# INVESTIGATING THE IMPACT OF HUNTING ON INSULAR NEWFOUNDLAND CARIBOU USING VIRTUAL POPULATION ANALYSIS

DANA ORLANDO PECKHAM

# Investigating the Impact of Hunting on Insular Newfoundland Caribou using Virtual Population Analysis

**Dana Orlando Peckham**

# Investigating the Impact of Hunting on Insular Newfoundland Caribou using Virtual Population Analysis

by

**© Dana Orlando Peckham**

A practicum submitted to the school of Graduate Studies

in partial fulfillment of the requirement for the Degree of

Master of Applied Statistics

**Department of Mathematics and Statistics**

**Memorial University of Newfoundland**

June, 2008

**St. John's** **Newfoundland and Labrador** **Canada**

# Abstract

The key issue in wildlife management is developing strategies to maintain the long-term sustainability of a species. In order to develop a management strategy, we must first understand the make-up of the species including estimates of the stock abundance. The species we are concerned with, in this case, is insular Newfoundland caribou or *rangifer tarandus caribou.*

Hunting and trapping is often thought of as a recreational activity, but it also plays a crucial role in wildlife management. Hunting contributes to wildlife management in many ways that most people do not even realize. It is used to maintain a healthy species population, especially in cases where there are no major predators like wolves. The information gathered through hunter returns helps to determine the status of a population by things like how many animals they saw, what was the sex and age (calf or adult) of the animals they saw, and how many of these hunters were successful in their hunt. Hunter experiences also help in understanding the behavioral patterns of a species.

Other things that hunting does that may not be so obvious are things like its contribution to the economy of a region. For example, the U.S. Fish and Wildlife Service reported that in 2002, hunters and trappers contributed $847 million to state and wildlife management agencies via hunting and trapping licences and excise taxes. This does not even include revenues gained from pelt sales, outfitting and other spin-offs like hotels, gunsmithing and hunting apparel sales. Hunting and trapping is also a useful tool in

taking care of problem animals such as beavers that may cause extensive damage to roads, bridges and dams or coyotes killing livestock on a farm. There is also no arguing that wildlife-auto collisions would be significantly higher, were the population densities not controlled.

One method we are going to focus on in this paper is virtual population analysis, also known as VPA or cohort analysis. This technique uses catch-at-age data from hunters and using backward recursive formulas, estimates the number of animals alive for a specific cohort at a specific time. VPA has been used most extensively in fisheries analysis but can also be applied in other wildlife applications. There are other methods of abundance estimation as well, such as aerial surveys, which we will compare in the paper. The problem with aerial surveys is that they are time consuming and very expensive.

The research done in this paper will be facilitated using data provided by the Wildlife Division of the Department of Environment and Conservation, Government of Newfoundland and Labrador.

# Acknowledgements

I would like to express my deepest appreciation to Dr. Gary Sneddon for his help and guidance throughout this program as well as the undergraduate program before it. He made himself available for assistance whenever and wherever possible. He was always very encouraging and helped make the process a whole lot easier and more enjoyable with his positive attitude.

I'd also like to thank the Department of Environment and Conservation of the Government of Newfoundland and Labrador, in particular Mr. Shane Mahoney and Mr. Glenn Luther. The Department provided the data for this analysis and was very willing to provide any other materials necessary to facilitate the process. Mr. Mahoney and Mr. Luther came up with the idea for the project. They made themselves readily available at any time to offer suggestions or answer questions. It truly has been an honor to work with them on this, and other projects.

I am especially grateful for my parents who have always been there to support any endeavor that I have undertaken. Without them, none of this would have been possible.

Finally, I am thankful for my wife, Julie, for her patience and emotional support. She may not know exactly what it is that I do, but she understands my passion for statistics and the wildlife of this great province, Newfoundland and Labrador.

# Contents

# List of Tables

# List of Figures

# Chapter 1

# Introduction

Maintaining a healthy and stable population is the main objective in wildlife management when it comes to any species. To manage a population, it is necessary to understand the background and current status of the species. The caribou hunt in Newfoundland has existed for the last several decades back as far as the 1960's. The sustainability of this harvest is a testament to the hard work and dedication of wildlife managers and particularly to the people of the province.

It is important to understand the demographic structure of the island's caribou herd to determine the state of the population. A commonly used method of obtaining an estimate for the total population is through aerial surveys. However, the cost and time associated with this method make it difficult to keep a consistent count on the herds throughout the entire island of Newfoundland. Although this method is effective when time and money allows, one drawback is that it gives no information on the age distribution of the animals. It is possible to determine the sex of the animals, particularly adults, but the age of animals greater than two years cannot be determined from a helicopter.

The purpose of this study is to provide an alternate method of estimating the total population and to give the demographic make-up of the island herd through the study period from 1980 to 2003. Virtual population analysis, when used under the right circumstances, is an effective tool used mainly in fisheries management but can be used in other wildlife species if the proper information is available. It is also a much cheaper technique than aerial surveys.

The main ingredients needed in VPA are catch-at-age data and natural survival rates. Catch-at-age data, or age of harvested animals, can be determined from hunter jawbone returns using pre-established scientific methods. Natural survival rates are acquired through radio collars placed on various individuals.

It is hoped through this study that this method will continue to be used in future in studying not only Newfoundland caribou but other species in this and other jurisdictions. Any gaps that are revealed in this study will hopefully lead to putting more emphasis on the importance of maintaining good data collection practices and hence provide better estimates going forward.

This paper will first define VPA and how the methodology will be applied to the case of Newfoundland caribou. The results of the cohort analysis will then be presented. Based on these results, we will investigate what else these results can tell us about the population. The virtual population determined will then be compared to another population estimating technique, aerial surveys. The final sections then will discuss other possible outcomes of the VPA had things been done a little differently as there were a couple of challenges that became apparent and we had some decisions to make on how to deal with them.

# Chapter 2

# Methods

## 2.1 Data Collection

The data used throughout this paper come from various sources of information collected on the island's caribou herd over the study period between 1980 and 2003.

With each caribou licence issued on the island of Newfoundland, there is a questionnaire issued with it that the hunter is asked to complete once the hunt is over. The questionnaire contains a number of questions regarding their hunt. Information obtained from the returns includes, but is not limited to, whether or not an animal was harvested, and if so, the date and location of the harvest, how many days hunted and how many animals seen. From 1980-2003, there were 83,931 caribou licences issued with 51,614 licence returns returned to the Wildlife Division for a return rate of 61.5%.

Another request issued by the Inland Fish and Wildlife Division of Newfoundland and Labrador is that successful hunters return the lower jawbone of the animal harvested to the division. From the lower jawbones, biologists can determine the age of the animal. For calves and yearlings, the age of the animal is determined based on tooth eruption (Peterson 1955). The ages of caribou greater than one year old are determined after careful inspection of cementum annuli in the first incisor (Sergeant and Pimlott 1959).

Over the study period, approximately 22% of the jawbones of all harvested animals were returned (21% for females and 22% for males).

Radio-telemetry data was also used which is a popular technique among wildlife biologists to estimate sex and age-specific survival rates. A radio collar is attached to individual animals and that animal's location can be tracked. There are two types of collars – GPS & satellite. GPS collars have to be searched after a pre-determined time-period to determine its location and death of the animal. Information from satellite collars can be downloaded from the manufacturer's web-site. The GPS collar information gives you a location every 2 hours and is more accurate. These collars will also notify when no movement has been detected over a specified period of time, indicating the possible death of the animal. Biologists can then determine the time, location and once found, the possible cause of death. During the study period, there were 1470 caribou collared at various times for various ages of both males and females.

One other source of information used throughout this study will be herd composition surveys, or classifications. Classifications are observations made on the ground of a particular herd determining the sex and age make-up of the herd, antlered and non-antlered animals, and numerous other calculations based on these observations. There were 234 classifications done between 1980 and 2003 on the various individual herds throughout Newfoundland.

## 2.2 Standard VPA Methodology

Virtual population analysis, or cohort analysis, is commonly used in fisheries and is a recursive method of estimating the age and gender make-up of a harvested wildlife population (Fryxell 1988, Sparre and Venema 1998). If a population were to be observed over a long enough period of time, eventually, the death of each animal could be recorded. Based on these recordings, we then know when each animal was recruited into the population and we can construct the population dynamics through time. The problem is that we cannot track the death, both natural and harvested, of every individual.

The number of harvested animals is estimated from sample hunter jawbone returns and hunter questionnaire returns. The jawbones are examined and the age and sex are determined from the mandibles. This information, combined with the success rates of all licences obtained from the returned hunter questionnaires, gives an estimate of the total number of animals harvested in each cohort. This number is then revised upward by 20% to account for additional crippling loss. Crippling loss is the wounding of an animal by a hunters' bullet and eventually dies but is never found by the hunter.

Natural death was calculated using data collected from radio-collared animals by a method known as the Heisey-Fuller method (Heisey and Fuller 1985). Radio telemetry is a popular technique used in the biological sciences to gauge natural survival rates of wildlife. When an animal is collared, its age, sex and date of collaring are recorded. Each twenty-four hour period of an animal wearing a radio collar, is known as one transmitter day. The survival days are then known from when the animal dies. Annual survival rates were calculated for both male and female calves, yearlings, two-year-olds and adults. For the purpose of calculating natural death, animals that died due to hunting, slipped collars or malfunctioning collars are considered censorship cases. Note, however, that poached animals are included in the calculation as deaths even though poaching is not really a "natural" cause of death. Deaths that are due to calf abandonment following collaring as well as deaths due to the collaring procedure are also classified as censorship cases.

Often, the exact date of death is not known. In such cases, the midpoint procedure is used to estimate the time of death. This is simply the mid-point between the last date known to be alive and the first date the collar was in mortality mode or not detected.

Survival rates among caribou, especially calves, can be quite different for different seasons. Therefore, since collaring took place at various times throughout the year, it makes sense to estimate survival for spring, summer, fall and winter separately. Otherwise, seasons with the largest samples will be most influential in the estimates. Season dates for this study were set as May 1 to June 30 for spring, July 1 to September 30 for summer, October 1 to November 30 for fall and December 1 to April 30 for winter. Calves, however, are born around the first week of June so their annual season is based on just 334 days, to April 30th of the following year. The lengths of the intervals can differ without affecting the annual estimates.

The seasonal estimates are calculated as:

$$S_i = [(x_i - y_i) / x_i]^{L_i},\ i = 1, 2, 3, 4 \qquad (2.1)$$

where $x_i$ is the total number of transmitter-days, $y_i$ is the total number of deaths, $L_i$ is the length of the interval and *i* is the season (Trent and Rongstad 1974). The associated variance of this estimate is calculated as (Johnson 1979):

$$\mathrm{Var}(S_i) = [\ (x_i - y_i) * y_i\ ] / (x_i)^3,\ i = 1, 2, 3, 4 \qquad (2.2)$$

This seasonal estimate, $S_i$, is actually the maximum likelihood estimate (MLE) of $S_i$ (Heisey and Fuller 1985, Bart and Robson 1982).

*Proof that* $S_i$ *is the MLE of* $S_i$*:*

*If we assume the daily survival, s, remains constant through the season, then the probability of surviving an interval of one day is*

$$P(d_i = 1) = s,$$

*where* $d_i = 1$ *if the individual survives the interval and 0 otherwise. Therefore, the random variable* $d_i$ *has a Bernoulli distribution with*

$$g(d_i) = (s)^{d_i}(1 - s)^{1 - d_i}.$$

*A sample of size N intervals then gives a log likelihood of N such Bernoulli probabilities of*

$$ln \prod_{i=1}^{N} g(d_i) = ln \prod_{i=1}^{N} [\,(s)^{d_i}(1 - s)^{1 - d_i}\,]$$

$$= \sum_{i=1}^{N} d_i \, ln(s) + \sum_{i=1}^{N} (1 - d_i)\, ln(1 - s)$$

*Maximization of this log likelihood then involves taking the derivative with respect to s and set equating to zero where we get:*

$$\sum_{i=1}^{N} d_i\,(1/s) + \sum_{i=1}^{N} (1 - d_i)\,[\,1/(1 - s)\,]\,(-1) = 0$$

$$(1/s) \sum_{i=1}^{N} d_i = [\,1/(1 - s)\,] \sum_{i=1}^{N} (1 - d_i)$$

$$(1 - s)/s = [\,N - \sum_{i=1}^{N} d_i\,] \;/ \sum_{i=1}^{N} d_i$$

$$s = \sum_{i=1}^{N} d_i \,/\, N,$$

*so we have* $s = \sum_{i=1}^{N} d_i / N$. *But* $d_i$, *the number of individuals surviving, is the same as* $x_i - y_i$, *the number of transmitter days less the number of deaths, in equation (2.1) above, and* $N$, *the number of intervals, is simply the total number of transmitter days,* $x_i$, *in equation (2.1). Therefore,* $s$ *is the MLE of* $s$ *and, due to the invariance property of MLE's (Kalbfleisch and Prentice 1980),* $S_i$ *is the MLE of the survival rate for season* $i$, $S_i$.

The annual survival rate over all four intervals is then calculated as:

$$S^* = \prod_{i=1}^{I} S_i,\ i = 1, 2, 3, 4.$$

Virtual population analysis assumes that there are a negligible number of animals that survive the harvest beyond a certain age and that a population with a short hunting season has a negligible number of natural deaths that occur during the hunting season. For this analysis, the terminal age of 13 was used since less than 5% of the total kills in each year from 1980 to 2003 were made up of individuals over twelve years of age. Using this assumption, the total number of animals of age 13+ in any given year is equal to the number of 13+ year olds harvested. Working in a recursive fashion, this number, plus the number of 12 year olds that died both naturally and through the harvest in the previous year, gives the total 12 year old population in that previous year. Eventually this procedure will provide each cohort that has passed through the population.

The above can be expressed mathematically by using the following equations (Fryxell et al. 1988):

$$N_{i+1,t+1} = (N_{i,t} - K_{i,t})p_i, \qquad (2.3)$$

$$N_{i,t} = (N_{i+1,t+1} / p_i) + K_{i,t} \text{ and} \tag{2.4}$$

$$N^{*}_{i+1,t+1} = K_{i+1,t+1} \text{ ,} \tag{2.5}$$

Where $N_{i,t}$ is the number of animals of age $i$ ($i = 0,1,\ldots,13$) present prior to the hunt in year $t$ ($t = 1980,1981,\ldots,2003$); $K_{i,t}$ is the number killed in the hunt; $p_i$ is the natural age-specific survival rate in the absence of hunting and $N^{*}_{i+1,t+1}$ is the number of terminal aged animals. Since we want to construct the population backwards through time, we need equation (2.4) which is simply equation (2.3) rearranged to solve for $N_{i,t}$. However, one must first calculate equation (2.5) to get the starting point, then substitute that into equation (2.4) and work recursively.

This process will give all cohorts that have passed through the population. However, we need another procedure to get the cohorts that have not. Commonly used in fisheries applications (Baranov 1918, Ricker 1940), the following two equations are used to estimate cohorts that still have surviving animals:

$$K_{i,t} = N_{i,t} \left( 1 - e^{-f_{i,t}} \right) \text{ , and} \tag{2.6}$$

$$f_{i,t} = q_i E_t \text{ ,} \tag{2.7}$$

where $f_{i,t}$ is the instantaneous rate of hunting mortality; $E_t$ is the total effort exerted by hunters (which in this case is expressed as simply the number of hunters); and $q_i$ is the vulnerability coefficient defined as the proportion of the population killed by one unit of hunting effort, i.e. by one hunter. Age-specific vulnerability coefficients are estimated from the completed cohorts. This, when combined with the total hunting effort, gives the instantaneous rate of hunting mortality for each cohort in the final year of the study. Estimates of $N_{i,t}$ are then calculated after rearranging equation (2.6) and obtaining the kills by age for this latest year. Subsequent years are then calculated as before, using equation (2.4).

In the terminal year we must calculate the incomplete cohorts. Since we are missing a piece of information that we had in previous years, we expect that these estimates may not be as accurate as the previous years', particularly those years not based on any cohorts in this final year.

## 2.3 VPA Methodology Modifications

The above procedures are the normal steps to follow in a virtual population analysis. However, for reasons that will be explained, there were modifications that were needed for the analysis of the caribou population. First, after initially running this procedure for both males and females it became evident that the age-specific populations for females were quite variable through time. This was due to the fact that harvest rates for females were quite a bit lower than those of males. Using classifications and census estimates, the harvest rate for males averaged about 9%, ranging from 5.0% in 1986 to 20.5% in 2002. For females the rate averaged around 1%, ranging from 0.5% in 1992 to 2.4% in 2002. Combine this with the fact that only 21% of all females harvested have their jawbones returned, and we realize that the small sample of age-specific female harvest numbers can be very erratic from year to year. For this reason, assuming that the sample of 13 year old females harvested is an accurate reflection of the total number of 13 year olds in the population may not be a valid assumption.

Since this estimate of 13 year olds was supposed to be the starting point, we must determine a new, appropriate starting point for females. We know from classifications that, at birth, the sex ratio of males to females is known to be 1:1. So, instead of working from the terminal age of 13+ and working backward like we did for males, we will work in the opposite direction. It is then known through the male VPA analysis how many male calves there are and so we assume the same number of female calves. Then we calculate the proceeding years using the same formula as before except using it in its original state, equation (2.3):

$$N_{i+1,t+1} = (N_{i,t} - K_{i,t})p_i.$$

Now for males, we had to use the vulnerability coefficient in the final year of the study to estimate cohorts that had not passed completely through the population. Here, for females, we must use this same vulnerability coefficient to estimate cohorts in the initial year of the study. The remaining cohorts can then be calculated from equation (2.3) using these estimates from the initial year of the study.

One other modification was made concerning estimates of calves and yearlings in the final year for males and in the initial year for females. Vulnerability coefficients were not used for calves and yearlings as they were for all other cohorts. The reason is that harvest rates for these cohorts are less than 1%. Since the population estimates based on vulnerability coefficients are highly dependant on the harvest estimates, even very small differences in harvest estimates can give drastically different results. For example, the estimate for male calves killed in 2003 is zero. Therefore, if we base the population estimates on this and vulnerability coefficients, then the estimate for total calves in 2003 would be zero, which we know is not correct.

Therefore, in the final year, calves were estimated from spring classifications which show the percent of adult females, two years or older, that have calved in that year.

This percentage is then applied across the adult females already calculated, to get the number of calves. Yearlings in the final study year were also calculated using spring classifications in that same year. Yearlings as a percent of total adult caribou is estimated from the classifications and then used to estimate the number of yearlings for the VPA.

## 2.4 Assumptions

As with most statistical procedures there are a number of assumptions that must be made in VPA. As stated earlier, it is assumed that there are a negligible number of animals that survive the harvest beyond a certain age. There is no evidence to suggest that hunters avoid harvesting older animals so this seems to be a reasonable assumption. We also trust that the hunter return questionnaires provide accurate and knowledgeable information. Other analyses using Newfoundland hunter return data have shown remarkable similarities to more scientific methods, so this also seems a safe assumption. We must also assume that estimates of natural survival rates are not affected by changes in weather patterns, population density and habitat conditions. Although this assumption may not hold, provided that deaths due to harvesting greatly exceed natural deaths, biased estimates of age-specific survival rates should not affect trends in abundance from cohort analysis (Ulltang 1977). This procedure also assumes a closed population with no immigration or emigration, which of course, in this case is certainly the case as Newfoundland is an island with great distances to other lands.

For the cohorts that have not passed completely through the population, there are additional assumptions to be made. If vulnerability coefficients do not remain constant through time or if the proportion of age $i$ animals harvested is not a linear function of the vulnerability coefficients, then significant biases may exist in the population estimates of recent years (Pope 1972). One final assumption is that the estimated age *i* animals harvested in the final year of the study based on $q_i$ and $E_t$ are not drastically different than the true harvest. If so, then not only will the final year population estimates be inaccurate, but so will the preceding years.

This final assumption may not hold for calves and yearlings since the harvest rates at these ages are significantly lower (<1%) than at other ages and hence the small sample of harvest estimates may not be accurate. However, the modifications for calculating calves and yearlings for both males in the final year and females in the initial year should account for this problem.

# Chapter 3

# Analysis and Discussion

## 3.1 Natural Age-specific Survival Rates

Natural age-specific survival rates were based on a sample of 1470 radio-collared animals between 1979 and 1998. Heisey-Fuller estimates were produced for calves, yearlings, two-year olds and adults for both males and females.

For all cohorts other than calves, the method is, as stated in Chapter 2, calculated for each season for both males and females and the annual survival rate is then calculated based on the seasonal rates. The results are presented in Table 3.1. For yearlings and adults, there is not a great deal of difference in the survival rates of males compared to females. However, for two-year olds there appears to be a higher survival rate amongst females than that of males.

**Table 3.1: Age-specific Heisey-Fuller Survival Rates by Season**

| Cohort | Sex | Season | | | | Total | 95% C.I. |
|---|---|---|---|---|---|---|---|
| | | Spring | Summer | Fall | Winter | | |
| Yearlings | Male | 0.9472 | 0.9611 | 0.9760 | 0.9369 | 0.8324 | (0.8318, 0.8331) |
| | Female | 1.0000 | 0.9342 | 1.0000 | 0.8966 | 0.8376 | (0.8371, 0.8380) |
| Two-year Olds | Male | 1.0000 | 0.8850 | 0.9668 | 0.9469 | 0.8102 | (0.8093, 0.8111) |
| | Female | 1.0000 | 0.9499 | 1.0000 | 0.9762 | 0.9272 | (0.9268, 0.9276) |
| Adults | Male | 0.9824 | 0.9672 | 0.9931 | 0.9252 | 0.8730 | (0.8728, 0.8732) |
| | Female | 0.9710 | 0.9613 | 0.9855 | 0.9557 | 0.8792 | (0.8791, 0.8793) |

For calves we had to make some modifications to get the survival rates for different periods. The results are presented in Table 3.2. From 1980-98, there was very little variation in the survival rates of both male and female calves, where estimates were between 60% and 65% for the sexes. Then from the period from 1999 through 2002, no data was collected on survival rates for all cohorts. In 2003, when estimates were obtained again, a dramatic difference was observed in the survival rates of caribou calves. The low survival rates of calves in 2003 continued for the next few years and was about 9% in those years for both males and females.

The issue then became estimating calf survival in the years where no information is available. We do know that rates didn't vary much from 1980-98 and from 2003-07, although the rate was much lower in the 2003-07 time period. Therefore, we assumed that the rates declined linearly between 1998 and 2003. Hence, we used different survival rates for calves, depending on the time period.

**Table 3.2: Calf Heisey-Fuller Survival Rates by Year(s)**

| | | Season | | | | | |
|---|---|---|---|---|---|---|---|
| Year | Sex | Spring | Summer | Fall | Winter | Total | 95% C.I. |
| 1980-1998 | Male | 0.8243 | 0.9202 | 0.9713 | 0.8683 | 0.6398 | (0.6387, 0.6408) |
| | Female | 0.8217 | 0.9280 | 0.9515 | 0.9018 | 0.6544 | (0.6536, 0.6552) |
| 1999 | Male | N/A | N/A | N/A | N/A | 0.5296 | N/A |
| | Female | N/A | N/A | N/A | N/A | 0.5417 | N/A |
| 2000 | Male | N/A | N/A | N/A | N/A | 0.4194 | N/A |
| | Female | N/A | N/A | N/A | N/A | 0.4290 | N/A |
| 2001 | Male | N/A | N/A | N/A | N/A | 0.3092 | N/A |
| | Female | N/A | N/A | N/A | N/A | 0.3163 | N/A |
| 2002 | Male | N/A | N/A | N/A | N/A | 0.1991 | N/A |
| | Female | N/A | N/A | N/A | N/A | 0.2037 | N/A |
| 2003-2006 | Male | 0.4755 | 0.3341 | 0.8814 | 0.6346 | 0.0889 | (0.0860, 0.0960) |
| | Female | 0.3749 | 0.3141 | 0.7727 | 1.0000 | 0.0910 | (0.0842, 0.0936) |

Another popular technique for estimating survival rates is the Kaplan-Meier (1958) or product-limit method. The survival rate in this method is expressed as the survival function:

$$S(t) = M_t/N_t,$$

where $M_t$ is the number of individuals surviving longer than time *t* and $N_t$ is the total number of individuals at risk at time *t*. The estimate of S(t) is calculated as:

$$S(t) = \prod_{t_i \le t} [\, 1 - d_i/n_i \,],$$

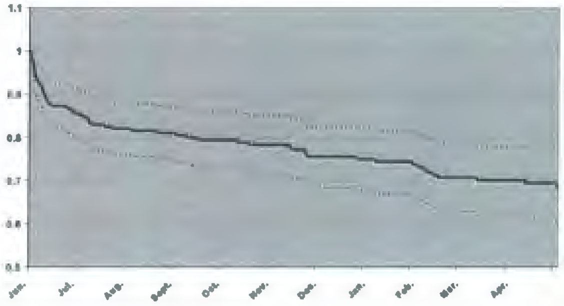
where $t_i$ is the study duration at point $i$, $d_i$ is the number of individuals that failed (or died) at time $t_i$ and $n_i$ is the number of individuals at risk prior to time $t_i$. The variance of S(t) is then estimated by the method of Greenwood (1926), where:

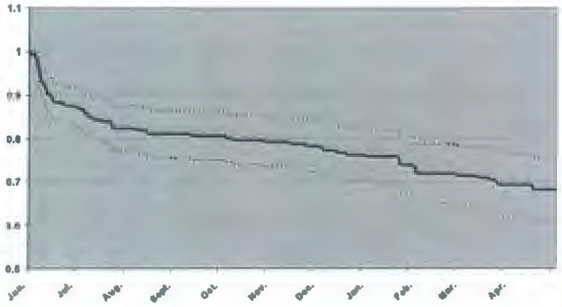
$$\mathrm{Var}\,[\,S(t)\,] = S(t) \sum_{t_i \le t} [\,d_i / n_i(n_i - d_i)\,].$$

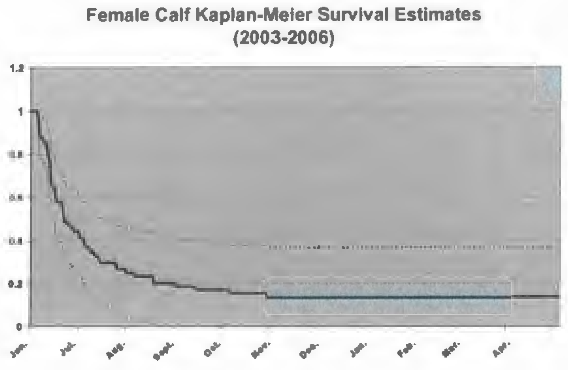
Using Kaplan-Meier estimates, the annual survival estimates are comparable to the original Heisey-Fuller estimates, as we can see when comparing Tables 3.1, 3.2 and 3.3. In fact, in each case, the 95% confidence interval based on the Kaplan-Meier estimates contained the Heisey-Fuller estimates. The product-limit method also allows us to easily graph the survival function throughout the entire year, so we can see how the survival changes through the year (Figure 3.1).

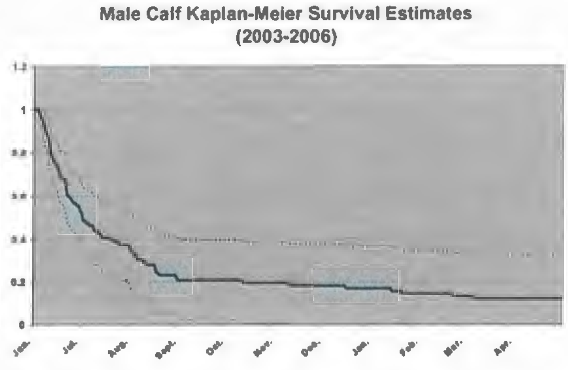
## Table 3.3: Age Specific Kaplan-Meier Survival Estimates

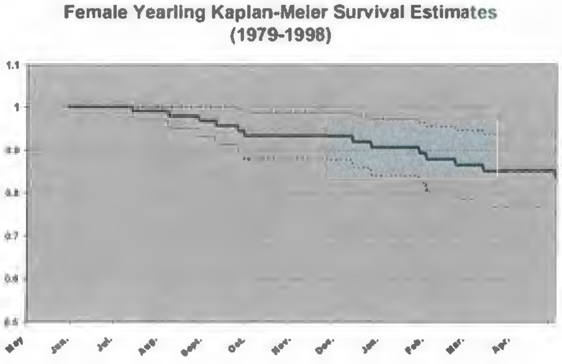
| Cohort | Period | Sex | Survival | 95% C.I. |
|---|---|---|---|---|
| Calves | 1980-1998 | Male | 0.6812 | (0.6067, 0.7557) |
| | | Female | 0.6802 | (0.5969, 0.7636) |
| | 2003-2006 | Male | 0.1184 | (0, 0.3218) |
| | | Female | 0.1344 | (0, 0.3652) |
| Yearlings | 1980-1998 | Male | 0.8596 | (0.7804, 0.9389) |
| | | Female | 0.8368 | (0.7471, 0.9265) |
| Two-year Olds | 1980-1998 | Male | 0.8045 | (0.6443, 0.9647) |
| | | Female | 0.9304 | (0.8610, 0.9997) |
| Adults | 1980-1998 | Male | 0.8731 | (0.8327, 0.9135) |
| | | Female | 0.8790 | (0.8582, 0.8998) |

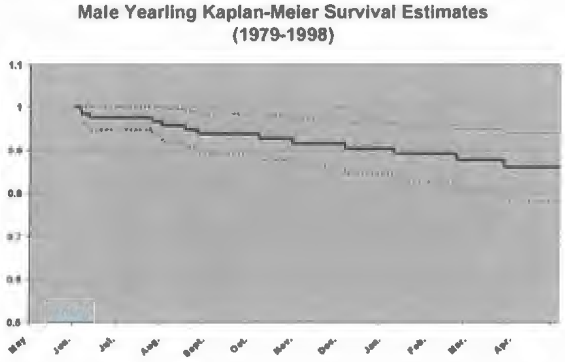

Female Calf Kaplan-Meier Survival Estimates
(1979-1998)

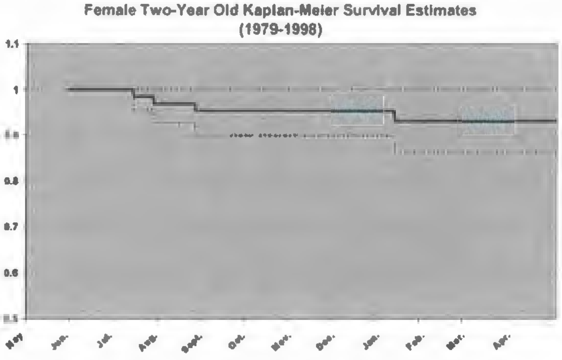

Male Calf Kaplan-Meier Survival Estimates
(1979-1998)

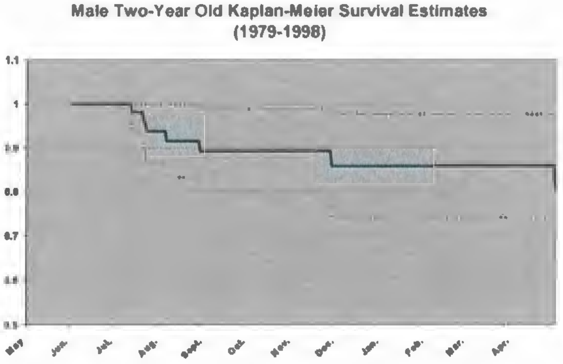

Female Calf Kaplan-Meier Survival Estimates (2003-2006)


Male Calf Kaplan-Meier Survival Estimates (2003-2006)


Female Yearling Kaplan-Meier Survival Estimates (1979-1998)


Male Yearling Kaplan-Meier Survival Estimates (1979-1998)


Female Two-Year Old Kaplan-Meier Survival Estimates (1979-1998)


Male Two-Year Old Kaplan-Meier Survival Estimates (1979-1998)

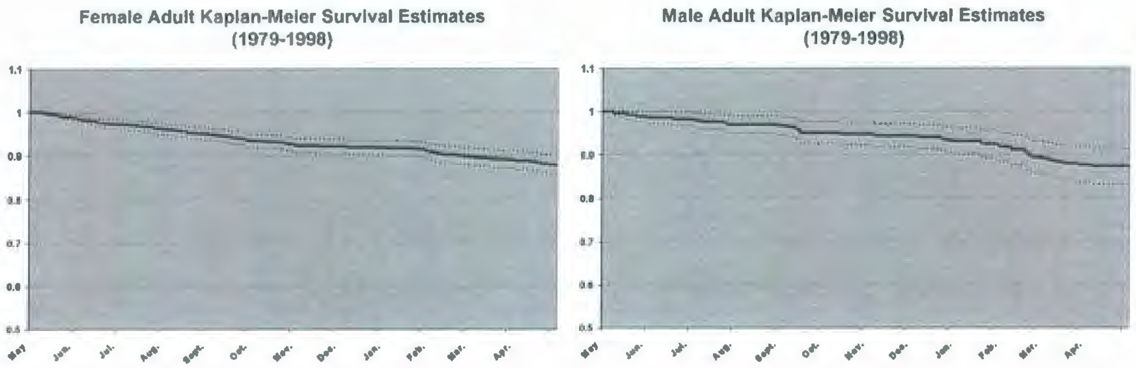


Figure 3.1: Kaplan-Meier Survival Estimates by Age and Sex

## 3.2 Virtual Population (1980-2003)

Virtual population analysis was based on 19,462 hunter jawbone returns collected from 1980-2003 from all harvested herds of caribou on the island of Newfoundland (Table 3.4). Results indicated that caribou abundance on the island peaked in 1995 at 116,604 animals. In 1980, estimates were just under 49,000 animals and steadily increased until 1995 and then went into a decline to just over 77,000 in 2003.

**Table 3.4: Population by Age**

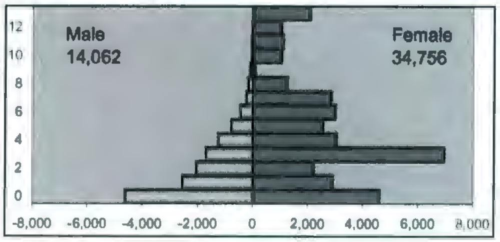
| | Age | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **0** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **12** | **13+** | **Total** |
| **1980** | 9,236 | 5,468 | 4,273 | 8,661 | 4,301 | 3,333 | 3,468 | 3,096 | 1,415 | 192 | 1,086 | 1,140 | 1,063 | 2,084 | **48,818** |
| **1981** | 9,970 | 5,946 | 4,566 | 3,673 | 7,320 | 3,501 | 2,760 | 2,909 | 2,616 | 1,159 | 126 | 919 | 969 | 917 | **47,350** |
| **1982** | 13,337 | 6,430 | 4,953 | 3,931 | 2,969 | 6,208 | 2,932 | 2,290 | 2,462 | 2,262 | 986 | 93 | 794 | 844 | **50,490** |
| **1983** | 16,019 | 8,622 | 5,363 | 4,220 | 3,304 | 2,486 | 5,287 | 2,435 | 1,919 | 2,101 | 1,948 | 838 | 55 | 688 | **55,288** |
| **1984** | 16,391 | 10,351 | 7,189 | 4,636 | 3,567 | 2,740 | 2,021 | 4,445 | 2,027 | 1,593 | 1,767 | 1,696 | 728 | 45 | **59,198** |
| **1985** | 16,420 | 10,601 | 8,625 | 6,194 | 3,882 | 2,968 | 2,306 | 1,596 | 3,685 | 1,696 | 1,363 | 1,527 | 1,482 | 631 | **62,976** |
| **1986** | 16,743 | 10,625 | 8,829 | 7,425 | 5,274 | 3,194 | 2,415 | 1,843 | 1,248 | 3,150 | 1,405 | 1,163 | 1,323 | 1,297 | **65,933** |
| **1987** | 20,077 | 10,809 | 8,863 | 7,574 | 6,316 | 4,388 | 2,553 | 1,905 | 1,479 | 973 | 2,732 | 1,226 | 1,003 | 1,158 | **71,055** |
| **1988** | 22,075 | 12,967 | 8,985 | 7,609 | 6,303 | 5,222 | 3,559 | 2,021 | 1,538 | 1,178 | 784 | 2,369 | 1,035 | 882 | **76,527** |
| **1989** | 20,765 | 14,274 | 10,828 | 7,756 | 6,418 | 5,242 | 4,204 | 2,825 | 1,590 | 1,194 | 952 | 642 | 2,040 | 898 | **79,629** |
| **1990** | 29,263 | 13,412 | 11,915 | 9,348 | 6,511 | 5,354 | 4,244 | 3,316 | 2,251 | 1,245 | 932 | 754 | 503 | 1,763 | **90,811** |
| **1991** | 30,893 | 18,835 | 11,199 | 10,292 | 8,045 | 5,488 | 4,374 | 3,366 | 2,638 | 1,779 | 956 | 780 | 625 | 419 | **99,669** |
| **1992** | 31,124 | 19,986 | 15,715 | 9,675 | 8,756 | 6,596 | 4,506 | 3,602 | 2,697 | 2,135 | 1,414 | 776 | 659 | 534 | **108,176** |
| **1993** | 27,295 | 20,117 | 16,689 | 13,647 | 8,226 | 7,302 | 5,341 | 3,652 | 2,911 | 2,180 | 1,738 | 1,158 | 654 | 555 | **111,464** |
| **1994** | 27,318 | 17,631 | 16,794 | 14,472 | 11,645 | 6,801 | 5,921 | 4,221 | 2,868 | 2,254 | 1,702 | 1,374 | 921 | 534 | **114,456** |
| **1995** | 26,761 | 17,639 | 14,714 | 14,502 | 12,372 | 9,739 | 5,606 | 4,690 | 3,275 | 2,259 | 1,729 | 1,387 | 1,136 | 794 | **116,604** |
| **1996** | 24,370 | 17,294 | 14,725 | 12,677 | 12,373 | 10,402 | 7,476 | 4,536 | 3,782 | 2,612 | 1,785 | 1,410 | 1,183 | 979 | **115,606** |
| **1997** | 22,778 | 15,718 | 14,210 | 12,264 | 10,727 | 10,361 | 8,544 | 6,104 | 3,615 | 3,036 | 2,113 | 1,455 | 1,191 | 985 | **113,101** |
| **1998** | 16,025 | 14,739 | 13,125 | 12,299 | 10,080 | 8,587 | 8,278 | 6,781 | 4,852 | 2,828 | 2,413 | 1,715 | 1,184 | 999 | **103,905** |
| **1999** | 16,530 | 10,369 | 12,302 | 11,306 | 10,147 | 8,321 | 6,801 | 6,490 | 5,239 | 3,844 | 2,233 | 1,932 | 1,430 | 1,000 | **97,943** |
| **2000** | 16,839 | 9,786 | 8,522 | 10,548 | 9,164 | 8,061 | 6,614 | 5,232 | 5,116 | 4,079 | 3,007 | 1,749 | 1,623 | 1,189 | **91,529** |
| **2001** | 13,687 | 9,041 | 8,174 | 7,105 | 8,414 | 7,319 | 6,369 | 5,177 | 3,917 | 3,889 | 3,147 | 2,412 | 1,425 | 1,377 | **81,452** |
| **2002** | 20,803 | 6,594 | 7,490 | 6,771 | 5,487 | 6,641 | 5,602 | 4,894 | 4,022 | 2,995 | 3,120 | 2,507 | 1,998 | 1,198 | **80,125** |
| **2003** | 27,417 | 4,137 | 5,344 | 6,056 | 5,105 | 3,952 | 5,045 | 4,350 | 3,710 | 3,100 | 2,417 | 2,562 | 2,123 | 1,702 | **77,019** |

The population pyramids show the age and sex distribution of a species through time (Figure 3.2). For our purposes, the vertical axis shows the age cohort and the horizontal axis shows the number of animals. The bars at the bottom represent the calf population and above that the 1 year olds, and so on until the top bar represents the number of animals ages 13+. The red bars on the right are the females and the yellow bars on the left are the males. A healthy population would have a pyramid shaped distribution with the majority of animals being younger and fewer animals at older ages.

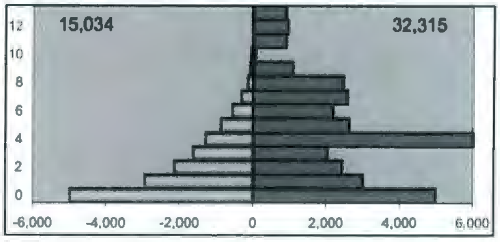
Early on we see that about 2/3 of the total population is female and also that the male age distribution seems fairly close to that of a pyramid. The female distribution is a little more erratic. Keep in mind, however, that 1980 is the year that the initial estimates for females started, and the estimate for this year will not be as good as those in subsequent years. Couple this with the fact that the sample sizes for females are smaller than the male samples, and we would expect to have less confidence in this initial estimate.

The population appeared to be quite healthy and stable from the early 1980's through to the late 1990's with an overall increasing population during that period. However, we notice in 2000 and even somewhat in 1999, that things started to change. In 2002 and 2003, we see that the age distribution does not resemble a pyramid at all.

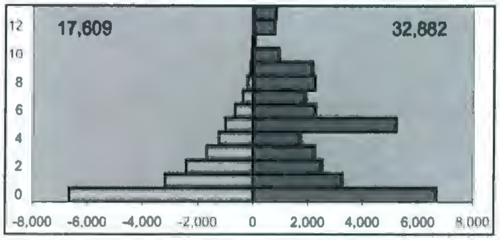
1980



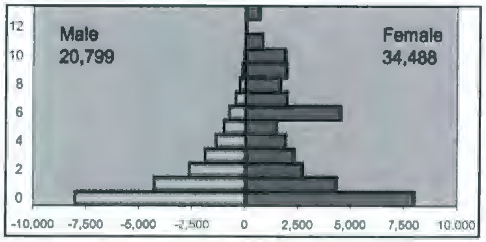
1981



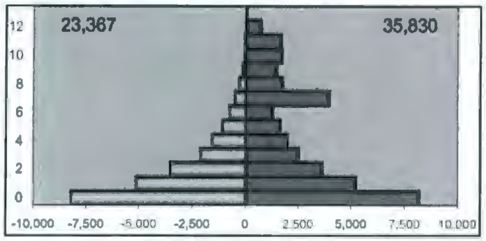
1982



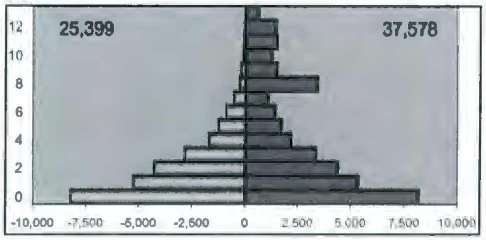
1983



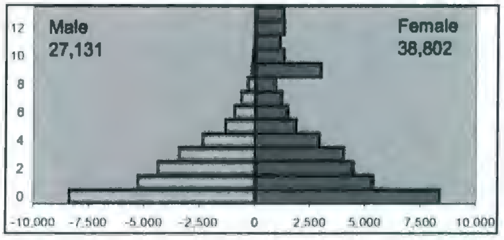
1984



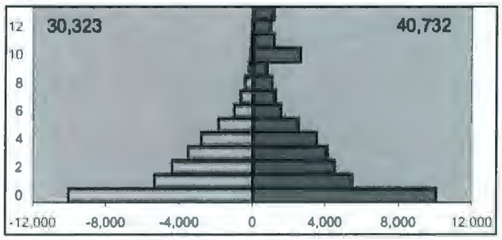
1985



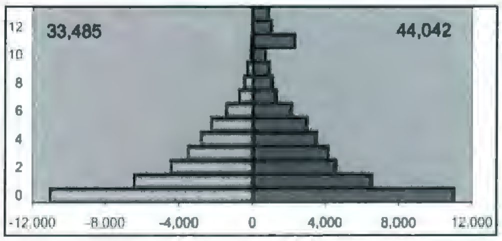
1986



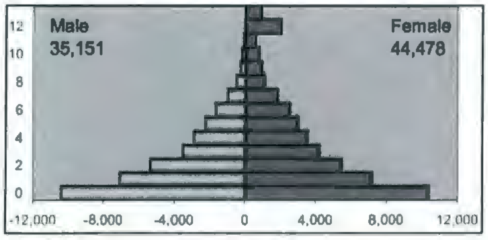
1987



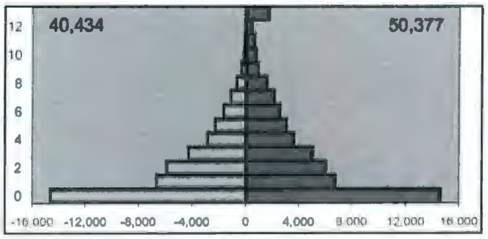
1988



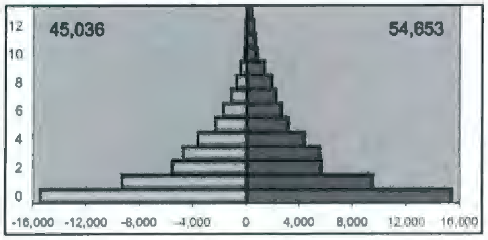
1989



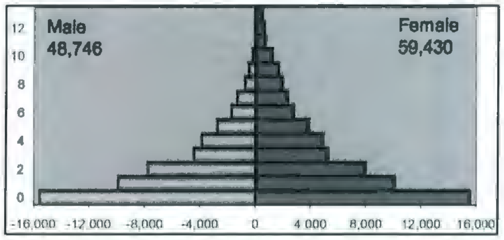
1990



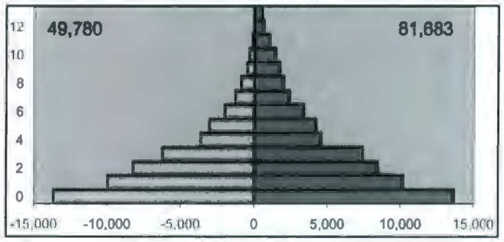
1991



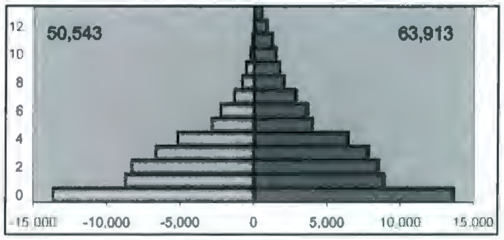
1992



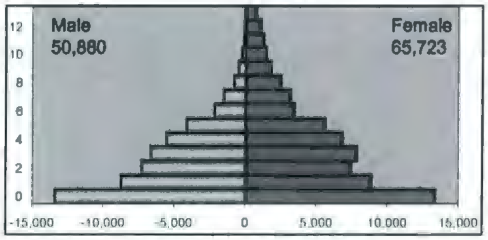
1993



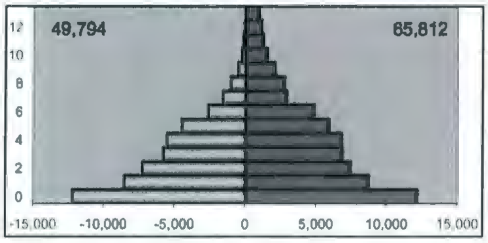
1994



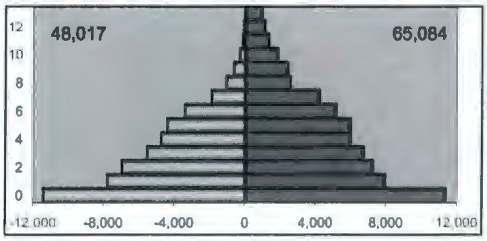
1995



1996



1997



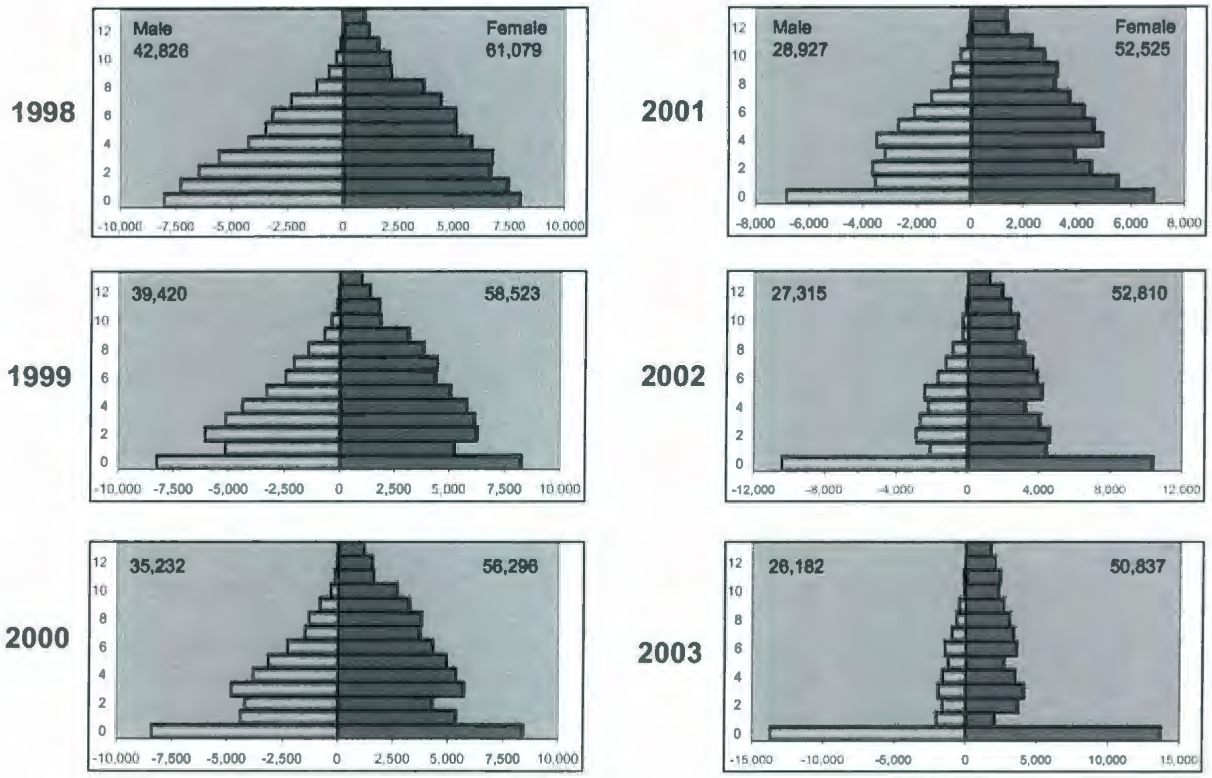


Figure 3.2: Population Pyramids by Year

Based on the resulting annual population, we can determine the percentage of animals that were harvested in a given year for a given sex, since we also have an estimate for the number of animals harvested for each year and sex (Figure 3.3). The results show that from 1980 to 1996, the annual harvest overall was between 2 and 4%. Since then, however, the harvest rate has steadily increased, peaking at almost 8% in 2002 and dropped back to about 6.5% in 2003. This is a direct result of the fact that the population was in a declining state, while the number of licences issued was increasing.

Similar results are seen for the sexes as well, especially for males. From 1980 to 1996, male harvest rates hovered between 4 and 8%. The increase then went as high as almost 18% in 2002 and stood at about 15% in 2003. Harvest rates for females were not so drastic, simply because there are fewer hunters harvesting female caribou. Rates in this case were always below 2% until 2001 and were above 2% in each of years 2001, 2002 and 2003.

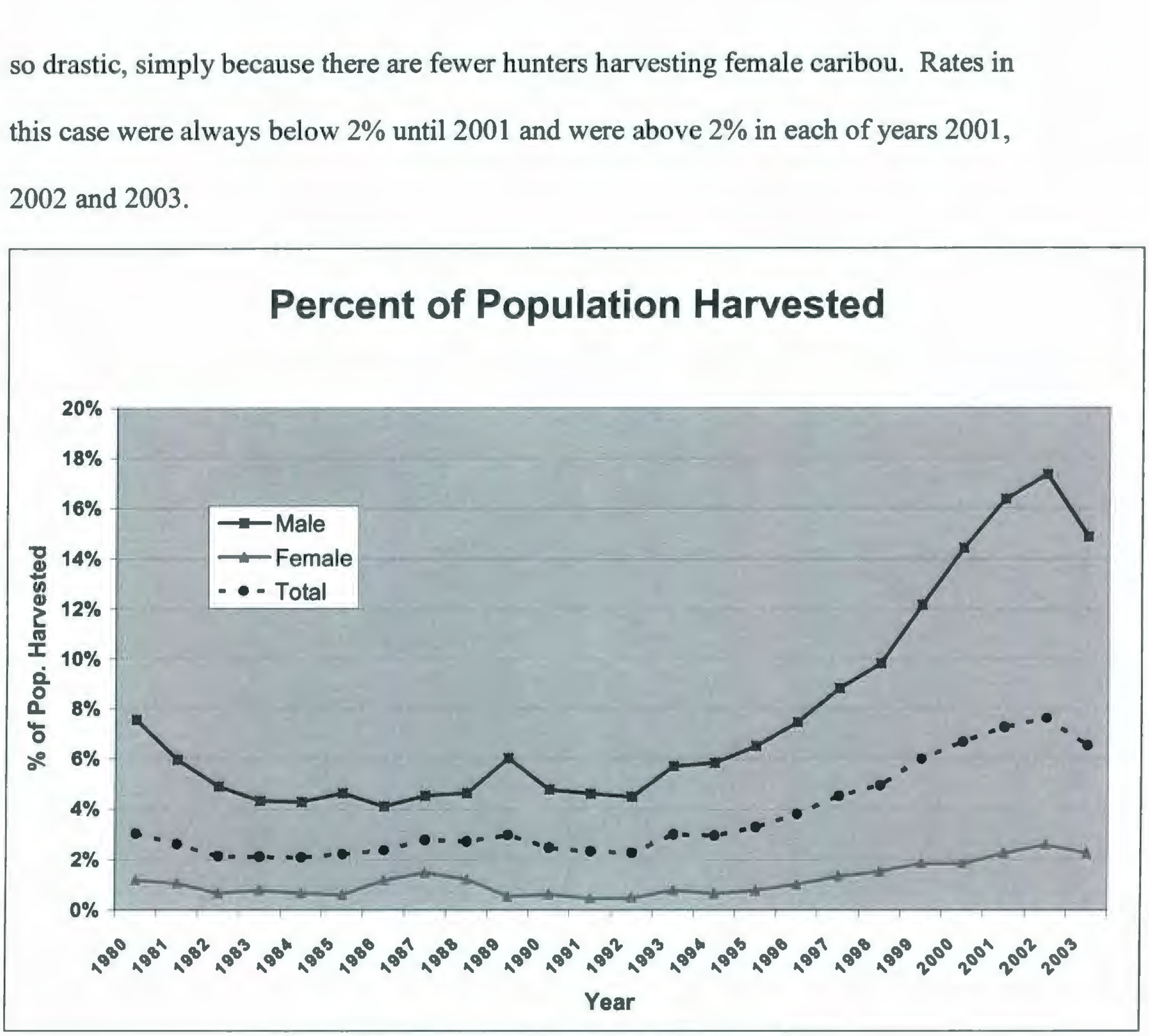


Figure 3.3: Percent of Population Harvested

From the results of the VPA, one can also investigate the mean age- specific harvest rates and vulnerability coefficients for both males and females (Figure 3.4). Mean age-specific harvest rates for males increased with each increment in age. That is, the older the animal the higher the proportion of the age group is harvested. This is not surprising since it is well known that hunters preference of animals are the big males with the trophy antlers and, for the most part, this coincides with the older males between the ages of 5 and 8 years. The really old males in the later stages of their life cycle do not have as big a set of antlers that they once had, but at the same time, one might believe they are also less agile and with their senses not being what they once were, they may be more likely to be harvested.

The mean age-specific vulnerability coefficients for males follows a similar pattern, increasing with age. The exception being that 10 year olds are less vulnerable than 9 year olds, and the same for 12 year olds being less vulnerable than 11 year olds.

For females, the situation is a little different. A very small percentage of females in each cohort are harvested each year, less than 3% in all cases. Harvest rates for calves and yearlings are negligible and rise slightly after that. The likely reason is that these animals are considerably smaller in size than the others and hence would provide less meat, so they are avoided. The same pattern can be seen in age-specific vulnerability rates for females as they are also well below the corresponding numbers for males.

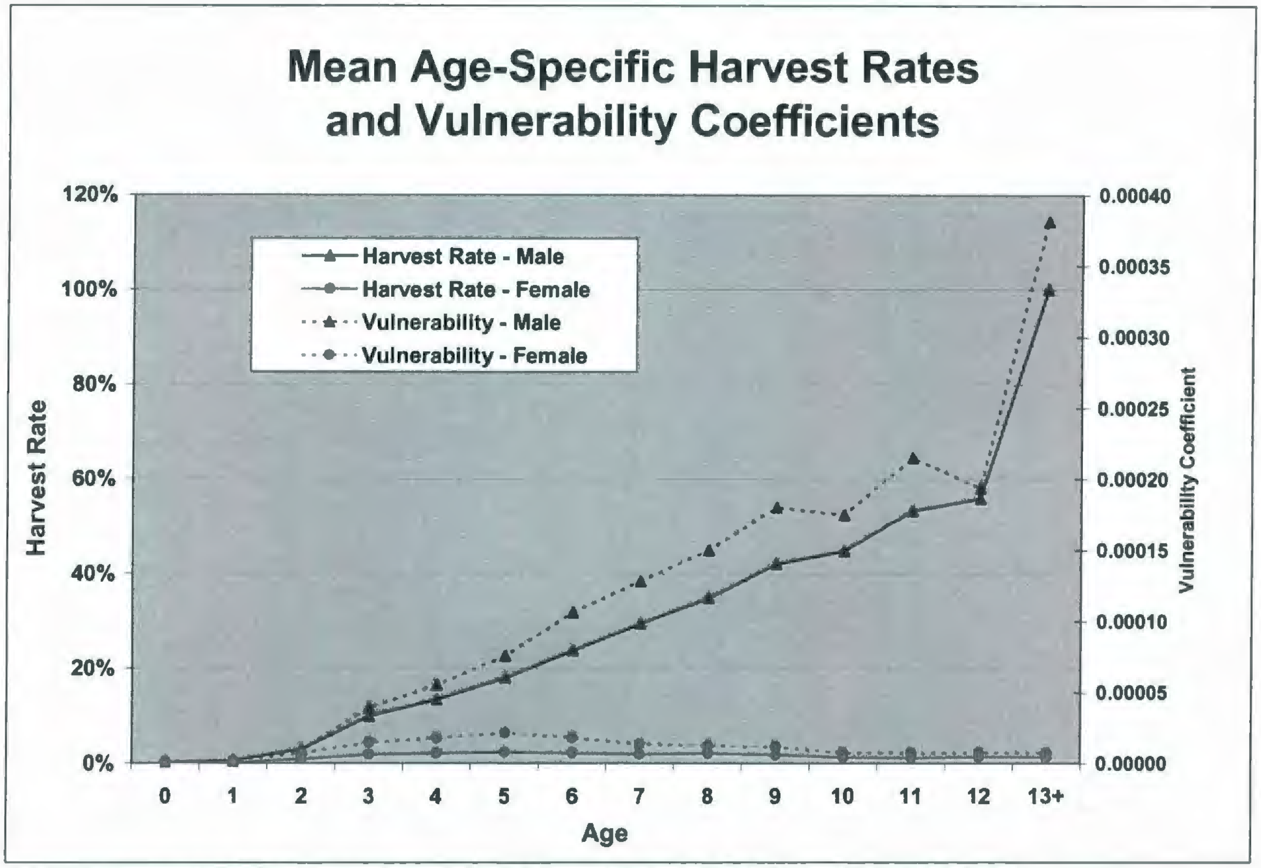


Figure 3.4: Mean Age-Specific Harvest Rates & Vulnerability Coefficients

It is a well known fact among wildlife biologists that a single male caribou can breed several females, and so it is not necessary to maintain a perfect 50/50 ratio of bulls to cows. White et al. (2001) did a study on the effect of adult sex ratios on productivity in elk and mule deer. They concluded that changes to the male:female sex ratio has little if any impact on subsequent population productivity. It is also known, as we discussed earlier, that hunters prefer males over females, to get that trophy antler rack but also because they are bigger and hence yield more meat for consumption. Therefore, since biology allows for it and hunters prefer it, there is more opportunity to harvest a male with male-only and either-sex licences being the only options. The question though becomes, what are the limits to how far we can skew the male-female ratio?

Although we will not provide an answer to this question, we will investigate these proportions from the results (Figure 3.5). From the beginning of the study period, 1980, results indicate that 71% of the population was female. This percentage then slowly declined year-over-year until it was just under 55% in 1991. It then rose again in the proceeding years to the point where it stood at 66% in the final year of the study, 2003. In comparison, the estimated male harvest during the period averaged 81% with a range between 70% and 90%.

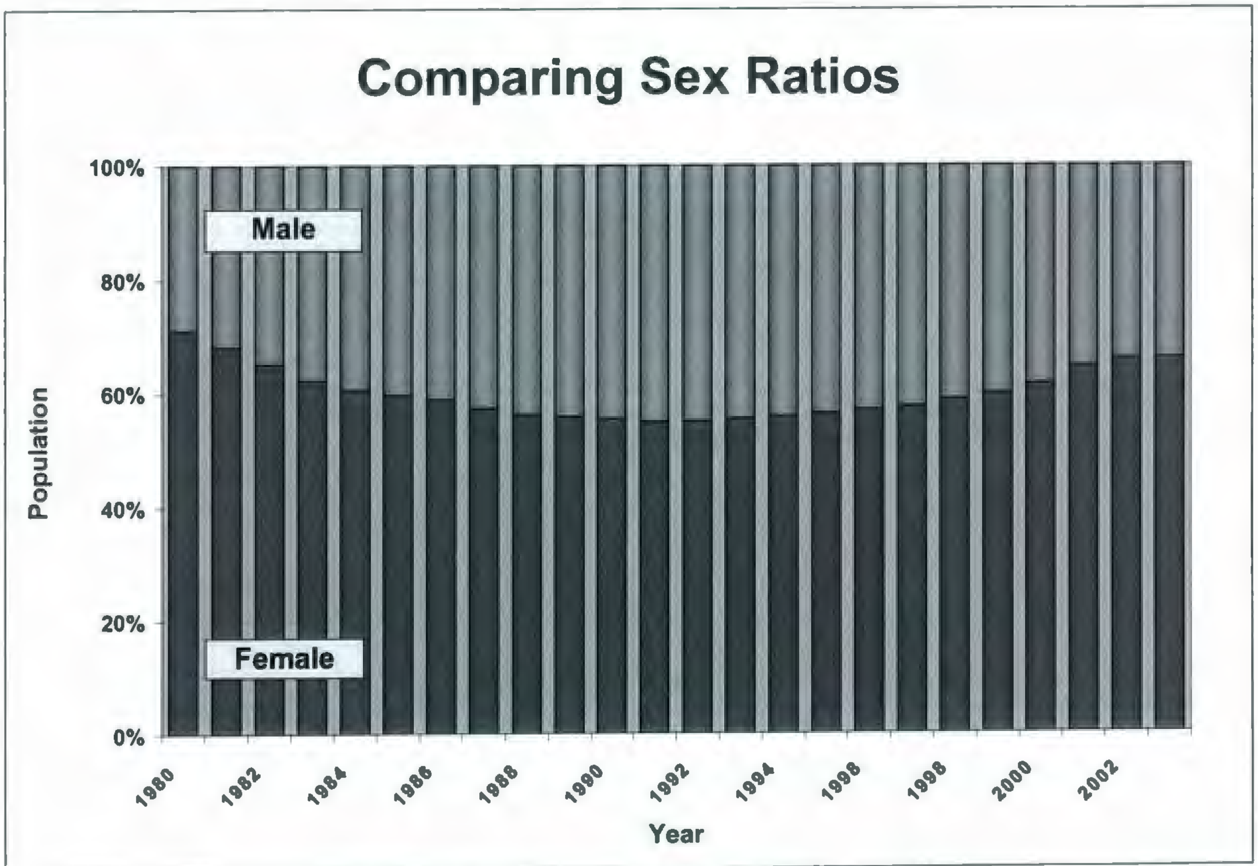


Figure 3.5: Comparing Sex Ratios

A healthy population in any species is normally thought of one that has an abundance of youth, and caribou are no different. Older animals tend to be weaker and hence are more prone to predators and less likely to survive harsh winter conditions.

Males tend to expend a great deal of energy fighting off other bulls during the rutting season, so they must be strong. At the same time, an impregnated female also needs to be healthy and strong in order to give birth to and nurture a healthy calf.

The mean age of the animals in 1980 was 3.9 years (Figure 3.6). This number came down as low as 2.5 in 1991 where it turned upward again to reach a peak of 4.3 in 2001. The trends were similar for both males and females although the mean age for males is much lower than that of females. This is not surprising, given the fact that stags are more heavily hunted and so do not have as great a chance to live as long a life as a doe. The mean age of harvest did not show any real pattern as it was up and down throughout the period. The mean age of harvest averaged 5.6 years with a range of between 5.0 and 6.1 years.

The mean age for males in 1980 was 2.1 and remained relatively stable through the rest of the 80's until it climbed to 3.0 in 1998 and remained there for the next couple of years. It then dropped quite rapidly to 2.6 in 2002 and 2.0 in 2003. The mean age for females declined from 4.7 in 1980 to 2.9 in 1991 and then rose again to 5.0 in 2001. The following two years saw moderate decreases to 4.8 and 4.7 respectively. So the mean age of each sex in 2003 ended up being almost exactly what it was when it started out in 1980.

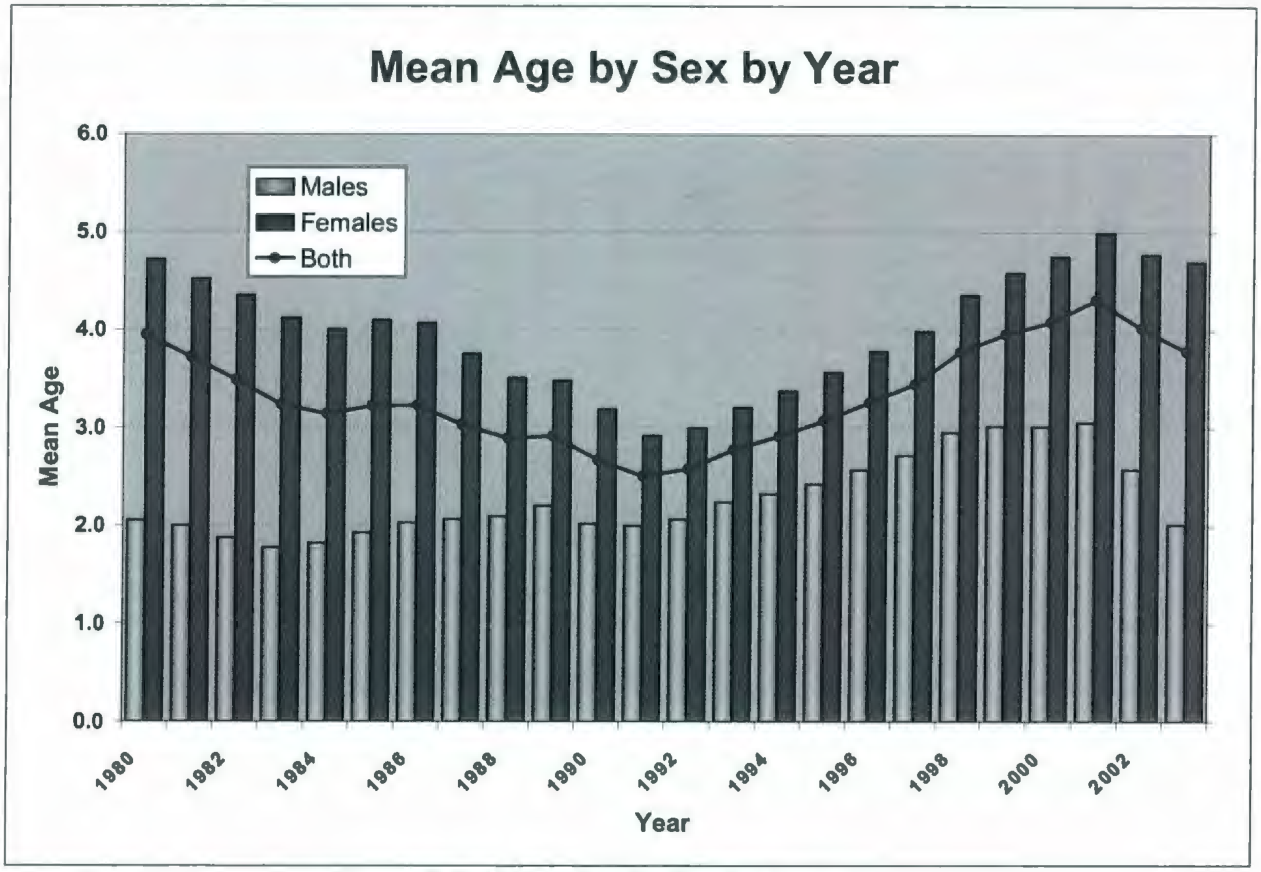


Figure 3.6: Mean Age by Sex by Year

## 3.3 Comparisons to Other Population Estimates

Another estimate used in estimating the population in big game animals is what is known as census counts. Census counts are actual manual counts of animals done using aerial surveys from a helicopter or fixed wing aircraft by traveling in a specific pattern to cover the home range of a given herd. There are different methods of conducting a census or aerial survey which include line transecting, blocking and mark-recapture (Krebs, 1999). The biggest problem with doing census counts is that it is a very expensive process. Particularly, as in this case, when there are several different herds to count over a vast land area. Because of the cost associated with each census count, it can only be done as funding is available. So in actual fact, it is not a true census count because there is never a time when all herds are counted in one specific year. There are certain herds chosen at various times. Therefore, the counts in years in between census counts have to be interpolated. Then, of course, the entire island wide herd is estimated by totaling each individual herd.

As one can see from Figure 3.7, the VPA results and the census estimates are quite similar. The cohort analysis estimates are consistently higher than those of the census, with the biggest gap being in the early-to-mid 1990's. The trends appear to be the same, showing population growth through the 1980's and early 1990's, and then a population decline in the late 1990's to the latest year, 2003. The estimates from the VPA indicate that the population peaked in 1995 at over 116,000 animals while the census estimates indicate a population peak one year later in 1996 at a little over 96,000. While the difference in the estimates is 20,000, we note that is no standard error estimate available for these estimates.

The correlation analysis certain shows that the two estimates have a very strong relationship to each other. The Pearson correlation coefficient between the two estimates was 0.973 with a p-value of less than 0.001 for the 24 years of data. Most census counts seem to be an accurate reflection of the true population, although as with any survey, there is always a margin of error associated with any count. So if we accept that the census counts are accurate then the VPA results appear to be pretty impressive.

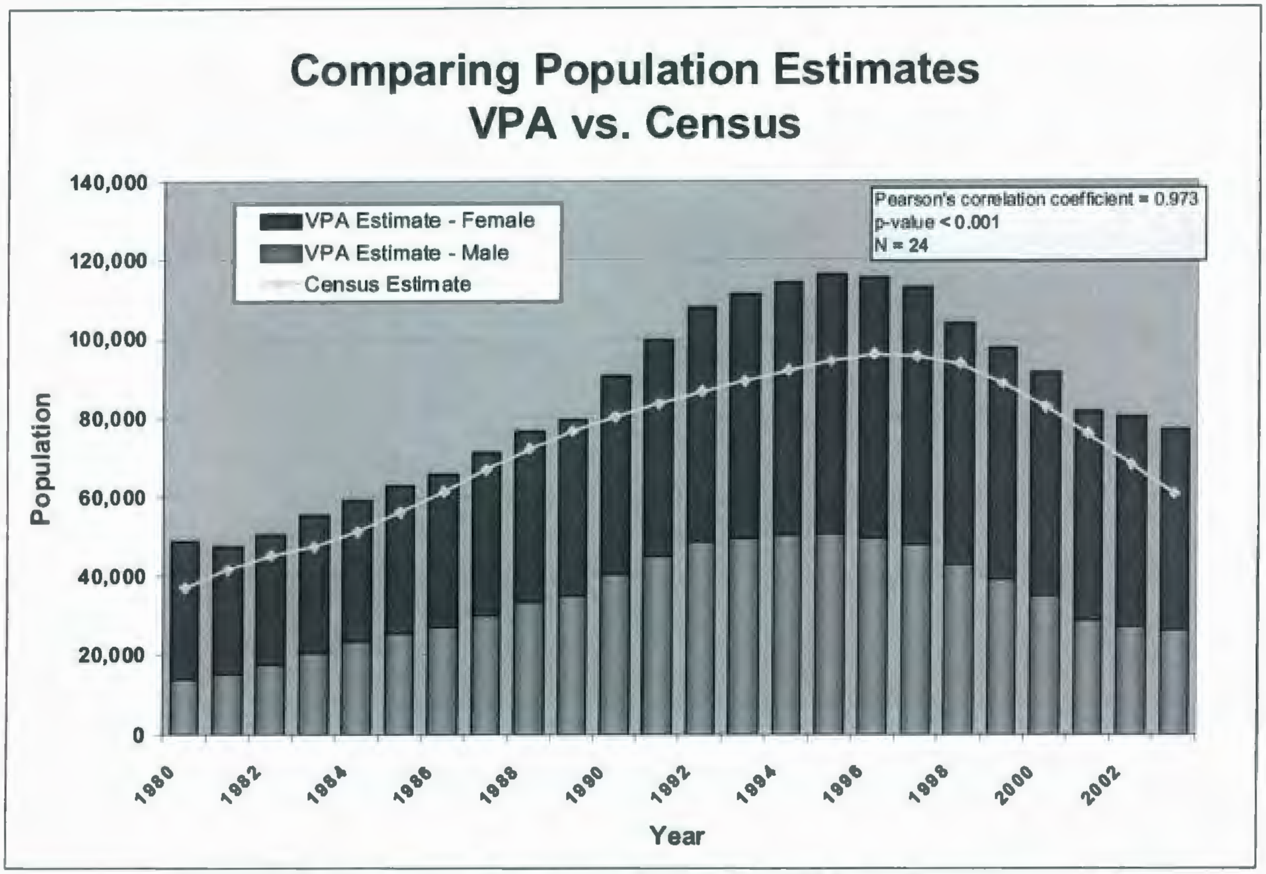


Figure 3.7: Comparing Population Estimates – VPA vs. Census

## 3.4 Using Hunter Returns to Estimate Population

Hunter returns provide invaluable information about what is happening with caribou herds. It has been shown in other studies (Filion 1981) that information provided by hunters can provide a good indication of what is happening in a wildlife population. We would like to know if hunter return data can be a good predictor of caribou herd populations in Newfoundland.

Some of the more commonly used statistics from hunter questionnaires is the average number of days hunted for successful hunters, the number of caribou seen per day by hunters and the success rates of hunters. Using these three indicators, we attempt to predict the population of the island's caribou herd.

Using stepwise linear regression, we get a model that appears to be a good fit, with a p-value of 0.006 (Figure 3.8):

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + e,$$

where $x_1$ is the number of days hunted, $x_2$ is the number of caribou seen per day and $x_3$ is the adjusted success rate of hunters. The variable chosen in the model turned out to be the adjusted success rate only. We can see from the corresponding correlation table (Table 3.5) that this success rate is highly correlated with the VPA population estimate. Although the other two variable possibilities are also highly correlated with the VPA population, they are also highly correlated to the success rate. Therefore, these variables are not necessary since these variables would then be accounting for variance in the model that has already been accounted for. The positive sign on the $\beta_1$ coefficient, shows that as the population increases, success rates of hunters also increases. The same is then true for a decrease in hunter success indicates a declining population.

**Table 3.5: Population and Hunter Return Correlations**

| Variable | | POP | DH | CS | AS |
|---|---|---|---|---|---|
| **Virtual Population (POP)** | Pearson Correlation | 1.000 | -0.422 | 0.539 | 0.543 |
| | Sig. (1-tailed) | . | 0.020 | 0.003 | 0.003 |
| | N | 24 | 24 | 24 | 24 |
| **Days Hunted (DH)** | Pearson Correlation | -0.422 | 1.000 | -0.258 | -0.347 |
| | Sig. (1-tailed) | 0.020 | . | 0.112 | 0.048 |
| | N | 24 | 24 | 24 | 24 |
| **Caribou Seen Per Day (CS)** | Pearson Correlation | 0.539 | -0.258 | 1.000 | 0.769 |
| | Sig. (1-tailed) | 0.003 | 0.112 | . | 0.000 |
| | N | 24 | 24 | 24 | 24 |
| **Adjusted Success (AS)** | Pearson Correlation | 0.543 | -0.347 | 0.769 | 1.000 |
| | Sig. (1-tailed) | 0.003 | 0.048 | 0.000 | . |
| | N | 24 | 24 | 24 | 24 |

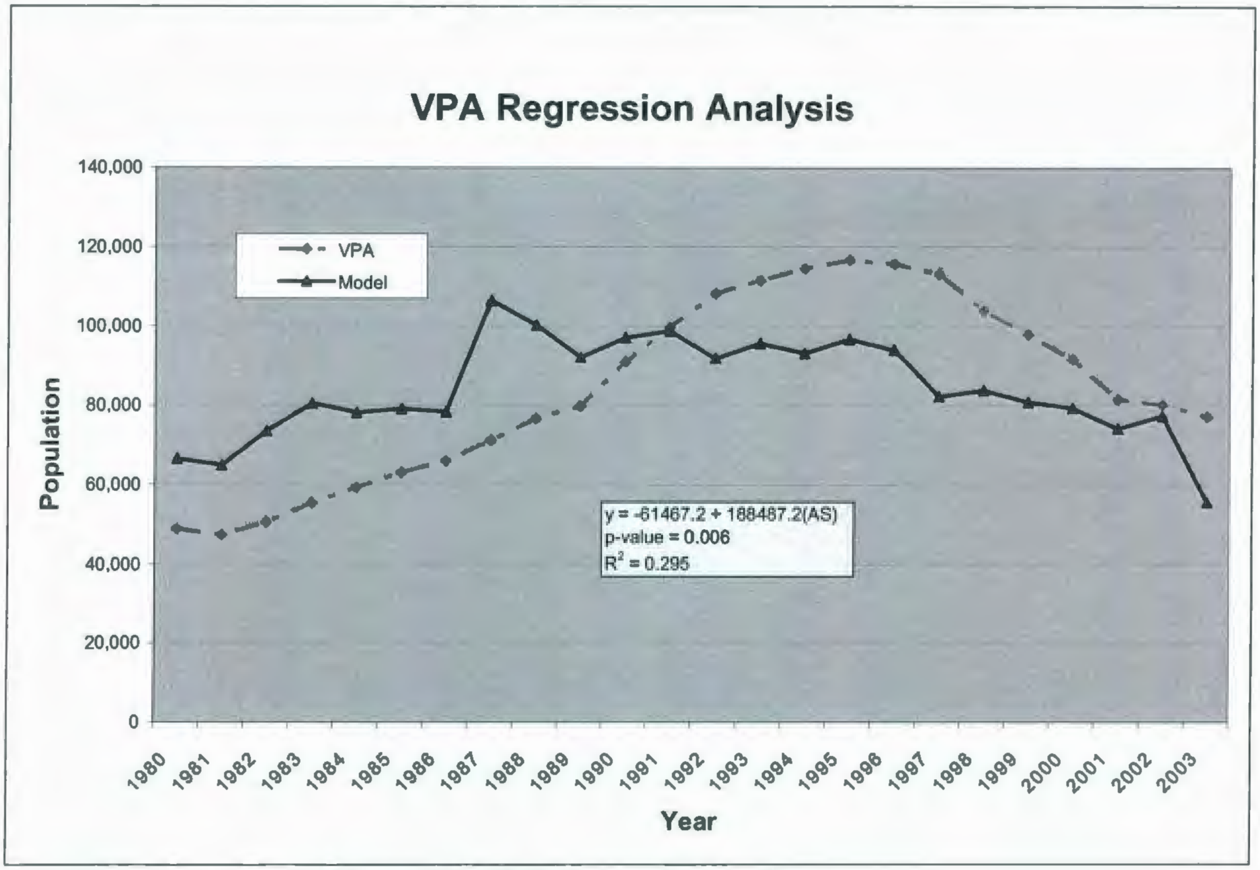


Figure 3.8: VPA Regression Analysis

A brief look at a residual plot (Figure 3.9) from the regression analysis shows no distinct patterns. There are only 24 points to the residual plot but the points appear scattered about randomly with both positive and negative values and no apparent outliers. This gives us confidence that the residuals look to be normally distributed and having a constant variance. Thus giving no reason to believe that the regression analysis is inaccurate.

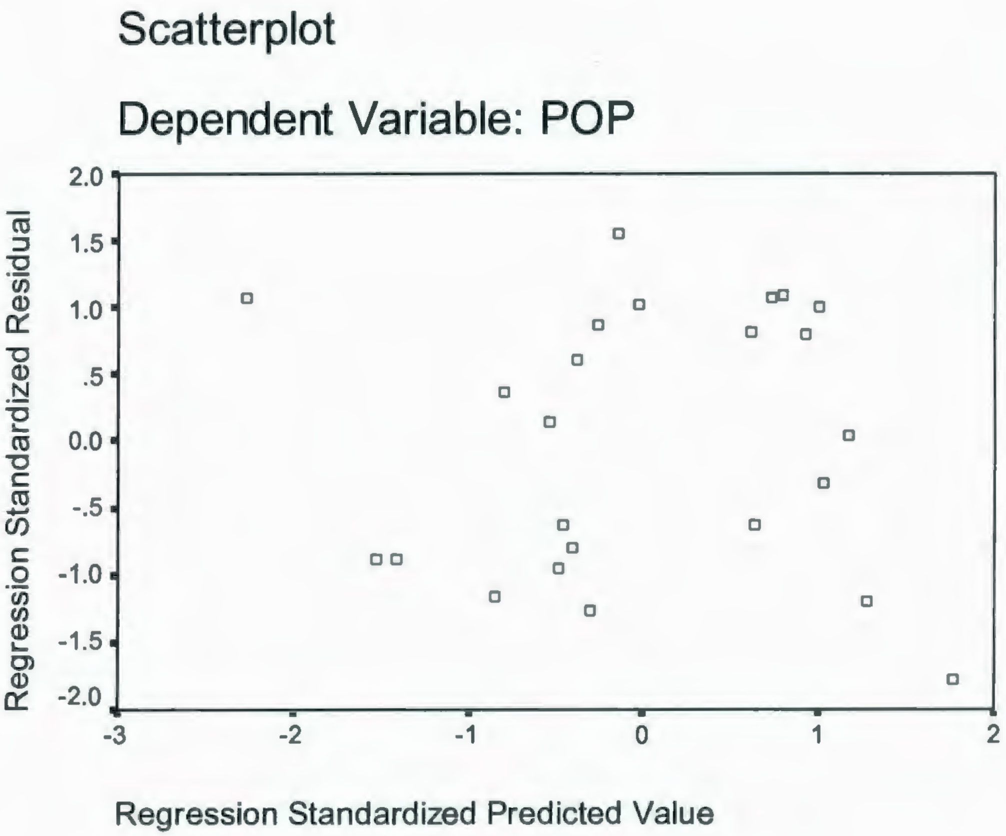


Figure 3.9: Plot of Residuals versus Predicted Values

## 3.5 Time Series Analysis

Essentially, the estimated population is a time series or a stochastic process made up of 24 data points, $x_1, x_2, \ldots, x_{24}$. So here we will investigate the population dynamics and attempt to fit an appropriate time series model.

The first thing to look at in any time series analysis is the plot of the original dataset to observe any important characteristics. Two important issues are whether the data appear to be stationary and does there appear to be any seasonal pattern to the data. For a process to be stationary, we require that the mean and variance of the data do not depend on the time, t, and the covariance be a function of the lag only. We can see from Figure

3.7 that the caribou population is clearly not stationary. The population in 1981 was just over 47,000 and increased annually until 1995 where it stood at over 116,000 animals and then declined each year after to about 77,000 in the year 2003. Therefore, the population size clearly depends on the time, *t*.

To account for this apparent nonstationarity, we took the first difference of the data,

$$\nabla^d x_t = (1 - B)^d x_t$$

$$= (1 - B)^1 x_t$$

$$= x_t - x_{t-1},$$

where d=1 and B symbolizes the backshift operator.

Seasonality only occurs in data that is collected at various times throughout the year, like monthly or quarterly, so since the observations are observed at only one specific time in each year, seasonality is not applicable in this case.

The plots of the auto-correlation function (ACF) and the partial autocorrelation function (PACF) of the first difference of the data suggest a first order auto-regressive model might be a good choice of time series model (Figures 3.10 and 3.11). This comes from the fact that the ACF decays slowly and the PACF cuts off after the first lag.

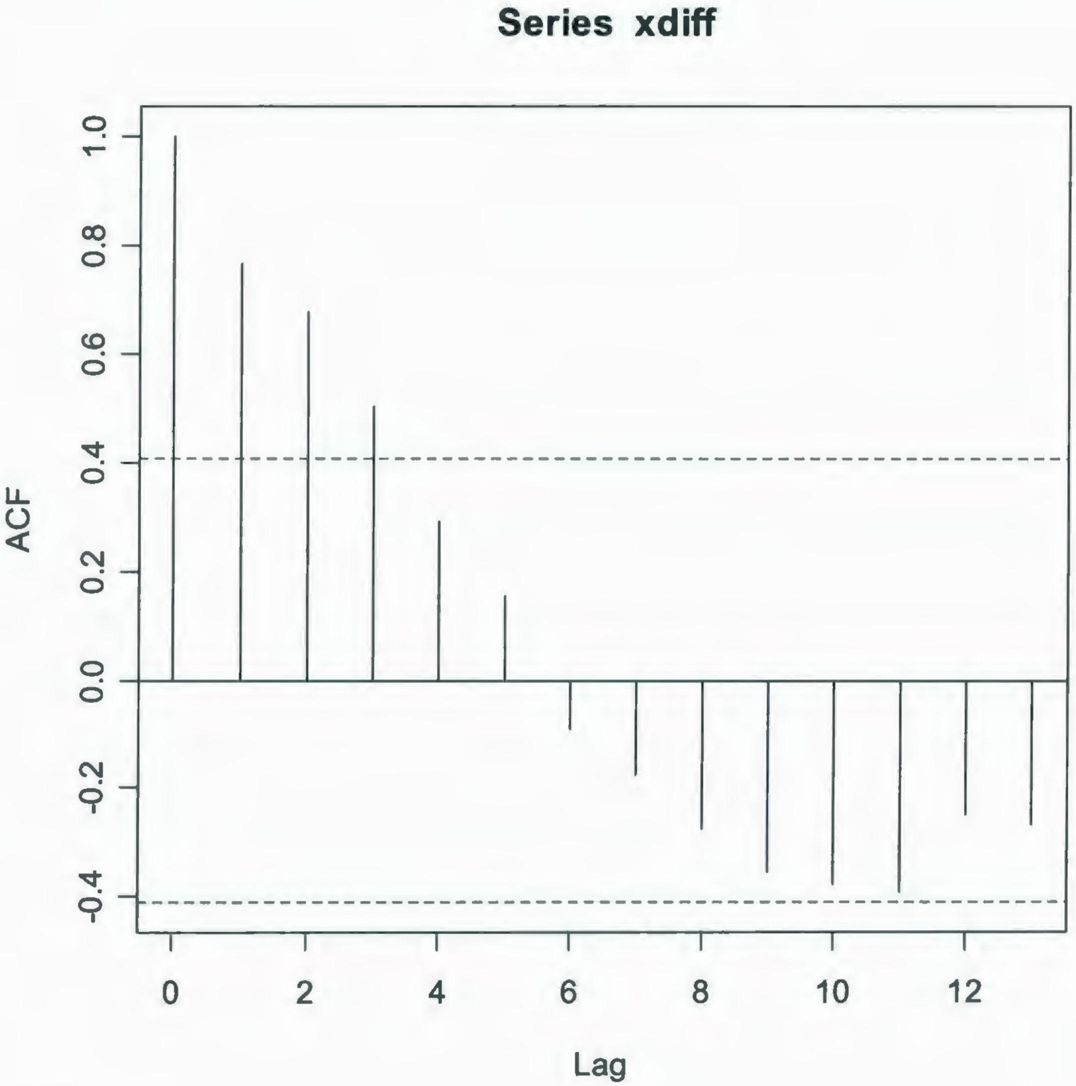


Figure 3.10: Auto-Correlation Function (ACF) of 1st Difference

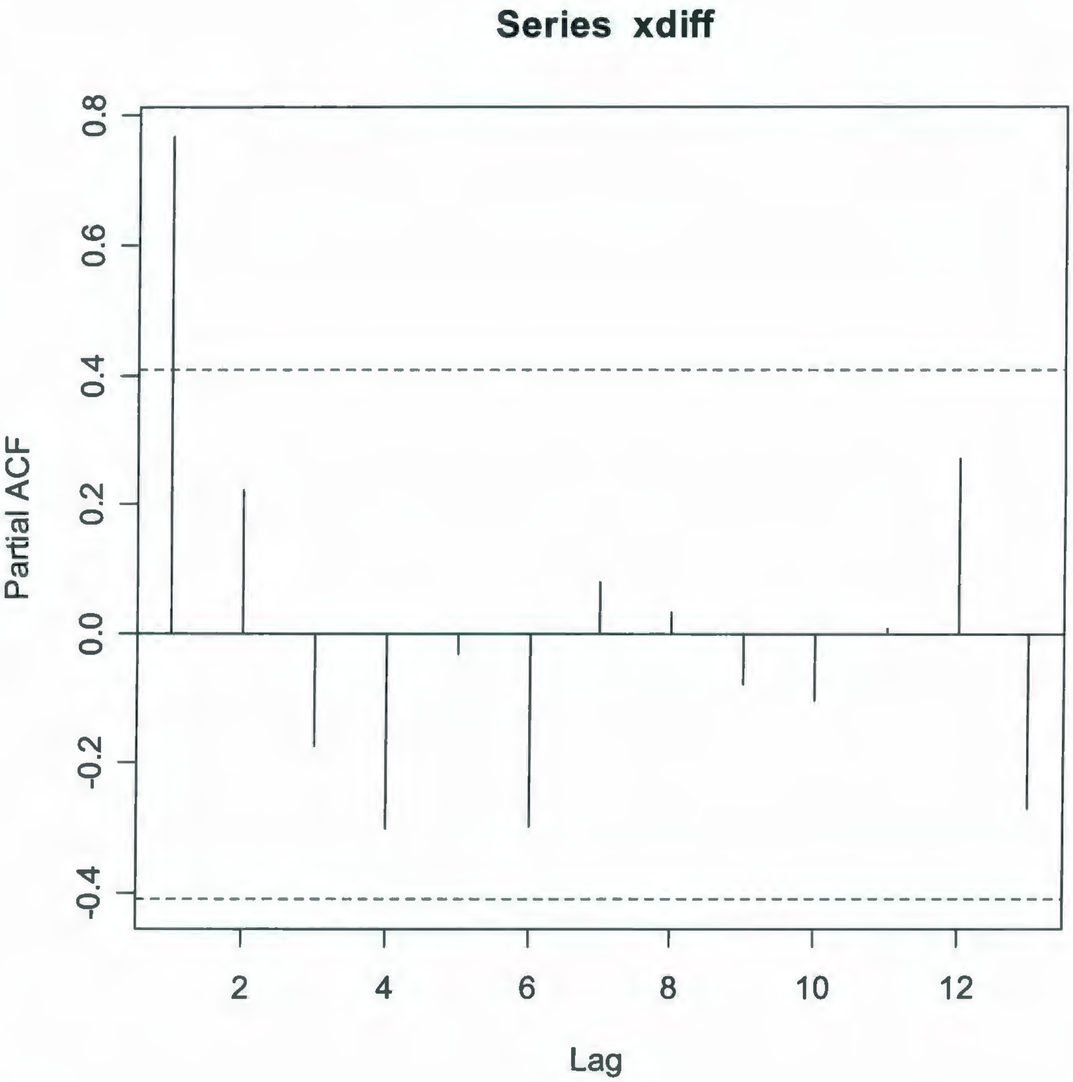


Figure 3.11: Partial Auto-Correlation Function (PACF) of 1st Difference

So based on the exploratory analysis of the population estimates, we have chosen to fit an ARIMA(1,1,0) model. For comparative purposes, an ARIMA(2,1,0) and an ARIMA(0,1,1) were also fit. Results from the model fits can be seen in the following table (Table 3.6):

**Table 3.6: Time Series Models for Population Size**

| Model | n | Coefficient(s) +/- SE(s) | AIC |
|---|---|---|---|
| ARIMA(1,1,0) | 23 | 0.7648 +/- 0.1229 | 446.34 |
| ARIMA(2,1,0) | 23 | 0.6042 +/- 0.2055<br>0.1979 +/- 0.2057 | 447.43 |
| ARIMA(0,1,1) | 23 | 0.6595 +/- 0.1727 | 455.4 |

Of the three models shown above, the model that seemed most logical in the first place, the ARIMA(1,1,0), gave the lowest Akaike's Information Criteria (AIC) value, so it appears to be the best fit. The 95% confidence interval for the model parameter was (0.5239, 1.0057), which does not include 0, so the parameter is significant.

Using the selected model, we can forecast where the population will go in the following years. Looking ahead 3 years showed population estimates of 80,636, 81,276 and 81,765 with 95% prediction intervals of (73,952, 87,320), (72,862, 89,690) and (72,487, 91,043), respectively. So the fitted model predicts that the population of the island herd will remain relatively stable over the next 3 years with a slight increase each year.

Another common approach to testing how good a model works is to remove the last two observations from the original data set. One then finds an appropriate time series model for this reduced data set, use it to predict the outcome at the next two time points and compare the predictions to what actually happened according to the observations that were removed.

So in this case, after removing the last two observations, we had a stochastic process made up of 22 observations. After comparing possible models for this data it was determined that the model of best fit according to the AIC criteria, was again an ARIMA(1,1,0) model. The following two predicted values of the population were 74,235 and 75,295 with 95% prediction intervals of (68,026, 80,444) and (67,099, 83,491), respectively. Both of these prediction intervals contain the two removed observations of 80,126 and 77,019 animals, adding confidence to the assumption of a good model fit.

Model diagnostics were investigated to determine if there appeared to be any problems with the model. The plot of the standardized residuals (Figure 3.12) did not show any real distinct patterns, which is what we hope to see. The ACF of the residuals (Figure 3.13) shows that all residuals outside of lag 1 are inside the limits of being considered white noise. Finally, the Ljung-Box-Pierce plot (Figure 3.14) shows that all p-values are outside of the rejection region meaning that the assumption of a good model fit under the null hypothesis is reasonable. Therefore, based on the model diagnostics, there is no reason to believe that the fitted time series model is inadequate.

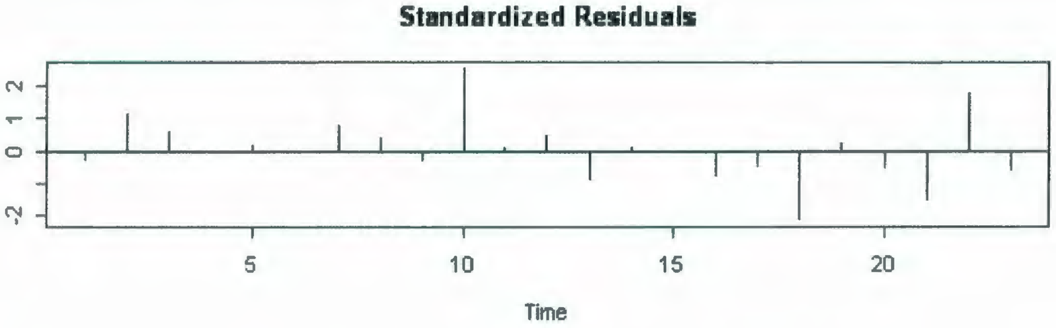


Figure 3.12: Plot of Standardized Residuals

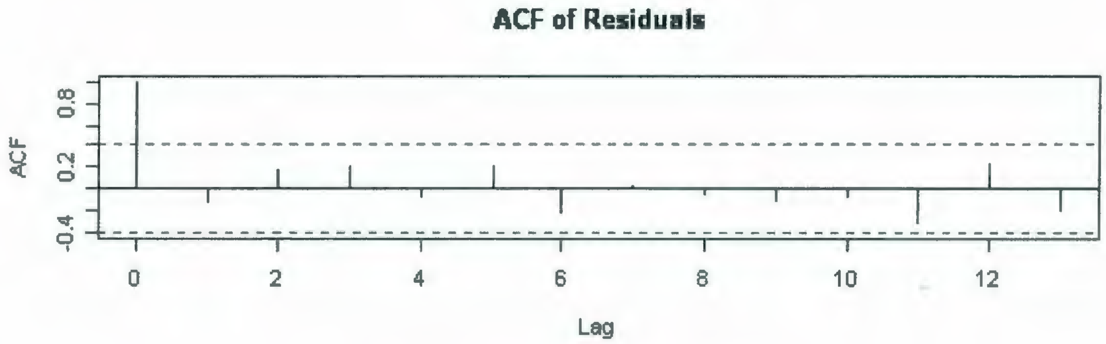


Figure 3.13: ACF of Residuals

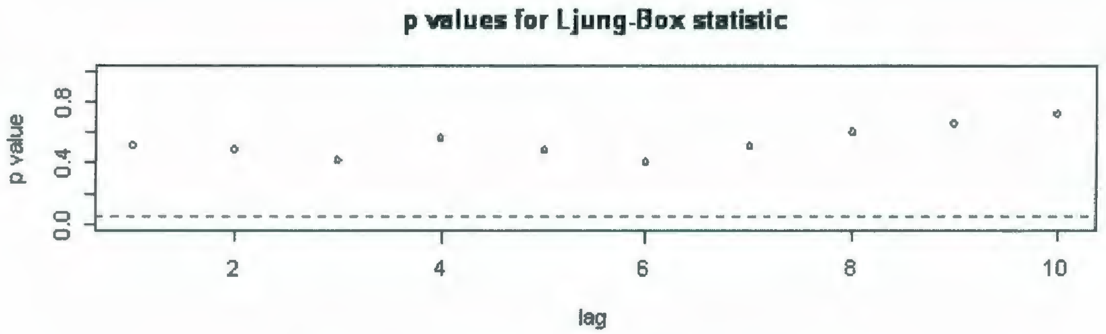


Figure 3.14: P-values for Ljung-Box Statistic

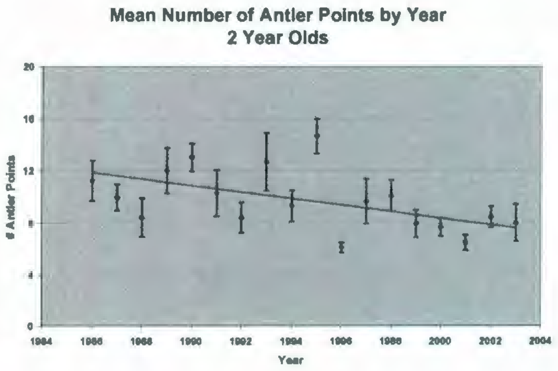
## 3.6 Trends in Antler Point Data

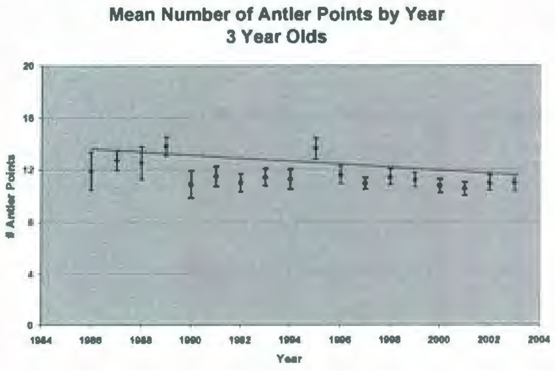
Although not related to VPA, it is of interest to investigate other aspects of population change in caribou. Another characteristic, other than the size of the herd, that would indicate the status of the caribou, is the antlers of the males. The size and number of antler points on a male caribou can be attributed to the type and amount of food they eat. A good diet would likely mean more antler points.

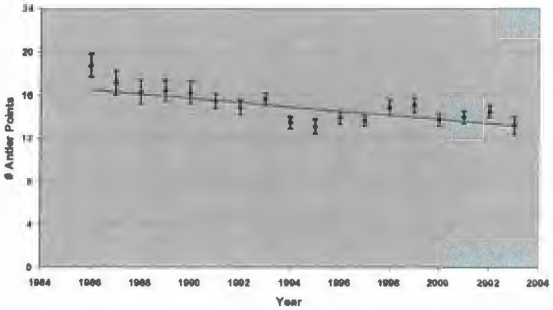
It would not make much sense to look at the average number of antlers points of kills by year because in years where there more mature animals, the average number of antler points would be expected to be higher. This would not give an accurate reflection of what we are interested in here as more antler points may simply mean older individuals and not healthier individuals. Therefore, we looked at the average number of antler points by age through time.

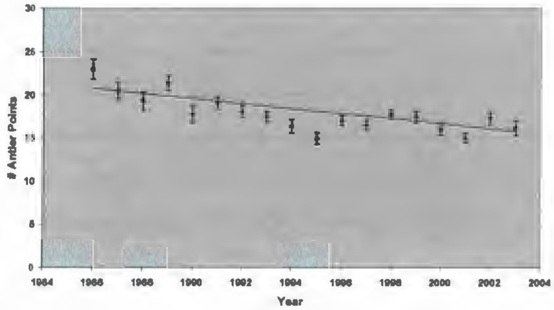
The number of antler points of harvested males was obtained from the hunter return questionnaires, which was then cross referenced with the licence numbers of the submitted hunter jawbone returns to get the number of antler points by age from the harvest. Unfortunately, the licence number on hunter returns were not recorded until 1986 so the time period of this analysis could only be performed from 1986-2003. Also, sample sizes were too small for animals aged 0, 1, 12 and 13+ so they were excluded.

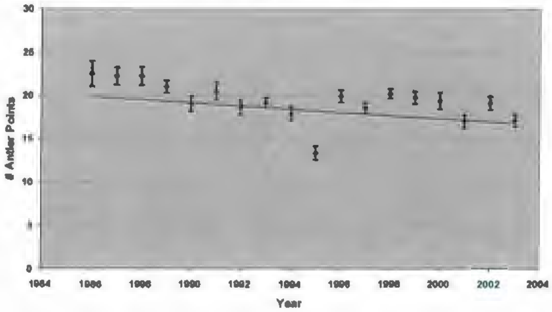
The charts that follow (Figure 3.15) show the mean number of antler points by year for each of animals aged 2 through 11 along with the associated standard error bars. Simple linear regression models are also shown as a red line where the model showed significance at the a = 0.05 level. Ages 8-11 did not show a significant linear trend as the variation between years was inconsistent and the standard errors were greater as the sample sizes were smaller. Each of ages 2 to 7, however, showed a significant linear trend from 1986 to 2003. In each case we saw a steady decline in the mean number of antler points through time.

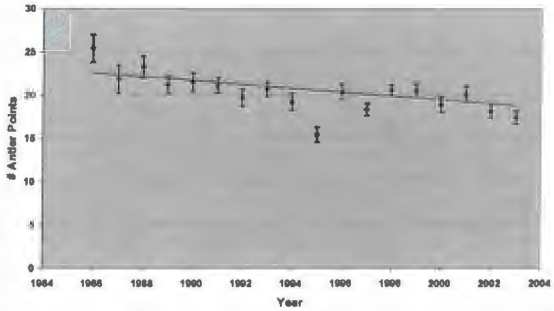


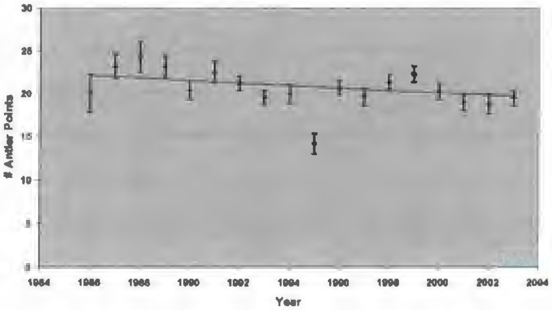


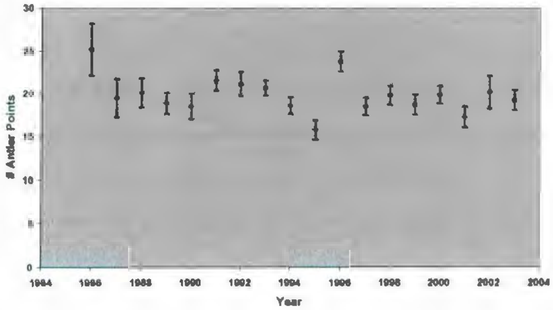

Mean Number of Antler Points by Year
4 Year Olds


Mean Number of Antler Points by Year
5 Year Olds


Mean Number of Antler Points by Year
6 Year Olds


Mean Number of Antler Points by Year
7 Year Olds


Mean Number of Antler Points by Year
8 Year Olds


Mean Number of Antler Points by Year
9 Year Olds

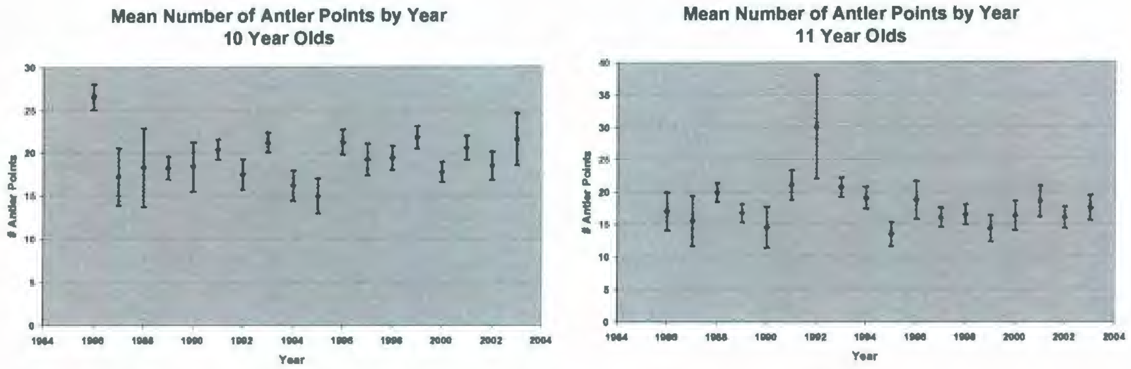


Figure 3.15: Mean Number of Antler Points by Year and Age

## 3.7 VPA Results Without Modifying Female Methodology

Initially, the analysis was carried out for females the same way it was done for males, using the standard VPA methodology. The results, however, turned out less than ideal and after thinking about the model and its assumptions, it was understandable why this was the case. Harvest rates for females is around 1% and so the assumption that 100% of all 13 year olds and older are taken in the harvest is likely not a valid assumption.

The results without the modification can be seen in Figure 3.16. On average the total population using this method was 53% lower than the total population observed after making the modification to the methodology for females. Of course, this difference is accounted for solely in the female portion of the population because the technique, and hence the results for males is exactly the same in both analyses. So the female population without altering the method, is on average 86% lower than what was found after making the change.

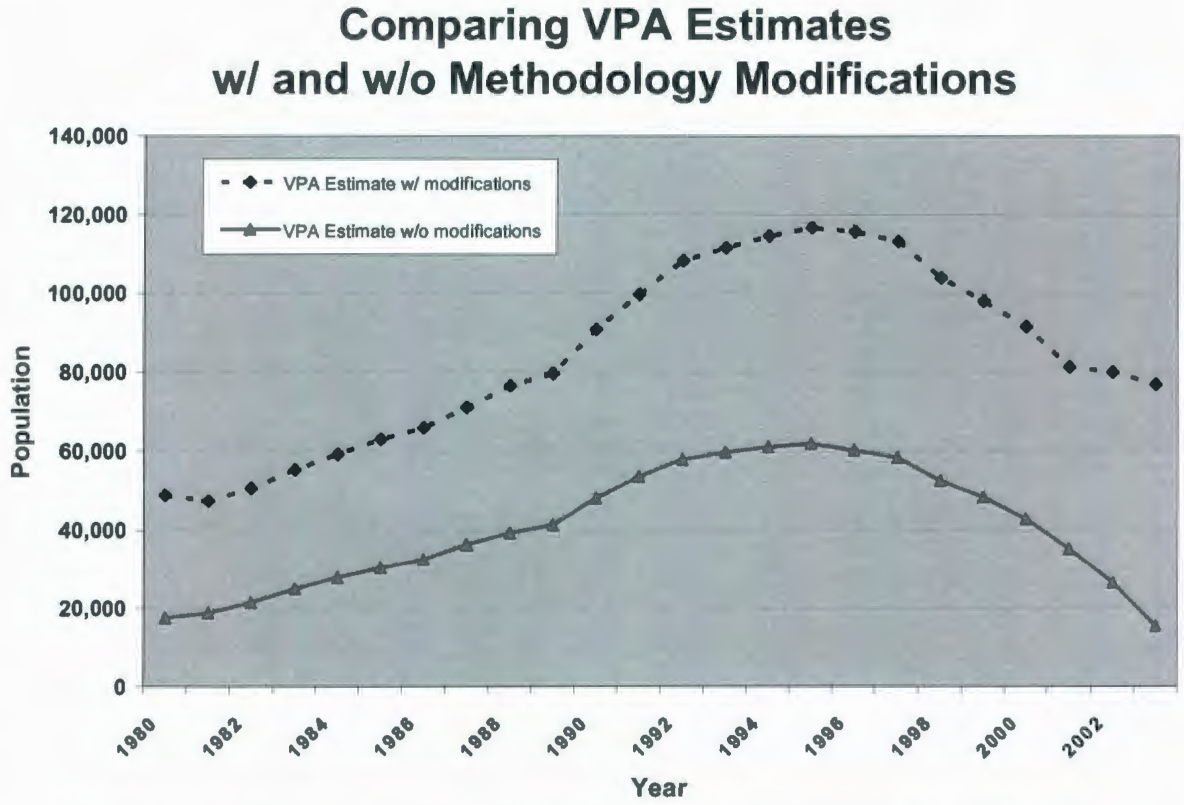


Figure 3.16: Comparing VPA Estimates w/ and w/o Methodology Modifications

There were two glaring points that led to the realization that something was wrong. First of all, the overall sex ratio ranged between 15% and 25% for females with an average of 18% between 1980 and 2003. We had already seen that the natural survival rates do not differ very much between the sexes. Add to this the fact that the number of males harvested in each year is drastically higher than that of females, and it was obvious that these sex ratios didn't make sense.

Further, the sex ratio for calves alone is known through classifications and other scientific methods to be a 50/50 ratio. The results here indicated a proportion of about 15% for female calves and 85% male calves, a far cry from the expected 50%. This is what led us to the decision to use a different approach for the female cohort. Simply equating the number of female calves to the number of male calves and reversing the VPA equations to get the proceeding female cohorts was the approach taken to achieve the desired goal.

## 3.8 Effects of Using a Different Terminal Age

There are no strict rules behind how to select the terminal age to be used in a virtual population analysis. The literature simply states that as long as the terminal age accounts for a small percentage of the total harvest, the results should not be too heavily affected. A threshold often used is that the terminal age account for less than 5% of the total harvest.

In this paper, we used a terminal age of 13, of which 13 year olds made up between 0.0% and 3.5% of the total harvest between 1980 and 2003 with an average of 0.8%. We could have also used age 11 as the terminal age as it constituted an average of 3.7% of the total harvest during the study period. It ranged, however, from a low of 1.5% to a high of 5.6% so that is the reason the threshold of 13 was chosen to be on the safe side. Figure 3.17 shows what the results are if we choose 11 as the terminal age.

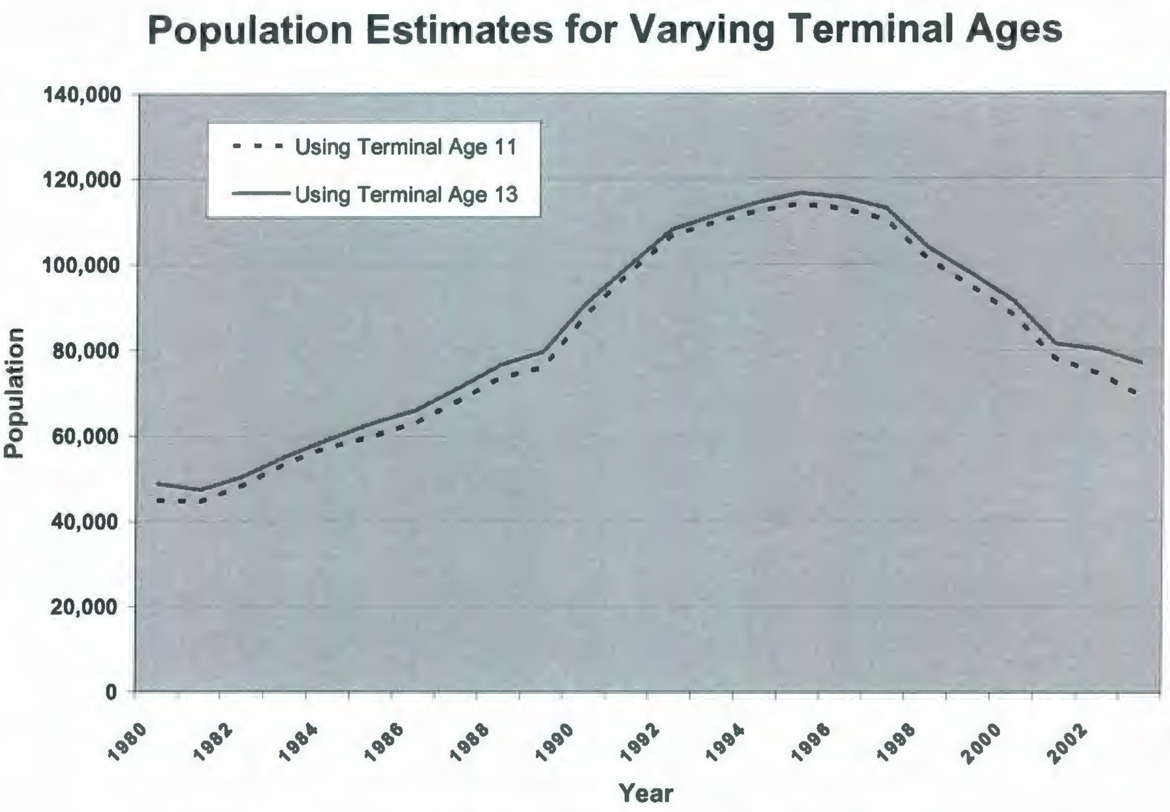


Figure 3.17: Population Estimates for Varying Terminal Ages

As can be seen from the figure, the difference going from 13 to 11 as the terminal age does not make a great deal of difference. The two are very close to giving the same results with the model using age 11 as the terminal age being, on average, 4% lower than the results using 13 as the terminal age. So as long as the terminal age used accounts for a very small percentage of the total harvest, then the results shouldn't vary a great deal.

# Chapter 4

# Conclusions

Virtual population analysis is a very useful method for estimating the demographic make-up of a wildlife population under the right circumstances. Though it is more commonly known to be a technique used for fish populations, others have shown that it can be adapted and applied to other species' as well.

The results here have proven to be a very viable option in the estimation of Newfoundland caribou herd populations. The results strongly agree with what has been seen previously for aerial survey or census counts. Although the overall population counts under VPA are a little higher than the census counts, the trends are strikingly similar. Having said that, census counts, as was mentioned earlier, do not give perfect results themselves due to their infrequency. So it may be argued that, under ideal conditions, virtual population analysis provide better estimates. One thing that is not arguable though, is that VPA is the far cheaper method.

Looking at the mean age of the caribou herd through time showed a comparable pattern to what happened with the population counts. The population increased from 1980 through 1995 and then declined from then on through 2003. The pattern was similar for the mean age, although in the opposite direction. The mean age first experienced a decrease from 1980 to 1991 and then increased back up to where in 2003 it was very close to where it stood in 1980.

We saw that using Kaplan-Meier survival estimates would not have made a considerable difference to the results we achieved through using Heisey-Fuller survival estimates. Both methods are widely used and acceptable although Kaplan-Meier is the more widely known in the statistics field. Also, using a terminal age of 11 instead of the chosen age of 13 did not have a great impact on the final estimates. The total population under the model of 13 as the terminal age produced results that were only slightly higher than under a model with 11 as the terminal age. Therefore, provided harvests in the terminal age account for a very small proportion of the total harvest, estimates should differ only marginally.

As with any analysis, the better the data quality, the more accurate and precise the results will be. We had to make a couple of adjustments here to account for some data gaps. Firstly, there was a five year gap in the radio telemetry data from 1998 to 2002. Unfortunately, this was at the time when calf survival became uncharacteristically volatile. So without this data, the calf survival rates at this time are less reliable than for other years. Also, with the female harvest rates being as small as they are, the higher the rate of return of questionnaires and jawbones the better.

There are a couple of areas in this particular case that could be worked on to strengthen the confidence we have in future results. The first is encouraging the importance to hunters of both their hunter return questionnaires and the jawbones. Most people don't realize that it is actually required by law that the questionnaires be returned to the Wildlife Division. Also, keeping a consistent dataset, where possible, on radio telemetry studies or any other datasets of value is imperative. Having said that, the process of this study has shown that using VPA for the purposes of Newfoundland caribou and really any game species were the appropriate data is available, is a very real alternative to aerial surveys or other population counting techniques.

# Bibliography

Baranov, F. I. 1918. On the question of the biological basis of fisheries. Nauchnyi Issled. Ikhtiol. Inst. Izv. 1:81-128.

Bart, J., and D. S. Robson. 1982. Estimating survivorship when the subjects are visited periodically. Ecology 63:1078-1090.

Caughley, G. 1974. Interpretation of Age Ratios. J. Wildl. Manage. 38:557-562.

Deriso, R. B. 1980. Harvesting strategies and parameter estimation for an age structured model. Can. J. Fish. Aquat. Sci. 37:268-282.

Filion, F. L. 1981. Importance of Question Wording and Response Burden in Hunter Surveys. Journal of Wildlife Management. 45 (4): 873-882.

Fry, F. E. J. 1949. Statistics of a lake trout fishery. Biometrics 5:27-67.

Fryxell, J. M., W. E. Mercer, and R. B. Gellately. 1988. Population dynamics of Newfoundland moose using cohort analysis. Journal of Wildlife Management. 52 (1): 14-21.

Fryxell, J., Falls, J.B., Falls, E.A., Brooks, R.J., Dix, L. and Strickland, M. 2001. Harvest dynamics of mustelid carnivores in Ontario, Canada. Wildl. Biol. 7:151-159.

Garrison, L. and J. Link. 2003. An Expanded Multispecies Virtual Population Analysis Approach (MSVPAX) to Evaluate Predator-Prey Interactions in Exploited Fish Ecosystems. Atlantic States Marine Fisheries Commission. 81pp.

Greenwood, M. 1926. The natural duration of cancer. Reports on Public Health and Medical Subjects. London: Her Majesty's Stationery Office. 33:1-26.

Hauge, T. M. and L. B. Keith. 1981. Dynamics of moose populations in northeastern Alberta. J. Wildl. Manage. 45:573-597.

Heisey, D. M. and Fuller, T. K. 1985. Evaluation of survival and cause-specific mortality rates using telemetry data. J. Wildl. Manage. 49 (3): 668-674.

Johnson, D. H. 1979. Estimating nest success: the Mayfield method and an alternative. Auk. 86: 651-661.

Kalbfleisch, J. D., and R. L. Prentice. 1980. The statistical analysis of failure time data. John Wiley & Sons, Inc., New York, N.Y. 321pp.

Kaplan, E.L., and P. Meier. 1958. Nonparametric estimation from incomplete observations. Journal of the American Statistical Association. 53:457-481.

Krebs, C. J. 1999. Ecological Methodology. 2nd ed. Benjamin Cummings, Menlo Park, California. 620pp.

Lassen, H. and P. Medley. 2001. Virtual Population Analysis - A Practical Manual for Stock Assessment. FAO Fisheries Technical Paper 400. 129pp.

Peterson, R. L. 1955. North American moose. Univ. Toronto Press, Toronto, Can. 280pp.

Pope, J. G. 1972. An investigation of the accuracy of virtual population analysis using cohort analysis. Intern. Comm. Northwest Atl. Fish. Res. Bull. 9:65-74.

Ricker, W. E. 1940. Relation of "catch per unit effort" to abundance and rate of exploitation. J. Fish. Res. Board Can. 5:43-70.

S.M. Nurul Amin and M. Zafar. 2004. Studies on Age, Growth and Virtual Population Analysis of *Coilia dussumieri* from the Neritic Water of Bangladesh. Journal of Biological Sciences 4 (3): 342-344.

Sergeant, D. E., and D. H. Pimlott . 1959. Age determination in moose from sectioned incisor teeth. J. Wildl. Manage. 23:315-321.

Solberg, E. J., B-E. Saether, O. Strand and A. Loison. 1999. Dynamics of a harvested moose population in a variable environment. Journal of Animal Ecology 68:186-204.

Solberg, E. J., A. Loison, J.-M. Gaillard and M.Heim. 2004. Lasting effects of conditions at birth on moose body mass. Ecography 27: 677-687.

Sparre, P. and S. C. Venema. 1998. Introduction to Tropical Fish Stock Assessment - Part 1: Manual. FAO Fisheries Technical Paper 306/1 Rev. 2. 408pp.

Trent, T. T., and O. J. Rongstad. 1974. Home range and survival of cottontail rabbits in south-western Wisconsin. J. Wildl. Manage. 38:459-472.

Ulltang, O. 1977. Sources of errors in and limitations of virtual population analysis (cohort analysis). J. Counc. Int. Explor. Mer. 37:249-260.

White, G. C., D. J. Freddy, R. B. Gill, and J. H. Ellenberger. 2001. Effects of adult sex ratio on mule deer and elk productivity in Colorado. J. of Wildl. Manage. 65:543-551.