=1200,) higher than the adults ($\widehat{h_A}$=0.09; *SE*=5.07x$10^{-7}$and were influences more by season length (Figure 2.35) than adults (Figure 2.34) (0.09 ≤$\widehat{h}_J$ ≥0.24 cf. 0.05≤$\widehat{h_A}$ ≥0.11) also were higher for the equivalent SL (P = 0.014).

Excluding the 43 days season there was evidence that increasing season length increased harvest rate (Table 2.10) in the juveniles for the 30-57 day SL (One-tailed *t –test*; $t_{(.05)(1)(557)}$=-5.643, P<0.001,) and 57-71 day SL ($t_{(0.05)(1)(765)}$=-46.4918, P<0.001). A difference in adult harvest rates 30-57 day SL was not evident ($t_{(0.05)(1)(124)}$=-1.430, P=0.77), but was significant for the 57-71 day SL ($t_{(0.05)(1)(223)}$=-8.457, P<0.001).

The correlation between total mallard and parera harvest and season length is not significant (R=0.36, $t_{(10)}$=1.2251, P=0.248). This will be due in part to the extended period (2002-2006) where season length was fixed to 57 days.

Harvest Rate and Effort: The correlation between effort and harvest rate was not significant for adults (R = 0.371, $t_{(10)}$ = 1.2628, P = 0.235, Figure 2.36) or juveniles (R=0.326, $t_{(10)}$ = 1.0907, P = 0.301, Figure 2.37). However when the 1997 data point is removed[1] for the adults the degree of significance improves (R = 0.595, $t_{(9)}$ = 2.2219, P= 0.0534). Removing the 1997 data point made little difference to the juvenile correlation. I thought this may be due to juveniles roaming further once banded but the percentage of recovered adults (both regions) further than 100 km from the banding site was higher than the juveniles (10.0 cf. 6.2%). Because the correlation between harvest and effort is so strong (R=0.76, $t_{(10)}$ = 3.7756, P = 0.004, Figure 2.28) I assume that reporting rate variation is affecting interpretation of the affect effort is having on harvest rate?

The linear model of adult harvest rate as a function of effort for a given season length (SL) is:

$$h_{A,SL} = 0.000565 + 0.00000091E_{SL} \qquad [2.34]$$

and juveniles:

$$h_{J,SL} = 0.0154 + 0.00000151E_{SL} \qquad [2.35]$$

The relationship between SL and h was probably compromised by effort per day increasing with shorter season length. This is evident in the 43 day season where the highest harvest rate was recorded for both the adults (0.109, SE=0.0007) and juveniles (0.209, SE=0.00076) (Table 2.10).

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[1] 1997 was the first year of the study, where only 265 adults were banded, over half of which were banded around Lake Rotorua which receives very little harvest pressure. It is likely that the true harvest rate is higher than the estimate.

Table 2.10. Average harvest rates in the Eastern Region as a function of season length (SL).

SL	$\hat{h}_A$	$\sigma^2_{\hat{h}_A}$	$\hat{h}_J$	$\sigma^2_{\hat{h}_J}$
30	0.070	0.000372	0.113	0.000265
43	0.109	0.000294	0.209	0.000448
57	0.078	0.000062	0.126	0.000057
71	0.087	0.000078	0.164	0.000141

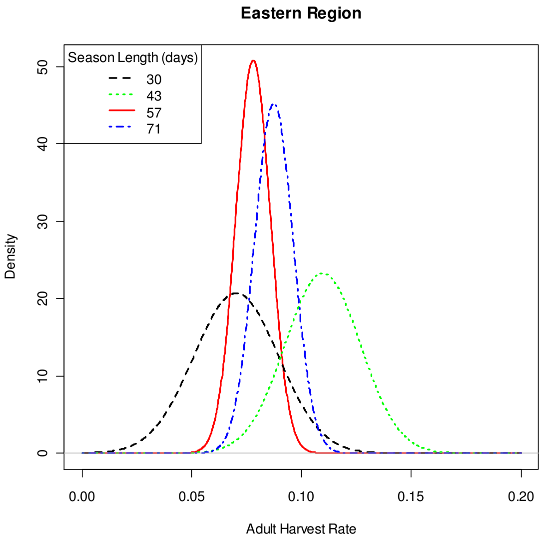

Figure 2.34. Annual adult harvest rate reported as a function of season length (days); Eastern Region

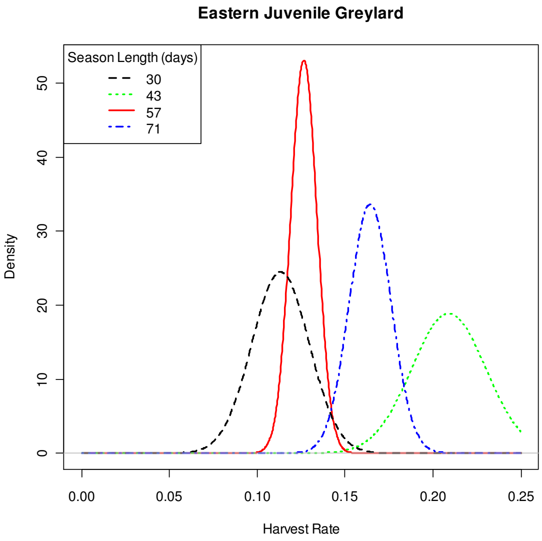

Figure 2.35. Annual juvenile harvest rate reported as a function of season length (days); Eastern Region.

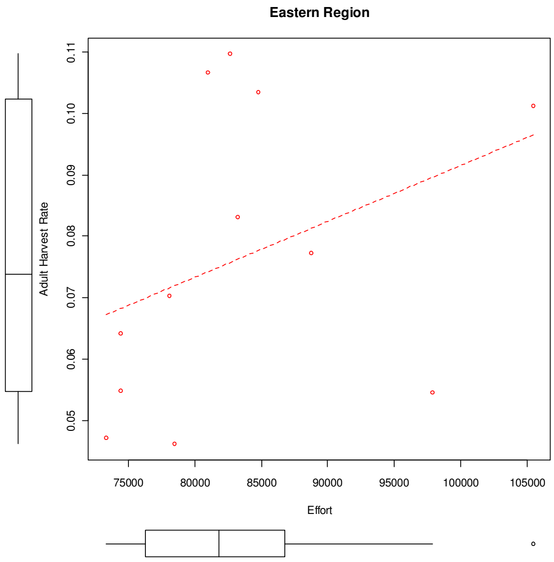

Figure 2.36. Scatterplot of adult harvest rate against total effort (hours hunting waterfowl) for the Eastern Region ($R = 0.371$, $t_{(10)} = 1.2628$, $P = 0.235$).

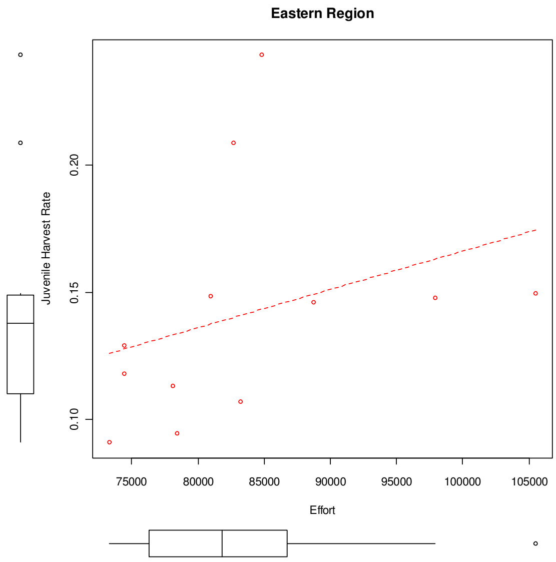

Figure 2.37. Scatterplot of juvenile harvest rate against total effort (hours hunting waterfowl) for the Eastern Region (R=0.326, $t_{(10)} = 1.0907$, $P = 0.301$).

Harvest (H = 60828-2168.2Yr) and effort (E = 98209-2255.9Yr) are declining in parallel (R=0.85, $t_{(10)}$=5.3193, P<0.001; Figure 2.38). Harvest rate shows a similar, if not precarious, trend (Figure 2.39).

There was no correlation between hen bag limits (R = -0.017, P = 0.956) or drake bag limits (R = 0.015, P = 0.961) and harvest rates. In fact the second highest hen bag limits corresponded with the lowest harvest rate (Figure 2.40). An examination of harvest rate during the period of fixed season length (57 days) shows a higher harvest rate during the restricted bag limit (<10) seasons (Figure 2.41).

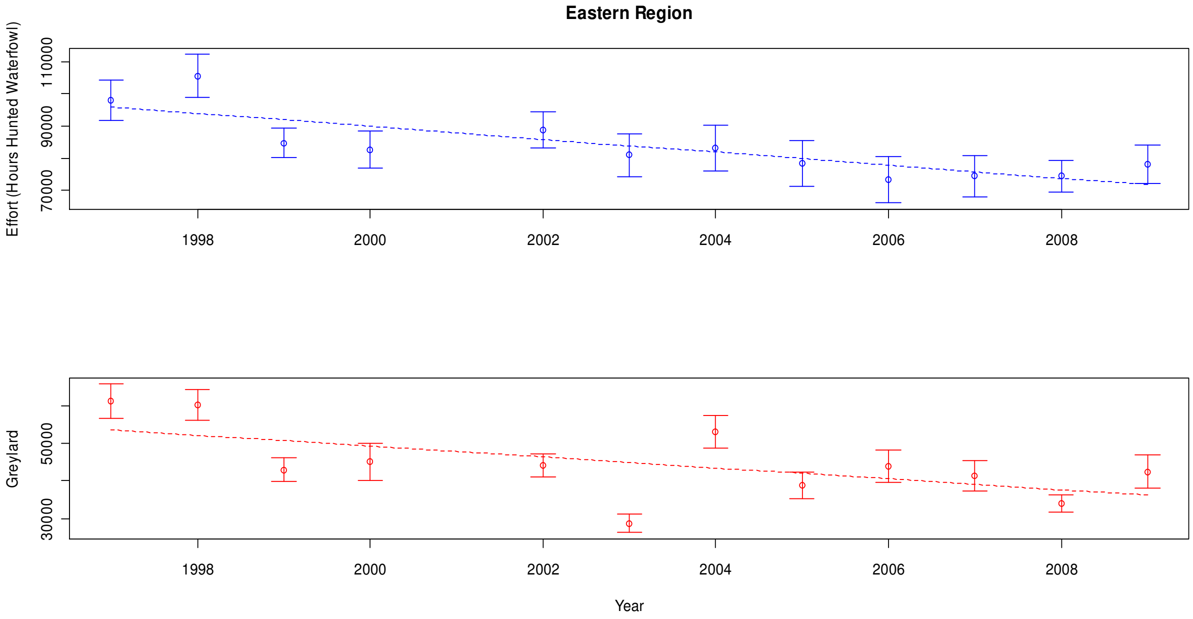

Figure 2.38. Total hours hunting waterfowl, effort (top graph) and annual mallard harvest (bottom graph) for the Eastern Region (year 2001 data not included). The fitted regression lines (dashed lines) show similar trend.

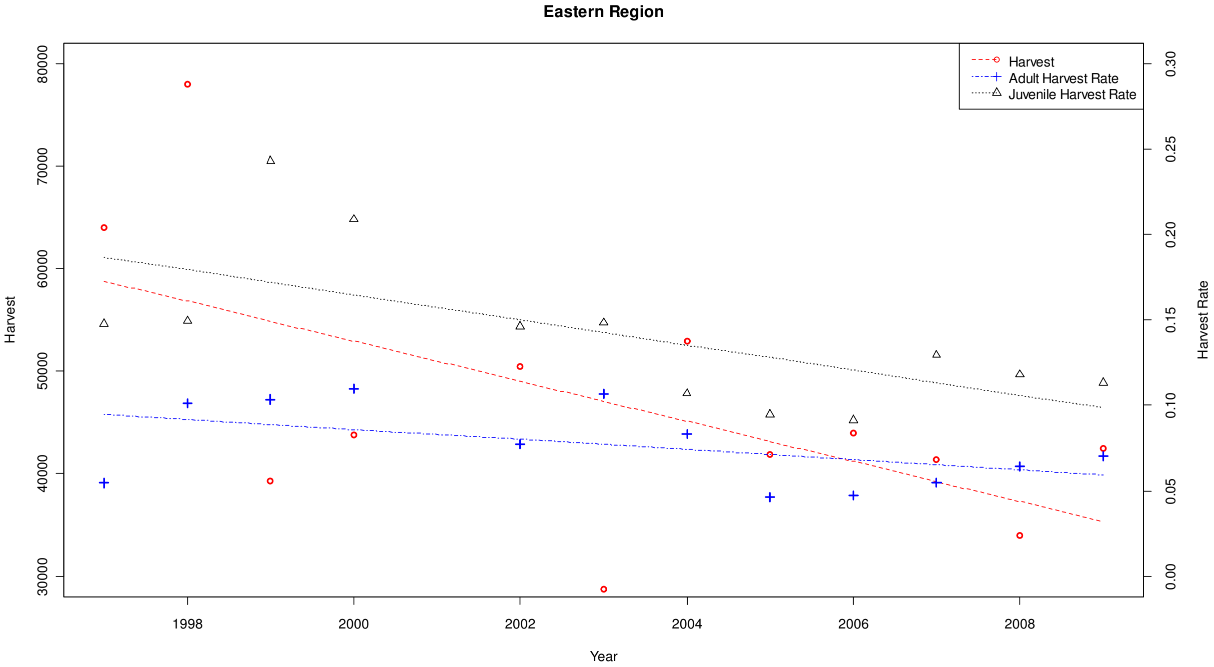

Figure 2.39. Harvest (red points), adult and, juvenile harvest rate (blue and black marks); for the Eastern Region (year 2001 data not included) with fitted regression lines (dashed) show a downward trend over the study period.

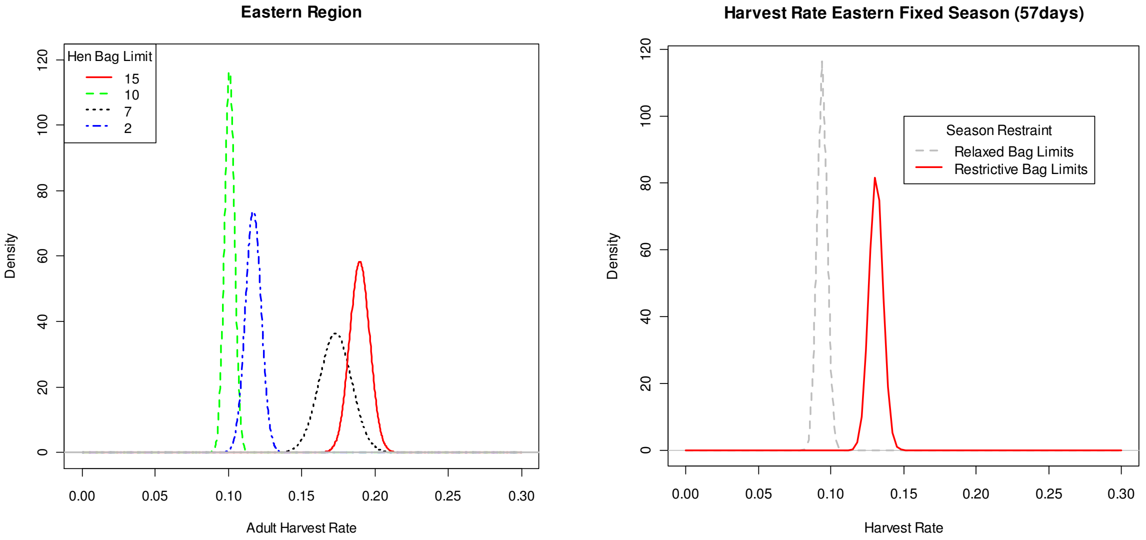

Figure 2.40 Annual harvest rate reported against hen bag limits Eastern Region

Figure 2.41. Annual harvest rate under different bag limits for a fixed season length of 57 days in the Eastern Region. The more restrictive bag limits (<10) have a higher harvest rate.

Hawke's Bay

Reporting Rate: There was no evidence that Hawke's Bay reporting rate (0.466; SE=0.0018) was changing with time (*R*=0.13, *P*=0.738). Therefore I defined the harvest rate as:

$$\hat{h}_i = \frac{\hat{f}_i}{0.466} \qquad [2.36]$$

Where $\hat{f}_i$ is the recovery rate in the year of banding.

It appears that a geographical variation in reporting rate exists between Eastern and Hawke's Bay regions ($t_{(0.05)(2)(633)}$=70.9; P<0.001).

Juvenile harvest rate ($\hat{h}_J$ =0.189; SE=2.99x10^{-6}) is significantly greater ($t_{(6775)}$=534, P=0) than adults ($\hat{h}_A$ =0.123, SE=3.64x10^{-6}).

Season Regulations: There was not a positive correlation between season length and harvest rate for either adults (Figure 2.42) or juveniles (Figure 2.43). As season length increased estimated juvenile harvest rate ($\hat{h}_J$) decreased (R=-0.658, P=0.0387) while adult harvest rate was positive but not significant (R=0.034, P=0.925) (Table 2.11). In case the anomalous negative *SL* - juvenile harvest rate correlation was a function of reporting rate I allowed reporting rate to vary annually. The correlation changed to positive but was not significant (R=0.145, P=0.709; Figure 2.45). Allowing reporting rate to vary annually improved correlation between SL and $\hat{h}_A$ but again it was not significant (R=0.346, P=0.362; Figure 2.44). If the 2001 data were removed (the lowest reporting rate in the survey; 0.22) the adult harvest rate approached significance at the 5% level (P=0.08) but made little difference to the correlation between season length and juvenile harvest rates.

There was no correlation between harvest rate in the Hawke's Bay and hen bag limits for adults (R=0.04, P=0.952, Figure 2.48) or juveniles (R=0.16, P=0.833; Figure 2.49).

Table 2.11. Average harvest rates in the Hawke’s Bay Region as a function of season length. Reporting rate was fixed at 0.47.

SL	$\hat{h}_A$	$\sigma^2_{\hat{h}_A}$	$\hat{h}_J$	$\sigma^2_{\hat{h}_J}$
30	0.132	0.00079	0.334	0.00437
43	0.093	0.00059	0.189	0.00061
57	0.127	0.00021	0.193	0.00029
71	0.113	0.00114	0.136	0.00053

Harvest Rate and Effort: There was also no obvious cause and effect between effort and harvest rate for Hawke’s Bay juveniles (*R*=0.0176, $t_{(8)}$ = 0.0499, *P* = 0.96, Figure 2.47) or adults (*R*=0.346, $t_{(8)}$ = 1.0435, *P*= 0.327, Figure 2.46). Because of the good correlation between harvest and effort (*R*=0.76, $t_{(8)}$ = 3.3878, *P* = 0.0095, Figure 2.33) and season length and harvest (*R*=0.66, $t_{(8)}$ = 2.4931, *P* = 0.037, Figure 2.33) I suspect that reporting rate variability is compromising the harvest rate results. Reporting rate in the first year of the Hawke’s Bay study (2000) was the highest (0.875) which also coincided with the shortest season (30 days). The second lowest reporting rate coincided with the only 71 day season (0.4) which explains the relatively low harvest rate. The combination of these two data points explains the negative season length - harvest rate correlation for the juveniles.

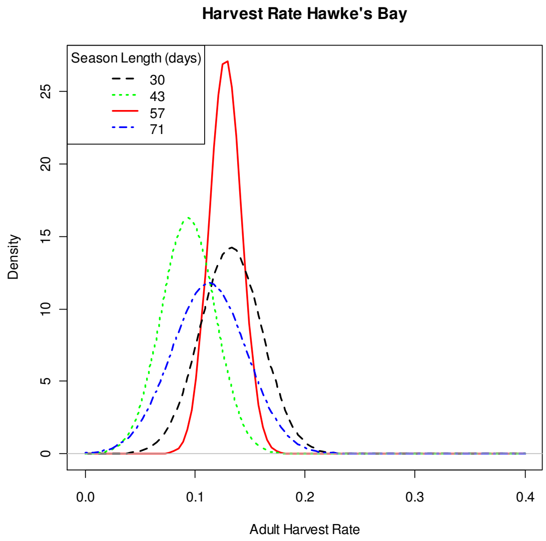

Figure 2.42. Annual adult harvest rate reported against season length (days) Hawke’s Bay Region.

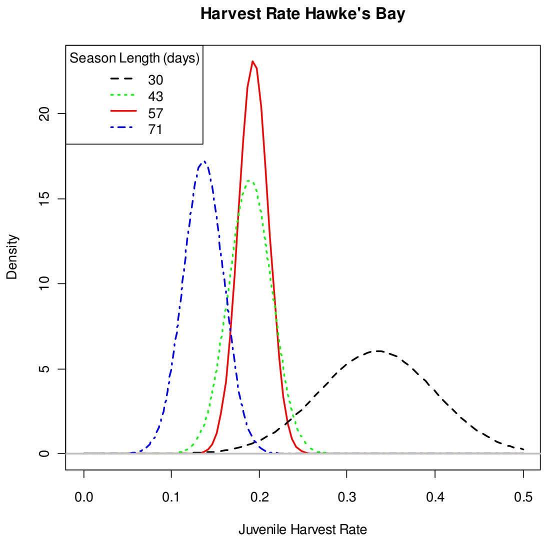

Figure 2.43. Annual juvenile harvest rate reported against season length (days) Hawke’s Bay Region.

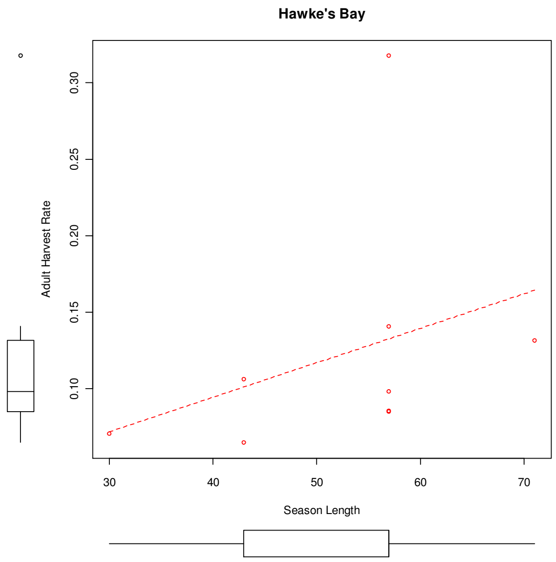

Figure 2.44. Scatterplot of Hawke’s Bay adult harvest rate against season length (*R*=0.346, *P*=0.362). Reporting rate was allowed to vary annually.

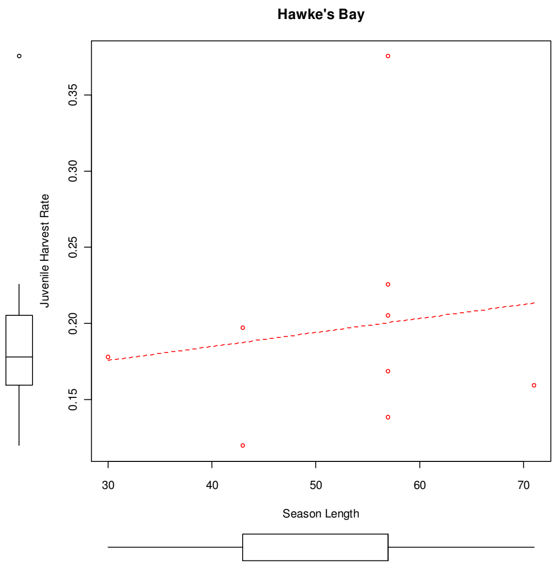

Figure 2.45. Scatterplot of Hawke’s Bay juvenile harvest rate against season length (*R*=0.145, *P*=0.709). Reporting rate was allowed to vary annually.

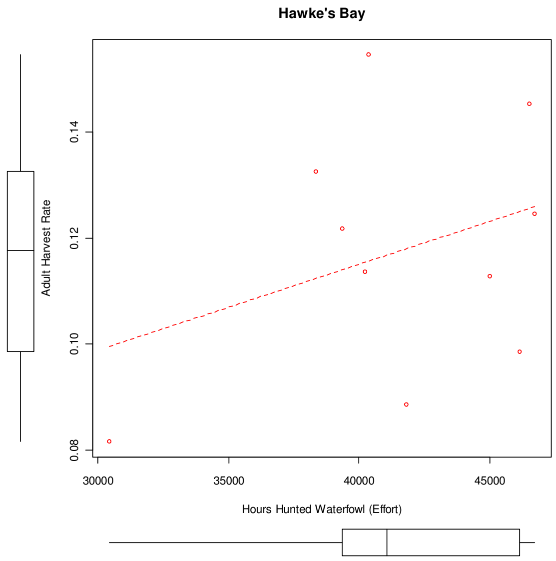

Figure 2.46. Hawke’s Bay annual adult harvest rates against total effort (hours hunted waterfowl) (*R*=0.35, $t_{(8)}$ = 1.0435, *P* = 0.327).

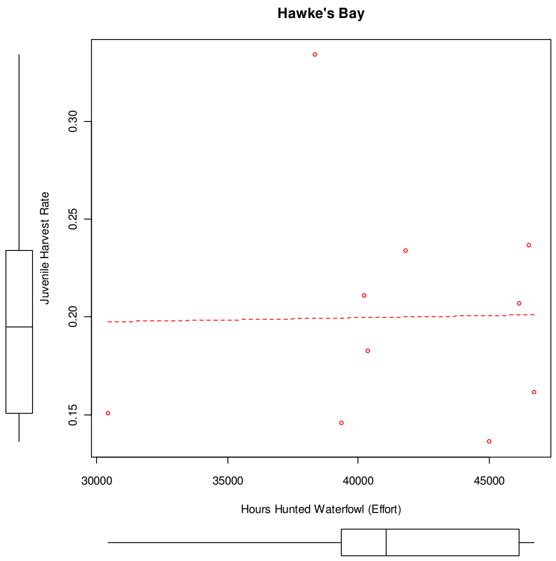

Figure 2.47. Hawke’s Bay annual juvenile harvest rates against total effort (hours hunted waterfowl) (*R*=0.017, $t_{(8)}$ = 0.0499, *P* = 0.96).

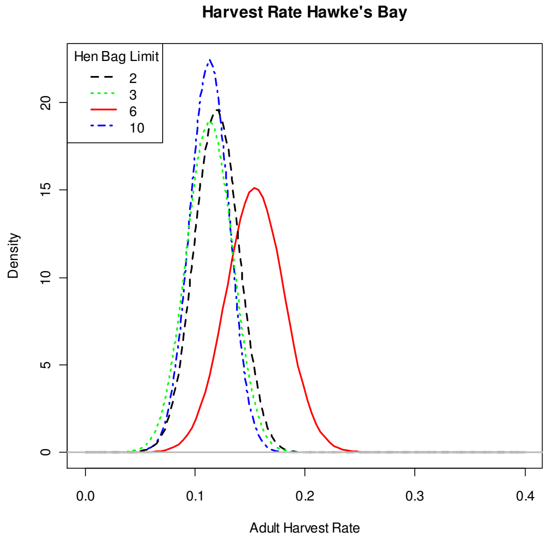

Figure 2.48. Adult harvest rate reported against hen bag limits; Hawke's Bay region.

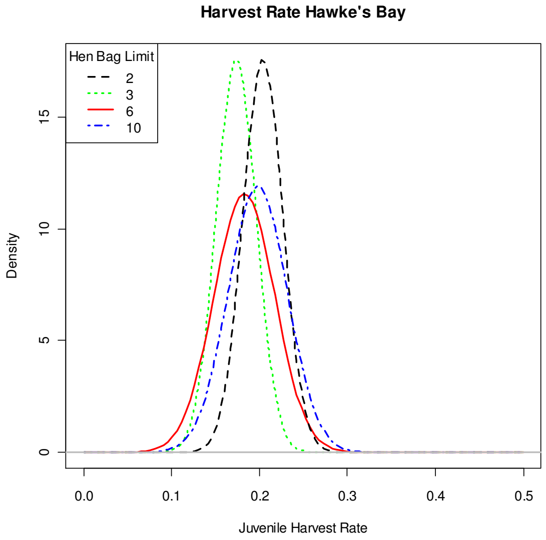

Figure 2.49. Juvenile harvest rate reported against hen bag limits; Hawke's Bay region

Season Regulations and Recruitment

There was no difference in the ratios of juveniles in the trap sample between the short and the long seasons for Eastern Region ($t_{0.05(2)(11)} = -0.649,\ P = 0.53$) or Hawke's Bay ($t_{0.05(2)(8)} = 0.2094,\ P = 0.839$) suggesting amongst other possibilities that season length is not effecting breeding or; the trap sample is not an appropriate measure of productivity.

Partial Management Control in Summary

Murray Williams once noted that a psychologist would fare better than a waterfowl biologist in setting season regulations. The results bear testament to these musings. The heuristic harvest model (Figure 2.1) proposes season regulations control hunter behaviour; primarily through season length governing the amount of hours a hunter can expend during the season. But because opportunity is not saturated there is latitude to increase effort in short seasons. Furthermore the effect of effort on survival is apparent but this is not easily transposed to harvest rates.

The key findings for the Eastern Region are:

- There is a strong relationship between effort and harvest, as effort increases so does harvest (page 62).

- But there is an insignificant linear relationship between season length and harvest (page 62). This is probably due to season length being fixed at 57 days from 2002 – 2006. But may also be due to short season not constraining total hours hunted?
- On average hunters hunted for more hours in the shorter season (Figure 2.9) and a greater proportion of the shorter seasons (Figure 2.12). Only in two 71 day seasons did Eastern hunters hunt more hours on average than the 30 day season. Because the average number of days hunted in the 30 day season (Figure 2.4 and Figure 2.10), were less than any other season, hunters must have hunter more hours per day. There were 5 seasons that hunters hunted on average more hours than the 43 days season hours but unlike the 30 day season this was due to them hunting more days (they hunted more days than 72% of the longer seasons). Nevertheless the confidence intervals on Figure 2.7 suggest that total effort is not significantly different between years.
- Despite a good relationship between effort (total hours hunted) and harvest (page 64) there is only a poor relationship between effort and estimated adult harvest rates and no relationship with juvenile harvest rates (page 62).
- When conditional reporting rate (r) (an indicator of harvest rate) was constrained to a linear function of effort (not shown in the model set) for the top ranked *Effort* model there was no additional support; in fact ranking fell to the fifth lowest in the model set (deviance was the fourth highest).
- Nevertheless as effort increases so does harvest rate. It is suspected that variation in reporting rate is compromising harvest rate estimates.
- Harvest rates, however, do appear to be influenced by season length. Excluding the 43 day season (the highest harvest rates corresponded with the 43 day season), there were significant differences for the juveniles between all season lengths while adult harvest rates were only significantly different between the 57 and the 71 days seasons (page 61).
- The poor relationship between effort and harvest rate, and the contrary significant relationship between season length and harvest rate gives weight to the proposition that the implementation of restrictive season length when the population was low (and vice versa) was appropriate. That is, the monitoring

estimates of population size may be reasonable; in years when the population was low harvest rates were low.

- Hunter effort explains changes in survival better than any of the other candidate models which further corroborates the premise that reporting rate is compromising harvest rate estimates (page 50).

The key findings for the Hawke's Bay Region are:

- Over the study period there was a strong positive correlation between effort and harvest (page 67).
- Average effort per day was higher in the shorter seasons (Figure 2.19). The total days hunted over the 30 day season exceeded one 43 day and three 57 day seasons but not significantly (Figure 2.18).
- There was no significant relationship between effort and harvest rates (page 67) or season length and harvest rate (page 66). I suspect that the lack of significance in either estimate is a function of poor estimates of reporting rate as there was a significant positive correlation between Hawkes Bay season length and harvest (page 67).
- But no apparent relationship between bag limits and harvest (page 66).
- Of the candidate models changes in survival in the Hawke's Bay were best explained by season length and effort (hours hunting waterfowl) (page 57). Climate covariates, spring and summer temperature, also had a small degree of support.

Management Suggestions:

- An alternative means of assessing reporting rate should be implemented such as reward bands or telemetry studies.
- Variance around measures of effort (hunter survey) makes it difficult to decipher the effects of regulations. A power analysis should be done to determine if more precision can be reasonably achieved.

Structural Uncertainty

Postharvest Population size and survival

It was anticipated that survival and productivity would decrease as post- harvest population size increases if density dependence is evident.

Eastern: There was a strong significant inverse relationship between total post-harvest population (N_{PoH}) and post- harvest survival (δ_{jt}) for the juvenile females (*R* = -0.667, $t_{(11)}$ = -2.9687, *P* = 0.013), and juvenile males (*R* = -0.791, $t_{(11)}$ = -4.2949, *P* = 0.001). A similar result was obtained for the adult males (*R*= -0.733, $t_{(11)}$ = -3.5766, *P* = 0.004) while post- harvest survival in the adult females was not as closely correlated (*R*=-0.490, $t_{(11)}$ = -1.8656, *P* = 0.089) it was considerably more significant for the post-harvest adult population (*R*=-0.632, $t_{(11)}$ = -2.7055, *P* = 0.020).

There was no improvement in correlation (total post-harvest population and survival of the respective cohorts) when an exponential or logarithmic model was fitted to the data.

The lowess line (locally weighted scatterplot smoothing (Kutner et al., 2005)) (Figure 2.50) possibly indicates static adult female post-harvest survival (0.73) up to a population size of 250,000 – 300,000 and then decreases with increasing post-harvest population size consistent with smaller populations showing compensatory mortality and additive mortality above some threshold? I tested a quadratic function of the form:

$$\hat{\delta}_{AF,t} = -8.77\text{E-}13\hat{N}^2_{PoH,t} + 3.58\text{E-}7\hat{N}_{PoH,t} + 0.689 \qquad [2.37]$$

on the adult females (Figure 2.50) to try and replicate a static survival at the lower population levels and then decreasing survival at higher population levels. The linear model proved to be the more parsimonious with 86% support (ΔAICc =3.26 for the quadratic).

The remaining cohorts are consistent with additive mortality in this post-harvest population range (Figure 2.51, Figure 2.52 and, Figure 2.53).

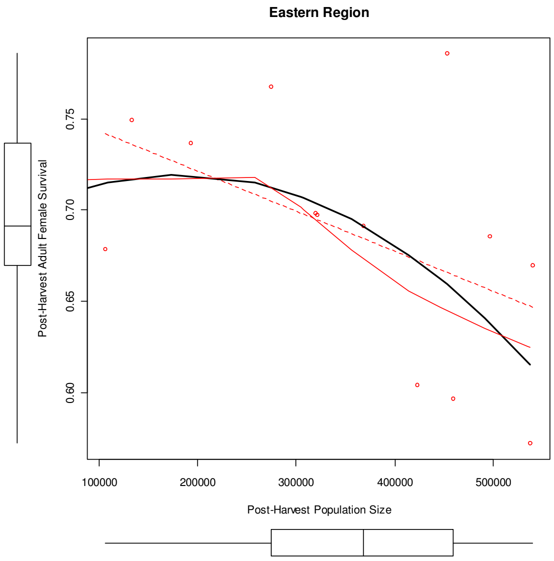

Figure 2.50. Post-harvest adult female survival against estimated post-harvest population size with fitted linear regression line, polynomial quadratic and lowess models.

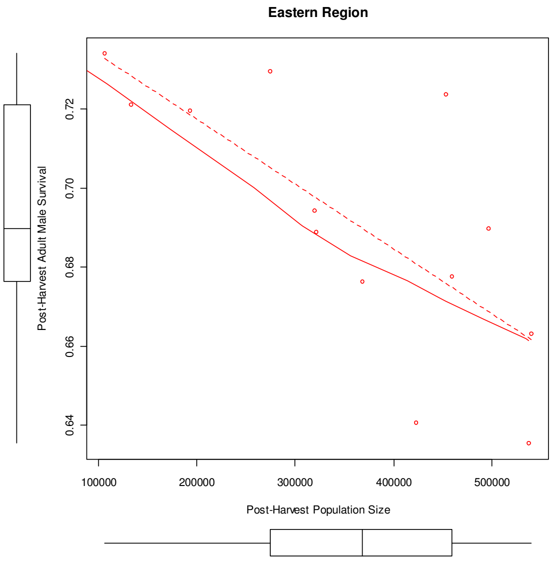

Figure 2.51. Post-harvest adult male survival against estimated post-harvest population size with fitted linear regression line and lowess model.

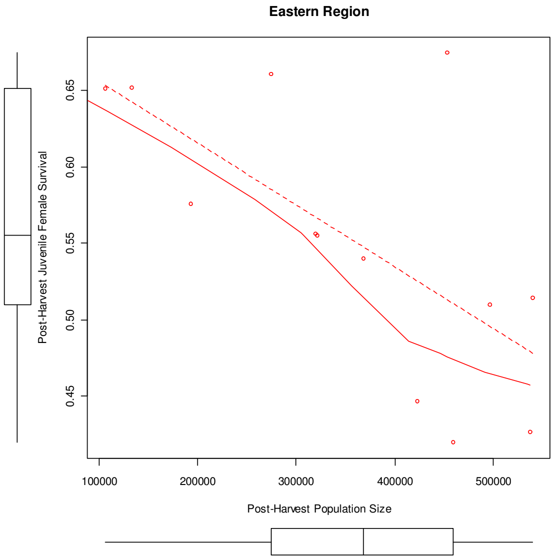

Figure 2.52. Post-harvest juvenile female survival against estimated post-harvest population size with fitted linear regression line and lowess model.

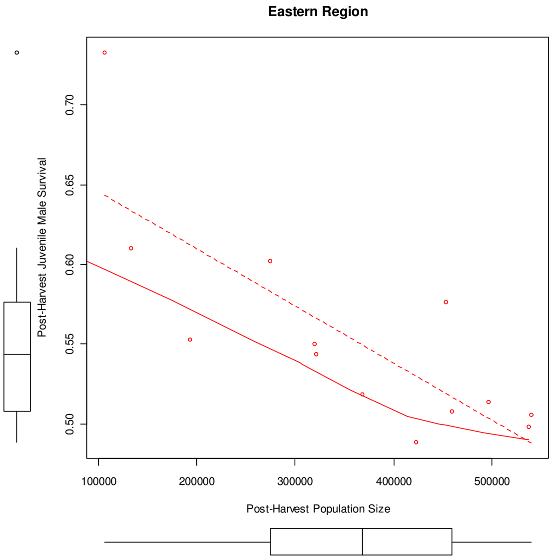

Figure 2.53. Post-harvest juvenile male survival against estimated post-harvest population size with fitted linear regression line and lowess model.

The post-harvest linear survival models for adult female, adult male, juvenile female and, juvenile male are respectively:

$$\hat{\delta}_{AF,t} = 0.765 - 2.18\text{E-}07\hat{N}_{PoH,t} \quad [2.38]$$

$$\hat{\delta}_{AM,t} = 0.750 - 1.64\text{E-}07\hat{N}_{PoH,t} \quad [2.39]$$

$$\hat{\delta}_{JF,t} = 0.696 - 4.03\text{E-}07\hat{N}_{PoH,t} \quad [2.40]$$

$$\hat{\delta}_{JM,t} = 0.681 - 3.57\text{E-}07\hat{N}_{PoH,t} \quad [2.41]$$

Hawke's Bay

Consistent with the Eastern findings all cohorts in the Hawke's Bay showed a significant negative correlation between post-harvest survival ($\delta_{j,t}$) and post-harvest population size ($\hat{N}_{PoH,t}$) (seeTable 2.12).

Table 2.12. Correlation between post-harvest population estimate and estimated post-harvest survival for Hawke's Bay.

	R	t	Df	P
Adult Female	-0.730	-2.8295	7	0.025
Adult Male	-0.658	-2.3146	7	0.054
Juvenile Female	-0.748	-2.9834	7	0.020
Juvenile Male	-0.735	-2.8648	7	0.024

However when the post-harvest survival is compared with the starting population ($\hat{N}_t$) (as opposed to the post-harvest population $\hat{N}_{PoH,t}$) the correlation is even more pronounced for the Hawke's Bay birds suggesting density dependent factors apply early in the season or, season regulations are having an impact (a high population results in a liberal set of season conditions which in turn has a high harvest rate and consequent low survival)?

There was no obvious indication that post- harvest mortality had anything but an additive effect (see Figure 2.54, Figure 2.55, Figure 2.56, and Figure 2.57)

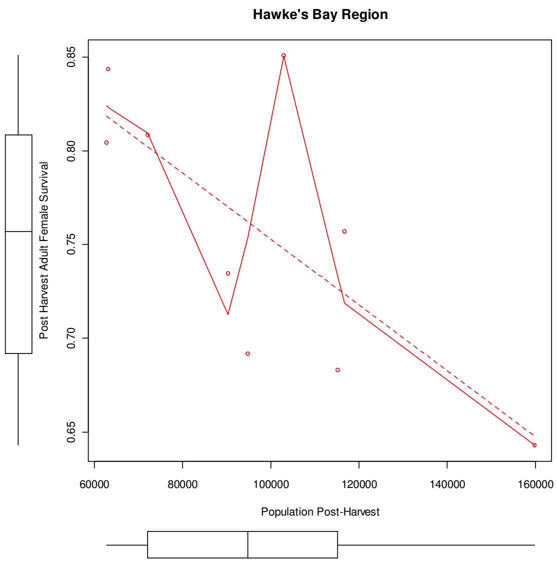

Figure 2.54. Scatterplot of Hawke's Bay post-harvest adult female survival against estimated post-harvest population size; showing the regression (red dashed) and lowess lines (red solid line).

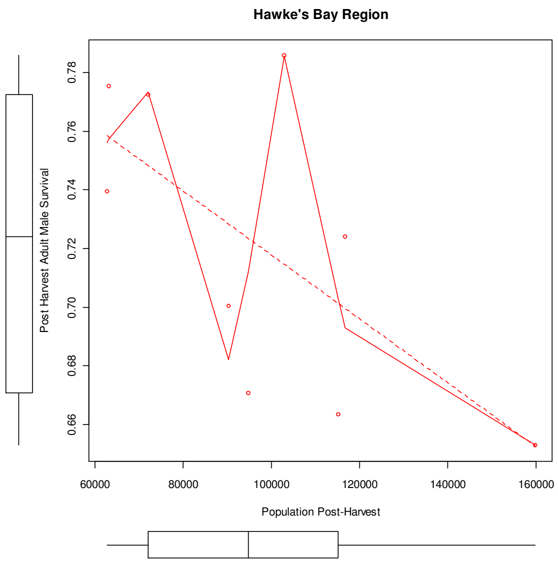

Figure 2.55. Scatterplot of Hawke's Bay post-harvest adult male survival against estimated post-harvest population size; showing the regression (red dashed) and lowess lines (red solid line).

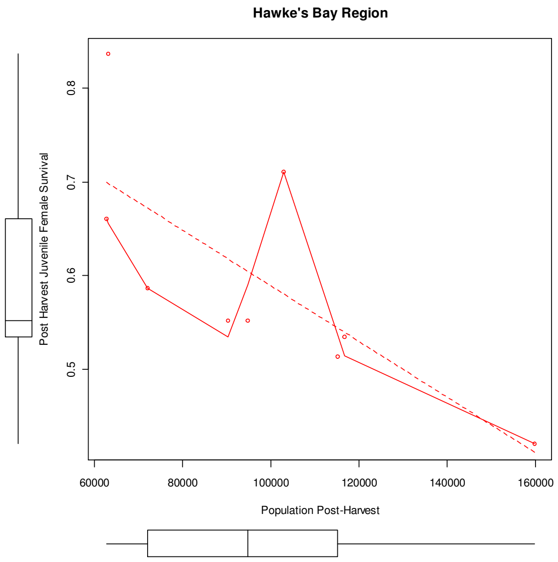

Figure 2.56. Scatterplot of Hawke's Bay post-harvest juvenile female survival against estimated post-harvest population size; showing the regression (red dashed) and lowess lines (red solid line).

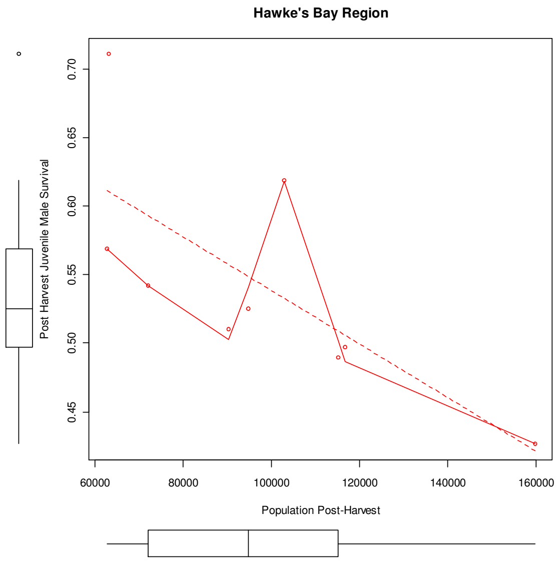

Figure 2.57. Scatterplot of Hawke's Bay post-harvest juvenile male survival against estimated post-harvest population size; showing the regression (red dashed) and lowess lines (red solid line).

The relationship between post-harvest population size and post-harvest survival in Hawke's Bay can be portrayed:

$$\hat{\delta}_{AF,t} = 0.929 - 1.76\text{E-}06\hat{N}_{PoH,t} \qquad [2.42]$$

$$\hat{\delta}_{AM,t} = 0.734 - 1.96\text{E-}07\hat{N}_{PoH,t} \quad [2.43]$$

$$\hat{\delta}_{JF,t} = 0.885 - 2.96\text{E-}06\hat{N}_{PoH,t} \quad [2.44]$$

$$\hat{\delta}_{JM,t} = 0.734 - 1.96\text{E-}06\hat{N}_{PoH,t} \quad [2.45]$$

I was concerned that the correlation between post-harvest survival and post-harvest population size may have been an artefact of season regulations (i.e. due to correlation between estimated population size and corresponding season regulations). For example a high population result in relaxed season conditions that in turn cause low survival. To test this the period of fixed season length regulations (57 days) in the Eastern Region (2001-2006) was examined. These years showed a highly significant correlation between estimated post-harvest population size and post-harvest survival rates ($\delta_{AF,t}$ =-0.959, $t_{(4)}$ = -6.7667, P = 0.0025; $\delta_{AM,t}$ =-0.918, $t_{(4)}$ = -4.6199, P = 0.0099; $\delta_{JF,t}$ =-0.918, $t_{(4)}$ = -4.6341, P = 0.0098; $\delta_{JM,t}$ =-0.922, $t_{(4)}$ = -4.7658, P = 0.0089) showing that the correlation was not attributed to a confounding effect of season length.

I was also mindful that there will be a correlation between the population size estimate and survival due to the same band returns being used in both estimates. To reduce this potential confound banded birds were randomly allocated to two groups and one group was used to calculate population size and the other survival[1]. In addition all survival models that incorporated any harvest data were removed and the survival analysis rerun.

The results from this analysis didn't alter much for the juveniles ($\delta_{JF,t}$ =-0.786, $t_{(11)}$ = -4.2105, P = 0.0014; $\delta_{JM,t}$ =-0.847, $t_{(11)}$ = -5.2765, P = 0.0003), but were not as significant for the adults ($\delta_{AF,t}$ =-0.484, $t_{(11)}$ = -1.8333, P = 0.094; $\delta_{AM,t}$ =-0.542, $t_{(11)}$ = -2.1393, P = 0.056).

The lowess line in the juvenile scatterplots (not shown) were reasonably linear while the adults (Figure 2.58 and Figure 2.59) could, with a liberal amount of licence, be

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[1] Nichols and Hines (1983) used a similar approach to investigate the relationship between harvest rates and survival.

interpreted as static survival up to 250,000 and then a declining with increasing population? This would also explain the relatively poor correlation.

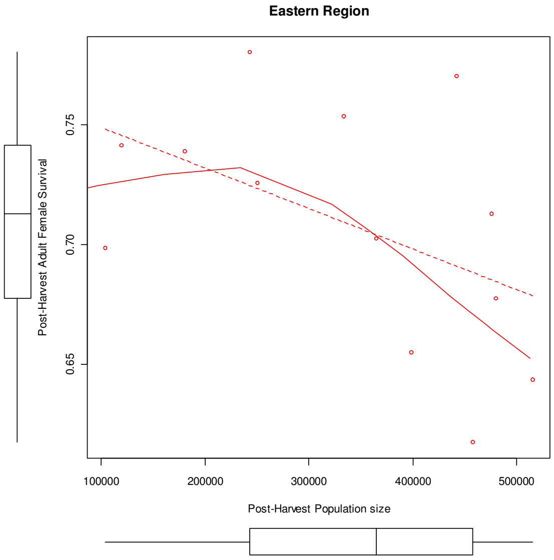

Figure 2.58. Scatterplot of Eastern Region post-harvest adult female survival against estimated post-harvest population size. Band data randomly allocated to harvest rate and survival analysis; regression (red dashed) and lowess (red solid) lines.

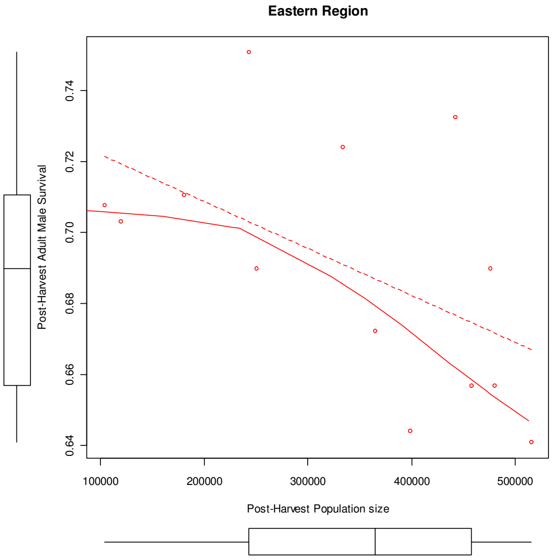

Figure 2.59. Scatterplot of Eastern Region post-harvest adult male survival against estimated post-harvest population size. Band data randomly allocated to harvest rate and survival analysis; regression (red dashed) and lowess (red solid) lines.

Structural Uncertainty and Survival Rates; In Summary.

Based on the above analysis I am reasonably comfortable that post-harvest population does have an inverse effect on post-harvest survival rates. Nevertheless I would be more comfortable with the population size estimates being derived from independent data such as aerial dabbling duck counts or similar.

Estimates of pre-harvest survival (α) were based on the telemetry work in the Eastern Region. It is clear that more work needs to be done over this period, it is unlikely that adult survival for this period will be 1 a greater sample is required for any degree of confidence in the estimates.

It appears that any mortality after harvest is in addition to harvest mortality (additive); for the juveniles but Eastern Region adults may show some compensatory mortality below populations of about 250,000.

Post Harvest Population size and recruitment

Average productivity (juvenile females to adult females) over the study period, as measured by the trap sample, for Eastern and Hawke's Bay was 1.84 and 1.63 respectively.

Rearranging equation [2.30] gives the expected productivity ($E(B)$) provided population change remains constant ($\Delta N = 1$) over the study period. Average survival rates for adult ($\bar{S}_{AF}$) and juvenile females ($\bar{S}_{JF}$) were derived from components of variance analysis of the top survival model for Eastern and Hawke's Bay regions (Table 2.7 and Table 2.9 respectively):

$$E(B) = \frac{(1 - \bar{S}_{AF})}{\bar{S}_{JF}} \qquad [2.46]$$

For Eastern:

$$0.9104 = \frac{1 - 0.641225}{0.394076}$$

with an approximate productivity range of 0.41≤0.91≤1.92, estimated from the upper and lower average survival rate estimates.

And Hawke's Bay:

$$0.628 = \frac{1 - 0.7074}{0.4660}$$

with a range 0.27≤0.63≤1.31.

The trap productivity estimates were 2.02 and 2.59 times higher than the ΔN=1 productivity estimates of 0.91 and 0.62 for the Eastern and Hawke's Bay Regions respectively. This suggests:

(1) The trap sample is biased and adult females are underrepresented or;

(2) The populations of mallard in Eastern and Hawke's Bay are increasing (if this were so harvest might be expected to be increasing which is not the case) or;

(3) Survival rate estimates may be too high or;

(4) Some combination of the above.

Many adult females are moulting at the time of banding (late January) and further not all of the banding sites are good moulting sites hence adult females may be under represented at these sites, lending credence to the first suggestion. However I am also sceptical of the Hawke's Bay average adult female survival rate ($0.58 \leq 0.70 \leq 0.83$; $S \pm$ 95% CI), which appears particularly high compared with the adult males ($0.63 \leq 0.64 \leq 0.65$; $S \pm$ 95% CI).[1]

Eastern Region Recruitment

There was no correlation between total post- harvest population size and productivity (R=0.378, $t_{(11)} = 1.3558$, $P = 0.2024$) but there was a reasonable negative correlation with post- harvest adult population size and female productivity (juvenile female: adult female) (r=-0.538, $t_{(11)} = -2.1165$, $P = 0.058$; Figure 2.60). The correlation between the ratio of juvenile males: adult females with post- harvest adult population was negative but not significant (r=-0.497, $t_{(11)} = -1.8999$, $P = 0.084$; Figure 2.61).

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[1] Previous studies of four areas in New Zealand found adult male survival rates were higher than adult females (Caithness et al., 1991, Nichols et al., 1990).

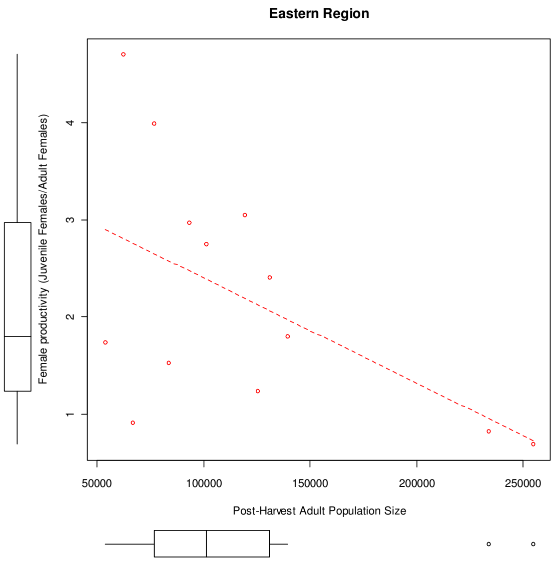

Figure 2.60. Eastern Region Post-harvest adult population against female productivity (juvenile female: adult female) (*R*=-0.538, *t* = -2.1165, *Df* = 11, *P* = 0.058).

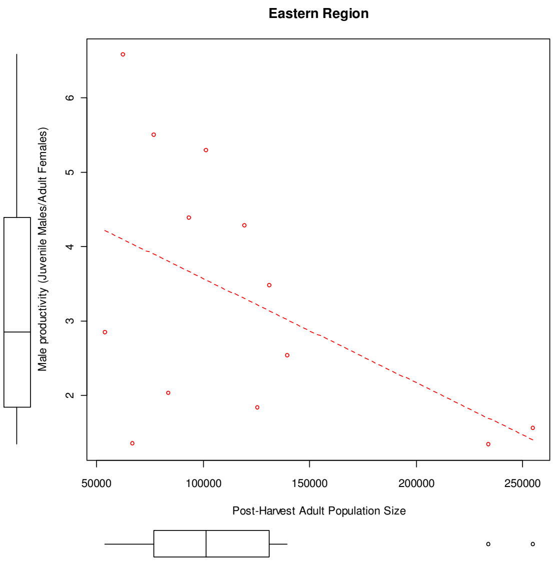

Figure 2.61. Eastern Region Post-harvest adult population against male productivity (juvenile male: adult female) (*R*=-0.497 *t* = -1.8999, *Df* = 11, *P* = 0.084).

The linear model for the juvenile female: adult female productivity (B_{JF}) is:

$$B_{JF} = 3.479 - 0.00001079 N_{AP} \qquad [2.47]$$

(Significance of parameter estimates: intercept *P*<0.001; and regression coefficient *P*=0.058 and N_{AP} = post-harvest adult population size) and juvenile male: adult female productivity (B_{JM}):

$$B_{JM} = 4.98 - 0.000014 N_{AP} \qquad [2.48]$$

(Significance of parameter estimates: intercept *P*<0.001; and regression coefficient *P*=0.084).

Structural Uncertainty Key Findings:

- Post-harvest survival in the Eastern Region is strongly negatively correlated with post-harvest population size for all cohorts except adult females (page 72). Adult females may show some compensatory survival below population size of 250,000 – 300,000 (Figure 2.58).
- The correlation between productivity and post harvest adult population size was negative but not significant at the 5% level (page 79). These results may be confounded by trap bias (page 78).

Regulation Strategy

The problem of developing an appropriate regulation strategy was approached in two different ways. In the first instance Eastern Region's current process of setting season regulations was utilised whereby regulation constraint is a function of some population size, coined *Threshold Management*. In the second illustration, a *State Dependent Strategy* (SDS) is presented. The SDS is where the regulation decision is a function of the current population size, assessed productivity and, harvest is maximised under the constraints:

- that total effort is not allowed to go below the lowest level recorded in the hunter survey during the study period and;
- $N_{(F)(t+1)} \geq N_{(F)(t)}$; where $N_{(F)}$ is the female population.

Threshold Management

To reiterate, *Threshold Management* is a set of regulation constraints that apply depending on the size of the assessed population. For example if the pre-harvest population is assessed at or above the maximum threshold then a relaxed set of regulations are implemented. Below this level but above the minimum threshold an intermediate regulation set applies, and below the minimum threshold a restricted regulation set is implemented.

The objective was to determine the threshold levels that maximised harvest over a 10 year period while ensuring the population of adult females in the 11th year was greater or equal to the starting population ($N_{AF(11)} \geq N_{AF(1)}$) based on a fixed productivity level.

For this a stochastic model in program R was compiled. A range of upper threshold levels (10,000- 225,000) were tested with the intermediate and minimum threshold set as a percentage of the upper threshold. For example, an upper threshold of 100,000 with the intermediate as 75% (75,000) of this level, and minimum 50% (50,000) of the top threshold (7550 denotes this strategy). For each of the upper thresholds the above 75%-50% strategy, 50%-25% (5025), and 90%-80% (9080) were simulated. The simulation was confined to adult and juvenile females.

The combined population size (adult and juveniles) were assessed against the relative thresholds, if the population was above the maximum threshold a set of relaxed season regulations applied. For example if the upper threshold was 100,000 and the female population was above this a 71 day season applied, if the population was below 100,000 but above the intermediate threshold (eg below 100,000 but above 75,000) a 57 days season was applied, and so on; below the intermediate threshold but above the minimum a 43 day season and, below the minimum a 30 days season.

For each season length an expected number of hours hunted was generated for that season length derived from a normal distribution with mean and standard deviation established from the hunter survey data.

For harvest rates bayes.lin.reg (Package Bolstad) was used to generate a stochastic harvest rate as a function of effort incorporating equations [2.34] and [2.35]. This ensured that the uncertainty associated with the model was incorporated into the estimated harvest rate.

For survival rate equations [2.25] and [2.26] were utilised but I found that survival rate was over estimated using an α of 1 for the adults and 0.91 for the juveniles (these estimates were derived from the telemetry study). A more realistic survival rate was achieved when α was set to 0.91 and 0.73 for the adults and juveniles respectively (interpolated from survival estimates as a function of total effort Figure 2.24 and Figure 2.25, page 51). For delta (δ) equations [2.38] and [2.40] were used for the adult female and juvenile females respectively and bayes.lin.reg to generate a normally distributed estimate as a function of the assessed post- harvest population size ($N_{PoH,t}$).

The end of year adult population size was calculated as the surviving adults ($N_{AF,t}S_{AF,t}$) plus the surviving juveniles (which would now be adults) ($N_{JF,t}S_{JF,t}$) where $S_{it} = \alpha_{it}\beta_{it}\delta_{it}$ (*i*= age). The new seasons (i.e year 2) juvenile population was calculated by multiplying this population by the productivity (*B*).

This population then carried over into year 2 as the starting population. This was repeated for 10 years and the simulation run 100 times. The result was a non-parametric distribution of cumulative harvest and final year (year 11) population size.

An example of the estimated cumulative harvest for the 10 year period is given in Figure 2.62. In this example the mean cumulative harvest was almost 82,000 based on an upper threshold limit of 100,000 a 75,000 intermediate level and, 50,000 minimum. Productivity for this example was set at 0.85 juvenile females to adult females.

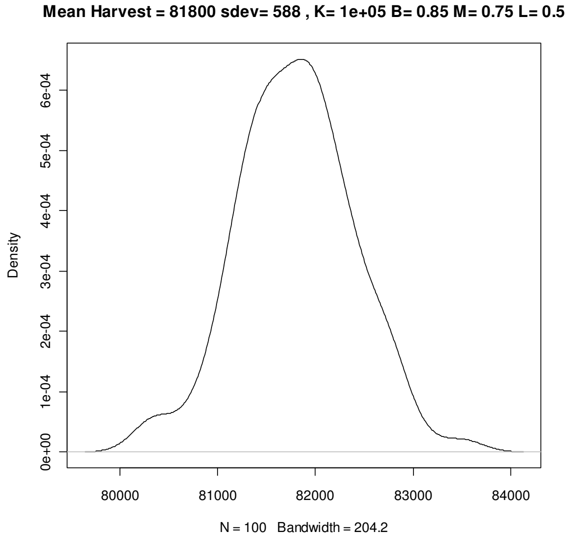

Figure 2.62. Non-parametric distribution of cumulative harvest based on an upper threshold level of 100,000 (k), and intermediate level of 75,000 and minimal threshold of 50,000. The starting population was 50,000 adult females and 50,000 juvenile females. Year 11 adult female population had a mean of 52,467 and season length was 57 days.

The respective strategies made very little difference to the cumulative harvest if productivity was above 0.8. Once productivity exceeded 0.8 the maximum threshold closest to the starting population (e.g. 100,000 in this example) produced the greatest cumulative harvest (Figure 2.63). When productivity was 0.8 or less no matter what the strategy (corresponding to a season length of 30 days for the entire 10 years), harvest was not sustainable i.e. the mean adult female population in year 11 was below the starting population of 50,000.

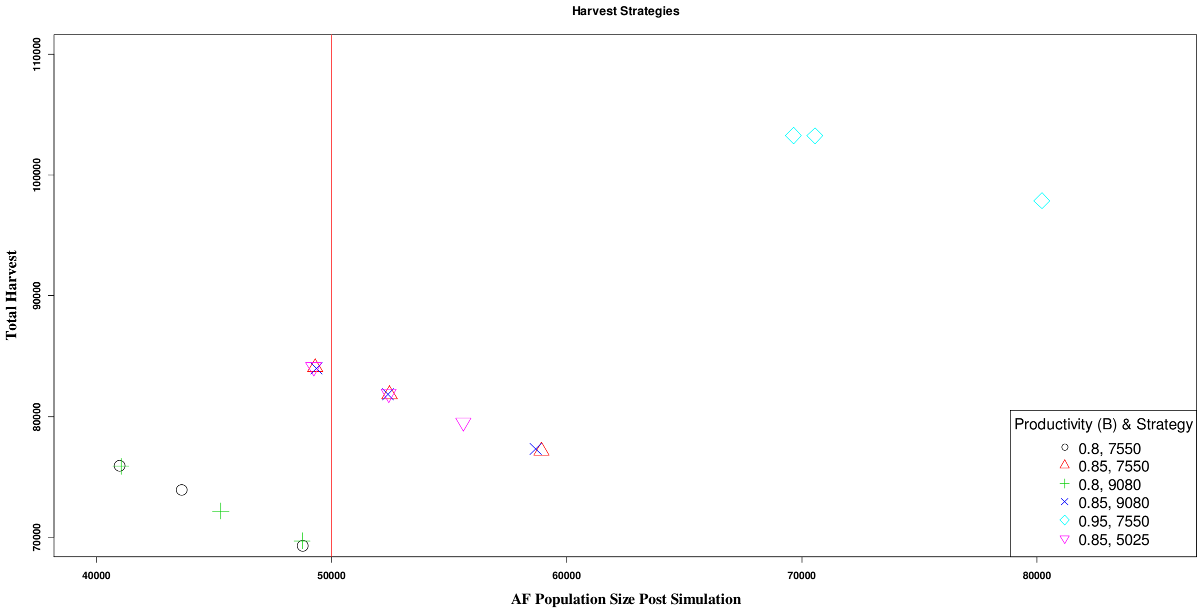

Figure 2.63. Simulated average total cumulative harvest (10 years; 100 iterations per estimate) constrained by different harvest regulation strategies and productivities (0.8, 0.85 & 0.95; juvenile females to adult females). The starting population is 50,000 adult females and 50,000 juvenile females. The x-axis shows the estimated mean adult female population in the 11th year. The points relate to the different strategies.

When productivity was 0.85 an upper threshold of 10,000 (strategy 7550) allowed a 71 day season for the full 10 years but resulted in a mean adult female population in year 11 of 49,303. The cumulative harvest (Figure 2.64) however, was about 4000 more than the more sustainable strategy - 100,000 upper threshold (57 day season for all 10 years). Governors might decide that the risk of reducing population over the 10 years by 700 or so birds may be worth it to have a 71 day season for all 10 years and a higher cumulative harvest?

Once productivity got to about 0.9 juvenile females to adult females a relaxed season (71 days) could be implemented with no long term consequences for population sustainability.

The findings that harvest is not sustainable even at a 30 day season once productivity (juvenile female: adult female) gets below 0.8 are very interesting. Adjusted productivity in 1997 and 1998 (see Appendix B) declined well below the 0.8 sustainable figure which also coincided with very low population levels (Figure 2.66, page 91) in the immediate following years and corresponding drop in harvest over the next 5 years in the Eastern Region.

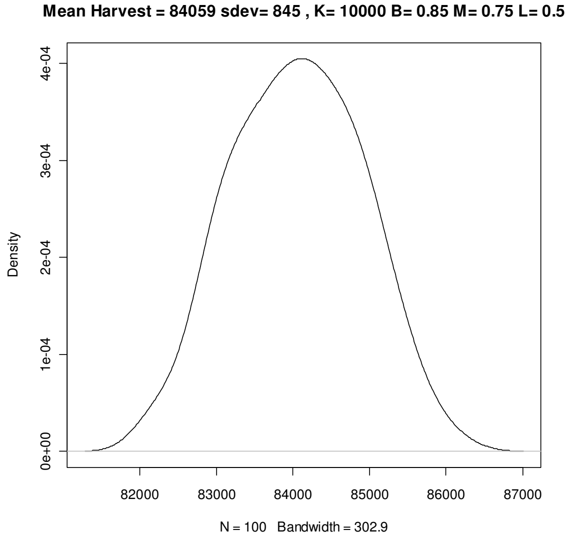

Figure 2.64. Simulated cumulative harvest over 10 years under a strategy of a 10,000 upper threshold (K), 75,000 intermediate (M) and, 50,000 minimum (L). Starting population was 50,000 adult female and 50,000 juvenile females. Productivity (B) was 0.85 juvenile female to adult female. Final (year 11) mean adult female population was 49303. Season lengths for the 10 years was 71 days.

State Dependent Strategy

The SDS is a two stage approach. In the first stage a deterministic model is used to maximise harvest and spawn a corresponding optimal effort as explained below. This optimal effort is then used to determine an appropriate season length. The second stage uses a partial stochastic model to estimate harvest and subsequent population size (N_{t+1}) as a function of the population size ($N_{(t)}$), effort under the selected season length, and assessed productivity. The partially stochastic model uses the same quantitative models as the deterministic model but include allowance for parameter variance. The stochastic model can be validated against harvest estimates from the hunter survey outside of the study period.

The following explains the deterministic model, followed by the stochastic model, and finally a worked example is presented.

Deterministic Model: Solver in Microsoft Excel is used to generate an optimal effort (E; total hours) that maximise harvest (the objective function), for a given productivity, and female (adults and juveniles) population size, subject to the constraints:

1. $N_{t+1} \geq N_t$ and;
2. Effort ≥the smallest number of hours reported in the Eastern hunter survey for hunting waterfowl:

$$\max H_t = \sum N_{it}\alpha_{age}h_{age};\ subject\ to:\begin{cases}E \geq 65000\_Hours\\ \sum N_{(i)t+1} \geq \sum N_{(i)t}\end{cases}$$

$$\text{where:}$$

$$h_{age} = a_{age} + b_{age}E \text{ (see Note 1)}$$

$$N_{t+1} = \sum\left(N_{it}\alpha_{age}\beta_{age}\delta_{age} + N_{it}\alpha_{age}\beta_{age}\delta_{age}B_{t+1}\right)\text{(see Note 4)}$$

$$\delta_{age} = c_{age} - d_{age}\left(N_{it}\alpha_{age}\beta_{age}\right)\text{(see Note 2)}$$

$$\beta_{age} = 1 - \left[\frac{h_{age}}{\bar{\gamma}}\right]\text{(see Note 8)}$$

$$B_{JF\,t+1} = \left[\frac{\left(3.479 - 0.00001079\left(N_{At}\alpha_A\beta_A\delta_A\right)\right)}{2.02}\right]\text{(see Notes 3, 9)}$$

$$i = adult\ \ females\ \&\ juvenile\ \ females$$

$$\gamma = 0.95\text{ (see Note 8)}$$

$$a = \begin{cases}adults = 0.000565\\ juvenile = 0.0154\end{cases}$$

$$b = \begin{cases}adults = 0.00000091\\ juveniles = 0.0000015\end{cases}$$

$$c = \begin{cases}adults = 0.765\\ juveniles = 0.696\end{cases} \qquad [2.49]$$

$$d = \begin{cases}adults = 0.00000218\\ juveniles = 0.00000403\end{cases}$$

And:

$$N_t = \sum\left(N_{it-1}S_{it-1} + N_{it-1}B_t\right)$$

$$\text{where:}$$

$$S_{it-1} = survival\ in\ year\ t-1\text{ (see Note 6)}$$

$$B_t = \frac{1}{2.02}\left[\frac{Juvenile\ Female_{(Trap\ Sample)}}{Adult\ Female_{(Trap\ Sample)}}\right] \qquad [2.50]$$

$$i = female\ only\ adults\ \&\ juvenile$$

$$A = adults$$

<u>Notes:</u>

1. Harvest rate is a function of effort (see Equations [2.34] & [2.35]). In this case "Solver" is used to derive the effort required to maximise harvest given the constraints that the population of females in year 2 will not be less than year 1 and that the total effort is not less than 65000 hours. Then it is up to the Council to determine what season length will achieve this effort (see Figure 2.65).
2. Post harvest survival is a function of post harvest population size. Larger populations result in lower post harvest survival (see Equations [2.38] - [2.41]).
3. Productivity is density dependent, as population size increases productivity (juvenile females: adult females) decreases (see Equation [2.47]).
4. Population size (N_t) and population variance were estimated from Equations [2.20] and [2.23].
5. The proportion of N_t attributed to each cohort was based on the proportion of the cohort in the trap sample.
6. Survival in the preceding year (S_{t-1}) was estimated using model averaging in Program MARK, and survival (S_t) using Equation [2.25]
7. The alpha estimate was set at 0.91 for juveniles and 0.95 for adults[1]. The alpha estimate was bias adjusted (from the telemetry study findings) to provide realistic survival rate estimates (see Figure 2.24)[2].
8. The beta estimate is defined in Equation [2.27] and harvest rates determined as above. Gamma was establish from the Eastern Region hunter survey ($\bar{\gamma} = 1 - \bar{c}$; where $\bar{c}$ =crippling loss = 0.05).
9. Productivity was based on the observed trap ratio of juveniles to adult females adjusted for trap bias ($\frac{B}{2.02}$) (see Post Harvest Population size and recruitment, Page:78).

The optimal effort required to maximise harvest is then used to determine a season length based on the *Effort-Participation* model Equation [2.31] and Figure 2.14. For example with a population of 50,000 adult female and 50,000 juvenile females and

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[1] The telemetary study found survival for all cohorts prior to the beginning of the hunting season was estimated at 0.95 (SE=0.07). No known adults died during this period.

[2] The difference between the Threshold Management simulations suggests alpha varies between adult males and adult females which is to be expected.

productivity levels of 0.65[1] harvest is maximised when effort is 65,952 hours. When this effort is mapped on to Figure 2.65, (the green dashed line), we see that it is below the mean hours for a 30 day season. Using the same data but this time productivity is 0.75 harvest is maximised when effort is 90,623 hours (blue dashed line on Figure 2.65). In this instance we can see that a season length of just less than 70 days produced an equivalent mean effort. If governors want to be about 90% confident that they don't exceed 90,000 hours it can be seen from Figure 2.65 that a season length of 30 days should achieve this. An example of when the restricted 30 day season may be appropriate is when the population is considered low or there is a high degree of uncertainty in the parameter estimates.

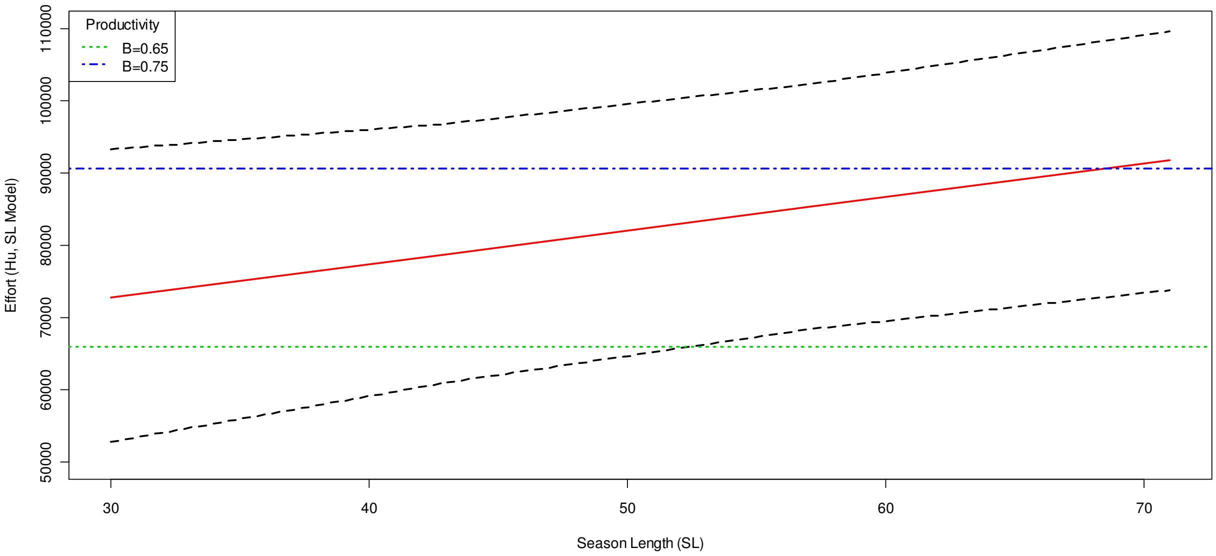

Figure 2.65. The red solid line is estimated effort from the *Hunter-Season Length* model. The black dashed lines are the 95% credible interval. The green and blue dashed lines represent two levels of effort under different levels of productivity (0.65 & 0.75 respectively) that maximise harvest for a population of 50,000 adult female and 50,000 juvenile female. The appropriate season length can be determined based on the level of risk the manager is comfortable with.

Partial Stochastic Model: Given the season length derived under the deterministic model and *Effort-Participation* model it is appropriate to predict harvest (H_t), and the size of population (N_{t+1}). This provides the opportunity to assess the performance of the structural models. The performance of the Equation [2.50], (estimation of N_t) can be evaluated against Equations [2.20] and [2.23] (the Lincoln-Petersen estimate and Seber's variance estimate) and H_t against the hunter survey results.

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[1] In this example I have fixed productivity whereas under the deterministic model productivity is a function of population size.

Predicting $N_{F,t}$ is particularly important as we are required to calculate $N_{F,t}$ in January following trapping and prior to setting the season regulations (wherein using the Peterson-Lincoln Estimate N_t cannot be determined until the end of the hunting season when all bands have been returned).

The number of adult and juvenile females in the population (N_t) are derived as their respective proportion of the trap sample. N_t is calculated from Equation [2.50]. Harvest (H_t) is predicted:

$$H_t = \sum N_{it}\alpha_{age} h^{`}_{age}$$

where:

$$h_{age} = a_{age} + b_{age}E_{SL} \text{ (see Note 10)}$$

$$h'_{Age,E} \sim N(m'_h, (s'_h)^2) \text{ (see Note 11)}$$

$$E_{SL} \sim N\left(\mu_{Sl}, \sigma^2_{SL}\right) \text{(see Note 10)}$$

$$\alpha_{age} = \begin{cases} adults = 0.95 \\ juveniles = 0.91 \end{cases} \text{(see Note 7)}$$

$$a = \begin{cases} adults = 0.000565 \\ juvenile = 0.0154 \end{cases}$$

$$b = \begin{cases} adults = 0.00000091 \\ juveniles = 0.0000015 \end{cases}$$

$$m'_h = m'_\mu$$

where:

m'_μ is posterior distribution of :

$$\mu_{n+1} = a_{(\bar{E})age} + b_{age}(E_{n+1} - \bar{E})$$

And the predictive variance of m'_h :

[2.51]

$$(s'_E)^2 = (s'_\mu)^2 + \sigma^2$$

$(s'_\mu)^2$ is the posterior variance of μ_{n+1}

plus the observation variance σ^2

And N_{t+1}:

$$
\begin{aligned}
& N_{t+1} \sim \left(\sum N_{it}\alpha_{age}\beta'_{age(E)}\delta'_{age}\right) + \left(\sum N_{jt}\alpha_{age}\beta'_{age(E)}\delta'_{age}B'_{t+1}\right) \\
& \text{Where:} \\
& \delta_{age} = c_{age} - d_{age}\left(N_{it}\alpha_{age}\beta_{age}\right) \text{(see Note 2)} \\
& \delta'_{age} \sim N(m'_{\delta},(s'_{\delta})^2) \\
& \beta_{age(E)} = 1 - \left[\frac{h_{age(E)}}{\gamma}\right] \\
& \beta'_{age(E)} \sim N(m'_{\beta_{age}},(s'_{\beta_{age}})^2) \\
& B_t = \left[\frac{\left(3.479 - 0.00001079\left(N_{At}\alpha_A\beta_A\delta_A\right)\right)}{2.02}\right] \text{(see Notes 3, 9)} \\
& B'_{t+1} \sim N(m'_B,(s'_B)^2 \\
& \alpha_{age} = \begin{cases} adults = 0.95 \\ juveniles = 0.91 \end{cases} \text{(see Note 7)} \\
& i = all\ cohorts \\
& j = adult\ females \ \&\ juvenile\ females \\
& A = Adults \\
& \gamma = 0.95 \text{ (see note 8)} \\
& c = \begin{cases} adults = 0.765 \\ juveniles = 0.696 \end{cases} \text{(see Note 2)} \\
& d = \begin{cases} adults = 0.00000218 \\ juveniles = 0.00000403 \end{cases} \text{(see Note 2)}
\end{aligned}
\qquad [2.52]
$$

N_{t+1} and *H_t*: algorithms were run in Program R for 1000 iterations to generate a non-parametric distribution of both estimates.

Notes (continued from page 86):

10. Age specific harvest rates derived from equations [2.34] and [2.35]. Mean effort ($\overline{E}$) and variance ($\sigma^2_{\overline{E}}$) was established from the hunter survey and function rnorm used to generate a normal effort distribution for a given season length ($E_{SL} \sim N\left(\mu_{Sl},\sigma^2_{SL}\right)$).
11. The predictive distribution of the harvest rate (h'_E) given a specific level of effort ($E_{SL} \sim N\left(\mu_{Sl},\sigma^2_{SL}\right)$) is derived using bayes.lin.reg (Package Bolstad)

(Bolstad, 2007). The linear form of the harvest rate model is as described in note 10. Normal priors with mean and precision for the slope and intercept were derived from harvest rate data for a given season length using program WINBUGs (1000 burn-in, 100,000 simulations).

12. bayes.lin.reg (Package Bolstad) was used to generate stochasticity around the delta and beta estimate utilising Normal priors with mean and precision for the slopes and intercepts derived from their respective model estimates using program WINBUGs (1000 burn-in, 100,000 simulations).
13. Population size (N_{t-1}) was estimated from a normal population and variance based on equations [2.20] and [2.23].
14. The proportion of N_t attributed to each cohort was based on the proportion of the cohort in the trap sample.
15. Survival in the preceding year (S_{t-1}) was estimated using model averaging in Program MARK. And survival (S_t) using Equation [2.25]

An example of the harvest distribution is given in Figure 2.68 and the following years population size (N_{t+1}) in Figure 2.69. This simulation is based on the 2009 population size (495,500; Figure 2.66). The mean simulated harvest (47,742; SD=8,485) is very close to the harvest estimated in the hunter survey for the 2009 year (42,424; 34,416-52,296; 95%CI, Figure 2.67).

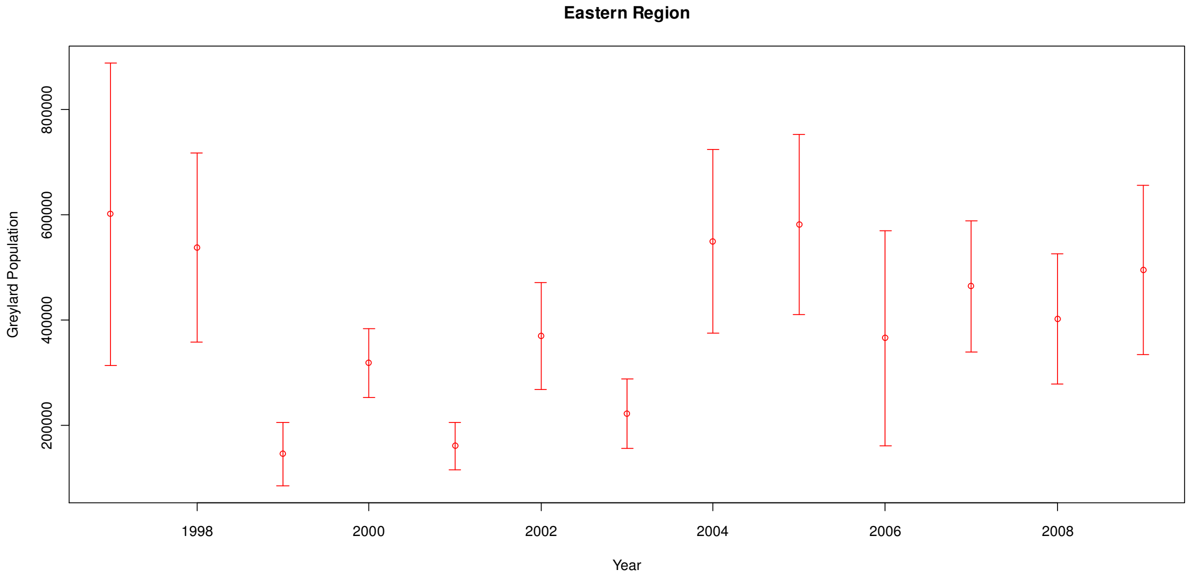


Figure 2.66. Mallard and parera population estimate for the Eastern Region.

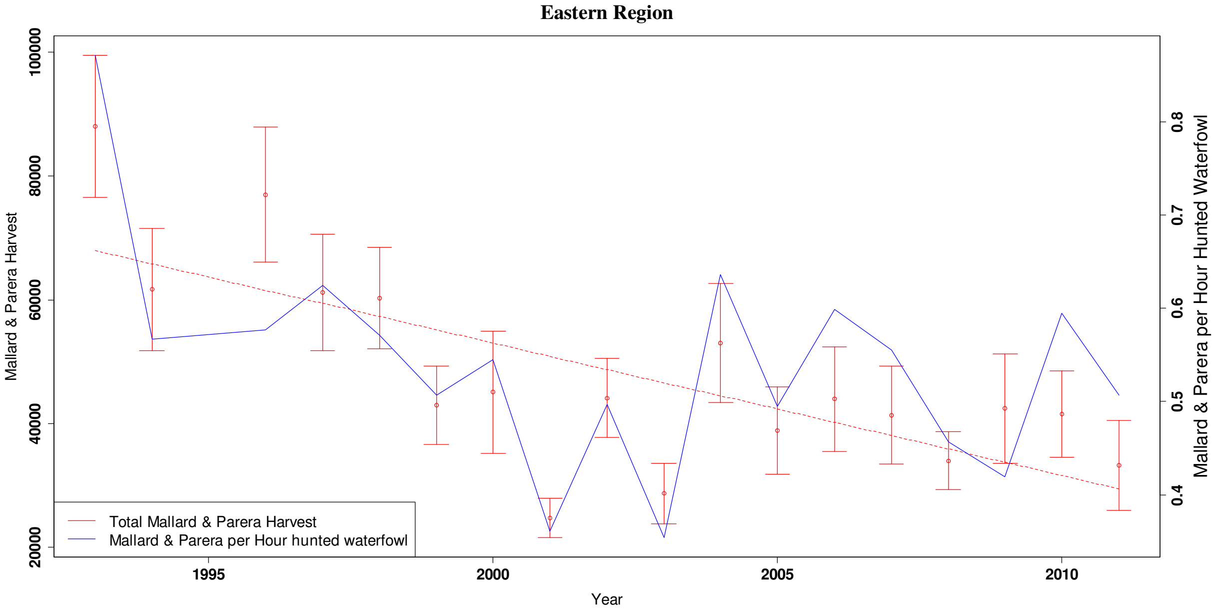


Figure 2.67. Eastern Region annual mallard & parera harvest (red dashed line is regression line on total mallard & parera harvest). Blue line is mallard and parera per hour hunting waterfowl. Note ducks (mallard & parera) per hour hunting waterfowl decrease at the beginning of the study (1997 & 1998) which coincided with unsustainably low levels of productivity (less than 0.8) and liberal season regulations (71 days). Ducks per hour remained low for the following 4-5 years.

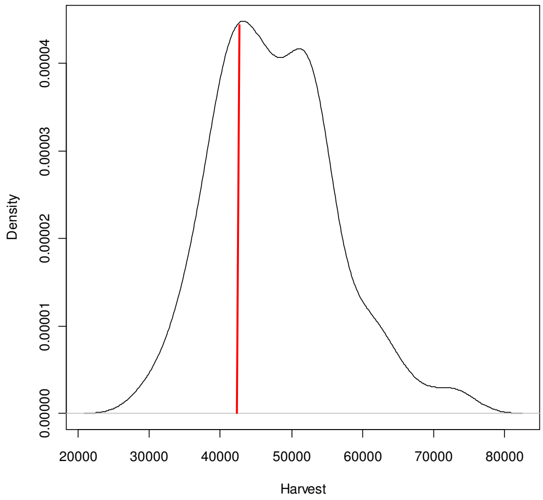


Figure 2.68. Predicted harvest under a 30 day season using the 2009 population size. The mean harvest estimate from the simulation is 47,742 (SD=8,485) and mode about 42,000. The hunter survey reports a mallard harvest of 42,424 (34,416-52,296; 95%CI).

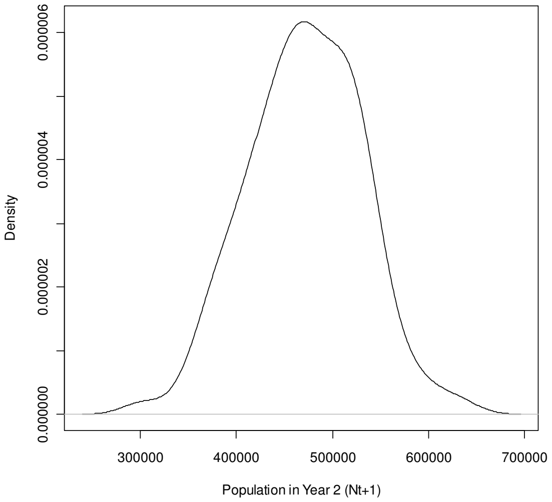
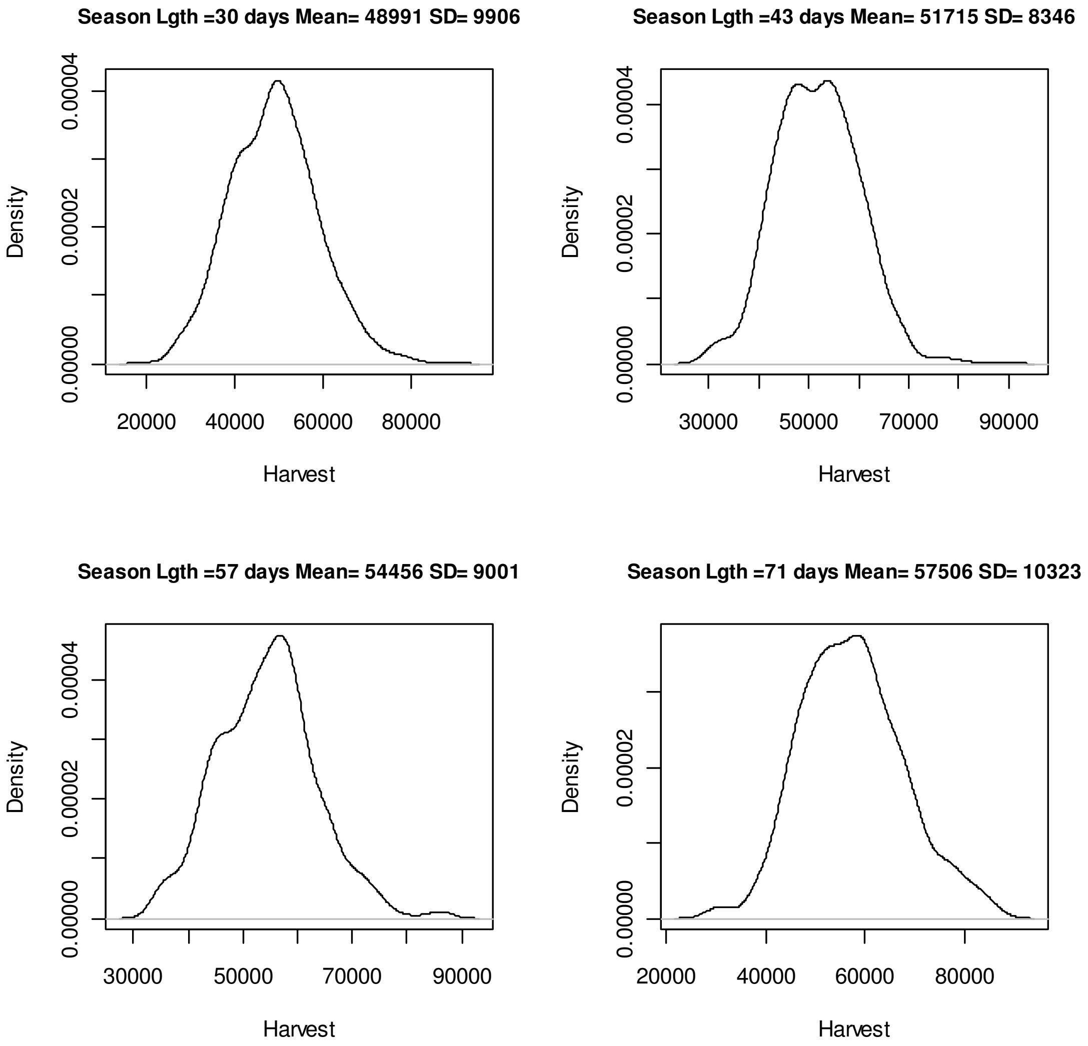


Figure 2.69. Predicted population size in 2010 following a 30 day season and harvest distribution given in Figure 2.68. The mean simulated population size is 471,486 (SD=59123). The population in 2009 was generated from a normal population (mean =495,512; SD= 82,136).

Figure 2.70 and Figure 2.71 show different harvest and, Year 2 population size, distributions respectively under 30, 43, 57 and, 71 day season lengths for equivalent starting population size (mean =495,512; SD= 82,136).



Season Lgth =30 days Mean= 48991 SD= 9906

Season Lgth =43 days Mean= 51715 SD= 8346

Season Lgth =57 days Mean= 54456 SD= 9001

Season Lgth =71 days Mean= 57506 SD= 10323

Figure 2.70. Expected harvest (from the stochastic model; 1000 simulations) under different season lengths based on the 2009 population.

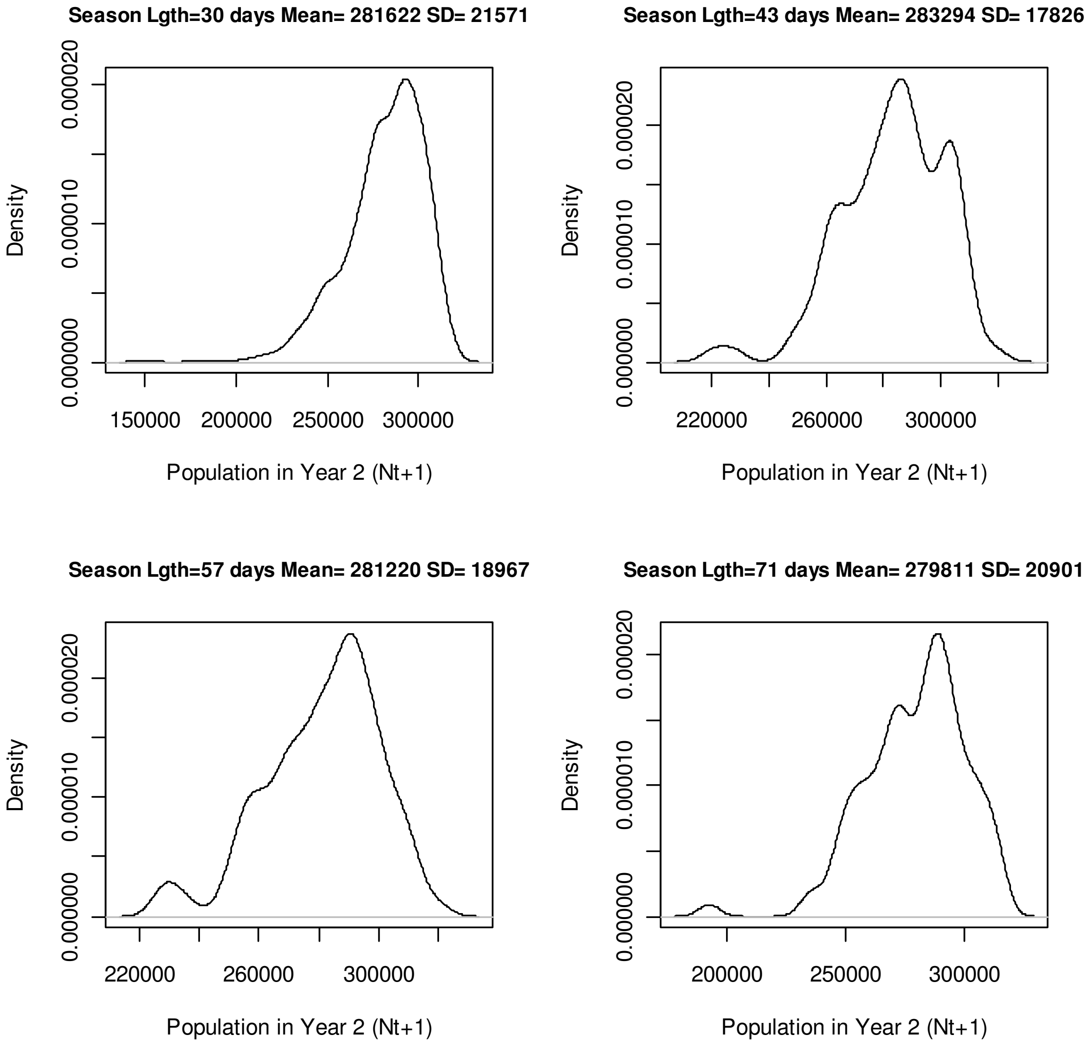


Figure 2.71. Expected year 2 population size (from the stochastic model; 1000 simulations) under different season lengths based on the 2009 (N_t) population (mean =495,512; SD= 82,136).

The population estimates in year 2 (Figure 2.71) were less than expected (~345,000) suggesting productivity correction factor (2.02) may be too high.

To predict harvest in 2010 the SDS simulation (Equation [2.51]; 1000 iterations) was run with $N_{(2010)}$ derived from Equation [2.52], and Program MARK to determine $S_{(i)(2009)}$ (model averaging; and rnorm in R was used to generate the uncertainty around the survival estimate). The simulated estimate of harvest (42,045; SE=1,992 (Figure 2.72) was on par with that recorded in the hunter survey (41,549; SE=3552).

To improve model realism and make it fully stochastic involves introducing stochasticity around the alpha estimates. This could be achieved through an additional banding session prior to the beginning of the hunting season or further telemetry work during this period. Program WinBugs could also be used to generate a distribution based on the known fate telemetry work.

Harvest and subsequent season population size models generate a non-parametric estimate that clearly portray uncertainty (although under estimated) associated with the appropriate regulation strategy. The models incorporate knowledge of density dependence, and effects of effort on harvest rates. What's more it allows competing models to be easily tested and their relative performance assessed. Perhaps this is not the traditional approach to assessing uncertainty but it provides an alternative that is easily updated and manipulated as better information comes to hand.

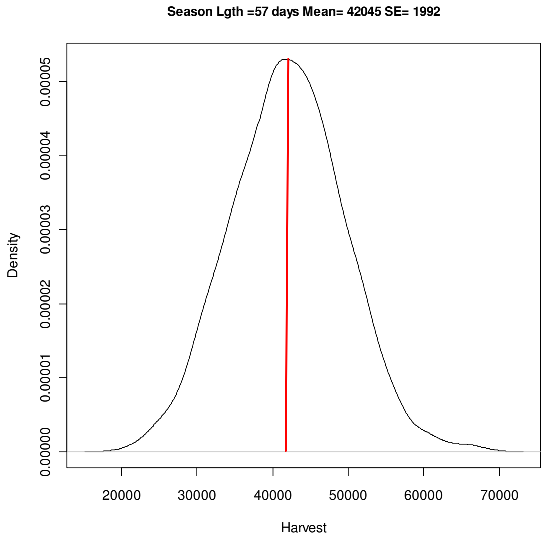


Figure 2.72. Simulated harvest (42,045; SE 1,992) for the 2010 season based on 2009 data. The 2010 mallard harvest estimate was 41,549 (34,586-48,511).

Comment on the two Regulation Strategies.

As a general comment the ease of use and interpretation of the State Dependent strategy make it more desirable than the Threshold strategy. The SDS provides governors with a process on which to decide an appropriate season length while having a visual interpretation of the risk or consequence associated with their decision. Risk in this case is the degree of confidence associated with the selected season length to deliver the hours that will maximise harvest subject to the constraints that the population in year $t_{+1} \geq t$. In addition the SDS achieves the goal to maximise harvest while not compromising the ability to maximise harvest in the following year.

The threshold technique does not seek to maximise harvest nor are there any restraints to ensure harvest is sustainable. What's more the analysis is not very accessible and in New Zealand Fish and Game governors like to have a hands on approach to setting season regulations.

Harvest Strategies; Key findings:

- Liberal season regulations in 1997 and 1998 in conjunction with low productivity in those years appear to have had a profound negative effect on subsequent population levels and harvest (Figure 2.66 and Figure 2.67).
- Modelling suggests harvest is not sustainable when productivity goes below 0.8 juvenile female: adult female in the Eastern Region (page 83).
- Once productivity ≥ 0.95 long term mallard harvest is viable under the most relaxed season constraint (71 days) in the Eastern Region.
- Assessing productivity from the trap sample is biased. The bias may be region specific (page 78).
- Threshold management does not confer sustainability (page 81).
- State Dependent Strategy provides a means for Councils to maximise harvest while ensuring sustainability (subject to productivity not going below 0.8 in the Eastern Region) under the current regulation set (page 85).

Management Implications:

- Simulations suggest that there is only a small difference in productivity between unsustainable and long term viable mallard harvest. Habitat management (provision of overhead cover from hawks) and perhaps predator control may be enough to ensure long term harvest viability.

SDS and Adaptive Management

There is a large degree of uncertainty in every aspect of the heuristic model and corresponding empirical models. One of the final steps in decision theory process is specifying constraints (Possingham et al., 2001). An important constraint in any harvest management program is cost. Improving certainty around estimates can be expensive. Therefore it is worth identifying where best to focus and prioritise monitoring. Further, some aspects of the heuristic model have a more pronounced effect on the decision process (regulation strategy), for example productivity, so it is prudent to formulate monitoring stratagems and management that will enhance understanding in these areas.

Effort: Season regulations rely on effort being constrained to control harvest rates in years when state variables such as population size or productivity are low. It appears however that the shorter seasons motivate hunters to make the most of the reduced opportunity and may hunt for more hours than some of the longer seasons. A second issue is the variance around estimates of effort, clouding the effect of season regulations. The hunter survey is designed to have a coefficient of variation (CV) of 6% around mallard estimates (Barker and MacKenzie, 1999). In 1998 the CV of hours hunted waterfowl in the Eastern Region was 6.5% (Barker and MacKenzie, 1999) which is reasonable. Improving precision by conducting more interviews runs the risk of survey fatigue (reluctance of hunters to be interviewed multiple times). To overcome this Barker and Mackenzie (1999) suggest tagging previous interviewees in order that they are not interviewed more than twice.

Increasing precision around effort estimates is probably one of the cheapest methods of improving uncertainty in the heuristic harvest model. Nevertheless it may be that effort is truly variable. Some hunters hunt everyday while others hunt for as little as half an hour per fortnight after opening weekend.

During this study I used contemporary regulations which have a minimum season length of 30 days and a maximum of 71 days. It appears that productivity seldom gets to the point where a 30 day harvest is not sustainable so introducing season lengths below this figure to determine if there is a saturation point (season length limits hours hunted) is probably not politically reasonable. At the other end of the spectrum, increasing the season length to determine if there is an impact on productivity requires a better measure of productivity than is currently available.

During the study period there were very few short seasons. Constraining season length when state variable indicate constraint is not necessary (to obtain a better idea of the effect of season length on effort) would be helpful in improving understanding but is probably not warranted in light of the overall goal to maximise harvest. Here a passive management approach may be more appropriate.

Harvest Rates: The relationship between effort and harvest rates in the Eastern Region approached significance at 5% for the adults once the 1997 data point was removed. The correlation was not significant for juveniles nor was it for any of the Hawke's Bay cohorts. But because effort explained changes in survival better than any other model (in the Eastern Region) and that the correlation between effort and harvest was highly significant, it is highly plausible that reporting rate is confounding the relationship between effort and harvest rate.

Despite poor correlation between effort and harvest rates it was interesting to note the significant difference between harvest rates of different season length. This was surprising given the above and that regulations had only a poor influence on effort. The significant relationship between season regulations and harvest rates implies that in years of low population harvest rates are low (season regulations were constrained when the population was assessed as low). There are three possible explanations for low harvest rates when the population is low: (1) The law of diminishing returns (Strickland (1996) i.e. in years the population is low hunters hunt less. (2) In years when the population is low hunters are less successful. (3) There is an association between harvest rates and population size (harvest rates and estimates of population size are inextricably linked; see equation [2.24]). However under this scenario we would expect harvest rates to increase as population decreased.

There was no evidence that hunters exert more effort if they are successful (or vice versa). Which suggests low populations produce low harvest rates which may explain how game bird hunting in New Zealand has remained sustainable over the years despite little regulatory intervention?

The lack of correlation between effort and harvest rates will in part be due to the impact of environmental conditions such as excessive ephemeral water[1] in some years reducing harvest regardless of effort (similar to some fisheries where catch rate is a function of density not population size; Beard et al., 1997). It would be useful to

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[1] Anecdotal evidence suggest climate conditions may have a large bearing on harvest rates. Apart from excessive ephemeral water other climate conditions include windy conditions (apparently in calm weather many ducks go out to sea), and fine conditions vs. heavy rain ducks fly higher and are less inclined to decoy, making them less susceptible to hunting.

incorporate some of these environmental variables where they are likely to influence density or concentration, in models of harvest rates to try and improve our comprehension of the harvest processes. Using season regulations to enhance understanding of harvest rates may be another option; for example fixing season length. The fixed 57 day season in the Eastern Region provided an opportunity to determine the effectiveness of bag limits on harvest rates. Manipulation of season length at different population sizes to evaluate the relationship between harvest rate and population size would be worthwhile provided an independent assessment of population size is initiated in conjunction.

It would also be beneficial to improve estimates and understanding around reporting rate. Why for example, do hunters in the Hawke's Bay report significantly fewer bands than the Eastern Region hunters? Nichols et al. (1995b) also found evidence of spatial variation in reporting rates between different areas in the U.S.. The difference in reporting rate between the two regions may be associated with the low level of compliance particularly early on in the study (unlicensed hunters may be less inclined to report a band fearing prosecution). It may also be an awareness issue that could benefit from an educational or promotional exercise? Increasing the number of bands reported is fundamental as poor reporting rate also has serious implications around precision of survival rate estimates (Sheaffer and Malecki, 1995). Reward bands[1] may be an alternative option to determine recovery rate (Pollock et al., 1994, Nichols et al., 1995b, Royle and Garrettson, 2005), but are expensive (Pollock et al., 1994). Royle and Garrettson (2005) observed that reporting rate of mallards in the U.S. increased from 0.33 to 0.72 with the inclusion of an 0800 number on the band. Auckland Waikato Fish and Game Region have set up an 0800 number to report bands which the Eastern and to a lesser extent Hawke's Bay Region benefit from already. It makes sense to publicize this more widely and investigate incorporating this 0800 number on the bands in all regions. I determined reporting rate through a randomised phone survey. Increasing the number of interviewees in the current reporting rate survey may increase precision associated with the estimates but because

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[1] A monetary value is attributed to a band. At some dollar value reporting rate approaches 1 with probability *h*. Once harvest rate of the reward band is established reporting rate and *h* of the non-reward bands can be calculated (Zimmerman et al., 2009).

of the small number of hunters that shoot a banded bird it might require a large increase (a power analysis would be useful).

Telemetry work is another option in establishing reporting rate and harvest rate, but will be expensive. Nevertheless it is probably the most effective method, and it may provide an opportunity to tease apart harvest rates and survival estimates.

Productivity: Low productivity during 1997 and 1998 in the Eastern Region (Appendix B) in conjunction with liberal season regulations (71 day seasons) were followed by an apparent population collapse (Figure 2.66) for the next 5 years. Of all the variables in the SDS simulation productivity has the greatest influence on the result. This is consistent with the observations of Anderson (1975b) where he notes population fluctuations are probably more closely associated with changes in productivity than with annual changes in survival rates.

It appears that a small shift in productivity (0.8 to 0.95 juvenile female: adult female) may be the difference between sustainable and viable harvest. Managers may consider this achievable through breeding habitat management (provision of overhead cover) and perhaps predator control in key breeding areas.

It is apparent, that the trap sample is overestimating productivity. This overestimation is probably due to some adult females not being available to trap as they are in the moult and juveniles may be more trap happy. To address this problem an alternative monitoring of breeding success could be incorporated into the SDS model to improve its accuracy; for example November brood counts. Brood counts would not be confounded by the moult but have their own set of issues. Alternatively basing the number of adult females on the adult males and using their respective survival rates to establish the trap bias.

A set of explanatory models (to predict *B*) could be developed and tested against subsequent harvests. The top productivity model is then incorporated in to the deterministic model to establish effort levels that maximise harvest.

Survival: An alternative hypothesis, or in addition to that of a trap bias (in favour of the juveniles), is that juvenile survival from trapping to the beginning of the game season (α_J) is lower than the telemetry work suggests.

The female only simulation (Threshold simulation) performed differently in predicting survival compared with the full cohort simulation suggesting the pre-harvest survival probability α_A differed between sexes. This is quite understandable as the females will have just finished moulting and breeding.

Obtaining a better understand of this period in the annual cycle will improve understanding of harvest mortality and post-harvest mortality. The latter is particularly important in establishing the effects of density dependence.

Population Size: To improve the robustness of many of the models it would be constructive to have an independent assessment of population size. Although the ideal time would be late autumn, before the game season, a more useful time would be prior to setting the season regulations in late January. Nevertheless detection issues in any survey (Nichols et al., 2000, Farnsworth et al., 2002, Pollock et al., 2002, Koneff et al., 2008, Pagano and Arnold, 2009b, Pagano and Arnold, 2009a) may be a major hurdle in Eastern and Hawkes Bay Regions, particularly in the hill country. Early in this study I tried random aerial transect counts of dabbling duck in the Gisborne hill country (to establish the feasibility of this method to estimate population size) but aborted it due to the difficulty of flying low enough to provide an accurate count. The Breeding waterfowl aerial transect counts in North America are conducted at 30-50m above the ground (Smith, 1995).

To overcome this dilemma an alternative approach may be to identify high density hunting areas in the lowlands and intensively monitor these areas on the assumption that the areas beyond these monitoring sites will be subject to less hunting pressure[1] and therefore less likely to succumb to over harvest. In other words regulations that are applicable to the high intensity hunting sites should be conservative for the

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[1] Barker (1990) found paradise shelduck were subject to less hunting pressure in hill country compared with the coastal areas in the Wanganui District.

unmonitored areas. Alternatively stratify sampling (hill country and lowland) and accept that the hill country counts may be less accurate.

An alternative estimate of population size would be particularly useful in assessing density dependent survival. This study suggested that survival appears to be additive for the juveniles and may be compensatory for the post harvest Eastern adults. However I am particularly uncomfortable with using the same bands to determine both population size and survival rates (despite my attempts to randomly separate the respective data). Because of the relatively sedentary nature of the mallards, an intensive annual census within a 50km radius of the trapping sites could be useful in addressing this issue.

An improvement on the method used in this study to determine population size and associated variance would be the Hypergeometric model (Seber, 1982) in Program WinBugs. Before this can be done a method to determine n (the number of hunters) used to derive the harvest estimate needs to be established.

Optimal Harvest Strategy Summary

In this study the goals of Fish and Game management were examined and it was suggested that the goal of sustainable game bird management espoused in many of the Sports Fish and Game Management Plans is understated. It was proposed that a goal to maximise harvest within a sustainable framework was probably more appropriate.

Consistent with this goal a harvest strategy that will facilitate maximum harvest while ensuring sustainability was developed. A state dependent strategy was considered the best option. A simple heuristic harvest model was devised which was then decomposed into a series of explanatory quantitative models. When multiple possible models existed, the one with the highest information value based on Akaike's information criteria was chosen or model averaging used.

The problem was divided into two parts (1) Partial Management Control and (2) Structural Uncertainty. The first dealt with the issue that harvest regulations only provide partial control over hunter behaviour (effort). While structural uncertainty looked at how the system performs when in different states, for example density dependent survival.

The harvest system was then reconstructed using a series of Markovian models (their performance was a function of the immediate state) and evaluated on how they performed under different strategies.

Correlation was used extensively in this study to investigate many of the components of the heuristic harvest model, but correlation does not necessarily mean cause and effect. The next step to advance understanding of the harvest process involves instigating management or experimentation to test associated empirical models, and any new theory, through passive, and/ or active adaptive management and/or experimental investigation. Top priority in order are:

1. A more accurate measure of productivity;
2. Develop a better measure of, and investigate what influences, harvest rates and;
3. Establish an independent measure of population size in particular to explore density dependent post-harvest survival.

Chapter 3

Can Climatic Covariates Explain Temporal And Spatial Variation In Mallard and Parera Survival In New Zealand?

Introduction

Creating management zones is probably the most important step in establishing a good management system (Strickland et al., 1996). Current mallard and parera management is confined to 12 separate Fish and Game Regions. Fish and Game regional boundaries in New Zealand were established on old Acclimatisation Society and catchment boundaries. These boundaries may not provide ideal management zones for mallard and parera as regional boundaries often encompass multiple environmental zones.

Environmental stochasticity is a significant issue in predicting the effects or consequences of management (Williams et al., 1996). Climate and climate events are likely to be an important component of environmental stochasticity. Furthermore we have seen in the previous chapter that hunter behaviour and, anecdotally, harvest rates, may be influenced by weather conditions. Therefore confining management to homogeneous climate zones, effectively stratify management, is prudent. This is particularly so in an adaptive management (AM) framework where mitigating uncertainty is a fundamental goal.

In addition to mitigating uncertainty there are other advantages of creating management zones around homogenous climate zones such as (1) improved regulatory management (regulatory consistency and less conservative management), (2) improved understanding of causal relationships between environment and state variables; and a better understanding of (3) regulatory consequences (partial controllability) and (4) structural uncertainty (e.g. density dependent survival and recruitment).

Land use, vegetation (Leathwick et al., 2003) and, to a lesser extent topography will also be a function of climate. Land use has impacted on waterfowl production in the U.S. (Bethke and Nudds, 1995) and topography in the U.S. explains changes in waterfowl richness (number of species) (Bethke and Nudds, 1993) so homogenous climate zones may be more instrumental in their effect on mallard dynamics through the synergistic effects of topography and land use.

Nevertheless New Zealand is considered to have a temperate climate and it could be argued that mallard may do well irrespective of the prevailing weather conditions. What's more, despite the assumptions above, other factors such as harvest, disease, land use, agricultural practices (Giudice, 2003), and topography may dominate any climate effects. There is already evidence that mallard survival varies geographically in New Zealand (Nichols et al., 1990) and that harvest effort is an important component of survival in at least two regions (see previous chapter). Many regions however have maintained consistent regulations over the years[1] so the impact of climate, if my premise is sound, will receive some support in explaining changes in survival.

Although survival probably contributes less to population change than productivity (Anderson (1975b), and previous chapter), and post- fledgling productivity is influenced by climate in the U.S. (Krapu et al., 2000, Miller, 2000), I am not confident that the trap sample would provide a reliable comparative measure of temporal and spatial productivity to evaluate the effects of climate on productivity. Therefore I have concentrated on survival rate and the influence of climate.

In this chapter a number of linear models are tested incorporating climate covariates against some more general models (including sex and time dependence). Support for the climate models infers climate explains changes in survival better than any of the other candidate models.

It is predicted that the relative influence of the climate models will differ geographically. For instance rainfall will be an important influence on survival in low

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[1] Consistent regulations do not necessarily imply consistent harvest.

rainfall areas (such as the Wairarapa and East Coast) whereas temperature will have an influence in the colder climes (such as Otago and Southland).

Geography and Climate

New Zealand is an archipelago situated in the Southern Pacific Ocean spanning 13^0 Latitude (34^0 S - 47^0 S) predominately made up of two large islands. The South Island is characterised by a central alpine fault that bisects its length, while the North Island has a less pronounced central divide (confined to the southern half). In Northland (latitude 34^0 south) the average yearly temperature is 16.1-18°C while Southland (latitude 47^0 south) is a little colder (8.1-10 °C) (http://www.niwa.co.nz). Precipitation is higher in the West particularly in the South Island (4000-10,000mm) and dryer in the East; for example Canterbury and parts of Marlborough (500-750mm). Eastern areas of the North Island particularly around the Hawke's Bay, Wairarapa and, Gisborne are often prone to drought. These key topographical features, prevailing westerly weather patterns, and latitudinal variance, have created a diverse array of climate and ecosystems (Williams et al., 2007).

Survival and latitudinal gradients

Spatial or latitudinal gradients explain changes in survival of mallard populations outside of New Zealand, for example prairie mallard in the United States (Sæther et al., 2008). Species richness (number of waterfowl species) also varies with latitude, and this relationship could be explained through changes in pond morphology (due to the effects of glaciations) (Bethke and Nudds, 1993). In New Zealand there is some evidence of latitudinal variation in species richness. The extinct finches duck (*Euryanas finschi)* appears to have been more prevalent in the South Island (Worthy and Holdaway, 2002). Other extant species such as the paradise shelduck (*Tadorna variegata*) were predominately found in the eastern part of the South Island (Williams, 1981) and were unknown west of the Ruahine and Kaimanawa Ranges (McAllum, 1965).

Determinants of mallard survival

Annual stress periods such as winter (Smith and Prince, 1973, Heitmeyer, 1988, Bergan and Smith, 1993), moult (Pehrsson, 1987, Panek and Majewski, 1990), migration (Heitmeyer, 1988) and, reproduction (Gloutney and Clark, 1991, Alisauskas and Ankney, 1992) may explain changes in mallard body condition.

Female mallard survival rates are higher in individuals in good body condition at time of capture (Bergan and Smith, 1993, Pollock et al., 1989). Some of the variation in survival rates may be explained through greater recovery rates (hunter killed) of mallards in poor condition (Hepp et al., 1986). Clinton et al. (1994) found no relationship between body condition and survival but suggest that there may be a threshold response between body condition and survival whereby survival is compromised once body lipid levels drop below a certain level. Loesch et al., (1992) reported that captive female mallards fed ad libitum lost weight over winter with no relationship to winter temperature and suggested that weight loss may have been endogenously controlled. Nevertheless they were unable to exclude the hypothesis that weight loss may have in part been due to the physiological cost of prebasic moult (particularly in the females) and pair formation.

Early winter rain of sufficient quantity in the Mingo Basin, Missouri, USA, progressed the female mallard annual cycle events and improved survival (Heitmeyer, 1988). Habitat and events occurring during the non-breeding period are also responsible for changes in survival and reproductive rates (Baldassarre and Bolen, 2006).

Climate effects on body condition can, in part, be explained through effects on basal metabolic rate (BMR). BMR is a function of size and temperature (Baldassarre and Bolen, 2006). For example there is a point known as the “lower critical temperature” (LCT)[1] where the core body temperature cannot be maintained without increasing BMR. For example an 1100 g female Mallard has a LTC of about 13°C.

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[1] For non-passerine birds $LCT = 47.17W^{-0.1809}$ where W is body weight in gms and LCT is °C. (Baldassarre and Bolen 2006).

It is conceivable that climate events (precipitation and temperature) in New Zealand exacerbate effects of body condition during stress events such as moult and reproduction (mallard are relatively sedentary in New Zealand which very likely reduce the annual endogenous energy requirements compared with migratory birds). Climate will also alter primary production, affecting exogenous energy supplies during pre and post-event periods.

The *a priori* candidate model set (Table 3.1) is based on climate events that may impact on key life cycle events (Figure 3.1) particularly of the female (sex effects) (Alisauskas and Ankney, 1992). For example, apparent sex effects in mallards are expressed as different strategies for dealing with harsh climate events, with females storing a greater proportion of their total weight as lipid (Boos et al., 2002).

Climate events at time of stress may be important but so may rainfall and temperature in the months prior to the year of interest (i.e. spring rain prior [*SppR*] and spring temperature prior [*SppT*]). Bethke and Nudds (1993) report that precipitation in the year(s) preceding a given breeding season on the prairies have shown to be better predictors of current wetland conditions and abundance of breeding ducks than have spring precipitation or aerial pond counts.

To keep the model set to a relatively small number (Anderson and Burnham, 2002) parameter structure of recapture and fidelity to the study site is left general (cohort and time dependent) despite the likelihood that they may be influenced by climate; birds may move out of the area during drought years and may be more trap happy when food is short. While the Climate models incorporate age specific recovery differences (New Zealand mallard and parera juveniles are more likely to be recovered than adults) (Balham and Miers, 1959, Nichols et al., 1990, Caithness et al., 1991).

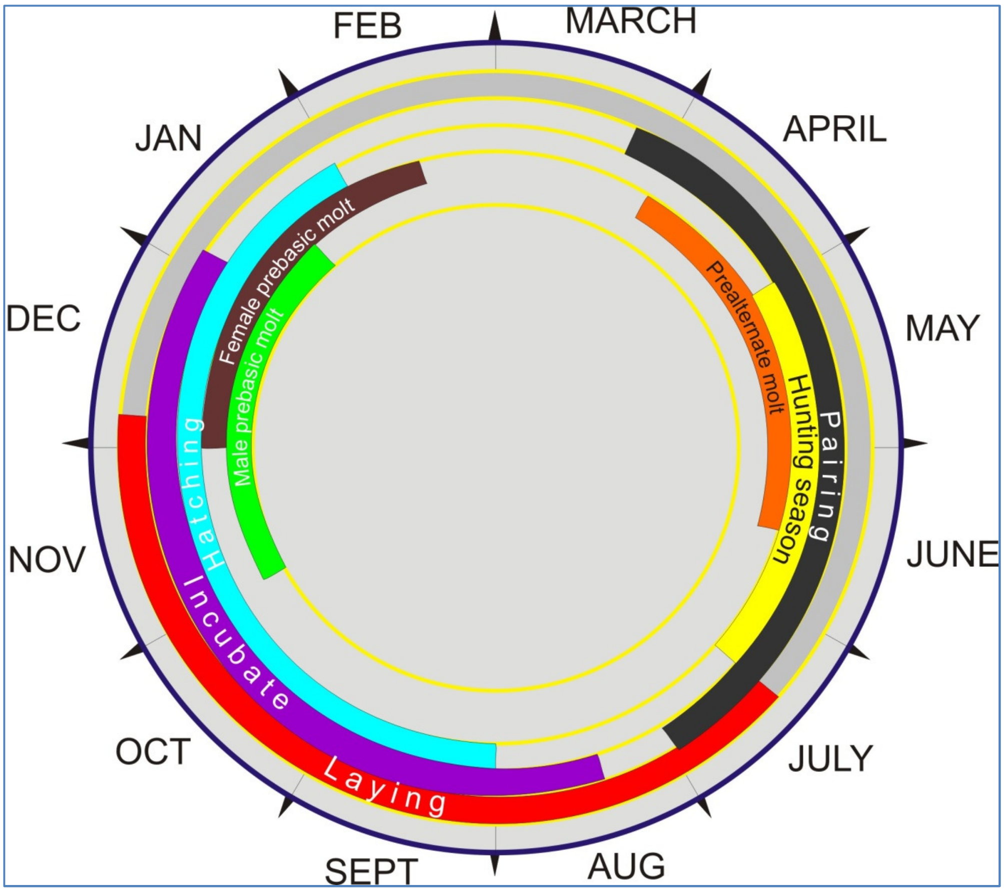


Figure 3.1: Annual Life Cycle of Mallard and Parera in New Zealand.

The set of alternate non-climate models included age and time dependent structure for survival and conditional reporting rates based on findings by findings of Nichols et al. (1990) and Caithness et al. (1991).

Methods

Between 1997 – 2009 18,862 mallard and 2,259 parera were trapped and banded at 26 discrete sites within the Eastern and Hawke's Bay Fish and Game Regions. In addition I extracted historic (1968-2009) band data sets from the Banding Office, Department of Conservation, comprising 69,000 mallard and parera banded predominately by Wildlife Service and Acclimatisation Society staff throughout New Zealand comprising about 13 different studies conducted either concurrently or at different times. The trapping and banding methods described in Appendix A were employed during this study but are also pertinent to the other studies. Survival analysis is described in the previous chapter (Survival, page 25).

Study Area

Band site location is recorded as one of 27 band regions (Department of Conservation Banding Office protocols, originally established on old Acclimatisation Society Boundaries) throughout New Zealand (Figure 3.2 and Figure 3.3). Results of the data analysis are reported against band region.

Figure 3.2: North Island Banding Regions

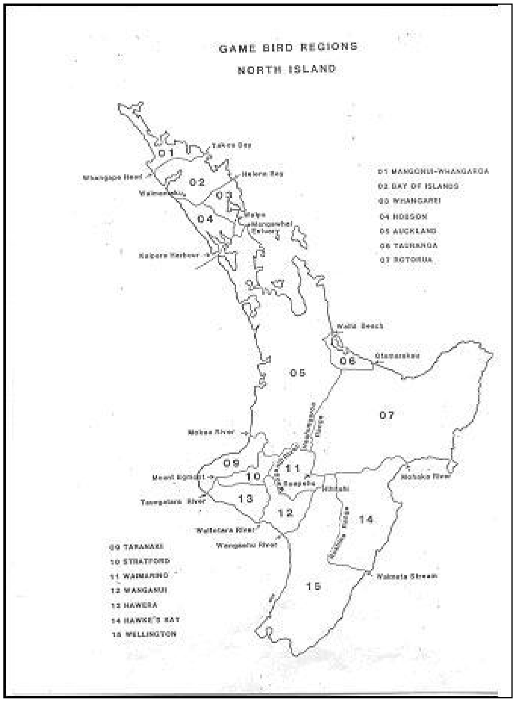
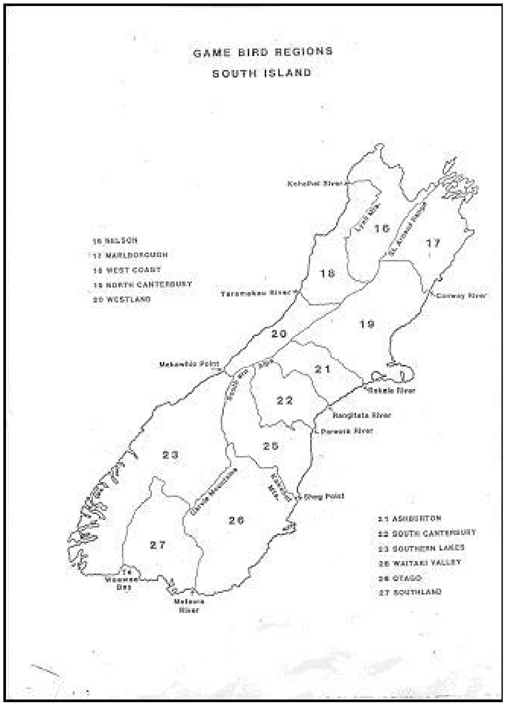


Figure 3.3: South Island Banding Region



Climate Data

Climate data were obtained from NIWA (now available directly from the website http://cliflo.niwa.co.nz/). Rainfall and temperature was extrapolated from the closest weather station to the band sites with a complete record for the relevant study period.

Season notation used did not always align exactly with the true seasons but rather coincided with key life cycle factors or periods during which the birds were assumed to be physiologically stressed, such as the moult, breeding season and, winter**.**

Climate covariates are recorded as Spring Rain (*SppR*) where the spring (October, November and, December) was the average rainfall for those three months prior to the banding year. Spring Rain for the year of banding (current year) (*SpcR*) was the average for September, October and November. Winter Rain (*WR*) was the average rainfall for June, July and, August. Summer Rain (*SuR*) was the average rainfall for January, February and March. Spring temperature for the spring prior to the banding year (*SppT*) and the current year, (*SpcT*) align with the spring rainfall periods mentioned above. Summer temperature (*SuT*) is the average temperature for December prior to the banding year and, January and February in the banding year. Winter temperature (*WT*) is the average monthly June-August temperature in the banding year.

Data Analysis

A set of candidate models were derived after careful deliberation of what are likely to be the most influential factors governing encounter probabilities (Table 3.1). Harvest models were not included as harvest data was not available for much of this study period. To reiterate (see Survival, page25), models represent the probability that the bird has survived (S), the bird is recaptured in subsequent trapping occasions (P), the band has been recovered and reported (r), and the bird showed fidelity to the band site (F). Parameters are group dependent (g; all cohorts; adult females, adult males, juvenile females and, juvenile males, banded in the same year), age (the effect of being a juveniles is additive to the sex effect), time dependent (parameter estimates change over time) and a function of eight different climate covariates.

The age time dependent reporting rate was used extensively (juveniles are more likely to be shot and therefore reported (Nichols et al., 1990, Caithness et al., 1991). Recapture and fidelity were kept as group time dependent in most models as they will be influenced by a multitude of factors that will differ between sex and age and over time (see discussion above).

Table 3.1: Candidate model set and Explanation

Model	Explanation
$\{S_{(g*t)}P_{(g*t)}r_{(g*t)}F_{(g*t)}\}$	Global Model: Survival (S), recapture (P), conditional reporting (r) and, fidelity (F) are group (cohort; adult female, adult male, juvenile female, juvenile male banded in the same year) and time dependent.
$\{S_{(g)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Group Model: Survival (S) is group dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile $^{(1)}$) and time dependent.
$\{S_{(sex)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Sex Model: Survival (S) is sex dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile at first capture$^{(1)}$) and time dependent.
$\{S_{(sex+age)*t}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Sex and age model: Survival (S) is sex, age (adult or juvenile at first capture), and time dependent
$\{S_{(g*t)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Time dependent group Model: Survival (S) is group and time dependent, recapture (P) and fidelity (F) are group and time dependent and, conditional reporting (r) is age (adult or juvenile).
$\{S_{(.)}P_{(.)}r_{(.)}F_{(.)}\}$	Fully constrained dot Model: Survival (S), recapture (P), conditional reporting (r) and, fidelity (F) are constant over time (dot model).
$\{S_{(.)}P_{(g*t)}r_{(g*t)}F_{(g*t)}\}$	Survival dot Model: Survival is constant over time. The rest of the parameters are group and time dependent.
$\{S_{(sex*cc+age)}P_{(g*t)}r_{(age*t)}F_{(g*t)}\}$	Climate Models: This model represents 8 different models where survival (S) is sex and age dependent and is constrained by one of the climate covariates (cc); winter rain (*WR*); spring rain in the preceding year (*SppR*); spring rain in the current year (*SpcR*); summer rain (*SuR*); spring temperature in the preceding year (*SppT*); spring temperature in the current banding year (*SpcT*); winter temperature (*WT*) and summer temperature (*SuT*).

1). Birds trapped as juveniles at first encounter are modelled as juveniles for the first year and then adults in subsequent years.

Climate models were constrained to a linear function on the logit scale where the response variable (logit(S)) was dependent on the sex, and a sex-climate interaction term, plus the age (adult or juvenile) of the bird. Therefore survival varied with the climate covariate and the respective juvenile survival was in parallel to the adults of the same sex. For example {S(sex*SuR+age)p(g*t)r(age*t)F(g*t)} suggests survival is dependent on the sex of the bird, whether it is a first year, and that it varies with summer rain.

Results

Results are reported against band region (Figure 3.2 and Figure 3.3, Page 110). Eighteen data sets (DS) comprising 91,500 mallard, parera, banded between 1968-2009 are examined. Only models with ΔAICc <10 or ΔQAICc <10 are presented. A summary of the key results are depicted at the end of this section in Figure 3.4 and Figure 3.5 (page 133).

Auckland (Region 5)

Region 5 runs from the Kaipara Harbour in the North (36°25′) to Mokau River (38°42′) in the South. The majority of Mallard and Parera were banded around the Waikato Lakes (37°28′) during three periods;

- Data Set 1 & 2, 1968 – 1974 (7000 mallard and 3327 parera; at 17 locations;
- Data Set 3 & 4, 1979 – 1983 (4265 mallard and 1114 parera; at Lake Whangape) and;
- Data Set 5, 2002-2009 (669 parera and 32,019 mallard at 26 sites) (Table 3.2 and Table 3.3).

Data for the climate covariate models was from Ruakura weather station (C75731) and Ruakura EWS (C75733). Model likelihood for all five data sets is presented in Table 3.7.

Table 3.2: Auckland mallards banded 1968 – 2009

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1968	65	72	93	162
1969	134	119	529	543
1970	179	197	263	280
1971	162	328	344	382
1972	102	131	289	367
1973	179	127	228	293
1974	285	383	378	386
1979	262	177	126	132
1980	152	230	184	263
1981	142	235	105	147
1982	281	394	297	354
1983	230	143	151	260
2002	99	182	519	661
2003	372	632	1570	2181
2004	264	186	1465	1798
2005	272	286	1641	2502
2006	572	952	1705	2594
2007	388	390	1664	2248
2008	241	148	1284	1668
2009	239	136	1290	1870

Table 3.3: Auckland parera banded 1968 – 2009

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1968	52	41	207	223
1969	51	55	223	311
1970	67	55	172	197
1971	41	60	78	88
1972	32	46	150	198
1973	61	41	154	174
1974	45	55	200	250
1979	10	13	16	23
1980	26	46	144	183
1981	36	22	45	56
1982	34	39	75	133
1983	36	47	61	69
2002	2	3	35	59
2003	5	4	18	18
2004	8	4	70	56
2005	11	12	39	46
2006	8	4	24	35
2007	12	10	47	55
2008	5	6	12	26
2009	3	2	12	18

Data Set 1 & 2: There was no support for the climate models in band period 1968-1974 for either mallard or parera data sets. The sex and age, time dependent survival models received 100% of the support in both analyses.

Data Set 3 & 4: Mallard summer rain (*SuR*) model received limited support (ΔAICc=2.215; w_i= 0.24; refer to footnote 1 page 28) in band period 1979-1983 while ΔQAIC for all the climate models for the parera data set were less than 7.7. The parera top ranked model, Spring Rain preceding year (*SppR*) received 66% of the total support.

Data Set 5: Seven of the climate models received some support (ΔQAICc<8.89) for the combined mallard and parera 2002-2009 data set. The top ranked climate model (*Summer Rain – SuR*; ΔQAICc=1.579) received 21% support. The top model was, like Data Set 1 & 2, the sex and age, time dependent survival model ($w_i = 0.46$).

Biological legitimacy of the top ranked model can be interpreted through an examination of the underlying Beta estimates for the survival parameters. Table 3.4 shows the Beta estimates for mallard survival for band period 1979-1983 (Data Set 3).

Table 3.4: Beta estimates for survival parameters {S(sex*SuR+age) p(g*t) r(age*t) F(g*t)} Auckland mallards 1979-1983

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept (β_1)	3.9968	0.3161	3.3773	4.6163
Sex (β_2)	1.0381	0.2827	0.4840	1.5921
Age (β_3)	-0.3432	0.1183	-0.5751	-0.1113
Summer Rain (SuR) (β_4)	-0.0347	0.0033	-0.0412	-0.0282
Sex*SuR (β_5)	-0.0132	0.0034	-0.0199	-0.0065

Interpolation of the Beta estimates[1] indicates for an average Ruakura summer rain (86mm; 1979-1983) we can expect an adult female survival of 76%; while a variation in summer rain of 10% above and below the average would produce 68 and 82% adult female survival respectively[2].

Beta estimates (Table 3.5) for the top ranked model in the parera 1979-1983 data set (DS 4; *SppR*; $w_i = 0.67$) suggest an average spring rainfall in the prior year (94.3mm) would result in an average adult female parera survival of 0.5057. An increase of 10% in this rainfall would cause adult female survival to decrease 12% (0.4425) and a 10% decrease in *SppR* would result in an equivalent increase in survival (0.5687). However the 95% CI for the Beta estimate of the *SppR* covariate span 0. Most of the effect of rain is a sex effect suggesting females are more susceptible to an increase in rain than the males.

The Beta estimates (not shown) for *SppT* model (11% support) suggest a 10% increase in average spring temperature in the preceding year (15^{o} C; Ruakura weather station) will increase average survival of adult females (0.5519) by 47%.

Table 3.5: Beta Estimates for survival parameters {S(sex *SppR+age) p(g*t) r(age*t) F(g*t)} Auckland parera 1979-1983

Parameter	Beta	Standard Error	95% Confidence Interval Lower	Upper
Survival Intercept	0.7838	0.6395	-0.4697	2.0373
Sex	1.7755	0.7514	0.3027	3.2482
Age	-0.7231	0.2305	-1.1749	-0.2712
SppR	-0.0086	0.0067	-0.0218	0.0047
Sex*SppR	-0.0183	0.0072	-0.0324	-0.0043

Summer Rain (*SuR*) model received 21% support (ΔQAICc<2) for the combined mallard-parera data set (DS 5) but interpolation of Beta estimates (Table 3.6) indicate

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[1] $Logit(S) = \ln \frac{S}{(1-S)} = \beta_1 + \beta_2 Sex + \beta_3 Age + \beta_4 SuR + \beta_5 Sex.SuR$ and the back transformation is given: $\hat{S} = \frac{e^{\beta_1 + \beta_2 Sex + \beta_3 Age + \beta_4 SuR + \beta_5 Sex.SuR}}{1 + e^{\beta_1 + \beta_2 Sex + \beta_3 Age + \beta_4 SuR + \beta_5 Sex.SuR}}$

[2] Model averaging provides a more robust method of estimating the true survival rate but in this study it is relative support of the model and the effect of the climate covariate on survival that is of interest rather than the value of the underlying parameters.

only a small change in survival with changing rainfall (less than 1% increase in survival for a summer rainfall 10% above the average [70mm; Ruakura EWS weather station]).

Table 3.6: Beta Estimates for survival parameters {S(sex *SuR+age) p(g*t) r(age*t) F(g*t)} Auckland mallard and parera 2002-2009

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.2646	0.1785	-0.6146	0.0853
Sex	-0.0104	0.2364	-0.4738	0.4531
Age	-0.6240	0.0672	-0.7557	-0.4923
SuR	0.0058	0.0023	0.0014	0.0103
Sex*SuR	-0.0039	0.0029	-0.0096	0.0019

Table 3.7: Auckland band region (5) model results

Data Set	Species	Band Period	Weather Station	Model	Q/AICc	ΔQ/AICc	AICc Weights	Model Likelihood	Number Parameters	Q/ Devi	C-hat	Goodness of fit (P)
1	Mallard	1968-74	Ruakura (C75731)	{S(sex+age*t) p(g*t) r(age*t) F(g*t)}	12275.29	0	1	1	126	394.7242	1.039	P<0.01
2	Parera	1968-74	Ruakura (C75731)	{S((sex+age)*t) p(g*t) r(age*t) F(g*t)}	5070.896	0	1	1	106	172.6319	1.000	P=0.47
3	Mallard	1979-83	Ruakura (C75731)	{S(sex+age*t) p(g*t) r(age*t) F(g*t)}	6667.672	0	0.75644	1	87	181.1284	1.000	P=0.25
				{S(sex*SuR+age) p(g*t) r(age*t) F(g*t)}	6669.964	2.2915	0.24054	0.318	65	229.0242		
4	Parera	1979-83	Ruakura (C75731)	{S(sex*SppR+age)P(g*t)r(age*t)F(g*t)}	2330.69	0	0.6686	1	79	134.5936	1.4688	P<0.01
				{S(Sex*SppT+age)P(g*t)r(age*t)F(g*t)}	2334.231	3.5413	0.11381	0.1702	79	138.1349		
				{S(sex*SpcR+age)P(g*t)r(age*t)F(g*t)}	2335.368	4.6778	0.06448	0.0964	79	139.2713		
				{S(.) p(.) r(.) F(.)}	2336.37	5.6807	0.03905	0.0584	4	298.2602		
				{S(sex*SuT+age)P(g*t)r(age*t)F(g*t)}	2336.513	5.823	0.03637	0.0544	79	140.4166		
				{S(sex*SpcT+age)P(g*t)r(age*t)F(g*t)}	2337.306	6.6158	0.02447	0.0366	79	141.2094		
				{S(sex*SuR+age)P(g*t)r(age*t)F(g*t)}	2337.636	6.9467	0.02074	0.031	79	141.5403		
				{S(sex*WT+age)P(g*t)r(age*t)F(g*t)}	2338.207	7.5169	0.01559	0.0233	79	142.1104		
				{S(sex*WR+age)P(g*t)r(age*t)F(g*t)}	2338.392	7.7024	0.01421	0.0213	79	142.2959		
5	Mallard & Parera	2002-2009	Ruakura EWS (C75733)	{S((sex+age)*t) p(g*t) r(age*t) F(g*t)}	23447.38	0	0.46994	1	89	190.4358	1.5872	P<0.01
				{S((sex*SuR+age) p(g*t) r(age*t) F(g*t)}	23448.96	1.5787	0.21342	0.4541	68	234.221		
				{S((sex*SuT+age) p(g*t) r(age*t) F(g*t)}	23450.43	3.0535	0.10209	0.2172	68	235.6964		
				{S((sex*WR+age) p(g*t) r(age*t) F(g*t)}	23451.3	3.9267	0.06597	0.1404	68	236.5692		
				{S((sex*SppT+age) p(g*t) r(age*t) F(g*t)}	23451.35	3.9696	0.06457	0.1374	68	236.6124		
				{S((sex*SppR+age) p(g*t) r(age*t) F(g*t)}	23451.47	4.0974	0.06058	0.1289	68	236.7401		
				{S((sex*WT+age) p(g*t) r(age*t) F(g*t)}	23454.02	6.639	0.017	0.0362	68	239.2819		
				{S((sex*SpcT+age) p(g*t) r(age*t) F(g*t)}	23456.27	8.8945	0.0055	0.0117	68	241.5368		

Rotorua-Tauranga (Regions 6 & 7)

Band Region 6 covers a small area (37°27' - 37°41') around Tauranga Harbour with few mallard and parera banded thus Region 6 and 7 were combined (Data Sets 6-9; Table 3.14).

Region 7 encompasses the Bay of Plenty and East Coast (Figure 3.2). Intuitively East Coast survival patterns will be more similar to Hawk's Bay than Bay of Plenty therefore band data from the East Coast portion of Band Region 7 were examined as part of Region 7 (DS 6 & 7; Figure 3.6), independently (DS 10; Table 3.14) and, combined with Hawke's Bay data (Region 15, Data Set 11, Table 3.16).

The 1968 - 1971 data-set (DS 6 & 7) included 2170 mallard (Table 3.8) and 2774 parera (Table 3.9) with a geographic spread covering ≈3° Latitude and 2° Longitude (five band sites). 10,566 mallard and parera (DS 8 & 9) were banded 1997 - 2009 (Table 3.10) at eighteen sites. Climate data were from Rotorua Aero 2 (B86131) (DS 6 & 7) as it was considered most central, and Whakatane Aero weather station (B76995) (DS 8 & 9). The Whakatane climate data were incomplete, so winter rain data were obtained from Te Puke EWS (B76838). Gisborne AWS (D87695) climate records were used for Data Set 10.

Table 3.8: – Rotorua mallard banded 1968 - 1971

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1968	46	54	76	96
1969	57	78	187	273
1970	56	57	183	188
1971	104	112	259	344

Table 3.9: Rotorua parera duck banded 1968 – 1971

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1968	51	79	210	276
1969	48	96	329	415
1970	53	61	248	262
1971	54	51	273	268

Table 3.10: Mallard and parera banded in the Bay of Plenty and Taupo areas

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1997	145	120	100	226
1998	124	123	134	176
1999	190	312	188	249
2000	142	178	198	312
2001	77	117	215	357
2002	70	101	398	535
2003	125	163	249	325
2004	60	67	237	427
2005	90	80	368	532
2006	82	107	346	473
2007	73	118	234	357
2008	34	48	192	289
2009	122	69	221	291

Data Set 6 & 7: The top ranked model for mallard recapture and recovery data banded 1968-1971 was the fully constrained *dot* model ,{S(.) p(.) r(.) F(.)} (w_i=0.98) with marginal support for the *WR* model (ΔAICc=8.54; w_i=0.013). Similar results were obtained for the parera data with the *dot* model receiving 99% support.

Data Set 8 & 9: Climate covariate models for mallard and parera banded 1997-2009 (DS 8) received no support (sex, age and, time dependent survival model; w_i=0.99). When the latitudinal range was restricted (Taupo birds removed[1] – DS 9) all eight climate models were supported (ΔQAICc≤6.92; $\sum w_i = 0.94$). The *WT* top model received 48% of total support.

Interpolation of the Beta estimates for the *WT* model (Table 3.11) for an average winter temperature of 9.6^o C (Whakatane Aero AWS) yields an adult female survival of 0.6781. A 10% increase in the average winter temperature will decrease adult female survival by 20%. Conversely a 10% decrease in winter temperature will increase survival by 17% of the average survival. We see from the Beta estimates the affect of winter temperature was predominantly sex based.

Table 3.11: Beta Estimates for survival parameters for the top ranked model {S(Sex*WT+age) p(g*t) r(age*t) F(g*t)}; Bay of Plenty Mallard & Parera, 1997-2009.

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	1.4726	1.3978	-1.2671	4.2122
Sex	5.1468	1.6862	1.8418	8.4519
Age	-1.0467	0.0900	-1.2230	-0.8703
Winter Temp (WT)	-0.0931	0.1430	-0.3733	0.1871
Sex*WT	-0.5160	0.1716	-0.8524	-0.1797

Data Set 10: QAICc ranking of models for mallard and parera banded 1998-2009 in Region 7 on the East Coast indicated no support for the climate covariate models with overwhelming support for the fully constrained *dot* model (Table 3.14).

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[1] It was considered that the Taupo birds would be subject to different a different climate than the other Rotorua Band Region birds.

East Coast – Hawke's Bay (Region 7 & 15)

Birds banded (11,958 mallard and parera) on the East Coast in Band Region 7 and Hawke's Bay (Region 15; nine sites) between 1998-2009 were amalgamated (Table 3.12 and Table 3.13). Banding did not commence in the Hawke's Bay until 2000.

Table 3.12: Parera banded in the East Coast and Hawke's Bay areas

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1998	22	18	20	27
1999	5	12	5	7
2000	14	23	21	24
2001	11	18	12	12
2002	7	6	27	43
2003	7	25	10	11
2004	12	14	11	30
2005	9	17	16	21
2006	8	17	9	9
2007	10	14	11	21
2008	9	6	13	13
2009	13	26	16	20

Table 3.13: Mallard banded in the East Coast and Hawke's Bay areas

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1998	172	207	107	223
1999	67	159	46	99
2000	210	405	168	215
2001	230	506	178	315
2002	79	139	290	423
2003	198	218	298	389
2004	121	153	344	575
2005	169	157	348	464
2006	133	237	148	285
2007	205	251	323	400
2008	140	277	253	277
2009	97	142	153	233

Table 3.14: Model Results Rotorua – Tauranga Band Region (6 & 7)

Data Set	Species	Band Period	Weather Station	Model	Q/ AICc	ΔQ/ AICc	Q/ AICc Weights	Model Likelihood	Number Parameters	Q/ Deviance	C-hat	Global Model Goodness of fit (P)
6	Mallard	1968-1971	Rotorua Aero 2 (B86131)	{S(.) p(.) r(.) F(.)}	3954.84	0	0.97902	1	4	222.7694	1.000	P=11
				{S(sex*WR+age) p(g*t) r(age*t) F(g*t)}	3961.39	8.5458	0.01365	0.0139	59	118.0065		
7	Parera	1968-1971	Rotorua Aero 2 (B86131)	{S(.) p(.) r(.) F(.)}	3582	0	0.98996	1	5	217.7925	1.556	P<0.01
8	Mallard & Parera	1997-2009	Whakatane Aero (B76995)	{S((sex+age)*t) p(g*t) r(age*t) F(g*t)}	12248.9	0	0.99966	1	155	556.2069	1.154	P<0.01
9	Mallard & Parera	1997-2009	Whakatane Aero (B76995)	{S(sex*WT+age) p(g*t) r(age*t) F(g*t)}	9965.08	0	0.47656	1	123	570.1762	1.367	P<0.01
				{S(sex*SuR+age) p(g*t) r(age*t) F(g*t)}	9967.33	2.2459	0.15504	0.3253	123	572.4218		
				{S(sex*SppT+age) p(g*t) r(age*t) F(g*t)}	9967.64	2.5546	0.13286	0.2788	123	572.7307		
				{S(sex*SpcT+age) p(g*t) r(age*t) F(g*t)}	9968.09	3.0118	0.10571	0.2218	123	573.1877		
				{S(sex*WR-Te Puke+age) p(g*t) r(age*t) F(g*t)}	9968.98	3.8985	0.06785	0.1424	123	574.0746		
				{S(sex*SpcR+age) p(g*t) r(age*t) F(g*t)}	9970.53	5.4509	0.03122	0.0655	123	575.6267		
				{S(sex*SuT+FY) p(g*t) r(age*t) F(g*t)}	9971.90	6.8167	0.01577	0.0331	123	576.9925		
				{S(sex*SppR+age) p(g*t) r(age*t) F(g*t)}	9972.00	6.9199	0.01498	0.0314	123	577.096		
10	Mallard & Parera	1998-2009	Gisborne AWS (D87695)	{S(.) p(.) r(.) F(.)}	6052	0	1	1	5	731.3065	1.3033	P<0.01

Data Set 11 & 12: East Coast and Hawke's Bay pooled mallard and parera (DS 11, Table 3.16) showed overwhelming support (100%) for the sex age, time dependent survival model. Separating the data into a more homogenous latitudinal range (DS 10 and 12) received varying empirical support. All eight climate models received some support (ΔQAICc $\leq$9.748) for the Hawke's Bay banded birds (DS 12, Table 3.16). The *SppT* model was the highest ranking model (w_i=0.79).

As reported above the East Coast data (DS 10) showed no support for the climate models.

DS 12; during an average Hawke's Bay *SppT* (15.6^{o}C) we can expect an adult female survival of 73% (Beta estimates Table 3.15). A 10% increase in average spring temperature in the preceding year would result in a 30% increase in survival where as a 10% decrease in temperature we could expect a 61% decrease in adult female survival. Beta estimates suggest that the temperature affect is predominately confined to the females.

Table 3.15: Beta Estimates for survival parameters for the top ranked model {S(Sex*SppT+age) p(g*t) r(age*t) F(g*t)}; Hawke's Bay mallard & parera, 2000 - 2009.

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-5.4152	3.8038	-12.8706	2.0401
Sex	-13.0470	3.9098	-20.7101	-5.3839
Age	-0.8736	0.1203	-1.1095	-0.6377
Spring preceding Temp	0.3891	0.2481	-0.0972	0.8755
Sex*SppT	0.8630	0.2522	0.3687	1.3573

Table 3.16: East Coast (Rotorua and Hawke's Bay Band Region 7 & 14) Model Results

Data Set	Species	Band Period	Weather Station	Model	QAICc	ΔQAICc	QAICc Weights	Model Likelihood	Number Parameters	Q Deviance	C-hat	Global Model Goodness of fit (P)
11	Mallard & Parera	1998-2009	Napier AWS (D96484)	{S((sex+age)*t) p(g*t) r(age*t) F(g*t)}	15853.02	0	1	1	155	822.5196	1.13	P<0.01
12	Mallard & Parera	2000-2009	Napier AWS (D96484)	{S(sex*SppT+Age) p(g*t) r(age*t) F(g*t)}	8529.151	0	0.78915	1	90	516.0854	1.159	P<0.01
				{S(sex*SpcT+Age) p(g*t) r(age*t) F(g*t)}	8533.17	4.0193	0.10577	0.134	90	520.1047		
				{S(sex*SuT+Age) p(g*t) r(age*t) F(g*t)}	8534.931	5.7804	0.04385	0.0556	90	521.8658		
				{S(sex*SppR+Age) p(g*t) r(age*t) F(g*t)}	8535.943	6.7921	0.02644	0.0335	90	522.8775		
				{S(sex*WT+Age) p(g*t) r(age*t) F(g*t)}	8537.544	8.3926	0.01188	0.0151	90	524.478		
				{S(sex*SpcR+Age) p(g*t) r(age*t) F(g*t)}	8537.813	8.6616	0.01038	0.0132	90	524.747		
				{S(sex*SuR+Age) p(g*t) r(age*t) F(g*t)}	8538.75	9.5994	0.0065	0.0082	90	525.6848		
				{S(sex*WR+Age) p(g*t) r(age*t) F(g*t)}	8538.899	9.7484	0.00603	0.0076	90	525.8338		

Wellington (Regions 15)

Wellington data were split into birds banded in the Manawatu – Wellington districts (west of the Tararua ranges, DS 13) (16,708 mallard and parera) and those banded in the Wairarapa (east of the ranges, DS 14) (7,044 mallard and parera) (Table 3.17) at 31 sites. Results are presented in Table 3.20.

Table 3.17: – Wellington mallard and parera banded between 1972 and 1990

Year	Adult Female	Adult Male	Juvenile Female	Juvenile Male
1972	136	39	235	168
1973	123	69	188	190
1974	374	149	331	306
1975	93	67	154	170
1976	266	272	246	211
1977	104	326	360	301
1978	234	320	590	372
1979	360	262	467	380
1980	697	460	398	539
1981	484	563	532	441
1982	408	241	376	311
1983	234	259	301	209
1984		1	1	2
1985		1		
1986	367	523	288	213
1987	507	408	438	502
1988	1026	965	395	497
1989	344	243	493	473
1990	816	437	249	247

Data Set 13: There was support for all eight climate models and the group model (ΔQAICc$\leq$5.2592, Table 3.20). The Summer Rain (*SuR*) and Summer Temperature (*SuT*) models received 35 and 32 percent support respectively (ΔQAICc$\leq$0.2082).

Beta estimates (Table 3.18) for the *SuR* model indicates an adult female survival rate of 0.4616 for an average summer rainfall (64mm). A 10% increase on the average rainfall for this period would cause a 3% increase in adult female survival (0.4761) and vice-versa. The evidence however appears equivocal as the 95% CI SuR parameter and the sex effect estimates span 0.

An increase of 10% in the average summer temperature (17.6^{o}C) would cause a 21% decrease in adult female survival based on the Beta estimates of the *SuT* model (not shown). As with the *SuR* model the climate covariate estimates span 0.

Table 3.18: Beta Estimates for survival parameters for the top ranked model {S(Sex*SuR+age) p(g*t) r(age*t) F(g*t)}; Manawatu Mallard & Parera, 1972-1992.

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.1286	0.2382	-0.5956	0.3383
Sex	-0.6101	0.2805	-1.1598	-0.0604
Age	-0.2677	0.0845	-0.4334	-0.1021
SuR	0.0018	0.0034	-0.0048	0.0084
Sex*SuR	0.0074	0.0040	-0.0005	0.0152

Data Set 14: Continuous temperature data were not available for analysis of the Wairarapa mallard and parera banded 1974-1991. All the rain models received substantial empirical support (ΔAICc≤2.827). The top ranked models *SppR*, *SpcR* and, *WR*, received 33, 31 and, 22% support respectively.

The top ranked model (*SppR*) indicate an adult female survival rate of 0.5254 following an average preceding year spring rainfall (59mm). A 10% increase in this rainfall would result in a 3% increase in adult female survival and vice versa. As with DS 13 results, the climate parameters are equivocal.

Table 3.19: Beta Estimates for survival parameters for the top ranked model {S(Sex*SppR+age) p(g*t) r(age*t) F(g*t)}; Wairarapa Mallard & Parera, 1974 - 1991.

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.4105	0.3838	-1.1627	0.3418
Sex	-0.2151	0.3503	-0.9016	0.4714
Age	-0.3105	0.1172	-0.5402	-0.0807
SppR	0.0113	0.0062	-0.0008	0.0234
Sex*SppR	0.0011	0.0054	-0.0094	0.0116

Table 3.20: Band Wellington Region 15 Model Results

Band Set	Species	Band Period	Weather Station	Model	Q/ AICc	ΔQ/ AICc	Q/ AICc Weights	Model Likelihood	Number Parameters	Q/ Deviance	C-hat	Global Model Goodness of fit (P)
13	Mallard & Parera	1972-1991	Palmerston North (E05363)	{S(Sex*SuR+age}p(g*t)r(age*t)F(g*t)}	18469.66	0	0.35445	1	183	796.24	1.671	P<0.01
				{S(Sex*SuT+age}p(g*t)r(age*t)F(g*t)}	18469.87	0.2082	0.31941	0.9011	183	796.45		
				{S(Sex*SpcR+age}p(g*t)r(age*t)F(g*t)}	18472.3	2.635	0.09492	0.2678	183	798.882		
				{S(g) p(g*t) r(age*t) F(g*t)}	18472.49	2.8288	0.08616	0.2431	183	799.076		
				{S(Sex*SppT+age}p(g*t)r(age*t)F(g*t)}	18474.47	4.8051	0.03207	0.0905	183	801.052		
				{S(Sex*WT+age}p(g*t)r(age*t)F(g*t)}	18474.62	4.9612	0.02967	0.0837	183	801.208		
				{S(Sex*WR+age}p(g*t)r(age*t)F(g*t)}	18474.74	5.0761	0.02801	0.079	183	801.32		
				{S(Sex*SppR+age}p(g*t)r(age*t)F(g*t)}	18474.81	5.1509	0.02698	0.0761	183	801.398		
				{S(Sex*SpcT+age}p(g*t)r(age*t)F(g*t)}	18474.92	5.2592	0.02556	0.0721	183	801.506		
14	Mallard & Parera	1974-1991	Martinborough, Riverside (D15142)	{S(sex*SppR+age)P(g*t)r(age*t)F(g*t)}	10345.09	0	0.33499	1	108	412.744	1.000	0.26<P<0.27
				{S(sex*SpcR+age)P(g*t)r(age*t)F(g*t)}	10345.23	0.1461	0.31139	0.9295	108	412.891		
				{S(sex*WR+age)P(g*t)r(age*t)F(g*t)}	10345.91	0.824	0.22187	0.6623	108	413.568		
				{S(sex*SuR+age)P(g*t)r(age*t)F(g*t)}	10347.91	2.8271	0.0815	0.2433	108	415.572		
				{S(sex) p(g*t) r(age*t) F(g*t)}	10348.99	3.9053	0.04753	0.1419	105	422.833		
				{S(g)p(g*t)r(age*t)(F(g*t)}	10354.72	9.631	0.00271	0.0081	108	422.375		

Canterbury (Regions 19, 22 &, 25)

Mallard recovery and recapture data for Waitaki, South Canterbury and, North Canterbury banding regions (Latitude 42° 35’ - 45° 28’) between 1968-1973 were pooled due to the limited number banded (3219;Table 3.21 at nine band sites).

Table 3.21: Mallard banded in Canterbury.

	Adult Female	Adults Males	Juvenile Female	Juvenile Male
1968	177	244	204	251
1969	387	480	263	263
1970				
1971	38	53	43	40
1972	99	96	73	47
1973	220	146	58	37

<u>Data Set 15</u>: Results of model computations (Table 3.23) for mallard in the Canterbury Region indicate overwhelming support (100%) for the fully constrained *dot* model. In case the *dot* model anomaly was a result of the large number of encounter periods (23) relative to band periods (over-paramatised), I re-ran the model set over 10 encounter periods (encounters beyond the 10 year cut-off were pooled with the encounters for those birds that were never seen again); effectively treating the data as a 10 year study period. The reduced encounter model set (not reported) showed similar support for the *dot* model.

Otago Mallards 1968-1971 (Region 20)

4758 mallard were banded in Otago 1968-1971 at Lake Tuakitoto.

Table 3.22: Mallard banded in Otago.

	Adult Female	Adults Males	Juvenile Female	Juvenile Male
1968	48	28	613	737
1969	282	506	333	325
1970	72	189	286	392
1971	167	356	206	218

Table 3.23: Model results Canterbury Band Region 19, 22 & 25.

Band Set	Species	Band Period	Weather Station	Model	Q/ AICc	ΔQ/ AICc	Q/ AICc Weights	Model Likelihood	Number Parameters	Q/ Deviance	C-hat	Global Model Goodness of fit (P)
15	Mallard	1968-1973	Christchurch Aero weather station (H32451).	{S(.) p(.) r(.) F(.)}	4967.205	0	1	1	4	286.053	1.26	0.01>P<0.02

Table 3.24: Model results Otago Band Region 26.

Band Set	Species	Band Period	Weather Station	Model	Q/ AICc	ΔQ/ AICc	Q/ AICc Weights	Model Likelihood	Number Parameters	Q/ Deviance	C-hat	Global Model Goodness of fit (P)
16	Mallard	1968-71	Dunedin Musselburgh (I50951)	{S(sex*SuR+Age) p(g*t) r(age*t) F(g*t)}	9365.625	0	0.33776	1	63	189.6919	1.000	0.43<P<0.44
				{S(sex)p(g*t)r(Age*t)F(g*t)}	9366.156	0.5315	0.25894	0.7666	60	196.3766		
				{S(sex*SpcT+Age) p(g*t) r(age*t) F(g*t)}	9366.36	0.7356	0.23382	0.6923	63	190.4275		
				{S(g) p(g*t) r(age*t) F(g*t)}	9368.852	3.227	0.06728	0.1992	62	194.9708		
				{S(sex*SppT+Age) p(g*t) r(age*t) F(g*t)}	9370.838	5.2131	0.02492	0.0738	63	194.9051		
				{S(sex*SppR+Age) p(g*t) r(age*t) F(g*t)}	9370.941	5.3162	0.02367	0.0701	63	195.0081		
				{S(sex*SuT+Age) p(g*t) r(age*t) F(g*t)}	9371.144	5.5188	0.02139	0.0633	63	195.2107		
				{S(sex*SpcR+Age) p(g*t) r(age*t) F(g*t)}	9371.73	6.1052	0.01595	0.0472	63	195.7971		
				{S(sex*WT+Age) p(g*t) r(age*t) F(g*t)}	9372.182	6.5574	0.01273	0.0377	63	196.2494		
				{S(sex*WR+Age) p(g*t) r(age*t) F(g*t)}	9374.76	9.1352	0.00351	0.0104	63	198.8272		

Data Set 16: All climate Otago models received some empirical support (ΔAICc$\leq$9.1352). The top ranked model (*SuR*) received 33% of the support while the next climate model (*SpcT*) was ranked third (ΔAICc=0.7356; w_i=0.23).

Beta estimates (Table 3.25) of the *SuR* model suggest an adult female survival of 0.4222 during an average summer (63mm rain). A 10% increase in the average summer rain coincides with a 5% increase in this survival and vice versa. The sex effect was however equivocal.

Table 3.25: Beta estimates for survival parameters for the top ranked model {S(Sex*SuR+age) p(g*t) r(age*t) F(g*t)}; Otago mallard, 1968-1971

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.7505	0.3988	-1.5321	0.0311
Sex	-0.4828	0.2986	-1.0680	0.1025
Age	-0.0999	0.1199	-0.3350	0.1352
SuR	0.0136	0.0053	0.0032	0.0240
Sex*SuR	0.0010	0.0041	-0.0069	0.0090

Southland Band Region (26 &27)

The Southland Band Region includes the Otago Band Region (26). But birds banded in this band region were confined to what is more traditionally referred to as Southland. 10,887 mallard (Table 3.26) were banded over two periods 1969-1974 (Data Set 17; 12 sites) and 1987-1991 (Data Set 18; 8 sites).

Table 3.26: Mallard banded in the Southland Region 1969-1991

Adult Female	Adult Male	Juvenile Female	Juvenile Male
1969	196	550	152
1970	342	544	79
1971	633	777	169
1972	534	564	251
1973	690	464	349
1974	617	280	325
1987	113	229	148
1988	202	224	415
1989	292	283	346
1990	282	245	154
1991	122	51	265

Data Set 17: Four climate models (Table 3.29) received varying empirical support (ΔQAICc≤9.2318). The three top ranked models *WR, SpcT* (ΔQAICc=1.574) and, *SuT* (ΔQAICc=2.883) accounted for 99% of the total support.

Interpolation of the Beta estimates (Table 3.27) for the top ranked *WR* model (w_i=0.59) suggests that a 10% increase in the average winter rain will result in a 5% increase in the survival of the adult female.

Table 3.27: Beta Estimates for survival parameters for the top ranked model {S(Sex*WR+age) p(g*t) r(age*t) F(g*t)}; Southland Mallard, 1969-1974

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.5963	0.4116	-1.4030	0.2104
Sex	-0.3401	0.2738	-0.8767	0.1965
Age	-0.6744	0.1086	-0.8871	-0.4616
Winter Rain (WR)	0.0180	0.0060	0.0062	0.0298
Sex*WR	0.0076	0.0036	0.0006	0.0146

Data Set 18: All the climate models received some empirical support (ΔQAICc≤6.2549). The three top ranked models *SuT, WT* (ΔQAICc=0.3177) and, *SppT* (ΔQAICc=0.6736) accounted for 79% of the total support.

Interpolation of the Beta estimates (Table 3.28) of the *SuT* model (w_i=0.31) indicate an adult female survival of 0.4508 during this period for an average summer temperature (13.3°C). An increase of 10% in the average summer temperature would correspond to 38% increase in its survival. The effect of summer temperature was strongly sex (female) dependent.

Table 3.28: Beta Estimates for survival parameters for the top ranked model {S(Sex*SuT+age) p(g*t) r(age*t) F(g*t)}; Southland Mallard, 1987 - 1991.

			95% Confidence Interval	
Parameter	Beta	Standard Error	Lower	Upper
Survival Intercept	-0.2045	3.1948	-6.4664	6.0573
Sex	-7.0389	2.4771	-11.8941	-2.1838
Age	-0.3131	0.1531	-0.6132	-0.0130
SuT	0.0221	0.2332	-0.4351	0.4792
Sex*SuT	0.5069	0.1838	0.1466	0.8672

Table 3.29: Model Results Southland Band Region 26 &27

Band Set	Species	Band Period	Weather Station	Model	Q/ AICc	ΔQ/ AICc	Q/ AICc Weights	Model Likelihood	Number Parameters	Q/ Deviance	C-hat	Global Model Goodness of fit (P)
17	Mallard	1969-1974	Invercargill Aero (I68433)	{S(sex*WR+Age) p(g*t) r(age*t) F(g*t)}	15995.17	0	0.58679	1	105	593.2785	1.284	P<0.01
				{S(sex*SpcT+Age) p(g*t) r(age*t) F(g*t)}	15996.75	1.5744	0.26706	0.4551	105	594.8522		
				{S(sex*SuT+age) p(g*t) r(age*t) F(g*t)}	15998.05	2.8831	0.13881	0.2366	105	596.1614		
				{S(sex*SppR+Age) p(g*t) r(age*t) F(g*t)}	16004.4	9.2318	0.00581	0.0099	105	602.5101		
18		1987-1991	Invercargill Aero (I68433)	{S(sex*SuT+age) p(g*t) r(age*t) F(g*t)}	7245.616	0	0.30771	1	45	157.9064	1.138	0.03<P<.04
				{S(sex*WT+age) p(g*t) r(age*t) F(g*t)}	7245.933	0.3177	0.26251	0.8531	45	158.2241		
				{S(sex*SppT+age) p(g*t) r(age*t) F(g*t)}	7246.289	0.6736	0.21972	0.7141	45	158.58		
				{S(sex+age)*t p(g*t) r(age*t) F(g*t)}	7249.005	3.3898	0.0565	0.1836	65	120.3464		
				{S(sex*SpcT+age) p(g*t) r(age*t) F(g*t)}	7249.663	4.0478	0.04066	0.1321	45	161.9543		
				{S(sex*SuR+age) p(g*t) r(age*t) F(g*t)}	7250.488	4.872	0.02693	0.0875	45	162.7784		
				{S(sex*WR+age) p(g*t) r(age*t) F(g*t)}	7250.735	5.1196	0.02379	0.0773	45	163.026		
				{S(sex*SpcR+age) p(g*t) r(age*t) F(g*t)}	7251.153	5.5371	0.01931	0.0628	45	163.4435		
				{S(g) p(g*t) r(age*t) F(g*t)}	7251.428	5.8123	0.01683	0.0547	45	163.7187		
				{S(sex*SppR+age) p(g*t) r(age*t) F(g*t)}	7251.87	6.2549	0.01349	0.0438	45	164.1612		
				{S(sex) p(g*t) r(age*t) F(g*t)}	7252.013	6.397	0.01256	0.0408	42	170.4159		

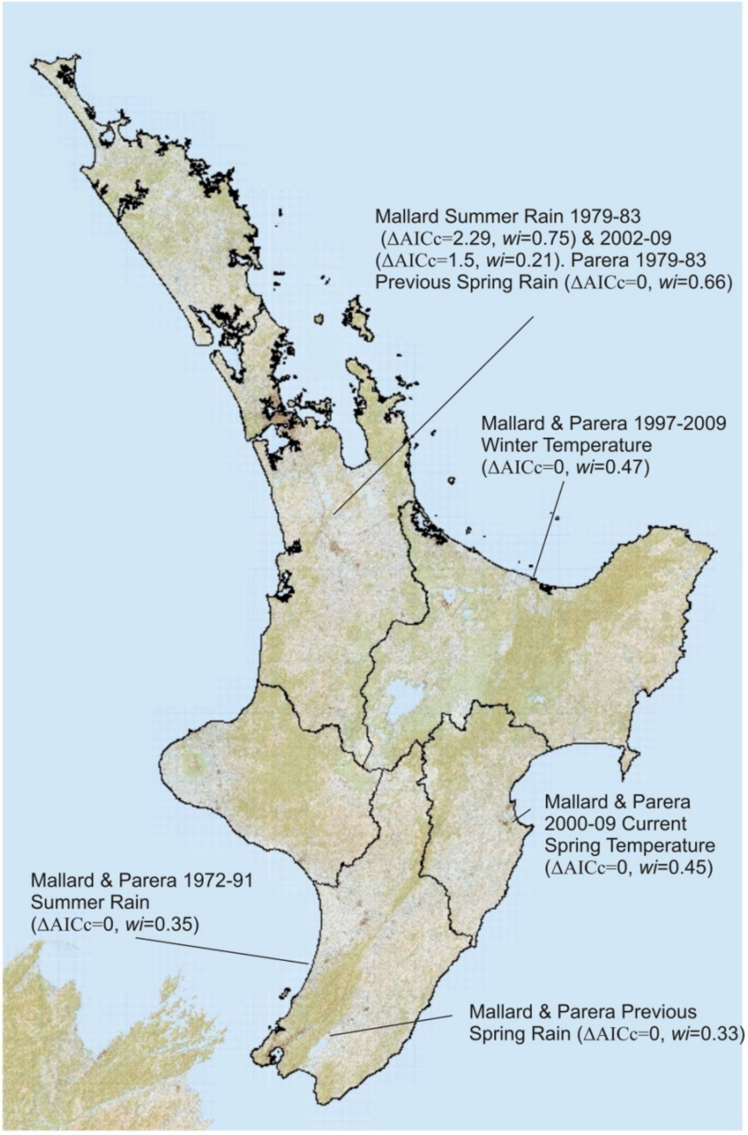

Figure 3.4. Selected Results from the top ranked climate models. The map shows North Island Fish and Game Regions.

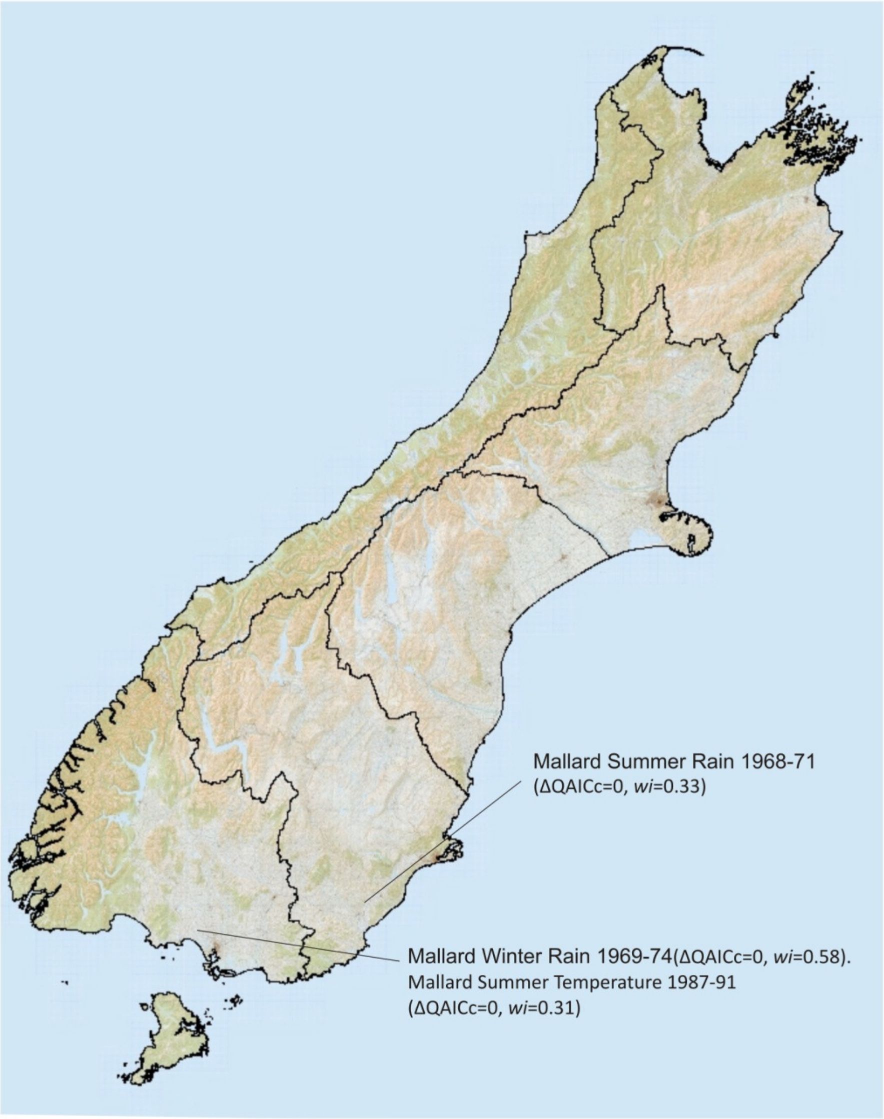

Figure 3.5. Selected Results from the top ranked climate models. The map shows South Island Fish and Game Regions.

Discussion

The objective was to determine if climatic variables explained temporal and spatial variation in survival rates. The premise was that it is more prudent to manage mallard populations subject to similar stochastic climatic influences and reduce environmental uncertainty.

New Zealand's longitudinal variance, topographical and, climatic diversity, provide a wide assortment of ecosystems. Subject to support for the climate models, this ecosystem diversity along with the relatively sedentary nature[1] of the mallard provide an opportunity to establish management units based on similar climate zones.

Auckland Band Region 5

It was expected that support for the climate models in the Auckland band region would be equivocal due to the prevailing temperate climate. Mallard and parera banded 1968-1974 showed no support for the climate models but subsequent band periods (1979-1983) indicated summer rain (*SuR)* (ΔQAICc=2.292, w_i=0.24) explained some of the stochastic variability in survival for the mallard and spring rain in the preceding year (*SppR*) explained some of the variation in parera survival (ΔQAICc=0, w_i=0.67). The 2002-2009 band period showed some support (ΔQAICc=1.579, w_i=0.21) for the *Summer Rain* (*SuR*) models for the combined parera and mallard data.

An increase in summer rain for the mallard and, spring rain prior to the season (*SppR*) for the parera, reduced survival in adult females. The key biological events occurring at these times are remex (flight feathers) moult and breeding season respectively (Marchant and Higgins, 1991).

Mallard lose body condition during remex moult probably due to reduced foraging opportunities (Panek and Majewski, 1990). Krapu et al. (2000) report mallard brood loss was 5.2 times more likely during rainy conditions. They attribute this to the possibility of increased risk of exposure. This in combination with poor body condition may reduce survival during high rainfall summers or reduce body condition going into the following year?

High rainfall events in the spring prior to the year of banding may promote re-nesting, as initial nests may be flooded (Caithness and Pengelly, 1973). This supposition, however, is more likely to hold for mallard than parera who don't nest as close to the water's edge (Balham, 1952, Heather and Robertson, 1996) and are therefore probably less susceptible to flooding. Re-nesting incurs a physical cost on the female.

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[1] Of approximately 3,000 recoveries (1997-2009) in Band Regions 6, 7 and, 14, eighty seven percent were recovered within 50km of the band site.

The cost of egg production is high relying on endogenous lipid reserves (Young, 1993) and exogenous protein and mineral sources in temperate regions (Baldassarre and Bolen, 2006). Hepp et al. (1990) report Wood Ducks (*Aix sponsa*) that were heavier at the end of one breeding season had higher survival rates in the following year. Birds in poor condition going into the band year are also expected to be more susceptible to harvest (Hepp et al., 1986, Dufour et al., 1993, Heitmeyer et al., 1993).

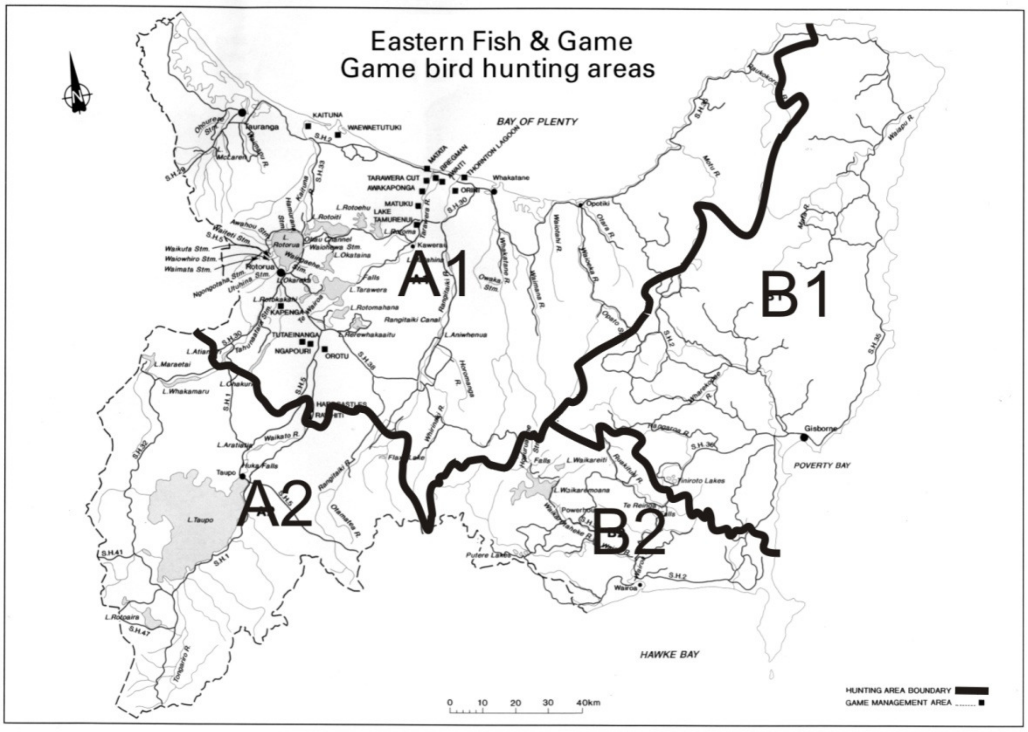
Another possible explanation may be the predisposition of botulism outbreaks in wet summers (Kadlec, 2002). In some years the Waikato (and Bay of Plenty) are susceptible to large avian botulism (*Clostridium botulinum*) die-offs over summer months.

Rotorua Band Region

Of the Rotorua data that showed support (1979-2009) for the climate models the result for the top ranked *Winter Temperature* model (*WT*; w_i=0.48) was on first observation counter intuitive. An increase in average winter temperature corresponded with a decrease in adult female survival. This however is consistent with the effort models in the preceding chapter which suggest hunters may hunt more in warm weather in the Eastern Fish and Game Region.

When the geographic range of Data Set 8 (Management Units A1, A2 and, B2 of the Eastern Region, Figure 3.6) was restricted, the climate models received 100% of the total support (DS 9 Table 3.14, page 121; Management Unit A1). This has interesting implications for the minimum size or, location of possible monitoring units and could be further interpreted as providing support for spatial disparity in survival rates as a function of climate.

Figure 3.6: Eastern Fish and Game Region showing game bird management units A1, A2, B1 & B2



Heterogeneity in mallard survival and recovery rates is usual and well accepted (Pollock and Raveling, 1982, Burnham and Rexstad, 1993), and consistent with previous studies in New Zealand (Balham and Miers, 1959, Nichols et al., 1990, Caithness et al., 1991). So complete support for the *dot* model (all parameters don't change over time) on the East Coast (DS 10, Table 3.14, Page 121) was unexpected. Particularly in light of the prevalence for drought in this region (Leathwick et al., 2003). On the East Coast one might anticipate temporal heterogeneity in all parameters (S, P, r and F) and across age and sex cohorts. It is expected that drought years may favour adults over juveniles (being more adept at foraging for food), and stress females at critical times (e.g. at the end of the breeding season; Young 1993) and through the remex molt (Panek and Majewski, 1990). Hepp et al. (1986) suggests birds in poor condition may be more mobile and I would therefore anticipate study site fidelity to decrease during dry years. Intuition also suggests mallard and parera in poor condition would be more susceptible to recapture however there is little evidence of such (Reinecke and Shaiffer, 1988).

Burnham (2001) suggests that when there is 10 or more years of data the fixed effects models where survival is fully constrained or allowed to vary annually in conjunction with a covariate, may be too restrictive. He suggests random effects models provide an intermediary. I ran a random effects model on the East Coast data set but this received very little support (not shown). The recovery and recapture array (Appendix C) appear reasonable with no indication that data is sparse. I therefore reran this analysis but included the 2010 recapture and recovery data and got a completely different result, the *SppT* model received 93% support.

Hawke's Bay

The *SppT* model also received the greatest support (w_i=0.79) for the Hawke's Bay data set (2000-2009) with lesser support for the *SpcT* mode (ΔQAICc=4.0193, w_i=0.11). Beta estimates suggest an increased adult female survival with warmer spring temperatures (preceding and current). Both scenarios have reasonable biological explanations; improved primary production and reduced metabolic requirements but the possibility of the onset of a drought following a hot spring could compromise these suppositions.

Unusually the Betas (Table 3.15) for the top ranked model (*SppT*) indicate that females have higher odds of survival than the males contrary to findings of birds banded elsewhere in New Zealand (Nichols et al. 1990; Caithness et al. 1991). What's more the survival rates are higher than I would expect.

Wellington Region 5

Temporal fluctuations in survival can be explained through changes in summer rain and temperature for the Manawatu data. The majority of the mallard and parera banded in the Manawatu were at band sites either contiguous with, or close to, the coastal dune lakes. Water levels in these dune lakes can fluctuate appreciably over summer (Caithness and Pengelly, 1973, Potts, 1977). During droughts in the early 1970s, Gibbs (1973), noted the complete disappearance of favoured mallard food, *Potamogeton pectinatus* and *P. crispus* (Potts, 1977). The impact of the pond drying out may in part be offset by re-colonisation of exposed literal zones which may improve invertebrate production associated with these areas in autumn and winter (Potts, 1977).

Mallard and parera banded in the Wairarapa showed support for all the precipitation models. Continuous temperature data were not available for this period.

The Wairarapa has a relatively low annual rainfall and is subject to summer droughts. The beta estimates are biologically sensible suggesting survival improves with spring rainfall prior to the season (refer discussion above) or conversely decrease with declining rainfall.

Otago Mallards

Temporal changes to summer rain explained fluctuations in survival rates for mallard banded 1968-1971 in the Otago Band Region (w_i=0.33). The *SpcT* model also received a measure of support (ΔQAICc=0.7356, w_i=0.23). The mallard were banded about 71km SW of Dunedin at Lake Tuakitoto. Dunedin is below the national average mean air temperature (http://www.niwascience.co.nz/edu/resources/climate/meanairtemp/) and the monthly water balance[1] is very low to low, (312mm; Leathwick et. al. 2003).

The Betas for the Summer Rain model (*SuR;*) are consistence with the expectation that survival will improve in years when summer rain is proficient. The work of Barker et al. (2005) concur to some extent with the precipitation models. They examined the influence of the Southern Oscillation Index (SOI) on Shoveler duck (*A. rhynchotis*) survival in the Otago Region and concluded large negative values of SOI were associated with higher survival. They suggest this may be a function of rainfall. The biological rational behind the effect of summer rainfall on mallard survival is explained above.

It was expected that low ambient temperature may exacerbate low temperature events but this does not appear to be the case. Beta estimates (not reported) for the spring current temperature model (*SpcT*) suggest a 10% increase in average spring temperature will depress adult female survival by 39%. I can only think that high spring temperatures are associated with subsequent summer droughts and reduced summer rainfall?

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[1] Water balance is difference between total evaporation and precipitation.

Southland

Invercargill has a relatively low mean annual temperatures (9.9 C^{o}), and annual rainfall doesn't appear to be limiting (1112mm) (Leathwick et. al. 2003; categorises the monthly water balance as low with pockets of moderate levels).

Southland mallards were banded at two different periods with differing outcomes. The *Summer Temperature* model (Data Set 18; 1987-1991) fits with biological expectations for this geographic location; adult female survival increases with an increase in average summer temperature. Average summer temperature in Invercargill 1987-1991 was 13.3°C, about the lower critical temperature (LCT) for an adult female mallard (the point where core body temperature cannot be maintained without increasing BMR; see previous discussion).

Mallard banded 1969-1974 showed some support for the temperature models (*SpcT;* ΔQAICc=1.574, w_i=0.26 & *SuT;* ΔQAICc=2.883, w_i=0.14), however the top model was the *Winter Rain* model (*WR;* w_i=0.59). The Betas for the *WR* model are difficult to explain; an increase in winter rain results in an increase in adult female survival. One theory consistent with my postulates regarding Whakatane winter temperature model and the Hawke's Bay analysis is that hunters hunt less in inclement weather.

Support for the climate models

18 data sets of mallard and parera banded 1968-2009 from Auckland to Southland were examined. Eight of these data sets showed no support for the climate models (AICc>5). Two data sets showed some support (AICc<5; but were not the top ranked model) and for eight data sets climate covariates explained temporal variation in survival better than any of the other candidate models.

Of the eight that showed no support one (the East Coast Data Set), showed support when 2010 recovery and recapture data were included. Of the 7 remaining no-support data sets, one was probably a function of sparse data (Canterbury) and 2 (Rotorua 1997-2009 & East Coast-Hawke's Bay 1998-2009) showed support once the geographic range was constrained. Some of the banding regions cover several obvious climate zones. For example birds banded in Band Region 7 (Figure 3.2) (DS 6 & 7,Table 3.14, page 121). The banding sites for these data set (DS 8[1] Table 3.14, page 121) were located just north of Wairoa, Taupo, and the Bay of Plenty (at least 1.2M ha), which probably comprise two and probably three different climate zones (extrapolated from Leathwick et al. 2003). When the geographic spread (location of birds banded) was restricted to just the Bay of Plenty (DS 9) the climatic models receive good support. Constraining the geographic range, however, sometimes had mixed results; for example when the East Coast – Hawke's Bay data set (DS 11, Table 3.16 page 118) (that received no support for the climate models) was separated into Hawke's Bay (DS 12, Table 3.16, page 123) and East Coast (DS 10, Table 3.14, page 121), model ranking changed. Hawke's Bay climate models received good support and unexpectedly, the East Coast climate model set received none (the top ranked model was the fully constrained *dot* model, but see comment above).

Constraining the geographic range of the band data were not always possible. In some instances there were not enough birds banded so the encounter histories were pooled across a wide area. Secondly, uncertainty of the exact location of the band site made it difficult to differentiate the data (e.g. Canterbury mallard DS 15, Table 3.23, page 128). Burnham and Anderson (2002) point out that relatively little information is contained in a small sample unless the effect is large; I expect that this is pertinent here and the DS encompassed a large geographical region.

This corroborated, to some extent, the premise of spatial disparity in survival as a function of climate. Support however, was not consistent between birds banded at different times in the same region. For example Southland mallard showed support for different climate models (Table 3.29) at different periods. This is not really surprising when temporal climatic stochasticity is considered.

The time, sex and, age dependent, models were the top ranked models in six of the data sets. This fits with the presumption that annual variations in effort and harvest (see previous chapter and (Caithness et al., 1991, Nichols et al., 1990) could explain these outcomes.

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[1] The Taupo-Reporoa birds were removed.

Management Implications

Should Fish and Game choose to review the way they currently manage mallard and parera in New Zealand, it appears that areas of similar precipitation and temperature would provide a legitimate foundation to create management zones. These climate covariates explain changes in survival in many instances and are likely to explain a proportion of productivity variability.

The intensity of monitoring effort will dictate to a large degree the size of the management zone and intensity of monitoring will be governed by a number of factors including, resourcing, threats, opportunities, information requirements and, management commitment. The relevance of these different factors will vary between Fish and Game Regions.

Land Environments New Zealand (LENZ) developed by Leathwick et al. (2003) provides an ideal framework to create management zones using areas of similar precipitation and temperature. They portray areas of similar environmental factors as layers where comparable environments are measured as an environmental distance.[1] The nice thing about this approach is that the scale or acceptable environmental distance can be increased or decreased. Therefore as there is a requirement to increase management intensity the environmental distance can be reduced, effectively decreasing the geographic range but keeping it within a common precipitation and or temperature band. For example I provide zones of common temperature and precipitation in Appendix D and then combine them to provide a common layer incorporating both variables. Finally I highlight zones of commonality which could be utilised as national management units based on a visual interpretation of the combined climate data.

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[1] Environmental distance is the average difference between two environmental variables

$$D = \frac{1}{n} * \sum_{i=1}^{n} \frac{[x_{ij} - x_{ik}]}{range(x_i)}$$

Where: D is the environmental distance between points *j* and *k* described by a set of variable $x_{1 \ldots n}$ (Leathwick et al., 2003)

Spatial variation in survival rates as a function of latitude was difficult to determine as similar regions showed different support for climate covariates at different times. A better way to assess special variation in survival rates would to be to use latitude as a covariate. At the time of preparation for this thesis the Department of Conservation did not have the recapture-recovery data in a form that would have allowed this.

Climate Covariate Summary

Mallard and parera band data were examined across seven band regions over a latitudinal range $36-46°S$. The hypothesis that climate covariates explained temporal and spatial variation in survival rates was tested on 18 combined recapture and recovery data sets. The 18 data sets were pared down to 17 as 1 was probably compromised by a combination of a lack of marked birds and large geographic area. Eleven of the 17 data sets showed some support for climate covariate models. Constraining the geographic range generally improved support for the climate models. It is likely that survival for some of the no-support data sets was a function of harvest.

In all but one instance, a sensible biological interpretation of the Betas in the linear models could be provided. Some variation in survival is attributable to hunter behaviour in relation to climate, disease (Avian Botulism outbreaks during wet summers), and climate exacerbating physiological stress points in the annual life cycle of mallard and parera (e. g. moult and nesting). The biological interpretation is not necessarily cause and effect but rather an explanation. Testing the support of the explanations is outside of the scope of this study.

I believe there is enough evidence to support the concept that climate influences survival of mallard and parera in New Zealand, and further that climate zones would provide a biologically sound basis to create management zones

Chapter 4

Conclusion

In many instances where wildlife populations are exploited there is a large degree of uncertainty around how the population responds to, harvest management, population states and, the influence of environmental factors. Mallard and parera management in New Zealand is no exception.

Fish and Game has as an objective sustainable harvest of game birds (Sports Fish and Game Management Plan, unpublished statutory document), but no measures of population parameters that can be used to determine sustainability. Further, it is likely that the aspirations of most Fish and Game Councils go beyond just sustainable harvest. That is they would like to maximise annual or cumulative harvest (combined harvest over many years).

This thesis set out to develop a better understanding of the consequences of regulations on mallard harvest in New Zealand, and provide an example of how Fish and Game Regions could use this information to maximise harvest. In doing so it highlights what information is required to implement a maximum harvest regime, uncertainty around the information, and what parameters have the greatest influence in the decision making. Furthermore, should Fish and Game choose to implement a nationally consistent approach to mallard management and embark on a path to maximise harvest. It is proposed that climate zones are a reasonable choice for creating management units which will assist in reducing the effects of environmental uncertainty.

The goal of maximising harvest was expressed as an objective function, whereby annual harvest is maximised subject to the constraints that hours hunted (effort) in any one year could not be less than that reported in the Eastern Fish & Game Region hunter survey (R. J. Barker University of Otago, unpublished data.) over the study period (6500 hours) and that the adult female population in year t_{+1} is greater than or equal to the adult female population in year t.

To solve this ecologically complex problem a simple heuristic model of mallard population change between year t and t_{+1} was formulated. The model inputs over the period t to t_{+1} were hunter effort and births. Outputs were pre-harvest mortality, harvest, and post harvest mortality.

For the input and outputs, a serious of explanatory models were tested on data collected from 1997 – 2009. Data included a total of 1024 recapture, and 3100 recoveries, from 22,500 mallard and parera banded annually over the study period. Annual reporting rate was estimated from random annual surveys of 120 hunters for 3 consecutive years (360 surveys per annual reporting rate estimate). Harvest data (hours hunting waterfowl, mallard and parera harvest) were derived from random fortnightly surveys (120 interviews per fortnight) over the game bird hunting season (R. J. Barker University of Otago, unpublished data). Annual survival and harvest rates were calculated from band recapture and recovery data (analysed in Program MARK; White and Burnham 1999). Pre-harvest survival was estimated from a study of 46 mallard duck fitted with transmitters in the Eastern Fish and Game Region and analysis was Known Fate procedure (Cooch and White, 2009) based on Kaplan-Meier methodology (Kaplan and Meier, 1958) and stagger entry design (Pollock et al., 1989a).

Hunter's behavioural responses to different regulation strategies are little understood and have received little research attention. Hunter effort (hours hunting waterfowl) was examined as a function of harvest regulation, climate, annual participation (number of hunters), trends in participation and success (the presumption that hunters would hunt for more hours in years when game was plentiful). It was established that in both Eastern and the Hawke's Bay Fish and Game Regions, hunters exerted proportionally more effort in the shorter seasons. Furthermore, it appears that bag limits do not limit effort and have no impact on harvest rates within the range utilised by either region over the study period. Existing harvest regulations are therefore a rather blunt instrument to constrain effort and harvest rates. Effort (E) was best explained as a function of the year (a downward trend in total effort over time Figure

2.13 and Figure 2.22). Nevertheless the second ranked effort model (*E=f(Year, Number of Hunters, Season Length= f(Yr, Hu, SL))* ($\Delta AIC_c = 5.305$) was used to examine the effect of regulations on the objective function for the Eastern Region, as it provided greater utility in determining maximum harvest.

Changes in annual survival 1997-2009 in the Eastern Region were best explained by hunter effort (Table 2.7, page 54) while season length and hunter effort received similar support in the Hawke's Bay 2000 – 2009 (Table 2.9, page 59).

The correlation between effort and harvest rates was poor in the Eastern Region (adults approached significance; P=0.053), or not evident in the Hawke's Bay. Because the correlation between hunter effort and harvest was good and that effort explained changes in annual survival it was assumed that reporting rate estimates were compromising harvest rate estimates.

The objective function was solved using a deterministic model (Equation [2.49], page 86). The deterministic model incorporates a series of linear models (female only) whereby harvest rates are a function of the age (adult or juvenile) and hunter effort. Pre-harvest survival was fixed and dependent on age (estimated from the telemetry study), while post-harvest survival was a linear function of post-harvest female population size, which in turn was estimated as the product of harvest rates (Equation [2.27], page 35) and pre-harvest survival estimates and initial population size. Post harvest-survival was estimated from annual survival divided by the product of survival over the hunting season and pre-harvest survival. Productivity was calculated as a function of end of year adult female population size. This productivity linear model was determined from the trap sample which was considered biased. A simple model proposed by Cowardin and Johnson (1979), (Equations [2.30], page 36, and [2.46] page 78), was used to assess average productivity estimates against average survival for the period. It was assumed no change in population over the study period. The average trap productivity estimate was 2.02 times higher than that required to ensure no change in population size for the study period. Therefore a bias adjustment of $\frac{1}{2.02}$ was made to the model. Harvest was estimated as the product of pre-harvest population size ($N_{age,t} \times \partial_{age}$) and harvest rate for both adult and juvenile female.

Solving the deterministic model spawned a level of effort that maximises harvest. To establish an appropriate season length corresponding to this level of effort I used the second ranked Effort model (*E=f(Yr, Hu, SL)*) which was run in Program WinBugs (Spiegelhalter et al., 2003) 10,000 times (1000 burn in). This generated a 95% credible interval around the expected effort given the season length (Figure 2.65, page 88).

The nice aspect of this approach is that managers are able to adjust the season length according to the level of risk they are prepared to accept to deliver the desired level of effort. In years that the population state is considered stressed (population size and or productivity is low) they may decide that a conservative approach is appropriate and select a season length that is highly likely to constrain effort to the desired level.

A partially stochastic model (Equation [2.51, page 89) compiled in Program R (R Development Core Team, 2005) was used to assess the performance of the underlying linear models (in the deterministic model) and to predict harvest. This stochastic model incorporated the same linear models used in Equation 2.53 (but included males) and introduced variance of the model parameters. The harvest estimate generated by the stochastic model could be assessed against harvest established by the hunter survey for the Eastern Region. The stochastic model was run 1000 times and used to predict 2010 harvest (outside of the study period). The model performed admirably, simulated harvest was 42,045 (SE=1,992), while the hunter survey derived estimate for the same year was 41,549 (SE=3552).

A similar model (to the stochastic model) was used to assess Eastern Fish and Games current process of establishing season regulations (Threshold Management, page 81). The results suggest sustainable harvest is dependent on productivity rather than the current regulation set (season length). Once juvenile female to adult female ratios went below 0.8, harvest over the long term (10 years) was not sustainable irrespective of the constraint imposed by harvest regulations (used by Eastern Fish and Game).

This simulation (Figure 2.63, page 84) highlighted that small changes in productivity (0.8 to 0.95) made very large differences to sustainable harvest strategies. The lower productivity levels were not sustainable while the upper (0.95) produced a surplus (mallards) under a relaxed regulation strategy (71 day season for the entire 10 year simulation).

Barker et al. (1991) examined compensatory verses additive harvest of parera in New Zealand and rejected the completely compensatory model but were unable to reject the completely additive model. Uncertainty around the impact of harvest regulations on mallard survival in the US has resulted in an adaptive harvest management approach wherein four competing models were formulated, two with additive and two with compensatory survival, with either strongly or weakly density dependent recruitment (Johnson et al., 1997, Nichols et al., 2007). In this study the relationship between population size and survival was examined by randomly allocating banded birds into two groups. One of these groups was used to calculate survival rates and the other an estimate of population size (using the Petersen-Lincoln estimate). The correlation between juvenile survival and population size was negative and highly significant (P<0.001, page 76) but not so significant for the adults (P=0.054 and P=0.094, adult male and female respectively). This inferior correlation can be explained following an examination of the scatterplot graphs (Figure 2.58 and Figure 2.59, page 77) which suggest that adult survival may not be linear. At lower population size survival appears constant, and decreases once the population is greater than 250,000.

There was also a negative marginally significant correlation (P=0.058) between adult population size and the ratio of juvenile females to adult females in the trap sample (Figure 2.60, page 80) suggesting density-dependent recruitment.

A number of models were formulated to predict annual survival, harvest rates, and productivity levels amongst others. The harvest rate models performed poorly given the data. Harvest rate estimates are confounded with reporting rate estimates which may explain why the data did not fit the models very well. There are alternative methods to assess reporting rate including reward bands and telemetry studies. It would be very worthwhile to accredit phone survey reporting rate estimates via one of these methods.

It is likely that harvest rate will be a function of many other variables other than those investigated. Some obvious omissions were ephemeral water prior to the opening of the shooting season (a lot of ephemeral water provides alternative habitat much of which is not hunted). This could be achieved by incorporating pre-harvest rainfall into the model. Hunter density which will be a function of licence sales and to some extent topography (hill country will likely have local effects on harvest rates as hunter density will probably be lower; See Footnote (1) page 101).

Productivity estimates from the trap sample appear biased (page 78). Alternative methods of assessing productivity should be evaluated; particularly in light of the large effect productivity can have on population change. These could be used to assess trap bias.

An independent measure of population size would be useful. Using band recapture and recovery data to estimate both population size, survival rates, and harvest rates is not good practice due to estimation errors being correlated.

The results in Chapter 2 provide a foundation to build an adaptive management approach for setting season regulations to maximise sustainable harvest. Focus has been drawn to important factors such as productivity and harvest rate estimates wherein future monitoring should concentrate. Councils may wish to test some of the model assumptions by altering season regulations for given population sizes. This combination of model theory, population monitoring, and harvest outcomes, given different harvest regulations are at the crux of adaptive management. Councils can be more forthright (active) or passive in their approach to learning how population states, and harvest regulations, influence maximum harvest and the consequences of their actions. Two examples are setting relaxed season conditions when the population is assessed as low, or alternatively, restrictive regulations when game is considered plentiful.

Should Fish and Game Councils wish to embark on a consistent national approach to harvesting mallard and parera, there is an opportunity to move away from administrative boundaries and create management zones that reflect populations subject to similar survival rates and probably recruitment influences. Chapter 3 explores the idea that environmental stochasticity has an important influence on survival rates. Clearly, it would be better to derive management units following an evaluation of mallard ecosystems. However, as Leathwick et.al., (2003) point out, this is difficult as ecosystems are complex, and lack natural boundaries. The easier alternative is to confine management units to areas of similar climate.

I extracted data (69,000 band records, 13 different studies) from historical recapture-recovery band studies throughout New Zealand (sourced from Department of Conservation Banding Office). In addition I used band recapture - recovery data from Eastern and Hawk's Bay Fish and Game Regions (reported in Chapter 2). A series of 8 linear climate models (Table 3.1, page 112) were analysed and assessed against 7 non-climate models.

The Department of Conservation record release sites as one of 27 different band districts (old Acclimatisation Society boundaries). These districts formed the basis for reporting the results and it was hoped may explain spatial variation in survival rates. The evidence for this was however, not clear. Band studies in the same band district but run at different times produced inconsistent results. Future studies may want to include latitude as a covariate and include all New Zealand mallard and parera band recapture-recovery data in one analysis.

The premise was that climate events could affect body condition (lipid reserves) which in turn may affect survival rates once levels get below a certain threshold (Clinton et al., 1994). Birds in better body condition have higher survival rates (Bergan and Smith, 1993, Pollock et al., 1989b). Climate variables were reported against periods when mallards are physiologically stressed such as winter (Smith and Prince, 1973, Heitmeyer, 1988, Bergan and Smith, 1993), moult (Pehrsson, 1987, Panek and Majewski, 1990), and reproduction (Gloutney and Clark, 1991, Alisauskas and Ankney, 1992).

Of the 18 data sets (13 from historical data and 5 from Eastern and Hawke's Bay Regions) one was excluded due to sparse data and large geographic range of band release sites (page 127). Eleven of the seventeen remaining data sets showed some support for the climate covariate models.

Given this overall support, I suggest Land Environments New Zealand (LENZ) (Leathwick et al., 2003) as an appropriate method of devising management units for similar climates. Precipitation and temperature layers (Appendix D, page 161) are combined (Figure 4.12) and similar environments highlighted. These similar environments provide a starting point to discuss management units for mallard and parera in New Zealand.

Ecologically consistent management units and a better understanding of partial management control and structural uncertainty put Fish and Game in a better position to assess whether mallard and parera harvest is sustainable. Should Fish and Game wish to expand their goal of sustainable harvest to one of maximum annual or maximum cumulative harvest of mallard, the findings in this thesis will go some way towards providing a platform to launch an adaptive management approach. This will assist in learning about complex ecological problems and provide an opportunity for rigorous thinking about the harvest system.

Appendix A

Trapping, Aging and Sexing Mallard and Grey Duck

Trapping

Mallard and Parera were trapped at secluded wetlands using walk-in funnel cage traps (Figure 4.2 & Figure 4.3).

I constructed 20 traps consisting of two rectangular compartments (1.840 x 1.840 and 1.840 x 0.92m) using welded 6mm reinforcing rod. A support strut was placed centrally on the 1.840x1.840m panels. Frames were galvanized and covered with 25mm chicken mesh, held in place by lacing wire. The entrance consists of a tapered mesh covered half funnel and the two compartments are linked by a funnel of the same construction. Cable ties join the panels and two compartments.

Traps were assembled over 10-15 days and are orientated so the second compartment appears as an escape route back to the water. Secluded trap sites are selected to avoid tampering and disturbance by the general public. Predator traps were set for cats and mustelids at the beginning of feeding out to minimise disturbance and deaths of grain enticed waterfowl.

Figure 4.1 Welded, netting covered, traps comfortably hold about 70 mallard duck.

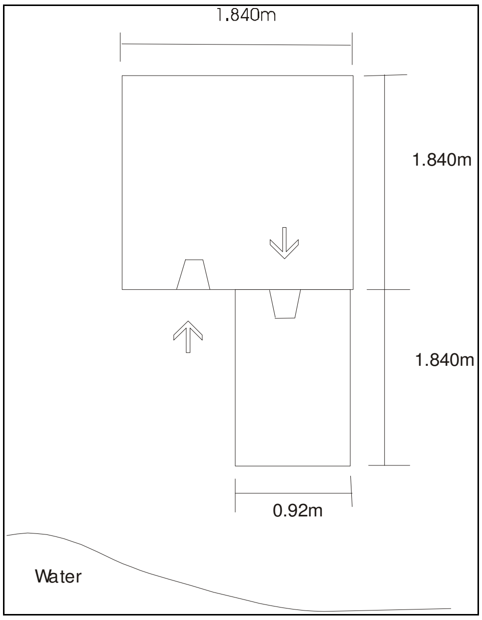

Figure 4.2 Plan view of trap.

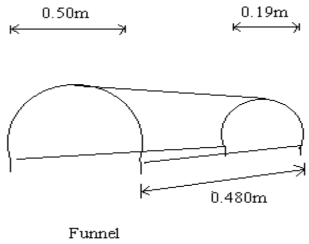

Figure 4.3. Funnel Plan

The trap sites are pre-fed for a minimum of 10 nights with 10 kg[1] of maize screenings per trap per night (some sites required more than 10 nights as the birds are often slow to initiate feeding). The number of traps set at each site is based on the amount of grain that is consumed overnight during the pre-feeding period. Additional traps can be added prior to close up but I have found that the birds tend to be wary of anything new and success with late added traps is low.

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[1] Eight ducks were killed by a dog in a closed up trap at Lake Aniwhenua in 1999. The weight of the maize screenings in their crops was measured. The range was 36-128gms with an average of 96gms. Based on this 10kg will feed about 100 ducks. 70 Mallard & Grey/Parera comfortably fit in a trap with a maximum of 100.

Generally about 1500kg of maize screenings was fed out in the Hawke's Bay and 3400 kg in the Eastern Region. Maize screenings proved cheaper than other feeds and as effective if not more so than other grains[1]. Pre-feeding commenced each year about the first week in January and banding finished about 31 January.

Aging and sexing

Cloacal features can be used to age and sex waterfowl (Taber, 1971, Baldassarre and Bolen, 2006). The bursa of Fabricius is a blind duct at the posterior of the cloaca which regresses with age (Ward and Middleton 1971). This allows females to be differentiated into adults and juveniles. The penis is small and unsheathed in juvenile males and large and sheathed in adult males.

An examination of 64 mallards in the Manawatu produced mean bursa depths of 0.71 (SD; 1.19) and 26.9 mm (SD; 6.1) for adult and juvenile females respectively (P Taylor pers. comm.). The range for adult females was 0 to 5mm and for juvenile females 10 to 37mm. These figures were used in this study to differentiate between adults and juvenile females. Adult and juvenile males were differentiated by an examination of penis size. Tail feather characteristics[2] (Figure 4.6) were also used where age differentiation was difficult (Taber, 1971, Baldassarre and Bolen, 2006).

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[1] In 1999 to determine feed preference a selection of feeds (maize, barley and wheat) were feed out in clumps and the preference was recorded based on the amount eaten overnight.

[2] Juveniles have a downy plume at the end of their tail feather that can break off leaving a notched tail feather.

Figure 4.4. Mallard and grey duck are aged and sexed and then an individually numbered stainless leg band is attached. Mallard and grey duck are differentiated based on their predominant plumage characteristics (many of the grey were obvious hybrids).

On two occasions I made a cloacal examination of 12 Mallard hatched on 9 September 2002. The first occasion was on the 14 November 2002. Bursa depth ranged from 16 to 29 mm (average 24.4, SD 6.08 mm). On the second occasion, 23 March 2003, female cloaca depth ranged from 23 to 29mm (average 25.8, SD 2.6mm). Two of the November measurements were shallower than the measurements taken 18 weeks later this could be a result of two different occurrences. Firstly the same probe was not used to age the birds. Therefore the depth may have been a function of the diameter of the probe. The second possibility was that the bursa had not yet reached its maximum depth by 9 weeks. This last supposition is supported somewhat by Ward and Middleton (1971) who found that bursa did not reach its maximum weight until about 9 weeks.

Figure 4.5. Measuring the depth of the Bursa.

What was interesting at the March examination was the tail notches that had been present in all birds at the earlier examination had disappeared indicating that the tail feathers had been molted by this age (27 weeks). Also the males' penises were fully sheathed making any differentiation between adult and juvenile males impossible based on these two techniques.

Figure 4.6: Juvenile Mallard showing notched tail feathers

Banding

Each bird was aged, sexed, and a leg band placed on the right leg of the males and left of the females. The bands were obtained from the Banding Office, Department of Conservation Wellington along with an Authority to band under Section 38 of the Wildlife Regulations. The bands are individually numbered with a return address to "Dept. of Conserv. P O box 10-420, Wellington, NZ[1].

Figure 4.7: Steel leg bands with an individually identifiable number and return address are placed on the captured ducks leg.

Schedules of birds banded were entered into a Microsoft Access Database and a copy sent to the Banding Office. Initially the schedule was filed with the Banding Office as a hard copy but later the Banding Office requested an excel spreadsheet.

The release site of the banded bird is recorded in both degrees and minutes of Longitude and Latitude and, as a 14 digit grid reference based on the NZMS 260 Map Series.

Demonstrable Mallard/Parera hybrids were categorised on their dominant features as a Parera or Mallard duck.

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[1] At the beginning of the study the address on the bands contained the address of the Wildlife Service (Wildlife Wellington NZ). Because of the inadequacy of this address it probably resulted in a number of early band returns going missing in the mail.

Figure 4.8. Many helpers ensure a quick turn over of the birds reducing stress.

Recoveries and Recaptureres

Band returns are predominately associated with harvest during the hunting season. A hunter shoots a duck and reports the band to the address on the band or to their local Fish and Game Office. A few are found on dead ducks outside of the hunting season.

Details of recoveries were entered into a Microsoft Access Database and a copy made available to the Banding Office. A letter was sent to the person who recovered the band providing specific details on the banded bird (distance and direction from banding site, age, sex, where banded and information on the banding project).

Recoveries are encouraged through a passive reward scheme. A draw was made for 10 free licences from those hunters that returned a band in the current year. This was advertised in the Fish and Game Magazine which is sent annually to every whole season hunter.

Band number and date of recapture of banded birds from previous band periods encountered in the trap sample was recorded in a separate table of the MS Access

Data base. I used Pivot Tables in Microsoft Excel to generate a summary of the encounter histories as input files for Program MARK.

Appendix B

Productivity Estimates

Table 4.1. Eastern Region productivity estimates from the trap sample. J=juveniles; A=adults; F=females. Adj=adjusted (adult female bias correction; 2.02)

	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
J/A	1.23	1.03	0.80	1.31	1.68	4.47	1.37	4.01	3.68	2.64	2.38	3.96	2.34
J/AF	2.25	2.16	2.27	3.07	4.60	11.29	3.57	8.05	7.34	7.36	5.88	9.49	4.34
JF/AF	0.69	0.82	0.91	1.24	1.74	4.70	1.53	2.75	3.04	2.97	2.40	3.99	1.80
J/F(Adj)	1.11	1.07	1.12	1.52	2.27	5.59	1.76	3.98	3.63	3.64	2.91	4.70	2.15
JF/AF(Adj)	0.34	0.41	0.45	0.61	0.86	2.33	0.76	1.36	1.51	1.47	1.19	1.97	0.89

Appendix C

East Coast Recapture & Recovery Arrays

Table 4.2: Recovery array for mallard and grey duck (all cohorts) banded on the East Coast 1998-2009 where $N_{(i)}$ is the number released.

Year	$N_{(i)}$	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
1998	796	52	29	19	5	3	6	1	3		1	1	
1999	400		28	13	3	2	1		1		2		
2000	554			32	12	8	7	2		1	1		1
2001	429				26	6	4		3	1	1		
2002	374					17	10	2	1	2	1	3	
2003	316						15	3	7	1	1	4	1
2004	466							28	7	1	3	1	1
2005	395								13	4	5	2	
2006	352									10	10	6	3
2007	505										21	12	5
2008	305											14	1
2009	289												15

Table 4.3: Recapture array for mallard and grey duck (all cohorts) banded on the East Coast 1998-2009 where $N_{(i)}$ is the number released.

Year	$N_{(i)}$	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
1998	796	23	11	4	1	6		1	1			
1999	400		13	6	1	1		1	1		1	
2000	554			52	4	13			1			
2001	429				12	6	4	1	2		1	
2002	374					27	8	4	2	3		2
2003	316						9	5	6	1		1
2004	466							15	9	3	1	1
2005	395								23	5	2	2
2006	352									14	4	4
2007	505										19	3
2008	305											8

Appendix D

National Mallard monitoring units

I used ArcMap-ArcView to layer LENZ (Landcorp New Zealand) Annual Temperature (Figure 4.10), October Water Vapour Deficit, Water Balance Ratio and Annual Water Deficit files (Figure 4.9) to produce a combined climate map (Figure 4.11).

Figure 4.9: New Zealand Map showing layered climate data incorporating Water vapour deficit. (Data source LENZ Landcare Research NZ)

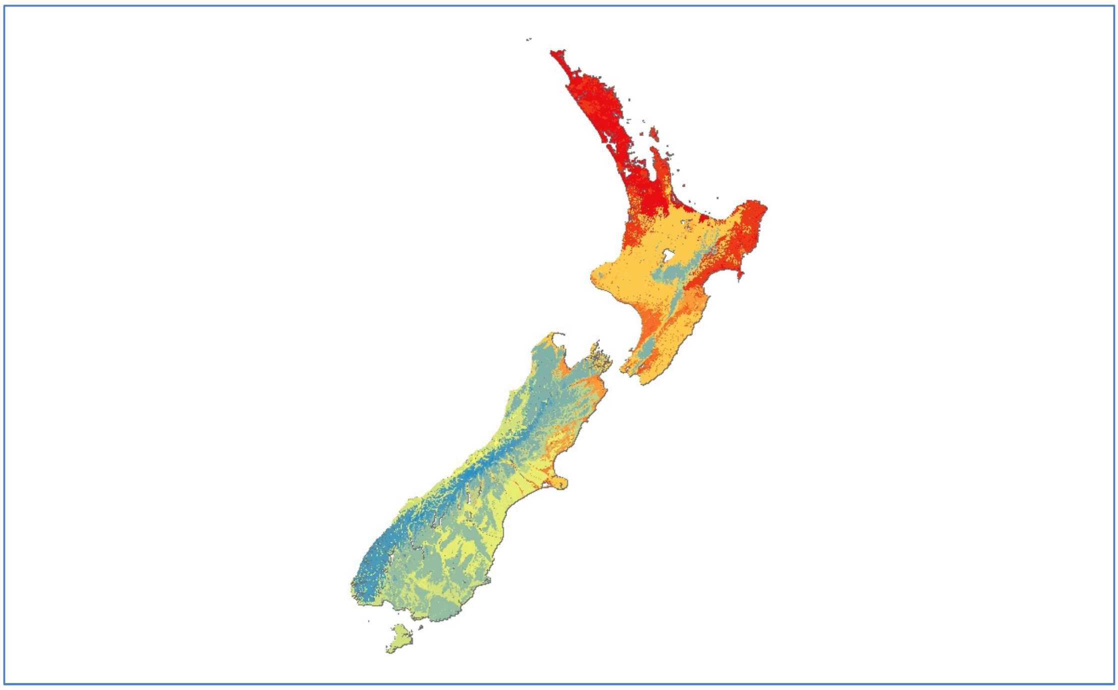

Figure 4.10: New Zealand Map showing Annual temperature data (Data source LENZ Landcare Research NZ)

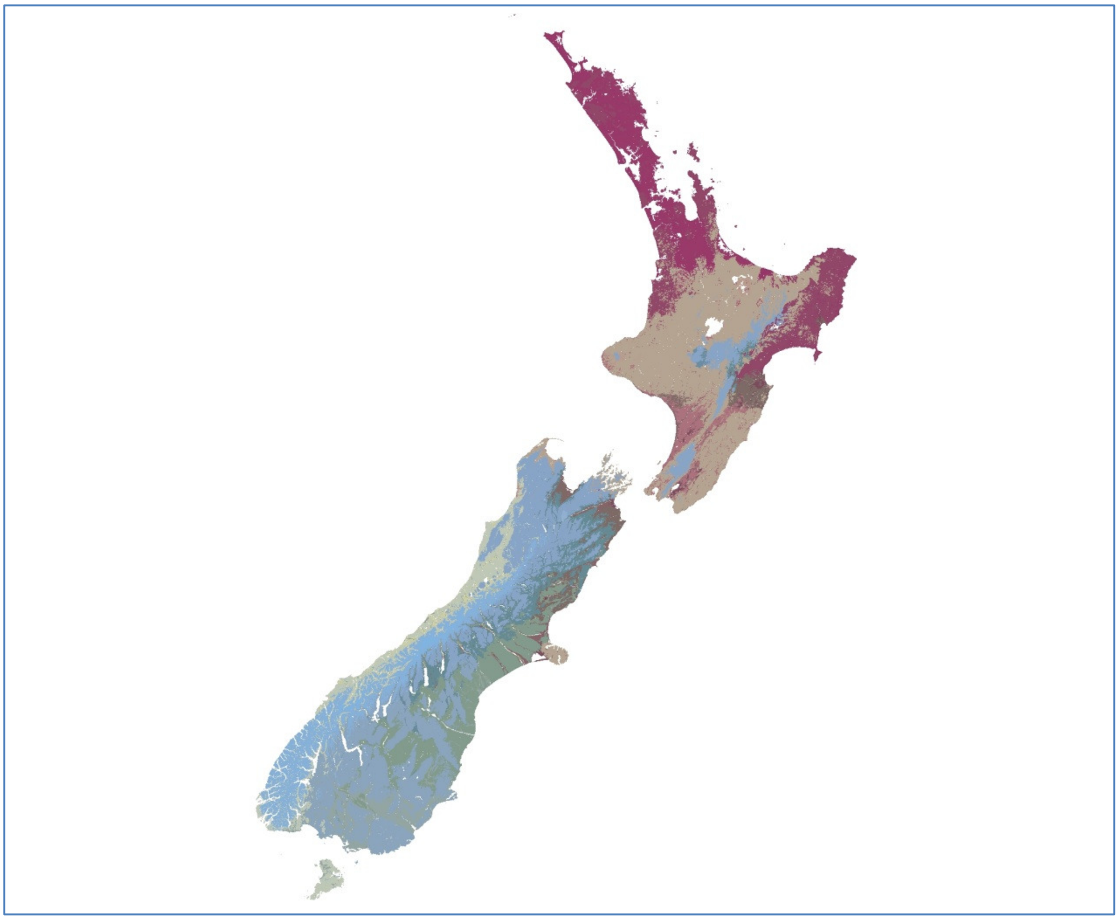

Figure 4.11: New Zealand Map showing layered climate data incorporating Water vapour deficit and temperature (Data source LENZ Landcare Research NZ).

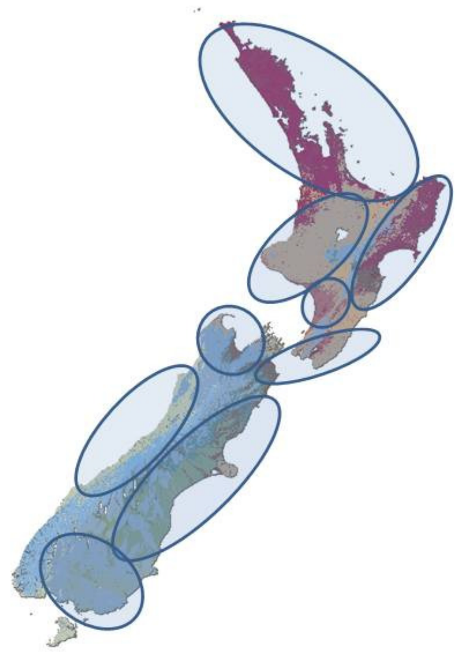

Figure 4.12: Monitoring boundaries based on layered climate data incorporating Water vapour deficit and temperature (Data source LENZ Landcare Research NZ).

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