

INCIDENTAL TAKE AND ENDANGERED SPECIES DEMOGRAPHY

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Doctor of Philosophy

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INCIDENTAL TAKE AND ENDANGERED SPECIES DEMOGRAPHY

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ABSTRACT

Incidental take is the permitted killing, harming, harassing or destroying habitat of an endangered species under endangered species law. Legislation assumes that endangered or threatened populations can somehow compensate for the potentially negative effects of incidental take. However if the species in question does not have the assumed capacity to compensate it is possible for incidental take to dramatically lower abundance and lead to decreasing population trends for a protected species. With this research I explored the concept of incidental take from an ecological and demographic stand point to evaluate the basis and application of the policy. I reviewed literature on incidental take, compensatory mortality, and harvest theory to elucidate the conditions under which incidental take might be sustainable from a demographic perspective. I also developed and used a predictive population model for Piping Plovers in the Great Plains to evaluate the potential effects of currently permitted take of eggs and chicks in the Missouri River on plover population viability. Lastly I examined the use of science and specifically quantitative population models in incidental take permitting decisions by reviewing biological opinions for Piping Plovers from throughout their range. Biological opinions are the official documents prepared and released by the US Fish and Wildlife Service detailing the ecology and biology of an incidental take action and determining whether and under what conditions incidental take can proceed.

Chapter 1

An introduction to incidental take, piping plovers and this dissertation.

With my dissertation research I examined the effect of incidental take on endangered species population dynamics and management. Incidental take is the permitted killing, harming, harassing or destroying habitat of a threatened or endangered species. Take is permitted under the US Endangered Species Act by Sections 10 and Section 7 of the Act and is regulated by the US Fish and Wildlife Service. However, there are take allowances under the endangered species protection laws of several countries including Canada and Australia. US legislation states that take can be permitted if the probability of the species' continued survival in the wild is not reduced; in other words, the take actions cannot "jeopardize" the continued existence of the species. The legislation and the supporting documentation does not explicitly define "jeopardy" or the concept of causing jeopardy. What happens if the species in question is already in decline? How could some additional mortality factor cause the jeopardy of a species that is in high jeopardy even with out incidental take?

I used predictive population modeling to examine the potential effect of permitted incidental take of piping plover nests in the Missouri River on population viability in the Great Plains. The piping plover is a small migratory shorebird species that was listed as threatened in the Great Plains in the mid 1980's. Piping Plovers breed throughout the Northern Great Plains. Females lay up to four eggs in a small scrape on the ground, those eggs take approximately 24 days to hatch and the precocial chicks take approximate 25 days to fledge (fly). The species requires vegetation free habitats with sand or gravel substrates and the birds nest on beaches at alkali wetlands, on riverine sandbars and on reservoir beaches on rivers behind dams. It was listed in large part due to widespread habitat loss or degradation throughout its range.

In the Missouri River, plover reproductive success is affected by water flow management by the US Army Corps of Engineers (USACE) through 6 major dams and reservoirs. The USACE manages water flow to enable barge trafficking in the lower reaches of the river, prevent flooding, generate hydro-electricity, enhance out-door recreation opportunities, and to benefit wildlife populations. These water management objectives are often in conflict. When the USACE releases water from the reservoirs during the piping plover breeding season rising water levels in the river reaches down stream floods plover nesting habitat and causes the destruction of nests. Alternatively if the USACE decides to retain water in the reservoir system for use later in the summer, rising water levels in the reservoirs will flood nesting beaches in the reservoirs and some level of nest destruction. To put it simply, the USACE's water management actions can lead to the destruction of piping plover nests which results in lower reproductive success and could possibly affect population dynamics and viability.

This take was permitted by the USFWS, but no demographic analyses were conducted to thoroughly evaluate the effect of take on piping plover recovery goals. Instead take levels were set and historic averages and maximums. This approach to establishing incidental take limits and permits raised several questions in my mind: 1) what affect do the currently permitted take levels have of piping plover population viability, 2) incidental take essentially applies a harvest paradigm to endangered species populations, but what if those population don't have the capacity to compensate for the harvest? and 3) what quantitative metrics typically go into making an incidental take management decision? Was using the historic levels of take to guide future take management really the best available science to managers during the permitting process?

To examine the effects of incidental take on population viability, I used, updated and improved an existing population model for Piping Plovers in the Great Plains developed by Mark Ryan and his graduate students since 1993. In that model, I applied take as a harvest of eggs or chicks that occurs immediately after laying or hatching. In this dissertation, I report on projected population growth, final abundances after 20, 30, and 40 years, and on the probability of quasi-extinction. As part of that modeling effort, I derived parameter estimates for fecundity, nest survival and chick survival using data provided by the USACE, the USFWS and by biologists with the Canadian Wildlife Service. With assistance from Dr. Joshua Millspaugh, Dr. Mark Ryan, and data from Casey Kruse and Greg Pavelka, I used a modified catch curve approach to estimate daily chick survival from 13 years of age based count data. This method is detailed in chapter 1 of my dissertation and the predictive population modeling is detailed in chapter 3.

In the second chapter I explore the concept of incidental take as a harvest applied to endangered or threatened species. I reviewed harvest theory and ask the important question of “Do endangered species have the capacity to compensate for incidental take?” The ecological capacity to compensate is an essential part of any successful harvest management. The language of the Endangered Species Act seems to imply that all wildlife species have the inherent ability to compensate for the negative affects of incidental take. I used simple population models for four federally protected species and examined the life history and annual cycle of Piping Plovers to look for opportunities for compensation. This chapter is written as a stand alone manuscript with Drs. Mark Ryan and Joshua Millspaugh is a co-authors.

The final chapter explores the question of what is the best available science in incidental take management. I evaluated the information resources used to write biological opinions concerning piping plover incidental take from across the species' range. Biological opinions are the formal documents that the USFWS writes to evaluate and determine the acceptability of an incidental take permit. I report on the types of resources referenced in the biological opinions and the diversity of those resources. I also closely examine the use of competing, published population models in the opinions to try and determine how the USFWS decided to rely on the results of one population model over another.

The policy of incidental take has not been widely addressed in an ecological context by the academic community. This work is one of the first attempts to develop a population model to explicitly address the potential effects of incidental take on a federally protected species. Predictive population models are perfectly suited for evaluating the potential effects of incidental take and models could be extremely useful in the permitting process. Furthermore, competing models of a species' population dynamics could be used in an adaptive management context to guide research and monitoring of incidental take. I hope that this application of a population model to a specific incidental take management proves useful to the USFWS in guiding piping plover recovery efforts and I hope that this modeling approach is used on other incidental take issues and for other taxa.

Hereafter, each chapter in this dissertation is written as a stand alone publication or manuscript. They are targeted for different journals and are therefore formatted for each of those journals. For example chapter 2 is written and formatted for Biological

Conservation, a European journal, and is therefore formatted according to that journals formatting requirements.

Chapter Two

Estimating chick survival for precocial birds using a catch curve and age
based count data

McGowan et al.
Estimating chick survival in precocial birds

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Estimating survival of precocial chicks in the pre-fledging period using a catch curve
analysis and count based age class data

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Abstract --- Estimating reproductive success for birds with precocial young can be difficult. We present an approach for estimating pre-fledging, chick survival using long-term, count based, age-class data, using piping plovers in the Missouri River as an example. We used a modified catch curve analysis, and data from 13 years for three five-day sampling periods near the middle of the breeding season. We used data from the US Army Corps of Engineers piping plover monitoring efforts in which field crews conducted weekly counts and aging of chicks between 1993 and 2005. Average daily survival estimates range from 0.895 (S.D. = 0.03) to 0.917 (0.02) depending on the sampling period. The probability of fledging ranged from 0.126 to 0.188. These estimates assume constant survival during the pre-fledging period and that there is no change in the age structure of the population. Our analyses were designed to minimize the probability of violating the methodological assumptions. This method of estimating chick survival may be useful for reproductive success research in a variety of precocial bird species when only count based age class data are available.

KEYWORDS: Catch Curve, Chick Survival, Count Data, Fecundity, Piping Plover, Reproductive success

Estimating fecundity or reproductive success is essential to understanding the demography and management needs of any wildlife species (Williams et al. 2001). In avian species, fecundity is often defined as the number of young fledged per breeding female (Ricklefs 1972). However, for bird species with precocial offspring that depart the nest within hours of hatching, estimating fecundity can be difficult. Detectability of pre-fledging birds is often low and determining how many chicks from one brood survived or even what brood belongs to which female is difficult (Bent 1929, Lukas et al. 2004). In these cases, estimating fecundity with any meaningful measure of variance is sometimes impossible.

Due to these difficulties, researchers often turn to estimating the probability of survival during the pre-fledging period (hereafter: chick survival) (Hitchcock and Gratto-Trevor 1997, Groen and Hemerik 2002, Colwell et al. 2007). Combining estimates of chick survival, with estimates of nest survival, clutch size, and re-nesting rate can provide a reasonable estimate of fecundity (Noon and Sauer 1992). Mark and recapture/resight or radio telemetry methods have been successfully used in conjunction with Cormack-Jolly-Seber models to estimate chick survival for some species of shorebirds (Hitchcock and Gratto-Trevor 1997, Groen and Hemerik 2002, Lukas et al. 2004, Ratcliffe et al. 2005, Colwell et al. 2007). But, the methods are data intensive and require substantial effort, time, and money to collect.

Recently Skalski et al. (2006) summarized many methods for estimating demographic parameters from age, sex structure, and count data that are much less detailed and less expensive to collect. Some of these estimation procedures might be

useful to avian biologists with access to extensive count data sets with limited choices for in field and analytical methods.

Here we present an alternative method for estimating chick survival for precocial birds species using a modified catch-curve analysis first described by Chapman and Robson (1960) and Robson and Chapman (1961). In this paper we estimate chick survival for piping plovers (*Charadrius melodus*) in the Missouri River system using age structure data collected by the US Army Corps of Engineers (USACE) between 1993 and 2005 (C. Kruse and G. Pavelka unpublished data). Piping plovers, small migratory shorebirds, are listed as Threatened in the Great Plains under the United States Endangered Species Act (USFWS 1985). The USACE monitors piping plover populations nesting the Missouri River system because of the effect that dam operations and water management have on nesting plovers (USFWS 2000). Up to this time the USACE has reported “fledge ratios” annually by dividing the number of observed fledglings at the end of the breeding season by the number of breeding females, counted near the middle of the breeding season (USFWS 2000). These estimates lack an associated estimate of variance and have limited value from a demographic modeling standpoint. Our estimates of chick survival based on the modified catch curve analysis could be beneficial to understanding the management needs of piping plover in the Great Plains especially when included in population viability analyses.

METHODS

The USACE monitoring efforts cover the Missouri River from the Gavin’s Point river reach starting at Ponca State Park in Nebraska, to the Fort Peck Reservoir in eastern Montana (USFWS 2000). Field crews conducted surveys for piping plover breeding

activity at all known nesting sites each week throughout the breeding season (approximately 20 May – 15 August) (C. Kruse and G. Pavelka, unpublished data). On each visit to a nesting site, field crews recorded the number of chicks observed in each of 5 age classes: 0 – 5 days old, 6 – 10 days old, 11 – 15 days old, 16 – 20 days old, and over 21 days old (C. Kruse and G. Pavelka, unpublished data). Age determination in the field was based on the known age of broods or on the size of chicks at the time of observation. There are documented relationships between age and mass or size of chicks in shorebirds (Hussell and Page 1976, Cairns 1982, Miller and Knopf 1993).

The catch-curve survival approach estimates survival based on the number of individuals in each age class that are caught in a harvest (Chapman and Robson 1960, Robson and Chapman 1961, Skalski et al 2006). This method was initially developed to use data from fisheries harvests to estimate survival and inform fisheries management (Skalski et al. 2006). The method has seven assumptions as stated by Skalski et al. (2006): 1) there is a stable age structure, 2) the population is stationary, 3) all animals have an equal probability of selection (equal detectability), 4) the sample is representative of the population of interest, 5) the fates of all animals are independent, 6) all ages are recorded accurately, and 7) Survival probability is constant across all age classes during the sampling period. We developed a modified catch curve analysis to apply to the Army Corps of Engineers database to account for these assumptions.

The data we used were not harvest data but observations of living individuals that were not individually marked. We created three “capture” periods that were 5 days long to represent a harvest period. The three “capture” periods were 26 June – 30 June, 1 July – 5 July and 6 July – 10 July. We started by identifying the approximate mid-point

of the breeding season (1 July – 5 July), and then added one capture period before and after that mid-point window. We selected the capture period length of five days to avoid double sampling of specific sites (visited on an approximate 7 day cycle) and to avoid double sampling individuals that could grow from one age class to the next within one capture period. The five day “capture” period design was intended to reduce the risk of violating assumptions one, two, and seven as listed above. We present survival estimates for each year individually, and for the mean of all 13 years. We analyzed individual years and overall averages rather than pooling data across years because the variance estimates were unrealistically small for the pooled data survival estimates. We arbitrarily used three capture periods to compare the survival estimates for different time periods in the season.

We tallied the number of individuals observed in each age class during that five day period. We calculated survival from these data according to Skalski et al. (2006) using:

$$\hat{S} = T / (n + T-1)$$

where T is the sum of all the ages of all the individuals in the sample, and n is number of individuals in the sample. To calculate T we assigned the median age to all the individuals in an age class. For example, all individuals in the 0 – 5 day age class were assigned an age of 2.5 days and those in the 6 – 10 day age class were assigned an age of 8 days. That procedure assumes that the average age of all the chicks in an age class is equal to the median. If the average age of the individuals in an age class is very different than the median value, the resulting survival estimates will be inaccurate.

To calculate the standard deviation of the survival estimates for each year we used the delta method presented by Chapman and Robson (1960):

$$SD(\hat{S}) = \sqrt{\hat{S} \times (\hat{S} - (T - 1)/(n + (T - 1)))}$$

We raised the daily survival estimates to the power of 20 to predict the probability of survival to fledging; it takes approximate 20 days for a piping plover chick to fledge (Haig and Elliot Smith 2004). To approximate a 95% C.I. for the estimate of survival to fledging we raised the upper and lower bound of the daily survival rate to the power of 20 as follows.

$$95\% \text{ C.I. for the probability of fledging} \approx (\hat{S} \pm (1.96 \times SD(\hat{S})))^{20}$$

This procedure mimics the method recommended by Hensler and Nichols (1983) for generating a 95% C.I. for a Mayfield estimate of hatching success (Mayfield 1961).

RESULTS

Over the 13 year period field crews observed a total of 2376 chicks between 26 June and 30 June, 1931 chicks between 1 July and 5 July, and 2594 chicks between 6 July and 10 July. The average daily chick survival ranged from 0.895 in the first capture period to 0.917 in the third capture period (Table 1). The estimates of daily survival and the probability of fledging increased from the first to the second and to the third capture periods in a near linear fashion (Fig. 1). However, the 95% confidence intervals for the average probability of fledging estimates for each capture period greatly overlapped (Fig. 1). The probability of fledging ranged from 0.11 to 0.18 (Table 1), meaning that between 11 and 18 percent of the chicks that hatched survived to 20 days of age.

The catch curves of the individual years generally showed indistinct patterns, but the sample sizes in some years were quite small (Table 1). Daily chick survival estimates

Table 1: The daily probability of survival, standard deviation of daily survival, and the probability of survival to fledging for piping plover chicks in the Missouri River between 1993 and 2005. Estimates were calculated using a modified catch curve analysis for three 5-day time periods.

Year	n ^a	T ^b	Survival	SD	Fledging ^c
26 - 30 June					
1993	28	364	0.931	0.049	0.239
1994	51	388	0.886	0.045	0.089
1995	25	189	0.887	0.065	0.092
1996	9	54	0.871	0.119	0.063
1997	12	52	0.825	0.114	0.022
1998	88	936	0.915	0.030	0.169
1999	36	402	0.920	0.046	0.188
2000	335	4007	0.923	0.015	0.202
2001	218	2219	0.911	0.019	0.155
2002	216	1583	0.880	0.022	0.078
2003	474	4190	0.899	0.014	0.118
2004	475	4070	0.896	0.014	0.110
2005	409	3506	0.896	0.015	0.111
Average			0.895	0.028	0.126
1 - 5 July					
1993	20	257	0.931	0.058	0.240
1994	13	76	0.864	0.099	0.053
1995	11	113	0.919	0.086	0.183
1996	22	142	0.871	0.073	0.063
1997	0	0			
1998	42	330	0.889	0.049	0.096
1999	34	404	0.924	0.046	0.208
2000	134	1537	0.920	0.023	0.190
2001	291	3217	0.917	0.016	0.178
2002	554	6685	0.924	0.011	0.204
2003	347	4120	0.923	0.014	0.199
2004	252	2265	0.900	0.019	0.122
2005	211	2383	0.919	0.019	0.185
Average			0.909	0.022	0.160
6 - 10 July					
1993	29	269	0.906	0.055	0.138
1994	26	155	0.861	0.069	0.050
1995	15	183	0.929	0.069	0.229
1996	6	63	0.926	0.117	0.217
1997	15	187	0.930	0.068	0.236

Table 1: Continued

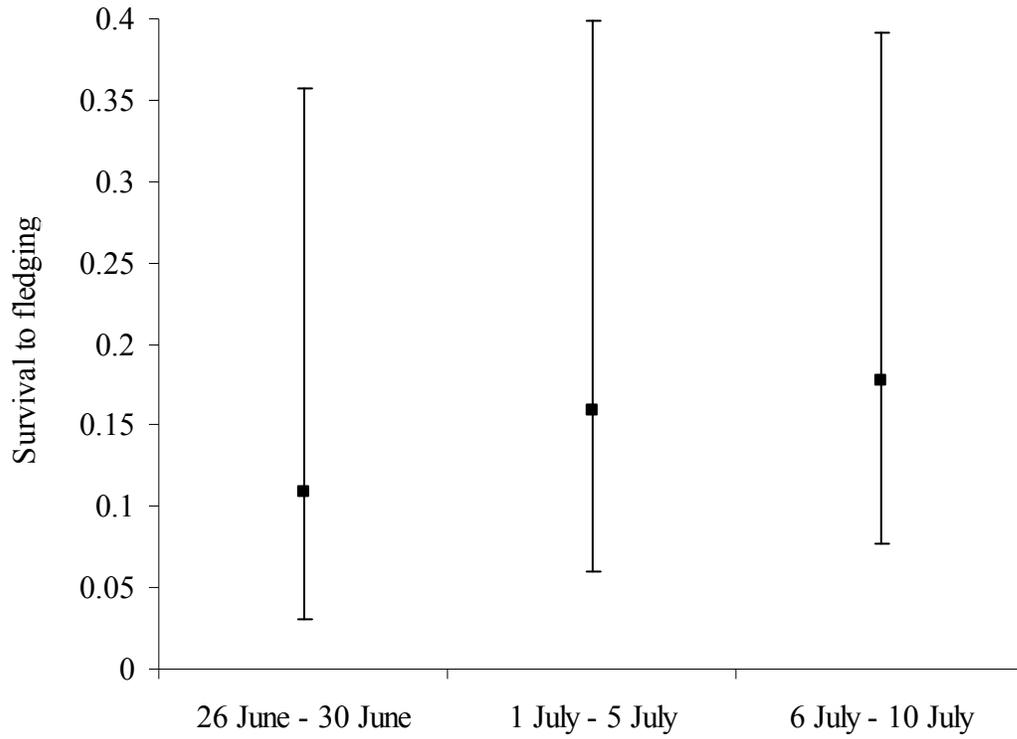
Year	n ^a	T ^b	Survival	SD	Fledging ^c
1998	107	1228	0.921	0.026	0.191
1999	147	1366	0.903	0.024	0.131
2000	201	2568	0.928	0.018	0.223
2001	326	3350	0.912	0.016	0.157
2002	322	4129	0.928	0.014	0.224
2003	455	5781	0.927	0.012	0.220
2004	470	5555	0.922	0.012	0.198
2005	475	6110	0.928	0.012	0.224
Average			0.917	0.019	0.188

^a The number of chicks observed

^b The sum of all the ages of all the chicks observed (in days)

^c The probability of fledging calculated by raising the daily survival to the power of 20.

Figure 1: Comparison of probability of survival to fledging estimates and the 95% confidence intervals for piping plover chicks in the Missouri River system from 1993 to 2005 between 26 June – 30 June, 1 July – 5 July, and 6 July – 10 July.



during the 26 – 30 June ranged from 0.825 in 1997 (n = 12) to 0.931 in 1993 (n = 28). For the 1 – 5 July period daily chick survival ranged from 0.864 in 1994 (n = 13) to 0.931 in 1993 (n = 20). For the 6 – 10 July period daily chick survival ranged from 0.861 in 1994 (n = 26) to 0.928 in 2000 (n = 201) and 2005 (n = 475). There were consistently more individuals observed in the years from 2000 to 2005 than were observed in the 1990's. There was no pattern to survival estimates across years or sampling periods. There were few statistical differences amongst the annual estimates, in part because the confidence intervals for the survival estimates in the 1990's were very large due to small sample sizes.

DISCUSSION

To our knowledge, this analysis is the first attempt to estimate survival of precocial bird chicks using a catch curve approach. However, count based survival estimation methods have recently been used in ornithological research (Beissinger and Peery 2007). The advantage is that the procedure enables researchers to estimate survival and variance using count data routinely collected by biologists and does not require individual animals to be marked. These results were obtained by utilizing a heretofore largely underused data base maintained by the USACE. A further advantage is that these data were relatively inexpensive to collect in time, money and labor; especially when compared to the effort needed to collect data for a radio-tracking or mark recapture analyses. Our method also allows for spatially broad scale investigations that are often valued by biologists, whereas individual marking studies provided detailed information about relatively small geographic regions. The data set we used covered 13 years and ranged from eastern Montana to north-eastern Nebraska. The use of count data

approaches offer opportunities to collect data over wider regions for less effort and cost than more labor intensive procedures that require making individuals (Skalski et al. 2006). These techniques have a long history of use in fisheries and wildlife science (Skalski et al. 2006) and we believe offer an option to mine existing data bases for demographic information as we have done in this study.

Despite these advantages, there are potentially limiting assumptions to using this approach in avian research and management (Chapman and Robson 1960, Skalski et al. 2006). The method assumes that the age structure of the population does not change over time (Chapman and Robson 1960, Skalski et al. 2006). However, we attempted to design the sampling periods to account for this stable age structure assumption. By making the sampling periods only five days, we can assume that the age structure does not change within one capture period. Factors such as weather, flooding, and water management in the Missouri River might affect average clutch initiations dates, or might cause high nest failures for first clutches (Espie et al. 1999, USFWS 2000), thus resulting in a younger chick population or an older chick population in some years during the selected capture period. However, we suggest that with 13 years of data our overall mean estimates capture that variation and incorporate it into the survival and variance estimates. Another assumption of the catch curve approach is that survival is constant over time. Again, the five day sampling period design means that the survival probability has to be constant within that five day period which seems like a reasonable assumption.

The other assumptions of this method (Robson and Chapman 1961, Skalski et al. 2006) are more easily met by the data set. For example, it is reasonable to assume that the chicks were aged accurately, because the age estimates were based on approximate

hatch dates for known broods and on size of the chicks. There are strong correlations between body size and age of chicks for several species of shorebirds (Hussell and Page 1976, Cairns 1982, Miller and Knopf 1993). If aging is inaccurate the survival estimates will also be inaccurate. Over estimating ages will result in survival estimates that are biased high and under estimating ages will result in survival estimates that are biased low. The method assumes that the population is stationary in size during the study (Robson and Chapman 1961, Skalski et al. 2006). The five day sampling periods were, in-part, designed to alleviate the constraints of this assumption.

Our application of the catch curve method to precocial chick survival estimation requires two further assumptions. The first is that detection probability of individuals is equal across all age classes. Unequal detectability across age classes will bias survival estimates. This assumption is not likely true for most precocial species because the older, more mobile chicks are bolder and more likely to be seen. It is important to modify searching techniques so that detection probabilities do not bias the results. In our case, field crews searched for older chicks (> 10 days) by scanning sandbars and beaches using binoculars from a distance and they searched for younger chick (< 10 days), by carefully walking through the breeding areas and looking for chicks hiding in the gravel and vegetation. The second assumption is that there is population level hatching asynchrony, so that there will be enough age differentials within the chick population to carryout the catch curve calculations. If all the chicks in the population under study hatch within a 5 day period, our use of the five day “capture” periods will not effectively estimate survival. The sampling design should consider the specific breeding biology of the species being studied.

The end result of these analyses is one survival estimate for chicks from the point of hatching to the point of fledging, which implies a constant survival value over that time period. Recent nest survival analyses have shown that daily nest survival varies with both nest age and with date (Dinsmore et al. 2002, Shaffer 2004). Evidence suggests that chick survival probability also changes with chick age (Colwell et al. 2007). As chicks age thermal stress becomes less of an issue, predation threats change, and so do energy requirements (Schekkerman and Visser 2001, Haig and Elliot-Smith 2004, Colwell et al. 2007). Our analyses did not attempt to account for these potential violations of the constant survival assumption. We cannot determine whether mortality is greater during the early or late stages and we cannot determine what factors affect survival at different stages of chick growth. However, we do assert that these estimates of survival represent the average daily survival of individuals throughout the pre-fledging period.

Our survival estimates are similar to, though slightly lower than, estimates reported by LeFer et al. (in press) who used a modified Mayfield approach (Mayfield 1961) to estimate survival for a small number of chicks in the Missouri River. Our results show a slight increase in survival and fledging probability from the first to the second and to the third capture period (Fig. 1). This may reflect some real change in survival probability over the season, but it might also reflect mere sampling differences in the capture periods. For example, the second period (1 July – 5 July) spanned a national holiday on which USACE field crews did not work. There were > 400 fewer chicks observed in this capture period than in the other two periods; in 1997 field crews did not observe any chicks in any age class during the second capture period. Considering non-

biological factors, such as the July 4th national holiday, when designing capture periods for this type of analysis, might have important impacts on survival estimates.

Another factor to consider is, if the surveys are carried out too soon, the survival estimates might be biased low and if carried out too late in the season, survival estimates might be biased high. In our data, as the breeding season progressed a greater proportion of the observed chicks were older chicks, because fewer new 1-5 day old chicks were being added to the population. In 2005, 51.2% of the nests monitored hatched on 25 June or earlier in the season, and 62.0 % of the nests had hatched by 1 July (C. Kruse and G. Pavelka unpublished data). At the start of the third capture period in 2005, 62% of the chicks in our study had already aged past the first age class. As fewer and fewer chicks are added to the population proportionally more of the chicks observed are in the older age classes, making survival probabilities appear to be higher. With this in mind it is important to select an appropriate capture period in which to survey the chick population. In retrospect an ideal capture period for use with these data would probably fall between 28 June and 2 July.

Despite the potential limitations to this methodology, as discussed, the use of catch curves to estimate survival of pre-fledged chicks for precocial bird species may have notable benefits. The estimates that we have presented here could be combined with nest survival data and used in a piping plover population viability analysis (Noon and Sauer 1994). Though the estimates come with caveats and should be used with appropriate caution, this approach could be used to inexpensively estimate chick survival for any precocial bird species. We also envision application of this method to any situation where short field surveys are conducted to count and age chicks present in the

population regardless of developmental mode (ie: colonial seabirds). It may also be applicable for annual survival estimates for species where individuals can be aged based on appearance and plumage patterns such as the larger gull species. However, as Conroy (2006) cautioned, researchers should not use this type of count based methodology “in lieu of other, more robust approaches.” Researchers should continue using use robust approaches when feasible (William et al. 2001). However, when only count data are available and interest is at a broad spatial scale, these methods provide an alternative that might offer useful insight into demographic properties provided assumptions are reasonable and robust to departures. Furthermore, large sets of count data, collected over numerous years (the type of data and many management agencies have collected for decades and remain largely underused) are potentially ideal for this type of survival analysis approach (Skalski et al. 2006)

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Chapter Three

Do endangered species have the capacity to compensate for incidental take?

Do endangered species populations have the capacity to compensate for incidental take?

Running head: Compensating for Incidental Take

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Abstract: Endangered species legislation allows for limited take of habitat or individuals of a protected species, as long as that take does not increase the “jeopardy” for that species. Inherent in these laws is the assumption that protected populations have the capacity to compensate for permitted take actions. Managers must be sure to explore whether a species can buffer incidental take through compensatory demographic responses. To demonstrate the extent to which endangered species might be affected by incidental take we present a case study of the Great Plains population of piping plovers and investigate that species’ capacity for compensation. Take of piping plover eggs is permitted in major river systems and we focus on the Missouri River, where managers have set take allowances based on historic levels of take. There is little evidence to support density-dependent mortality of juveniles or adults, an important component of compensatory mortality, and limited clutch size and low observed renesting rates, reduces the possibility of compensatory natality. To further explore the consequences of incidental take, we use simple demographic models for Yellowstone grizzly bears (*Ursus arctos*), loggerhead sea turtles (*Caretta caretta*), greater sage-grouse (*Centrocercus urophasianus*) and piping plovers (*Charadrius melodus*). We demonstrate that when compensatory mechanisms do not exist, incidental take can have important negative affects on endangered populations. Following the precautionary principle, managers should consider the capacity of a species to compensate for permitted actions and take steps to create compensation opportunities through mitigation and protection rather simply minimizing incidental take.

Key Words: compensatory mortality, endangered species, harvest management, incidental take, piping plovers

Endangered species laws of industrialized nations prohibit the killing, harassment, or habitat destruction of any species protected under those laws. The United States' the Endangered Species Act (US-ESA) provides for exceptions to those prohibitions through "incidental take" provisions in Section 10. It states "The Secretary may permit...any taking otherwise prohibited by section 9(a)(1)(B) if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity"(US-ESA of 1973. U.S. Code 16, section 668d). The Secretary of the Interior permits take if "the taking will not appreciably reduce the likelihood of the survival and recovery of the species in the wild" (US-ESA of 1973. U.S. Code 16, section 668d). The concept of incidental take is widely used in endangered species legislation of other countries. In the European Union, the Bern Convention on the conservation of the European wildlife and natural habitats prohibits destruction of protected species but permits exceptions in order "to prevent serious damage to crops, livestock, forest, fisheries, water and other forms of property..." as long as "the exception will not be detrimental to the survival of the population concerned" (E.U. Convention on the conservation of European Wildlife and natural habitats, 1981, Bern, 19.IX.1979, Article 9). The Australian Environment Protection and Biodiversity Conservation Act (Australian Environment Protection and Biodiversity Conservation Act, 1999 No. 91) and the Canadian Species at Risk Act (Canada Wildlife Act 1985. Revised Statues of Canada, chapter W-9, section 1) contain similar language for exceptions to endangered species protections. Despite the prevalence of incidental take allowances in endangered species legislation there have been few examinations of the incidental take concept in published academic literature. We focus on the US-ESA because there is an abundance of government publications on the implementation of this

policy (eg: USFWS, 1998). But, the concepts presented are applicable to all endangered species laws that contain incidental take allowances.

Under the US legislation, private citizens request permits to affect listed species via habitat conservation planning (Beatley, 1991; Aengst et al., 1997), whereas public agencies work with the U.S. Fish and Wildlife Service (USFWS) under the rules of Section 7 of the Act. In the USFWS Section 7 consultation handbook, managers are instructed to determine if the proposed actions would result in “increased jeopardy” of the listed species (USFWS, 1998). Inherent in the concept of “incidental take” is the notion that endangered populations can withstand a limited increase in mortality or a decrease in reproduction without affecting the chances for recovery.

Incidental take is essentially the concept of compensatory mortality commonly utilized in developing strategies for the harvest of natural populations (Nichols, 2000; Williams et al., 2001; Poysa et al., 2004). Even in situations where take permitting applies to habitat destruction, population demographics are affected in ways that must be compensated for. Compensatory mortality theory predicates that populations annually produce more individuals than the environment can sustain and that those excess individuals will be naturally culled at some density-dependent bottleneck (Troost, 1987; Nichols, 2000; Williams et al., 2001). Compensatory theory assumes that harvest mortality alleviates natural density-dependent mortality and that population size and growth are unaffected by some level of harvest (Troost, 1987; Nichols, 2000). Population level compensation can theoretically act via increases in reproductive success or decreases in juvenile or adult mortality (Nichols, 2000; Williams et al., 2001).

The concept of compensatory mortality in wildlife management seems to work well for most game species in harvest management scenarios. Game species tend to have high reproductive rates and it seems reasonable that the compensatory mortality paradigm might apply to those populations because of correspondingly high juvenile mortality (Baldassarre and Bolen, 1994; Williams et al., 2001). The relationship between hunting mortality and annual survival has been studied extensively for mallards (*Anas platyrhynchos*) in North America (Burnham et al., 1984; Trost, 1987; Nichols, 2000). Burnham et al. (1984) analyzed band recovery data for mallards between 1950 and 1979, and reported that the non-hunting mortality rate decreased in years when hunting pressure was high. This suggests that hunting mortality was compensated for by reduced density-dependent mortality factors. In a later analysis, Smith and Reynolds (1992) compared changes in mallard survival during periods of high and low hunting mortality and found evidence supporting additive mortality in this species. These results suggest that a population's capacity to compensate for a harvest varies over time.

Applying the compensatory mortality paradigm to endangered species management sounds like a well balanced compromise between the needs of economic development and rare species conservation. However, compensatory mortality and natality are theoretical concepts that do not apply to all species in the same way (Boyce et al., 1999; Xu et al., 2005). Compensation probably varies across species, and years and likely only exists for species with high reproductive capacity (Kokko et al., 2001; Poysa et al., 2004). Generally, species that are classified as threatened or endangered do not have excess reproduction but rather have high adult survival and low annual productivity (Rabinowitz, 1981; Purvis et al., 2000).

We believe that when considering the potential extinction of a species it is most appropriate to follow the precautionary principle (Kriebel et al. 2001) and prove that a species has the capacity for compensation rather than assuming it does. Central questions related to take of endangered species should be ‘Does the harvest management paradigm hold true for such a species?’ and ‘Does this species have the reproductive capacity or density-dependent mortality to buffer the effects incidental take?’ This is an especially important set of questions for species with declining population trends where the opportunities for compensation would be highly limited. In this paper we discuss the implications of applying a compensatory mortality paradigm to endangered populations. We present a case study examining the ecology and annual cycle of piping plovers (*Chardrius melodus*) in the Great Plains and show that despite currently permitted take occurring, there is little room for compensatory mechanisms to occur in this species. We further explore the potential affects of incidental take using simple population models of protected species with four different life history strategies and we impose a take component on each population. The models show how each life history strategy performs under the pressure of incidental take if mortality were entirely additive. These explorations demonstrate the importance of verifying the legislative assumption of compensatory mortality when permitting incidental take actions.

Case Study: Piping plovers in the Great Plains

Piping plovers, small, migratory, ground nesting shorebirds (Haig and Elliot – Smith, 2004), are listed as threatened under the US-ESA in the Great Plains, largely due to habitat alteration and destruction throughout their range (USFWS, 1985). Annually, the fate of hundreds of piping plover nests on major rivers in the Great Plains are

determined by the needs of downstream communities and power generation (Espie et al., 1999; USFWS, 2000). On the Missouri River, the USFWS allows flooding to continue as long as it affects a relatively small proportion of plover nests (USFWS, 2000). The USFWS assumes that one of two ecological conditions exist: 1) that flooded nests represent excess reproductive capacity for this population, or 2) that flooding of some nests in a specific river reach or reservoir is neutralized by an increase in reproductive success at some other location or time. For the first assumption to be true, the number of chicks produced in the absence of flooding would put the population over its ecological carrying capacity and result in higher depredation rates, disease rates, or starvation during migration and over wintering periods. This assumption is dependent upon density-dependent mortality factors of juveniles at some point throughout the annual cycle. For the second assumption to be valid there would have to be density-dependent factors affecting reproductive success. For example, perhaps nest failure decreases as the overall number of nests in the region decrease.

When establishing permissible levels of incidental take in the Missouri River, the USFWS worked with the Army Corps of Engineers, who are mandated to manage water flow in the system via 6 major dams. The agencies set allowable take levels at a five year running average of 8.3% of eggs (+/- 1.0%), with a single year maximum allowed to reach 42% of eggs (USFWS, 2000). These numbers were derived by examining the average and maximum values over the previous 7 – 10 years of take activities in the River, not by conducting modeling exercises to determine sustainable levels of egg harvest (USFWS, 2000; USFWS, 2003). The Corps are also charged with annually monitoring and measuring piping plover reproductive success (USFWS, 2000), however

there are no coordinated efforts to conduct range wide monitoring and population counts on a yearly basis. Without annual population trends it is difficult to determine how take in the River affects the population in the Great Plains as a whole. The effects of take and permitted take levels are not annually reassessed based on new data collected the previous breeding season.

Compensatory Mortality:

There is little evidence to support the existence of density-dependent mortality during the non breeding seasons in piping plovers (Drake et al., 2001) though few studies have examined mortality during this period. In fact, there are few published reports of mortality for piping plover after fledging (Haig and Elliot-Smith, 2004). Little is known about the migration route of piping plovers in the Great Plains but it is probable that most birds make the migration to the Gulf Coast in one, non stop flight (Haig and Elliot-Smith, 2004). If migration flights are non-stop, then most mortality during this time period would likely be caused by density-independent, weather-related factors. Pre-migratory body condition could also affect an individual's survival during migration, and that could be a function of food availability and density-dependent, conspecific competition in breeding or staging habitats (Alerstam and Lindstrom, 1990). There also may be mortality during migration due to raptor predation which could be a density-dependent factor. However, these factors are considered minor contributors to mortality in piping plovers (Haig and Elliot-Smith, 2004).

Drake et al. (2001) reported very high survival of piping plovers during the winter in coastal Texas. Despite high densities of migrating peregrine falcons (*Falco peregrinus*) no radio marked plovers were killed by raptors during the Drake et al. (2001)

study. It is most probable that there is a small amount of raptor predation on adults and juveniles, but Drake et al. (2001) suggested that the reluctance of piping plover to take flight when threatened by an avian predator reduces the occurrence of raptor predation compared to other shorebird species like Dunlins (*Calidrius alpine*) (Whitfield, 2003). The majority of mortality probably comes from density-independent weather related causes, such as late-season tropical storms along the Gulf and southern Atlantic coasts and other unpredictable events. Habitat and food could be limited during the over-wintering period, but little research has been conducted to evaluate the effect of habitat and food availability on survival. Piping plovers tend to feed in small flocks during the winter and, although flock size is correlated with prey density (Zonick, 2000), flocking behavior suggests a non-limiting food source that is abundant and not defensible (Brown, 1964). Therefore, density-dependent starvation is not likely to be a major source of mortality.

There have been a few documented records of density-dependent disease outbreaks in shorebirds. Adams et al. (2003) surveyed records of botulism outbreaks and reported that not one piping plover carcass has ever been recovered in post outbreak clean up efforts in the Canadian prairies. Closely related species like Killdeer (*Charadrius vociferous*) and other small sandpiper species were reported (Adams et al., 2003), so piping plovers are susceptible to the disease, but disease outbreaks do not appear to be major sources of mortality.

Compensatory Natality:

Consideration of plover reproductive biology and ecology would suggest that this species most likely cannot absorb the effects of incidental take by increasing reproductive

output or through improved reproductive success. Piping plovers, like most shorebird species, are limited to four egg clutches (Haig and Elliot-Smith, 2004). Rarely, plovers will lay five eggs in one clutch, but it is unclear if this results from egg dumping or increased reproductive effort (Haig and Elliot-Smith, 2004). Regardless, egg production per clutch highly restricts annual reproductive capacity and plovers cannot compensate for lost eggs due to flooding by producing more eggs in subsequent nesting attempts. Plovers unaffected by flooding will not increase egg production above four eggs per clutch as food competition is reduced. Furthermore, piping plovers are single brooded and will not renest if their first brood is lost (Haig and Elliot-Smith, 2004). In the Great Plains, piping plovers do not appear to have tremendous renesting capacity regardless of whether their first clutch hatches. Between 1986 and 2003, Missouri River piping plover reproductive success data collected by the Army Corps of Engineers shows that an average of 0.98 nests were monitored per female. These data indicate that piping plovers have a low capacity to renest and compensate for human induced nest failure.

It is also improbable that lowered nesting success in one area allows for increases in nesting success in other areas. The events that take place at one breeding site likely have no influence on events at other breeding sites. The flooding of one reservoir or one river reach in one river has nothing to do with flooding of reservoirs or river reaches in other rivers in the Great Plains. Nesting success in a river system has no effect on nest success at non-river, alkaline wetland sites. There is no reasonable causal link between the two habitats and no way to expect that lowered reproductive success on river sites would positively impact reproductive success at non-river sites. The actions at one point of a river could affect reproductive success downstream, meaning that flooding nests in

one portion of a river could prevent the need to flood nests in a downstream portion of that river. But, flooding nests up stream does not increase nesting success down stream; it means that downstream reproductive success is not depressed.

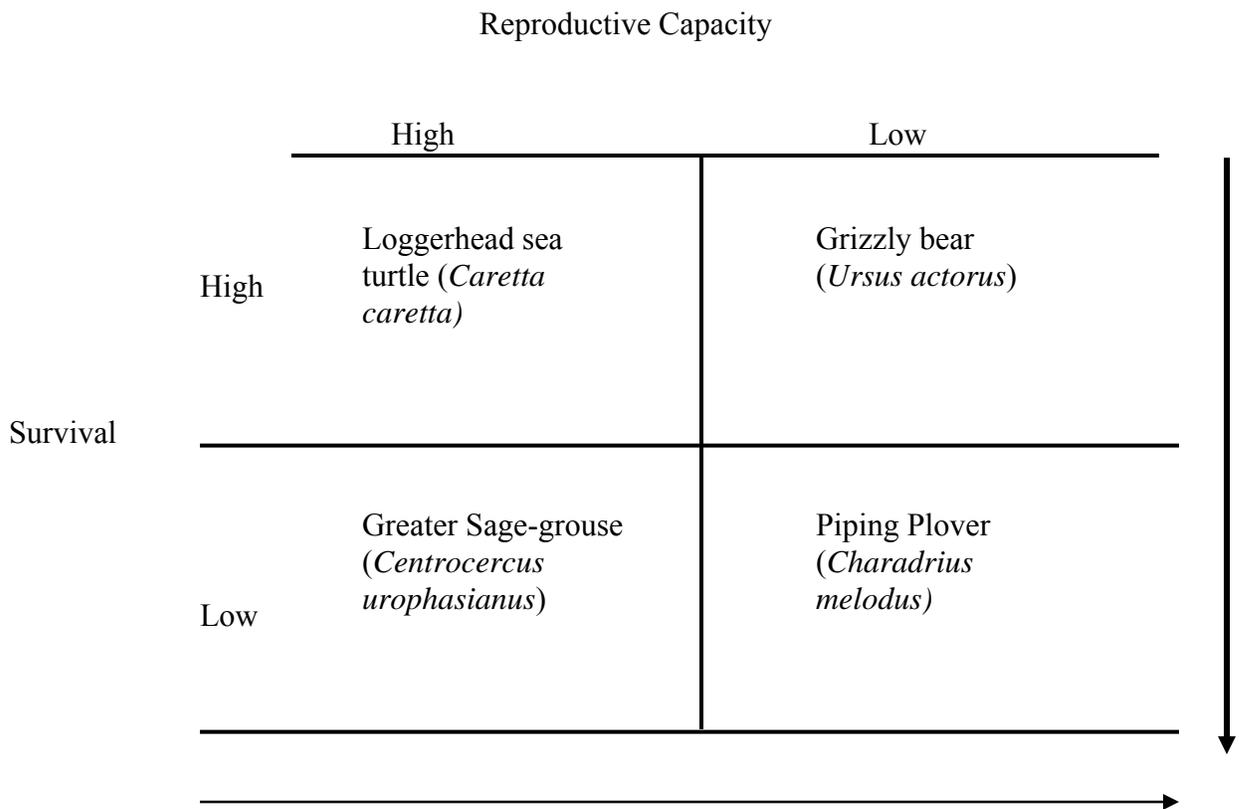
Piping plover Conclusions:

Through our exploration of piping plover ecology we showed that this species may not have an inherent capacity to compensate for incidental take. This review of piping plover ecology is not conclusive with regard to density-dependent mortality or natality, but we do show that piping plovers may not have the assumed capacity to buffer incidental take. The system established under the US-ESA that allows limited takings of plover nests, in what amounts to plover egg harvest, may be insufficient to enable plover populations to recover. We believe that it is important to follow the precautionary principle and place the focus on proving that compensation opportunities do exist instead of assuming they exist and trying to prove that they do not. Similar limitations might exist for a wide variety of protected species that do not have high levels of density-dependent mortality or excessive reproductive capacity. Managers may be able to create opportunities for compensatory mortality or natality, through habitat mediation, or enhancing reproductive success. In the remainder of this paper we use population models to explore the effects that take might have on protected species if compensation for take is not ecologically possible.

Modeling the effect of incidental take

The life history pattern of a protected species may predict its capacity to buffer incidental take (Fig. 1; Kokko et al., 2001). As adult survival decrease incidental take will have an increasing effect on the population (Fig. 1; Kokko et al., 2001). Further

Figure 1: Susceptibility to incidental take might depend on a species demography or life history. The arrows indicate the direction of increasing susceptibility to incidental take, and the thickness of the arrows indicates the strength of that trend. The species in each box are example taxa for that life history pattern.



more, as reproductive capacity decreases incidental take will also have an increasing effect on population growth and abundance. (Fig. 1; Kokko et al., 2001). We developed simple stochastic population models for four protected species representing four different life history strategies: the grizzly bear (*Ursus arctos*) is a long lived species with low reproductive rates, the loggerhead sea turtle (*Caretta caretta*) is a long live species with high reproductive rates, the greater sage-grouse (*Centrocercus urophasianus*) is a relatively short lived species with high reproductive rates, and the piping plover is a relatively short lived species with low reproductive rates.

We developed models in Microsoft Excel using the PopTools program add-ins (Hood, 2005). All models were developed for a post-breeding census. We used published demographic parameters for each species and published estimates of variance for each parameter. Variance estimates were not available for loggerhead sea turtles or greater sage-grouse so we set the standard deviation for the parameter estimate at 10% of the population average. Survival parameters varied according to a beta distribution bounded between zero and one, and fecundity parameters varied on a normal distribution. The populations were initially set to be approximately in a stable age distribution. These models are not comprehensive population models for these species. Our goal was simply to illustrate how take might affect species with differing life histories and to inform the take permitting process for all species. The results of these models should not be use for specific management actions on these species but rather to further our understanding of the incidental take policy.

We applied three different levels of take to each life stage for each species. We did not research and model specific permitted take amounts because we did not want this

paper to be a criticism of specific management actions. We chose levels of take that seemed reasonable to us as wildlife biologists. We calculated 10%, 15% and 20% of the juvenile life stages in the initial population, and 2.5%, 5% and 10% of the breeding life stages in the initial population. In separate simulations, we subtracted those amounts from the population each year for the first ten years, and the models were projected for 20 years total. We projected the populations 100 times for each take scenario and calculated the average abundance at each time step, the standard deviation at each time step, and the average population growth rate. We compare the final abundances with the average final abundance under a no take scenario, and we compare the average population growth rates while take was occurring with the population growth rates under a no take scenario (Table 1). For all species, take was modeled under fully additive mortality conditions to explore the effects of take under a worst case scenario.

Modeling results:

Grizzly Bears

We modeled a representation of the Greater Yellowstone grizzly bear population. After nearly a century of exploitation, grizzlies were hunted to near extinction in the United States by the early 1900's and have been federally protected since 1975 (Mattson and Reid, 1991; Primm, 1996; Schwartz et al., 2006). Adult females are long-lived ($\hat{S} = 0.92$), they delay reproductive maturity until the fourth year, and they only breed every third year thereafter (Schwartz et al., 2006). We used the Schwartz et al. (2006) three stage matrix model and used the same parameter and variance estimates for adult, cub and yearling survival, and annual reproductive rate. We set initial population size at 250 adult females, 31 yearlings and 50 cubs. This population size falls in the range modeled

by Schwartz et al. (2006) and approximates the mid 1990's estimate by Eberhardt and Knight (1996).

Without incidental take the population increased by six percent per year (Table 1). The final population abundance declined for all simulations of adult take. The population growth rate became negative and the final abundance was 68 percent smaller than the no take simulation when only 26 adults were removed from the population (Table 1). Under take of juveniles and young of the year, the population continued to increase and simulations showed little change from the no take scenario (Table 1).

Loggerhead sea turtle

Loggerhead sea turtles breed on beaches in the southeastern United States where habitat alteration, hunting, and fisheries by-catch have dramatically lowered populations (Crouse et al., 1987; Crowder et al., 1994). Turtles have relatively high adult survival ($\hat{S} = 0.81$), like grizzly bears have delayed sexual maturation and females only mate every two to three years once they reach adulthood (Crouse et al., 1987; Crowder et al., 1994). When sea turtles breed they lay large numbers of eggs and can produce up to 61 hatchlings per breeding female per year (Crouse et al., 1987). We used the Crowder et al. (1994) five stage matrix model for loggerhead sea turtles and we used the same demographic parameters as Crowder et al. (1994). We set the initial population size at 3000 breeding females to represent the breeding population of a typical south Florida county (USFWS, 1999).

The turtle population declined without incidental take (Table 1; Crowder et al., 1994). The taking of hatchlings had a minimally negative affect on population growth and resulted in small declines in final abundance after 20 years (Table 1). Small and large

Table 1: Population growth rates, total abundances, and percent declines of four different protected species exposed to various levels of take incidental take when compensation is not possible.

Take Scenario	<u>Grizzly Bear</u>				<u>Loggerhead Sea turtle</u>				<u>Greater Sage Grouse</u>				<u>Piping Plover</u>			
	λ^a	No. ^b taken	Final ^c abun.	% ^d decline	λ	No.	Final	%	λ	No.	Final	%	λ	No.	Final	%
No Take (S.D.)	1.06 (0.004)	0	1095 (198.0)	---	0.952 (0.003)	0	337483 (31971.9)	---	1.02 (0.009)	0	844 (214.2)	---	0.944 (0.013)	0	667 (229.8)	---
2.5% Adult (S.D.)	1.04 (0.006)	6	889 (163.6)	19	0.942 (0.011)	71	294358 (61035.7)	13	0.990 (0.009)	6	681 (195.5)	19	0.921 (0.014)	34	537 (172.0)	20
5% Adult (S.D.)	1.02 (0.006)	13	735 (145.7)	33	0.928 (0.011)	150	253997 (31585.3)	25	0.979 (0.015)	13	584 (177.0)	31	0.886 (0.027)	68	355 (151.6)	47
10% Adult (S.D.)	0.952 (0.016)	38	353 (105.4)	68	0.896 (0.031)	300	1166856 (45446.2)	51	0.908 (0.031)	25	248 (101.7)	70	0.635 (0.381)	136	0	100
2.5% Sub-Adult ^e (S.D.)					0.947 (0.004)	161	309781 (63538.5)	8								
5% Sub-adult (S.D.)					0.942 (0.006)	336	284058 (28936.6)	16								
10% Sub-adult (S.D.)					0.930 (0.018)	674	226134 (45446.5)	33								
10% Large juv. ^f (S.D.)	1.05 (0.004)	3	1024 (156.7)	6	0.936 (0.009)	9621	239635 (26830.5)	29								
15% Large juv. (S.D.)	1.05 (0.005)	5	974 (182.6)	11	0.927 (0.013)	14431	195802 (24237.5)	42								
20% Large juv. (S.D.)	1.04 (0.007)	6	955 (170.6)	13	0.920 (0.019)	19241	140722 (18513.4)	58								
10% Sm. juv. (S.D.)					0.894 (0.009)	62535	220683 (23373.4)	35								

Table 1 (Continued)

15% Sm. juv. (S.D.)					0.850 (0.031)	93802	172197 (23240.9)	49								
20% Sm. juv. (S.D.)					0.725 (0.179)	125069	116778 (21601.2)	65								
10% Young ^g (S.D.)	1.05 (0.004)	6 (178.1)	1030 (187.8)	6	0.940 (0.005)	22128	313247 (29467.3)	7	0.975 (0.014)	38 (164.9)	575 (172.3)	32	0.923 (0.007)	64 (206.2)	546 (160.0)	18
15% Young (S.D.)	1.05 (0.004)	9 (187.8)	1002 (187.8)	8	0.928 (0.006)	33192	291328 (31265.0)	14	0.960 (0.021)	57 (172.3)	489 (172.3)	42	0.901 (0.013)	95 (160.0)	427 (160.0)	36
20% Young (S.D.)	1.04 (0.004)	12 (147.2)	915 (147.2)	13	0.920 (0.007)	44255	279634 (30027.1)	17	0.928 (0.023)	76 (135.5)	374 (135.5)	56	0.884 (0.024)	127 (154.2)	347 (154.2)	48

^a λ values are the average population growth rate over the years of each simulation that take occurred, except for the “no take” scenario which are the average population growth rates over the whole simulation.

^b “No. Taken” are the number of individuals removed each from the corresponding age class of that take scenario.

^c “Final abun.” are the average population sizes after 20 years of each simulation.

^d “% decline” is the percentage that the population experiencing take declined below the population that had no take.

^e Sub-Adult designation is based on the Crowder et al. (1994) loggerhead sea turtle population model. These individuals have reached sexual maturity but do not yet breed with high frequency or effort.

^f Large a small juvenile differentiation are based on the Crowder et al. (1994) loggerhead sea turtle population model. Grizzly bears have only one juvenile stage and these data are presented in the space for Large Juveniles.

^g Simulations where young of the year were removed to simulate take.

juvenile take had a much larger effect than we anticipated, but our results do reflect the elasticity analysis of Crowder et al. (1994). Taking small juveniles caused a precipitous decline to between 35 and 65 percent below the no take scenario (Table 1), and in 37 of 100 simulations the population abundance declined to zero with 20 years. The taking of large juveniles had a substantial impact on the overall abundance and caused the population to decline at a faster rate than the no take simulation over the whole 20 year period (Table 1). Taking sub-adults and adults also had large negative effects on abundance and population growth rate. Taking hatchlings from sea turtle populations had a slightly larger effect (7% - 17% decline) than taking cubs from a grizzly bear population (6% - 13% decline).

Greater sage-grouse:

The greater sage-grouse once ranged widely across the western United States and prairie Canada in sage brush habitats. Its former range has been greatly reduced due to market hunting and widespread habitat alteration or destruction (Schroeder, Young and Braun, 1999). Some sub-populations are “candidates” for listing under the US-ESA in parts of its range (USFWS, 2006). Annual survival for adult females ranges from 55 percent (Swenson, 1986) to 75 percent (Connelly et al., 1994) and average clutch size ranges from 6.6 to 9.1 eggs (Schroeder, 1997; Schroeder et al., 1999; Aldridge and Bringham, 2001). We created a two stage matrix model and we chose survival and fecundity estimates that fell within the range of published values (adult survival = 0.60, hatchling survival = 0.28, fecundity = 1.5 female fledglings/breeding female). We set the initial population size at 250 adult females to represent a small, isolated, population (Schroeder, 1997; Schroeder et al., 1999; Aldridge and Bringham, 2001).

Without take, the population increased at 2% per year (Table 1). Take from both age classes had a large negative effect on projected population size after 20 years and on population growth rates (Table 1). Taking hatchlings caused the population to be 32 to 56 percent smaller than the no take scenario. Taking breeding females each year caused the population to be reduced to between 19 and 70 percent smaller than the no take simulations (Table 1). Taking fledglings from this population had a much greater affect on total grouse abundance than taking hatchling or cubs had on sea turtle and grizzly bear total abundance.

The Great Plains piping plover:

Piping plover populations in the Great Plains are declining by 3-12% per year (Ryan et al., 1993; Plissner and Haig, 2000; Larson et. al., 2002), which immediately suggests that the population does not meet the essential criteria for sustainable harvest (Williams et al., 2001). We used a two stage matrix model (Ryan et al., 1993) and used the demographic parameters and variances estimated by Larson et al. (2000), and Larson et al. (2002) in our simulation. Piping plovers have relatively low adult survival ($\hat{S} = 0.74$) and low reproductive rate. We set initial breeding female abundance at 1477 breeding females (Haig et al. 2005)

Similar to the greater sage-grouse simulation, take of either fledglings or adults had large negative effects on abundance and population growth rates (Table 1). Taking offspring caused the population to decline to between 18 and 48 percent smaller than a population without take (Table 1), and aking adults caused the population to decline to 20 to 100 percent smaller (Table 1). Sixty nine of the 100 simulations declined zero when 136 (10%) adults were removed from the population for the first 10 years.

Modeling Conclusions

Our modeling exercises show that even small amounts of incidental take can greatly affect endangered species populations under additive mortality. Though these models are not comprehensive representations of these species, they do elucidate how incidental take might affect generic endangered species that meet these life history characteristics. Taking breeding adults out of the populations had negative consequences on abundance for all life histories that we explored and in some cases caused population growth rates to go from positive to negative. Species with low adult survival are potentially highly affected by incidental take of any life stage regardless of the reproductive rate. However, taking all life stages from species with higher reproductive capacity resulted in greater population declines than a species with low reproductive capacity. Species with higher reproductive rates are likely more dependant on recruitment to sustain their populations than species with low reproductive rates, however in both cases presented here the species with low reproductive capacity had higher adult survival rates than their high reproductive counter parts. The patterns we identified reflect the published elasticity values for each population matrix of these species, and matrix elasticity may be a quick method for predicting the potential effects of incidental take on a population. Even simpler, managers could use the life history of a species as a rough guide as to what types of incidental take might be more or less applicable. However, where ever possible we recommend a population modeling approach where incidental take is incorporated as a harvest component.

The effect of incidental take on species recovery depends on how recovery is defined, but in most cases the substantial reduction in population size that resulted from

incidental take would greatly reduce recovery probability and could reduce population viability. Despite fully additive mortality in these models, grizzly bear and greater sage-grouse populations with positive growth rates ($\lambda > 1$) showed some capacity to recover if take actions were terminated in the future. Piping plovers and loggerhead sea turtles did not recover over time because the populations were declining even without incidental take. This raises the important question of ‘What is increased jeopardy?’ especially for a species that is already in decline. The general patterns identified in our simulations have important implications for endangered species management. Our results emphasize the importance of verifying the assumed capacity of a protected species to compensate for incidental take because populations could suffer large declines if the mechanisms do not exist.

Conclusions

We assert that an important, but not fully recognized, component of endangered species management is the legislative assumption that protected species will be able to compensate for low levels of incidental takings. Despite the prevalence of incidental take allowance in endangered species laws the applicability of a compensation paradigm to endangered populations has not been addressed. Compensation is an ecological phenomenon that does not occur equally among species or equally over time. We showed via the piping plover case study that compensatory mechanisms may not exist in populations of threatened species. We further demonstrated with simple modeling exercises the potentially dramatic effect of incidental take on four different life history strategies under additive mortality.

An additional question not explored in this paper is, when in the annual cycle is a population most capable of compensating for incidental take. Managers must consider the seasonality of take and how it fits into the annual cycle of the species being managed (Kokko and Lindstrom, 1998; Boyce et al., 1999; Kokko et. al., 1998; Xu et al., 2005). If the incidental takings occur in the annual cycle after the majority of density-dependent mortality has happened, the takings will be additive. Or, if take occurs before individuals have had the opportunity to breed, take will have a greater impact on the population (Kokko and Lindstrom, 1998; Boyce et al., 1999; Kokko et al., 1998; Xu et al., 2005). Our paper primarily considers when in a species' life cycle is it most able to compensate for incidental take.

Because a harvest paradigm is inherent within the incidental take allowances of many endangered species protection laws, wildlife managers must consider the applicability of harvest modeling theory to each endangered species. They should consider whether a species has the capacity to buffer incidental take through excess reproduction, or if the population likely experiences some density-dependent mortality after the take occurs. If neither of these buffers are likely to exist, they should be created via habitat management and species protection.

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Chapter Four

The effect of incidental take in the Missouri River on piping plovers in the Great Plains:

Does flooding eggs and chicks increase jeopardy?

The effect of incidental take in the Missouri River on piping plovers in the Great Plains:
Does flooding eggs and chicks increase jeopardy?

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Running Head: Quantifying the effects incidental take

Abstract --- Incidental take is the permitted killing, harassment, or habitat destruction of a protected species under endangered species law and is permissible as long as the take does not jeopardize the species persistence in the wild. However, take is seldom addressed in a quantitative or population modeling context and the criteria of “jeopardy” has no universal quantitative definition. We modified an existing population model to simulate the population abundance, growth, and quasi-extinction probability for Piping Plovers (*Charadrius melodus*) in the Great Plains. The model incorporated environmental stochasticity and variation due to sampling variance. Eggs and chicks were taken out of the population as a harvest to simulate incidental take that currently occurs in the Missouri River system. We used least squares regression and an AIC model selection approach to evaluate the population’s elasticity incidental take covariates. Even in the absence of take the population declined by 4 % over a 20 year period. Linear models that included combinations of egg take rate, renesting rate, an egg take squared term, and chick take as covariates received support to explain variation in population viability. The model selection analysis indicated that incidental take of eggs and chicks is likely to depress population viability of Piping Plovers in the Great Plains. Population growth, and final abundance were reduced and the probability of quasi-extinction was increased in simulations where egg take and chick take were applied. However, the question remains as to whether permitted levels of take cause jeopardy for Piping Plovers since the population declined even in simulations with no take allowances.

KEY WORDS: Incidental take, Piping Plovers, Population Viability, Model Selection, Elasticity

Incidental take is the permitted killing, harassment, or habitat destruction of a protected species under endangered species law. Take is allowed as long as it does not “jeopardize” the persistence of the species in the wild and as long as that take is incidental to some otherwise lawful activity (USFWS 1998). Take allowances are permitted under the provisions of Section 10 and Section 7 of the United States Endangered Species Act and endangered species protection laws of several countries worldwide have similar language (McGowan et al. in review A (Chapter 3 of this report)). Without careful quantitative and ecological consideration, incidental take could have substantially negative effects on population growth and abundance for protected species (McGowan et al. in review A (Chapter 3), Runge 2003). However, take is seldom addressed in a quantitative or population modeling context and the criteria of “jeopardy” has no universal quantitative definition. Properly crafted population models could be useful tools for evaluating the effect of planned take action on the population viability of a threatened or endangered species.

The Piping Plover (*Charadrius melodus*) is a protected species of migratory shorebird in the United States and Canada for which the incidental killing of eggs and chicks and destruction of habitat has frequently been permitted for water management, economic development, and human recreation purposes (Sidle et al. 1991). In the Missouri River system of the Great Plains, the US Army Corps of Engineers (USACE) is permitted to flood Piping Plovers nests as a side effect of normal water management activities (USFWS, 2000). The USACE manages water flow through six major earth-filled dams to control flooding, generate electricity, and facilitate barge traffic in the down stream portions of the River (Galat et al. 2005). However, water management

practices have altered the ecology of the river (Galat and Lipkin 2000, Galat et al 2005) and affect the reproductive success of Piping Plovers and Least Terns (*Sterna antillarum*) by flooding in-channel sandbar habitats during the breeding season, where both of these species nest on the ground in small sand scrapes (USFWS 2000).

In 1990, 2000, and 2003 the US Fish and Wildlife Service (USFWS) entered into consultation with Army Corps of Engineers under auspices of Section 7 of the Endangered Species Act to examine the effect of water management practices on Piping Plover populations. The goal of these consultations was to determine if, and under what conditions, the USACE could continue flooding plover nests (USFWS 1990, USFWS 2000, USFWS 2003). Both the 1990 and 2000 consultations concluded that the USACE's actions in the Missouri River were jeopardizing the Great Plains Piping Plover population (USFWS 1990, USFWS 2000). As a result of the consultation process, limits were established on the number eggs that could be destroyed each year by USACE and a set of water management guidelines were established to minimize the effects that late spring and early summer water releases have on nesting birds (USFWS 2000, USFWS 2003). In all of these consultations no effort was made to quantify or model the effect of flooding Piping Plover eggs on population level reproductive success parameters or on population viability.

In this paper, we modified an existing stochastic population model for Great Plains Piping Plovers to predict the effect of incidental take on future piping plover population growth and viability. All modeling efforts to date for this population indicate that the population is annually declining between 3% and 12% (Ryan et al. 1993, Plissner and Haig 2000A, Larson et al. 2002). Our model incorporates stochasticity into survival

rates, fecundity parameters, habitat distribution parameters, and annual levels of incidental take. We separated out the Missouri River nesting population on an annual basis to apply take allowances only to those birds nesting in Missouri River habitats. The goal of this modeling effort was to examine the effect of incidental take on population growth, abundance, and the probability of quasi-extinction in the future. We arbitrarily set the quasi-extinction threshold at 100 breeding females; a level of population size that would likely require urgent management actions to prevent complete extinction. Quasi-extinction is considered to be a more useful measure of population viability than full extinction, because it alleviates the need to consider allee effects and deleterious genetic effects in a population model (Gilpin and Soule 1986, Morris and Doak 2002). Setting a quasi-extinction threshold is also more useful in a wildlife management context because extinction is a management barrier that, once crossed, cannot be reversed (Beissenger and Westphal 1998, Morris and Doak 2002).

METHODS

The Model:

We used a modified version of the two stage Great Plains Piping Plover population matrix models developed by Ryan et al. (1993) and Larson et al. (2002). Our model split the population each year into three habitat types: Alkali wetlands, Great Plains rivers, and the Missouri River. These habitat types were differentiated by reproductive success and there was no spatial explicitness incorporated into the model. Plissner and Haig (2000A) developed a spatially explicit population model for piping plovers in the Great Plains, but the model was highly dependant on immigration and emigration rates between subpopulations (Plissner and Haig 2000A, Larson et al. 2002). Because there is little

empirical data available to measure or verify assumptions about immigration and emigration rates (Larson et al. 2002), we decided to exclude spatial explicitness from our population model. We set annual survival rates as equal for the entire Great Plains, because there is no evidence indicating that breeding habitat effects annual survival of adults or juveniles.

The basic algebraic formulation of the model was as follows:

$$N_{t+1} = \{[(N_t * P_w) * F_w] + [(N_t * P_{MR}) * F_{MR}] + [(N_t * P_{or}) * F_{or}]\} * S_i + (N_t * S_a)$$

Where N_{t+1} is the population size one time step into the future and N_t is the current population size. P_w is the proportion of the population nesting in alkali wetland habitats, F_w is the estimated fecundity for alkali wetland habitats, P_{MR} is the proportion of the population nesting in Missouri River habitats, F_{MR} is the estimated fecundity for Missouri River habitats, P_{or} is the proportion of the population nesting in other rivers in the Great Plains and F_{or} is the estimated fecundity for other river habitats. S_a is the estimated annual adult survival and S_i is the estimated annual survival for immature birds (yearlings) in the population. This model is a females only model because population growth in this species is limited by female productivity (Larson et al. 2002, Morris and Doak 2002).

For each simulation, we ran the model for 5000 iterations and calculated average growth rates, abundances and the probability of quasi-extinction. The model included environmental stochasticity on all survival and fecundity parameters, and where ever possible we separated sampling variance from temporal variance using the procedures recommended by White (2000). For all population survival and fecundity parameters where we thought sampling variance might be a concern, we incorporated sampling

variance into the iteration loop of the model so that for each of the 5000 replications a new distribution was created for each parameter from which annual values were randomly selected in each year of that simulation (Boyce 2001, White 2000). We believe it is more conservative to include all the sampling variance in the iteration loop of the model, especially in the case of threatened species management, even though the results have increased variability.

Our model also included ceiling type density dependence, which set an upper size limit to the populations (Morris and Doak 2002). Density dependant reductions in survival or reproductive rates have not been empirically documented for the species. However, it is unlikely that the population could grow unimpeded by space and habitat availability. We used a simple ceiling type density dependence because the specific relationships of the population parameters to density are unknown, and this method allows us to apply an upper size limit to the population without speculating on the nature and form of those density dependant relationships (Morris and Doak 2002).

Parameterization:

Population Size: We set the initial population size at 2300 individual breeding females. The 2006 international Piping Plover Census counted a total of ~4600 birds in the northern Great Plains during the breeding season (E. Elliot-Smith, USGS, personal communication). We assumed a 1:1 sex ratio and set the initial population size at half of the total counted. We arbitrarily set the maximum population size for the ceiling type density dependence at 5000 females, or 10000 individuals. Increase the maximum population size to 10,000 breeding females had no effect on the modeling results over the first 20 years.

Annual Survival: We used adult and immature survival and standard deviation estimates from Larson et al. (2000) and from Larson et al. (2002). They used immature survival from east coast populations because empirical estimates of juvenile survival in the Great Plains were probably biased low due to low resighting probability and relatively low natal site fidelity (Melvin 1996, Larson et al. 2002). Our model selected mean survival values for each iteration from a beta distribution with a mean of 0.737 for adults and 0.48 for immatures with standard deviations of 0.00226. For each year in a given iteration the model selected an adult survival value from a beta distribution with the iteration mean and a standard deviation of 0.0445. Our model annually selected a juvenile survival value from a beta distribution with the iteration mean and a standard deviation of 0.0445. We used the values for temporal (SD = 0.0445) and sampling (SD = 0.00226) variance for adult and immature survival calculated by Larson et al. (2000).

Habitat distribution: We included stochasticity in the habitat distributional parameters. The model selected a random value of P_w from a beta distribution with a mean of 0.6 and standard deviation of 0.2. The P_{MR} value in each year was equal to $(1 - P_w) * 0.8$ and the P_{or} value was equal to $1 - (P_w + P_{MR})$. These habitat distributions were based on empirical data from the four International Piping Plover Censuses (Plissner and Haig 2000B, Haig et al. 2005). Every year the population redistributed across the three habitats with no inter annual correlations.

Fecundity: We estimated fecundity for all three habitat types using empirical data. For alkali wetlands we used data from the USFWS widespread monitoring efforts in the Northern Great Plains, as well as data from the Canadian Wildlife Service at Big Quill Lake, the Saskatchewan Coteau (C. Gratto-Trevor, Canadian Wildlife Service,

unpublished data) and Chaplin Lake (C. White, Canadian Wildlife Service, unpublished data) (Table 1). With these data we report the average number of chicks that fledged for each breeding pair. This approach tracks known pairs and broods over the breeding season and is different from previous methods of reporting a fledge ratio where the total number of fledged chicks was divided by the number of breeding pairs. In the end, the results appear similar, however our approach enables us to calculate a corresponding variance estimate for fecundity. The model selected a random value of alkali wetland fecundity for each simulation from a log-normal distribution with a mean of 0.614 and a standard deviation 0.280 (Table 1). Attempts to separate temporal variance from sampling variance returned a negative result, indicating that either #1) there is no temporal variance or #2) that the sampling variance is too large to be able to measure temporal variance (Gould and Nichols 1998). As a result we did not include any inter-annual variation in alkali fecundity in the model, all variation was incorporated into the iteration loop.

To estimate fecundity in the Missouri River system we combined nest survival, chick survival, number of breeding pairs in the river, and the number of nests per female following the recommendations of Noon and Sauer (1992) for estimating fecundity in a population model. The calculations were as follows:

$$F_{MR} = (N_t * P_{MR} * R * CL * NS * ChS) / (N_t * P_{MR})$$

Where N_t is the current female population size and P_{MR} is the proportion nesting in the Missouri River. R is the number of nests per breeding female, CL is the average clutch size, NS is the probability that a nest survives to hatching, and ChS is the probability that a chick survives to fledging. We estimated daily nest survival to hatching from USACE

Table 1: Number of fledglings and the average number of fledglings per breeding pair produced at alkali wetland sites in the Northern Great Plains and Prairie Canada. Data were provided by Adam Ryba of the US Fish and Wildlife Service and by Cherri Gratto-Trevor and Cory White of the Canadian Wildlife service.

Year	No. Pairs	No. Fledglings	Average Fl	Var Fl
1994	177	212	1.197	1.614
1995	69	43	0.62	1.297
1996	120	144	1.2	1.657
1997	159	250	1.57	2.208
1998	165	165	1	1.707
1999	190	238	1.25	1.703
2000	250	337	1.35	1.955
2001	231	268	1.16	1.735
2002	328	364	1.12	1.9
2003	388	509	1.33	1.93
2004a	216	147	0.69	1.11
2005a	140	129	0.92	1.54
Total	2433	2806	1.117	0.075

^a Data from 2004 and 2005 are only from Canadian sites provided by Cherri Gratto-Trevor and Cory White

nest check records using the logistic exposure method (Shaffer 2004). We tested for annual variation in nest survival by comparing models of nest survival and using an information theoretic approach to select the best model from our proposed set of candidate models (Burnham and Anderson 2002). The candidates models contained inter-annual effects and within season (Julian date) effects. The most supported model had year and cubic effects of Julian date as covariates. We used the estimate of daily survival at the mid-point of the breeding season and averaged across years (Shaffer and Thompson 2007). We raised the average estimate of daily survival to the power of 24 to represent survival to hatching (Haig and Elliot Smith 2004). We excluded nests that the USACE had categorized as “taken” from this data set in order to estimate nest survival in the absence of take. Average daily survival of nests over 13 years between 1993 and 2005 was 0.977 (S.D. \pm 0.013) and the probability of survival to hatching was 0.553. We used estimates of daily chick survival calculated using USACE, count based, age class data and a modified catch curve analysis (Chapman and Robson 1960, Skalski et al. 2006, McGowan et al. in review B (See Chapter 2 of this report for details on this method)). Average daily survival estimates for chicks between 1993 and 2005 ranged from 0.895 to 0.917 and the probability of survival to fledging ranged from 0.11 to 0.19. We used 0.16 as the chick survival value in our model because it is an intermediate value between the maximum and minimum survival estimates. Renesting rate for piping plovers in this system has not been thoroughly measured. We modeled hypothetical levels of renesting to assess the effect that renesting has on population viability. For modeling scenarios that did not directly examining the interaction of renesting and incidental take, we held the number of nests per female at 1.05 with a standard deviation

of 0.05. Unpublished USACE data indicate that between 1998 and 2004 the USACE monitored 0.98 nests per breeding pair observed. Clutch size estimates were based on USACE Piping Plover monitoring data. The clutch size was not reported for many nests in that data base but of the nests where the data was recorded the average clutch size was 3.4 eggs with a standard deviation of 0.96. We set a maximum average clutch size of 5 (2.5 for a female only population model) because Piping Plover clutches rarely exceed 5 eggs (Haig and Elliot-Smith 2004).

To estimate fecundity for other rivers in the Great Plains we used the same approach as for the Missouri River calculations. For nest survival in this system we use the average annual value of nest survival for all nests in the Missouri River, including nests that were designated as “taken” by the USACE. This is intended to represent nest survival in other river systems where incidental take does occur. Average daily nest survival was 0.96, and hatching probability was 0.45. We used the same values for clutch size, renesting rate, and chick survival as we did in the Missouri River calculations.

For both Missouri River habitats and the other river habitats, we assumed that all the measured variance in nest survival was temporal variance, and we included that variation in the annual loop of the model. Because estimating nest survival uses a known fate modeling approach, we assumed there were no issues with detectability and therefore little or no sampling variation. Similarly, for the clutch size estimate we assumed that all variability was temporal variability, because detecting the number of eggs in a clutch should have little or no error once a nest is found. Therefore, all the measured variability in clutch size was included in the annual loop. Both of these decisions assume that the

nests monitored and included in the USACE database are a random sample of the overall river nesting population. For chick survival, we attempted to discern temporal variance from total variance, but as was the case with alkali wetland fecundity, the result was negative. Therefore, we incorporate all the measured variance in chick survival into the model as sampling variance in the iteration loop.

Incidental take: We modeled incidental take as a subtraction of specific numbers of eggs or chicks from the Missouri in each year. This approach reflects the permitting practices of the USFWS and the USACE in the Missouri River Section 7 consultation. Egg take and Chick Take were incorporated in to the Missouri River fecundity calculation as follows:

$$F_{MR} = \{((((((N_t * P_{MR}) * R) * CL) - ET) * NS) - CT) * ChS\} / (N_t * P_{MR})$$

Where ET is the number of eggs taken due to USACE actions in the Missouri River system, and CT is the number of chicks taken due to USACE action in the Missouri system. Egg Take and chick take were random variables selected from a log-normal distribution based on the prescribed mean and variance of take in the Incidental take permit. Initially egg take was set at the permitted average level of 9.2% of eggs in the Missouri River with a single year maximum not to exceed 294 eggs (147 female eggs). To calculate the annual number of eggs taken we first calculated the annual number of eggs laid:

$$\text{Eggs in the Missouri River} = ((N_t * P_{MR}) * R) * CL$$

We then multiplied that number by the randomly selected egg take value for that year:

$$\text{Egg Take} = (\text{Eggs in the Missouri River}) * (\text{Percent of eggs taken} / 100)$$

If the Egg Take value exceeded the 147 maximum, the value was automatically reset to equal 147 eggs.

There is no permitted taking of Piping Plover chicks in the Missouri River, however it is probable that chicks are lost due to USACE activities in the system but those losses go undocumented. We modeled some hypothetical chick take scenarios to assess the affect that flooding and drowning of chicks might have on population viability. We calculated the number of chicks taken from the Missouri River system the same way we calculated the number of eggs taken; first we calculated the number of chicks present in the river that year and then multiplied that value by the randomly selected proportion of chicks taken.

Modeling Scenarios:

Currently the USACE are limited to flooding 8.4% of eggs over a ten year running average. The average must fall with 10% of that limit, meaning that after 10 years take could range from 7.3% to 9.2%, without violating the incidental take agreements (USFWS 2000, 2003). The Corps manages for take at the upper end of that allowable range, 9.2% (C. Aron, USFWS, personal communication). Furthermore, the USACE are not allowed to exceed an upper limit of 294 eggs taken in the River system in any single year. The incidental take statement also phrases that upper limit as 42% of eggs in the River system (USFWS 2000, 2003). These numbers are derived from historic averages and maximums that the USACE has observed (USFWS 2003).

Our goal was to assess the effect of incidental take of Piping Plover eggs and chicks in the Missouri River system on population viability in the Great Plains. Our modeling scenarios attempted to isolate incidental take and the parameters that might

affect the impact of incidental take (See Table 2 for an organized list of all modeling scenarios). All means are the average percent of eggs in laid in the Missouri River that are subsequently flooded by USACE actions.

We modeled the population with no egg or chick take in the Missouri River (Scenario 1). We modeled the currently permitted amounts of egg take (mean = 9.2%, SD = 0.10, Max = 294 eggs) (Scenario 2), and we ran two scenarios of increased mean egg take by increasing the percent of allowable take by 15 % (Scenario 3) and 30 % (Scenario 4) above current levels and keeping the standard deviation and the maximum the same (mean = 10.58% and 11.96%). We increased the frequency of maximum egg take events by increasing the standard deviation of the mean to 1.0 (Scenario 5) and 2.0 (Scenario 6). We lowered the annual average by 15 % (Scenario 7) and 30% (Scenario 8) to 7.82% and 6.44%, holding the standard deviation and the maximum egg take constant. We further conducted exploratory analyses to determine what level of mean annual egg take resulted in no change in the population growth rate or the final abundances. We also modeled the single-year maximum allowable egg take as 42% (Scenario 9) of eggs in the River system, because the incidental take permit phrases the maximum as both 294 eggs and 42% of eggs.

Holding egg take at the currently permitted levels, we then added the hypothetical chick take components. Chick take was modeled at two levels where mean percent of chicks taken from the River was arbitrarily set at 3.0% (Scenario 10) and 5.0% (Scenario 11) and the standard deviation of chick take was held constant at 0.5. We set a single year maximum level of chicks take from the river at 100 individuals.

To assess the potential for renesting to alleviate any negative consequences of taking eggs or chick in the Missouri River we increased the mean number of nests per female from 1.05 to 1.15 (Scenario 12) and 1.25 (Scenario 13) keeping the standard deviation the same and modeled egg take at the currently permitted levels. 1.25 nests per female is approximately equivalent to 50% of females renesting after losing their first nest, when the probability of a survival to hatching is 0.553 (the estimated hatching probability for the Missouri River habitats). We also model egg take at currently permitted levels combined with the lower level of chick take and increased number of nests per female to 1.15 (Scenario 14) nests and 1.25 (Scenario 15) nests. We emphasize that the renesting rate of piping plovers in the Great Plains is largely unquantified. Most piping plover biologists believe that these birds will renest if a nest is lost before hatching. However, data indicate that the actual renesting rates are quite low (M. R. Ryan unpublished data, USACE unpublished data), and these birds will not renest after brood or chick loss, they are a single brooded species (Haig and Elliot Smith 2004).

Lastly we modeled the effect that the distribution of individuals across habitats has on the impact of incidental take. As more birds nest in the Missouri River system incidental take might have a larger affect on the over all population. To do this we decreased the proportion of the population nesting at alkali wetland sites from 0.6 to 0.55 (S.D. = 0.1) (Scenario 16), and 0.45 (S.D. = 0.1) (Scenario 17) while holding egg take at the currently permitted levels, chick take at the lower level and renesting at 1.05 nests per female.

For each scenario listed above we project the population 20, 30, and 40 years in to the future and we replicated the population 5000 times. We report the average final

abundance, the average population growth rate over the course of the simulation and the cumulative probability of quasi-extinction. We set the quasi-extinction threshold at 100 breeding females. We also conducted an elasticity analysis for the five parameters of interest (egg take, S.D. of egg take, chick take, number of nests per female, and proportion of the population nesting at Missouri River Sites).

Linear Regression and Model Selection:

Rather than use a more traditional sensitivity and elasticity analysis to evaluate the potential effects of incidental on population growth, we used simulated data for a regression based analysis similar to the method used and recommended by Wisdom and Mills (1997) and Wisdom et al. (2000). Because variables we are interested for this study are not matrix elements, but rather are population parameters that are components of matrix elements, the regression based approach will be more informative regarding the contribution of each specific population parameter (Wisdom and Mills 1997, Wisdom et al. 2000). This approach also enables us to look at the combined effects of take variables, potential non-linear effects of take variables on population viability. Furthermore this approach allows us to study the effects of take variables on population viability in a stochastic environment where parameters like adult or juvenile survival are allowed to vary.

We conducted additional simulations with 5000 iteration each and generated average growth rates, final abundances and quasi extinction probabilities for a variety of combinations of egg take, standard deviation of egg take, chick take, and renesting rate. The input values of egg take, variation in egg take, chick take, or number of nests per female for these extra simulations were chosen haphazardly to create a substantial range

in values. All other population parameters like adult survival, juvenile survival, Missouri River nest survival, were allowed to vary stochastically to reflect natural variation in those values. With the simulated data we compared a set of a priori linear regression models examining the relationship of the separate response variables (average population growth rate, average final abundance, and quasi-extinction probability) and the above take variables as covariates. We developed 17 models for each dependant variable, at 20 years, 30 years and 40 years (Table 3) to evaluate 4 research hypotheses: 1) Stochastic variation in adult survival is the primary factor affecting population growth, abundance and quasi-extinction probability, 2) Adult survival and incidental take affect population growth, abundance and quasi-extinction probability, 3) Renesting rate can compensate for the negative affects of incidental take, 4) Variability of incidental take affects population growth, abundance and quasi-extinction probability. We included linear models for all four variables and quadratic models for both egg take and chick take variables. The quadratic models would indicate if there are some low levels of take for which there are no effects on population growth, abundance or probability of quasi-extinction. We included adult survival in all models because previous research efforts concluded that the population was most sensitive to that parameter (Ryan et al. 1993, Larson et al. 2002). We tested the goodness of fit for the global model (a model containing all the covariates) and for the sub-global models for each of the 4 hypotheses (models containing all the covariate pertaining to that hypothesis).

We used an AIC approach to assess the relative support for the models under each hypothesis where the sub global model had reasonable fit to the data ($p < 0.2$). We also report the R-squared value for the least-squares regression models to demonstrate how

much of the variation in the data was explained by the supported models in the candidate set (Neter et al. 2000). We calculated the model averaged beta parameter estimates for each of the covariates in the top supported models (Burnham and Anderson 2002).

We predicted a negative relationship between the population growth rate or final abundance and egg take, and chick take at 20, 30 and 40 years. We further anticipated a positive relationship between the probability of quasi extinction and those same covariates. If low levels of egg take and chick are equivalent to no take then the squared terms for those covariates should have negative coefficients. We predicted that the standard deviation of egg take would have no effect on the 20 and 30 year population projections, but would have a negative effect on population growth and abundance and a positive effect on the probability of quasi-extinction at 40 years. Lastly we predicted a positive relationship between the population growth rate and final abundance with the number of nests per female, and a negative relationship between probability of quasi extinction and the number of nests per female.

RESULTS:

The modeled population declined by $\sim 7.1\%$ annually (average $\lambda = 0.927$) even when no eggs or chicks were taken from the Missouri River (Table 2). Without incidental take there was a 0.156 probability that the population will have less than one hundred females after 20 years which increased to 0.390 after 30 years and 0.519 after 40 years.

Population growth rates and final abundances were lower after 20, 30 and 40 under currently permitted take levels, and the quasi-extinction probability increased above the no take scenario for 30 and 40 year projections (Table 2). Currently permitted

Table 2: Average population growth rate, final abundance and probability of quasi-extinction for piping plovers in the Great Plains under different incidental take regimes.

	Mean	SD	20 years			30 years			40 years			
			λ	Final Abundance	Quasi extinction	λ	Final Abundance	Quasi extinction	λ	Final Abundance	Quasi extinction	
1. No Take												
Eggs	0	0	0.929	1033.0	0.156	0.923	904.2	0.390	0.920	826.8	0.519	
Chicks	0	0										
2. Current												
Eggs	9.2	0.1	0.924	959.0	0.161	0.921	813.8	0.392	0.917	807.0	0.544	
Chicks	0	0										
3. Elevated Egg Take 1 (15% increase)												
Eggs	10.58	0.1	0.926	971.0	0.161	0.922	816.3	0.392	0.916	775.0	0.556	
Chicks	0	0										
4. Elevated Egg Take 2 (30% increase)												
Eggs	11.96	0.5	0.923	919.5	0.172	0.920	836.0	0.403	0.918	833.0	0.535	
Chicks	0	0										
5. Elevated variability of egg take 1												
Eggs	9.2	1	0.925	970.8	0.167	0.922	844.3	0.393	0.917	801.1	0.538	
Chicks	0	0										
6. Elevated Variability of egg take 2												
Eggs	9.2	2	0.925	978.5	0.164	0.921	810.0	0.394	0.917	805.0	0.538	
Chicks	0	0										
7. Decreased Egg Take 1 (15% decrease)												
Eggs	7.82	0.1	0.926	977.0	0.159	0.922	854.7	0.385	0.917	823.0	0.536	
Chicks	0	0										
8. Decreased Egg Take 2 (30% decrease)												
Eggs	6.44	0.1	0.927	966.7	0.158	0.922	815.0	0.392	0.918	821.0	0.539	
Chicks	0	0										
9. Maximum Egg Take as percent (42%)												
Eggs	9.2	0.1	0.926	959.0	0.153	0.921	895.0	0.381	0.900	813.6	0.540	
Chicks	0	0										
10. Egg Take + Chick Take 1												
Eggs	9.2	0.1	0.922	916.3	0.175	0.922	843.3	0.392	0.918	836.7	0.530	
Chicks	3	0.5										
11. Egg Take + Chick Take 2												
Eggs	9.2	0.1	0.922	929.7	0.179	0.921	857.8	0.398	0.917	817.0	0.539	
Chicks	5	0.5										
12. Egg Take + Renesting 1												
Eggs	9.2	0.1	0.926	1008.8	0.161	0.925	934.8	0.363	0.920	906.0	0.524	
Chicks	0	0										
No. Nests	1.15											
13. Egg Take + Renesting 2												
Eggs	9.2	0.1	0.928	1100.1	0.160	0.927	998.5	0.362	0.921	950.7	0.512	
Chicks	0	0										
No. Nests	1.25											

Table 2: Continued

	Mean	SD	λ	20 years		λ	30 years		λ	40 years	
				Final Abundance	Quasi extinction		Final Abundance	Quasi extinction		Final Abundance	Quasi extinction
14. Egg Take + Chick Take + Renesting 1											
Eggs	9.2	0.1	0.925	945.2	0.160	0.924	936.7	0.383	0.920	880.0	0.531
Chicks	3	0.5									
No. Nests	1.15										
15. Egg Take + Chick Take + Renesting 2											
Eggs	9.2	0.1	0.927	1049.8	0.155	0.298	1049.0	0.355	0.924	982.1	0.504
Chicks	3	0.5									
No. Nests	1.25										
16. Egg Take + Elevated Mo. Riv. Prop 1											
Eggs	9.2	0.1	0.913	761.5	0.216	0.910	612.7	0.456	0.907	583.0	0.600
Chicks	3	0.5									
Alk Prop	0.55	0.1									
17. Egg Take + Elevated Mo. Riv. Prop 2											
Eggs	9.2	0.1	0.892	459.6	0.308	0.889	370.0	0.592	0.879	267.0	0.752
Chicks	3	0.5									
Alk Prop	0.45	0.1									

levels of egg take reduced population growth to be 0.5 % lower than the no take scenario over 20 years, 0.2% annually over 30 years and 0.3 % over 40 years. At 30 and 40 years, egg take only scenarios had lower population growth rates, final abundances and greater quasi-extinction probabilities than the no take scenario (Table 2).

A clear pattern of the relationship between egg take and population growth rate, final abundance or quasi-extinction probability did not emerge from the prescribed modeling scenarios (Table 2). For example, projecting the population 40 years into the future taking 11.96 percent of eggs annually resulted in a higher estimate of population growth ($\lambda = 0.918$ (- 8.2% annually)) than taking only 9.2 percent of eggs annually ($\lambda = 0.917$ (-8.3% annually)). Modeling the maximum level of egg take as a percentage (42%) resulted in a much lower population growth rate ($\lambda = 0.900$) then when it was modeled as a specific number ($\lambda = 0.917$) for the 40 year simulations (Table 2). The 20 and 30 year simulations showed no difference between modeling maximum egg take as a percentage verses a specific number.

Adding chick take to the simulations resulted in populations that were smaller and had lower growth rates than the equivalent egg take only scenarios at 20 years and the pattern was unclear for the 30 and 40 year simulations (Table 2). Adding increased re-nesting to the model appeared to neutralize any negative affects of both egg take and chick take. With 1.15 nests per female in the Missouri River, population growth was equal to or higher than the no take scenario at the 20, 30 and 40 year marks (though it was still negative overall population growth) and with 1.25 nests per female population growth was also higher at the 20, 30 and 40 year marks. For the 20, 30, and 40 year projections under both 1.15 and 1.25 nests per female final abundances were

Table 3. Candidate models predicting the effect of egg take, chick take, number of nests per female, and standard deviation of egg take on population growth, final abundance and the probability of quasi-extinction. In parentheses before each is the predicted sign of the coefficient. Each model was compared for each response variable at 20, 30 and 40 years.

Model Title	Model Structure
Null*	Intercept only, no covariates
Adult Survival*	(+) AS ^a
Egg Take 1*	(+) AS + (-) Egg Take
Egg Take 2*	(+) AS + (-) Egg Take + (-) Egg Take ²
Egg and Chick Take 1*	(+) AS + (-) Egg Take + (-) Chick Take
Egg and Chick Take 2*	(+) AS + (-) Egg Take + (-) Chick Take + (-) Egg Take ²
Egg and Chick Take 3*	(+) AS + (-) Egg Take + (-) Chick Take + (-) Egg Take ² + (-) Chick Take ²
Take and Renesting 1 ^b	(+) AS + (-) Egg Take + (+) RN
Take and Renesting 2	(+) AS + (-) Egg Take + (+) RN + (-) Egg Take ²
Take and Renesting 3	(+) AS + (-) Egg Take + (-) Chick Take + (+) RN
Take and Renesting 4	(+) AS + (-) Egg Take + (-) Chick Take + (+) RN + (-) Egg Take ² + (-) Chick Take ²
Renesting	(+) AS + (+) RN
Egg Take Variation 1 ^c *	(+) AS + (-) Egg Take + (-) ETSD
Egg Take Variation 2*	(+) AS + (-) Egg Take + (-) ETSD + (-) Egg Take ²
Egg Take Variation 3*	(+) AS + (-) Egg Take + (-) Chick Take + (-) ETSD
Egg Take Variation 4*	(+) AS + (-) Egg Take + (-) Chick Take + (-) ETSD + (-) Egg Take ² + (-) Chick Take ²
Global*	(+) AS + (-) Egg Take + (-) Chick Take + (-) ETSD + (+) RN + (-) Egg Take ² + (-) Chick Take ²

^a AS stands for the average Adult Survival value due to sampling variance over the 5000 simulations

^b RN stands for resenting and was modeled as the number of nests per female in the Missouri River.

^c ETSD stands for standard deviation of egg take.

* denotes models that were used in the parallel analysis for the data set with no variation in renesting rate.

approximately equal to or greater than the no take scenario and the cumulative quasi-extinction probabilities were smaller (Table 2).

When a greater proportion of the population nested in the Missouri river the simulated population exhibited lower population growth rates, lower final abundances and higher cumulative quasi-extinction probabilities. When Take is modeled as a percentage of eggs laid in the Missouri River, take has a greater affect on the Great Plains population as more individuals nest in the Missouri River habitats.

Linear Regressions and Model Selection: We ran the population model 34 times with varying combinations of egg take, standard deviation of egg take, chick take and reneating rate. The program randomly selected values of adult survival for ranging from 0.735 to 0.741, nests per female ranged from 1.0 to 1.26, egg take from 0% to 11.96%, chick take from 0% to 6.5%, and standard deviation of egg take from 0.01 to 2.0. The sample size for the all linear regression models was 34.

For the 20 year projections the global model and the sub-global models for each research hypothesis met the goodness of fit criteria for all three response variables. I proceeded with the information theoretic model selection analysis for all models in the candidate set. The most supported model to explain variation in average population growth rate had adult survival, egg take rate, and the number of nests per female as the only covariates ($\Delta AIC = 0$, $w = 0.47$, $R^2 = 0.87$, Table 4). The linear relationship between egg take and population growth rate was negative when all other variables are allowed to vary stochastically ($b = -0.0003$, $R^2 = 0.54$, Figure 1). However there was some model uncertainty because three additional models fell within five AIC units of the best model, and the top three models combined held 90% of the weight. All three of the top

Table 4: The most supported models from the candidate set to explain variation in average population growth, average final abundance, and probability of quasi-extinction after twenty years of simulation under a variety of incidental take conditions. Population growth, final abundance and quasi-extinction probability were the average values calculated from 5000 iterations of the Great Plains Piping Plover population model (n = 34 simulations)

Population Growth						
Model	SoS ^a	K ^b	AICc	Δ AIC _c ^c	w ^d	R ²
(+) AS (+) RN (-) ET	4E-05	5	-189.639	0.000	0.471	0.868
(+) AS (+) RN (-) ET (-) CT	3.4E-05	6	-188.772	0.867	0.305	0.881
(+) AS (+) RN (-) ET (+) ET2	3.8E-05	6	-187.215	2.424	0.140	0.868
(+) AS (+) RN	6.2E-05	4	-185.688	3.952	0.065	0.798
Final Abundance						
Model	SoS	K	AICc	Δ AIC	w	R ²
(+) AS (+) RN	18309	4	103.135	0.000	0.648	0.570
(+) AS (+) RN (-) ET	16568	5	104.423	1.28813	0.340	0.703
Probability of Quasi-extinction						
Model	SoS	K	AICc	Δ AIC	w	R ²
(+) AS (+) RN	0.00148	4	-138.902	0.000	0.655	0.734
(+) AS (+) RN (-) ET	0.00144	5	-136.543	2.359	0.202	0.733
(+) AS (+) RN (-) ET (-) CT	0.00135	6	-134.528	4.374	0.074	0.741

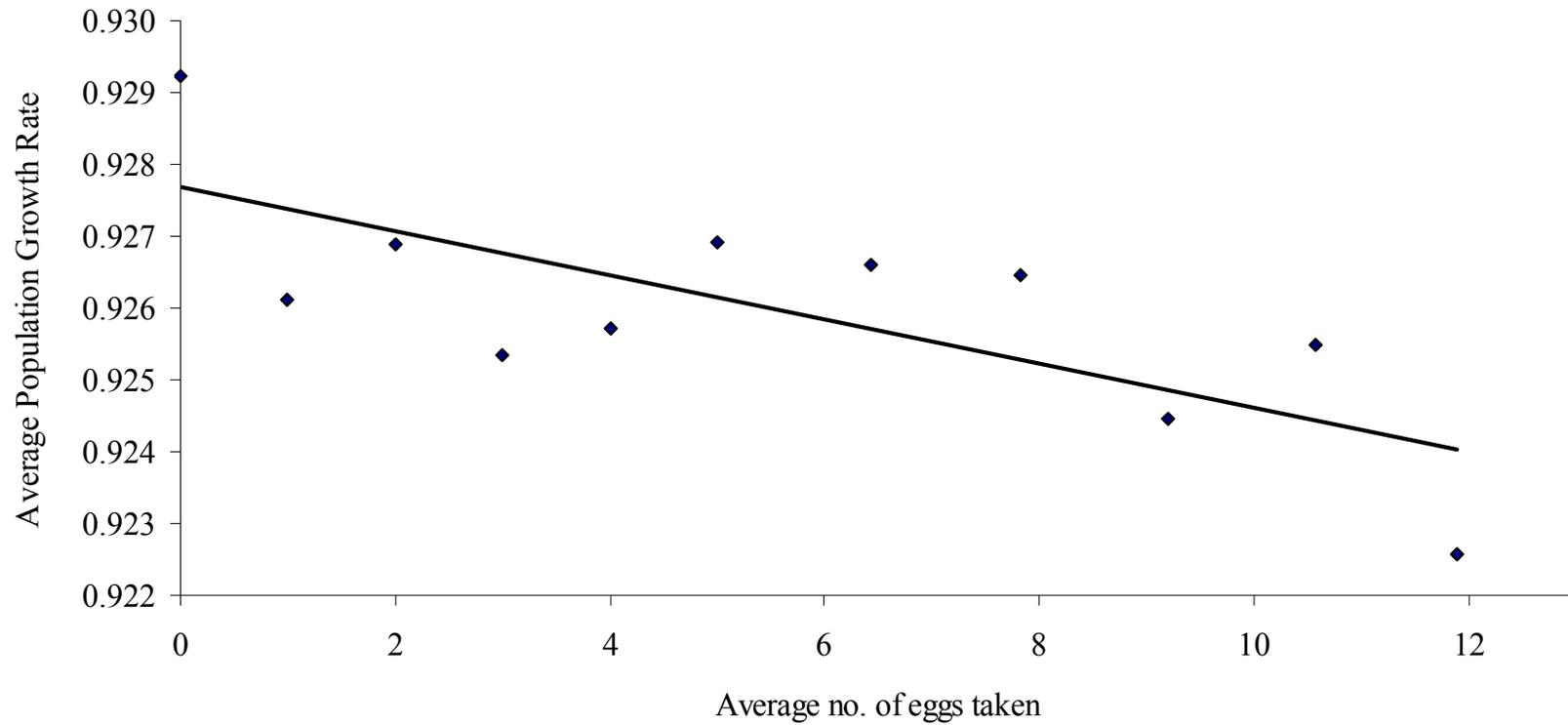
^a Sum of squares for the error term of the model

^b Number of parameters in the model, least squares have two more parameters than the number of coefficients in the model

^c Change in the AICc from the best model

^d The AICc weight of the model, estimates the probability that this is the correct model given the data.

Figure 1: A linear regression of the relationship between the input value of average egg take and the resulting average population growth rate from 34 simulations with 5000 iterations of piping plover populations in the Great Plains. All other variables in the population model were allowed to vary stochastically according to the population model programming.



models had adult survival, egg take rate, and number of nests per female as covariates. The second most supported model included a chick take covariate ($\Delta\text{AIC} = 0.867$, $w = 0.305$) and the third most supported model had an additional egg take squared term ($\Delta\text{AIC} = 2.424$, $w = 0.140$). The parameter coefficients were positive on adult survival, the number of nests per female, and the egg take squared covariates but negative for the egg take and chick take covariates (Table 4).

The most supported model to explain variation in final abundance after 20 years had adult survival, and the number of nests per female in the Missouri River as covariates ($\Delta\text{AIC} = 0$, $w = 0.648$). Again, there was some model uncertainty because that model only garnered 64.8% of the total weight and there was one other model that fell within five AIC units of the top model. The second most supported model had the egg take rate as an additional covariate ($\Delta\text{AIC} = 1.288$, $w = 0.34$). However the R-squared value for the model including egg take as a covariate was much greater ($R^2 = 0.70$) than the model where egg take was not a covariate ($R^2 = 0.57$) (Table 4).

There was also uncertainty in model selection among models to explain variation in the cumulative quasi-extinction probability. There were three models that fell within five AIC units of the best model. The most supported model again had adult survival and number of nests per female as the only covariates, but garnered only 65.5% of the AIC weight. Other competing models included egg take as a covariate ($\Delta\text{AIC} = 2.359$, $w = 0.202$), and egg take and chick take as covariates ($\Delta\text{AIC} = 4.374$, $w = 0.074$), but both of these models had higher R-squared values than the top model (Table 4).

For the 30 year projections the global model and the all sub-global models met the goodness of fit criteria. To explain variation in the population growth rate, the most

Table 5: The most supported models from the candidate set to explain variation in population growth, final abundance, and probability of quasi-extinction after thirty years of simulation under a variety of incidental take conditions. Population growth, final abundance and quasi-extinction probability were the average values calculated from 5000 iterations of the Great Plains Piping Plover population model (n = 34 simulations)

Population Growth						
Model	SoS ^a	K ^b	AICc	Δ AICc ^c	w ^d	R ²
(+)AS (+)RN (-)ET (-)CT	0.0000113	6	-205.141	0.000	0.931	0.910
(+)AS (+)RN (-)ET (-)CT (+)ET2 (+)CT2	0.0000109	8	-199.078	6.063	0.045	0.910
Final Abundance						
Model	SoS	K	AICc	delta	w	R ²
(+)AS (+)RN (-)ET (-)CT	7283.75	6	94.361	0.000	0.793	0.870
(+)AS (+)RN (-)ET	11662.00	5	98.343	3.982	0.108	0.800
Probability of quasi-extinction						
Model	SoS	K	AICc	delta	w	R ²
(-)AS (-)RN	0.00215	4	-135.967	0.000	0.446	0.66
(-)AS (-)RN (+)ET (+)CT	0.00128	6	-135.314	0.653	0.322	0.78
(-)AS (-)RN (+)ET	0.00171	5	-134.006	1.962	0.167	0.71
(-)AS (-)RN (+)ET (-)ET2	0.00167	6	-131.387	4.581	0.045	0.71

^a Sum of squares for the error term of the model

^b Number of parameters in the model, least squares have two more parameters than the number of coefficients in the model

^c Change in the AICc from the best model

^d The AICc weight of the model, estimates the probability that this is the correct model given the data.

supported model included adults survival, number of nests per female, egg take rate and chick take as covariates ($\Delta AIC = 0$, $w = 0.931$). There were no other models that fell within five AIC units of the best model. To explain variation in final abundance, the model with adult survival, number of nests per female, egg take and chick take as covariates was the most supported model ($\Delta AIC = 0$, $w = 0.793$). Additionally the model containing only adult survival, renesting rate and egg take as covariates fell within five AIC units of the best model ($\Delta AIC = 3.98$, $w = 0.103$, Table 5). To explain variation in the probability of quasi-extinction, the top model did not have any of the incidental take variables as covariates, however there were four partially supported models in the candidate set. Three of the top models had egg take as a covariate, one model had Chick take and one model had the egg take squared term (Table 5).

For the 40 year projections the global model met the goodness of fit criteria for all three response variables, but the only sub-global to pass the goodness of fit test was the model pertaining to the renesting research hypothesis. Therefore we only included the models that had the number of nests per female as a covariate in the candidate set. For all three response variables the model containing adult survival, number of nests per female, and egg take as covariates was the most supported model (Table 6). Again there was model uncertainty for all three response variables with three to four additional models falling within five AIC units of the best model (Table 6). With the population growth rate as the response variable the second most supported model included the egg take squared term and the third model contained the chick take term as a fourth covariate. For the final abundance and the quasi-extinction probability response variables the model with only adult survival and number of nests per female received some support (Table 6).

Table 6: The most supported models from the candidate set to explain variation in population growth, final abundance, and probability of quasi-extinction after twenty years of simulation under a variety of incidental take conditions. Population growth, final abundance and quasi-extinction probability were the average values calculated from 5000 iterations of the Great Plains Piping Plover population model (n = 34 simulations)

Population Growth						
Model	SoS ^a	K ^b	AICc	Δ AIC _c ^c	w ^d	R ²
(+) AS (+) RN (-) ET	4E-05	5	-189.639	0.000	0.471	0.868
(+) AS (+) RN (-) ET (-) CT	3.4E-05	6	-188.772	0.867	0.305	0.881
(+) AS (+) RN (-) ET (+) ET2	3.8E-05	6	-187.215	2.424	0.140	0.868
(+) AS (+) RN	6.2E-05	4	-185.688	3.952	0.065	0.798
Final Abundance						
Model	SoS	K	AICc	Δ AIC	w	R ²
(+) AS (+) RN	18309	4	103.135	0.000	0.648	0.570
(+) AS (+) RN (-) ET	16568	5	104.423	1.28813	0.340	0.703
Probability of Quasi-extinction						
Model	SoS	K	AICc	Δ AIC	w	R ²
(+) AS (+) RN	0.00148	4	-138.902	0.000	0.655	0.734
(+) AS (+) RN (-) ET	0.00144	5	-136.543	2.359	0.202	0.733
(+) AS (+) RN (-) ET (-) CT	0.00135	6	-134.528	4.374	0.074	0.741

^a Sum of squares for the error term of the model

^b Number of parameters in the model, least squares have two more parameters than the number of coefficients in the model

^c Change in the AICc from the best model

^d The AICc weight of the model, estimates the probability that this is the correct model given the data.

Table 7: Averaged beta parameter estimates for the covariates of least squares models of average population growth, average final abundance and quasi-extinction probability at 20, 30 and 40 years. Only parameters from models that comprised 90% of the AIC weight (Tables 4, 5 and 6) are presented. Population growth, final abundance and quasi-extinction probability were the average values calculated from 5000 iterations of the Great Plains Piping Plover population model (n = 34 simulations)

Covariate	Beta estimate	SD	Upper bound	Lower bound	Importance
20 years					
Lambda					
AS	1.27321	0.17994	0.92054	1.626	1.000
RN	0.02512	0.00308	0.01908	0.031	1.000
ET	-0.00033	0.00015	-0.00063	-0.00003	1.000
CT	-0.00009	0.00005	-0.00018	0.00000	0.333
ET^2	0.00000	0.00006	-0.00010	0.00011	0.153
Final Abundance					
AS	17709.525	4665.925	8564.312	26854.738	1.000
RN	373.741	99.125	179.456	568.027	1.000
ET	-2.629	1.850	-6.254	0.997	0.344
Quasi-extinction					
AS	-4.49695	0.95474	-6.36824	-2.6257	1.000
RN	-0.10514	0.01817	-0.14075	-0.0695	1.000
ET	0.00009	0.00016	-0.00022	0.0004	0.296
CT	0.00009	0.00010	-0.00011	0.0003	0.079
30 years					
Lambda					
AS	1.36601	0.15819	1.67606	1.05596	0.931
RN	0.02439	0.00154	0.02741	0.02137	0.931
ET	-0.00012171	0.00004427	-0.00003	-0.00021	0.931
CT	-0.00038014	0.0000623	-0.00026	-0.00050	0.931
Final abundance					
AS	23983.3486	4136.123557	15876.54642	32090.15077	1
RN	526.4857917	40.45379261	447.19636	605.77523	1
ET	-2.96244586	0.253076245	-3.45848	-2.46642	1
CT	-5.80819701	0	-5.80820	-5.80820	0.88

Table 7: Continued.

Covariate	Beta estimate	SD	Upper bound	Lower bound	Importance
Quasi-extinction					
AS	-9.0426	1.9342	-12.8337	-5.2515	1.0000
RN	-0.1444	0.0202	-0.1839	-0.1048	1.0000
ET	0.0005	0.0004	-0.0002	0.0012	0.8211
CT	0.0007	0.0005	-0.0003	0.0017	0.3441
40 Years					
Lambda					
AS	1.4787	0.3574	0.7782	2.17918	1.000
RN	0.0411	0.0031	0.0349	0.04726	1.000
ET	-0.0005	0.0001	-0.0007	-0.00020	0.882
CT	-0.0001	8.26773E-05	-0.0003	0.00005	0.386
ET^2	1.51743E-06	3.90889E-06	-6.144E-06	9.17886E-06	0.000
Final Abundance					
AS	16012.157	6093.308	4069.272	27955.041	1.000
RN	593.167	56.573	482.283	704.051	1.000
ET	-2.911	1.749	-6.339	0.516	0.761
CT	-1.231	1.058	-3.305	0.843	0.274
ET^2	0.011	0.048	-0.083	0.106	0.091
Quasi-extinction					
AS	-6.6929	1.1738	-8.9935	-4.3923	1.000
RN	-0.1493	0.0186	-0.1859	-0.1128	1.000
ET	0.0015	0.0006	0.0003	0.0027	0.902
CT	4.01207E-05	0.0003	-0.0005	0.0006	0.316
ET^2	-2.1912E-05	1.94661E-05	-0.00006	0.00002	0.074

The averaged parameter coefficients for the covariates in the top models of each candidate set were positive on adult survival, the number of nests per female, and the egg take squared covariates. The coefficients were negative for the egg take and chick take covariates (Table 7). That pattern was reversed for the models to explain variation in the probability of quasi-extinction (Table 7).

DISCUSSION:

Our data suggest that incidental take of eggs in the Missouri River has the potential to increase the probability of quasi-extinction and to reduce population growth and Final abundance for Piping Plovers in the Great Plains. Our population model predicted a 7% increase in the rate of decline when currently permitted levels of incidental take were applied. Under currently permitted take level population viability decreased. Increasing take above current average levels will likely cause further decreases in future population growth and abundance. From the prescribed take modeling scenarios, we conclude that chick take may impose a further additive decrease in population growth and abundance and that chick take levels should be empirically examined in the Missouri River system. Our modeling effort found no effect of increased egg take variability on the three population viability response variables. The increased egg take variation scenarios were intended to simulate an increased frequency of hitting the maximum level of egg take. It is possible that the levels of standard deviation that we modeled were not large enough to significantly increase the frequency of maximum take events. However modeling the maximum egg take as a percentage (42%) rather than a specific number of eggs (294 eggs) showed large declines in the population viability. It is possible, with a high a proportion of birds nesting in the Missouri River in any given

year that even low percentages of egg take could exceed 294 eggs. Thus restricting maximum egg take to 294 might frequently prevent very high levels of egg take in the early years of the population projections. This is a different effect than was intended when the incidental take limits were designed (USFWS 2000), but permitting maximum take as a specific number still likely reduces the negative impacts of egg take.

According to our population model, Piping Plovers might be able to compensate for incidental take via renesting efforts. McGowan et al. (in review, Chapter 3 of this report) identified a species' capacity to compensate for incidental take as an essential component of successful incidental take management plan. We must stress however, that all renesting scenarios modeled in this research effort were not empirically based. Renesting capacity for piping plovers has not been thoroughly quantified in the Great Plains, and limited data indicate that these birds do not have high renesting capacity. For example, USACE data indicate that the Corps monitored only 0.98 nests for each observed breeding female between 1998 and 2004 (C. Kruse and G. Pavelka, USACE, unpublished data). Mayer (1991) reported that there were 1.09 nests per breeding pair at non-river alkali wetland sites in North Dakota during the late 1980's. If renesting is going to be considered as part of an incidental take management plan for Piping Plovers it is important to first gather data and quantify renesting rates in Missouri River habitats (McGowan et al. in review, Chapter 3 of this report).

The model selection approach allowed us to examine the combined effects of different variables of interest in our population model and to better understand the effect that incidental take of eggs and chicks has on population viability. This approach has benefits over a more traditional elasticity and sensitivity analyses (Caswell 2001) because

the affects of minor population parameters, such as egg take or chick take in this case, may only become evident when two or more of those parameters act in conjunction. Furthermore using an AIC approach instead of a typical elasticity analysis also enabled us to examine potential non-linear relationships between incidental take variables and measures of population viability. Most importantly for conservation and mangement this approach allowed us to examine the effects of the incidental take parameters on population viability in a stochastic context (Wisdom and Mills 1997, Wisdom et al 2000).

The model selection procedure confirmed that egg take and reneating play an important role in explaining that variation in the measures of population viability. The parallel analysis of the no-variability in reneating data set served as confirmation of the results from the primary analysis. We carried out this additional analysis because, as previously noted, reneating rates have not been studied or quantified for piping plovers in the Great Plains. The results of this parallel analysis did not substantially vary from the primary analysis.

Our results indicated that over 20, 30 or 40 years egg take negatively affects all three measures of population viability and most likely in a linear fashion; according to our model results, egg take has a negative and additive effect on population viability. There was some limited support for the quadratic egg take model. If the coefficient was negative, as we predicted it would be, that would indicated that low levels of egg take were similar to or no different than zero egg take. However, the coefficient of the egg take squared term in our models was positive, indicating that negative affects of egg take flatten out at high levels of take.

Chick take was also present in some of the most supported models depending on the time frame and the response variable; reaffirming the importance of empirically quantifying the amount of chick mortality caused by water management activities in the River. Renesting also emerged from the model selection process as an important variable for explaining the variation in the three measures of population viability. The relationship was positive and linear, but again we stress that all renesting values we modeled were hypothetical and this covariate should be empirically studied.

According to the r-squared values for these regression models, 50 to 70 percent of the variation in population growth, final abundance and quasi-extinction probability was explained by the adult survival, nests per female, egg take and chick take covariates (Tables 2, 3, & 4). In some cases these strong correlations were surprising. But, the covariates included in these models either are or have major impacts on the primary demographic parameters of this population. Previous research indicated that the population model was highly sensitive to annual adult survival, and moderately sensitive to annual fecundity (Ryan et al. 1993, Larson et al. 2002). Renesting, egg take and chick take influence the annual fecundity value for on average of 30 percent of the modeled Piping Plover population. Thus, in retrospect high r-squared values should be expected given the structure of the population models and the subsequent regression models.

It is important to keep in mind that even though the coefficients for the egg take covariates and the chick take covariates were significantly negative, they were still very small effects on population growth and final abundance. The model average coefficient for the egg take covariate in the population growth linear models was -0.00033 . When compared to the model averaged coefficient for adult survival (1.273), the egg take

coefficient is dwarfed. This indicates that although egg take does have a negative effect on population viability it is a very small effect compared to factors that limit adult survival.

Resulting from these population modeling exercises the question still remains: Does incidental take of eggs and chicks in the Missouri River increase the jeopardy of Piping Plovers in the Great Plains? Quantifying the concept of jeopardy for a threatened population is a difficult task. In this case, the population model predicts that even with incidental take components set to zero, the population was still declining by over 7 % annually. Runge (2003) concluded that allowable incidental take due to water craft collisions for a sub-population of Manatees was zero individuals per year because that sub-population was already in decline by 1.1% annually. Runge (2003) found that take not only further decreased the population growth rate but also concluded that take reduced the probability of meeting recovery goals for that sub-population to near zero. Determining whether a 7.6 % annual decline is larger than a 7.1 % annual decline is possible given standard deviations and confidence intervals on those average estimates (Rohlf and Sokal 1995). Determining whether a population with a 7.6 % annual decline is at greater risk of extinction than a population with a 7.1 % annual decline is almost an arbitrary determination (Pimm et al. 1988, Boyce 1992, Lande 1993). Our model predicts an increase of 1.5% in quasi-extinction probability after 30 years and 3.7 % increase after 40 years under currently permitted take levels compared to a population with no take of eggs or chicks in the Missouri River. It is important to keep in mind that we are talking about a probability of quasi-extinction that increases from 51.6 % in a population with no take to 55.3 % when take is added over the first 40 years of

simulation. In this situation, if take of eggs is allowed in the Missouri River, the jeopardy determination becomes an issue of acceptable risk of extinction. What level of extinction risk for this species are we willing to tolerate in order to allow the status quo of Missouri River water management to continue?

The predicted quasi-extinction probabilities from our population model also indicate that Piping Plover management in the Great Plains continues to face tremendous challenges even without incidental take on the Missouri River Population (Ryan et al. 1993, Plissner and Haig 2000A, Larson et al. 2002, Larson et al. 2003). However, from these analyses we conclude that egg and chick take in the Missouri River will further depress population growth of Piping Plovers in the Great Plains. High renesting rates may be able to compensate for take, but limited data indicate renesting rates are low. Alternatively, managers may be able to create compensation by enhancing nest and chick survival via habitat management and predator control.

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Chapter 4 - APPENDIX 1: A user's guide for the 2008 Piping Plover Great Plains population model.

By Conor P. McGowan, Mark R. Ryan, and Morgan Weber

The Model is programmed in Visual Basic for Applications (VBA) for use with a Microsoft Excel interface. It can access and downloaded from the University of Missouri internet server at the following address: <http://web.missouri.edu/~RyanMR/2008pipl-model/PIPL-GP-IT-model-FINAL.xls>. The Model is embedded as a macro in the excel file "PIPL-GP-IT-model-FINAL.xls" and the code for macro can be accessed via the "tools" menu in that excel workbook. Select "visual basic editor" from the tools menu. The main process macro appears as "module1" in the macro list. We do not recommend accessing and editing the Macro for this program unless you are an experienced Visual Basic programmer.

The model parameter inputs are accessible and can be manipulated in the "Model Parameters" spread sheet of the excel file. The parameters for the model are (in order of appearance on the spread sheet): 1) initial population size, 2) maximum population size 3) Adult survival, 4) Juvenile survival, 5) Alkaline fecundity, 6) Other river nest survival 7) river clutch size, 8) Missouri River nest survival, 9) river chick survival, 10) number of nests per female, 11) Proportion of birds nesting at alkali wetlands, 12) Proportion of birds nesting at Missouri River sites, 13) Proportion of birds nesting at other river sites, 14) average egg take, 15) maximum egg take, 16) average chick take, 17) maximum chick take, 18) years in the model simulations, 19) Number of repeated simulations, 20) Quasi-extinction threshold. The average parameter estimates are in column C of the spread sheet, the standard deviations of the population means are in column D and the

variances are in column E. All parameter values can be altered in the spreadsheet except the variance estimates and the proportion of birds nesting at other river sites and proportion of birds nesting at Missouri River sites. Those two parameters are calculated from the value input for proportion of birds nesting at alkali wetlands and the variance estimates are calculated from the input standard deviation values. The initial survival and fecundity values in the model are based on published values or are empirical parameter estimates based on USFWS and USACE data (See chapter 4 for details on the parameterization of the model).

Using the model is quite simple. Open the Excel file and click “Enable Macros” in the message box that appears after opening the program. The user needs to set the input parameters at the desired levels on the “Model Parameters” spread sheet. Initial parameter estimates are saved for reference purposes on the spread sheet entitled “Initial Parameter Values.” On the “Model Parameters” spreadsheet there is a click button that says “Run Model.” Clicking on this will instruct the program to access and execute the population model macro. The macro uses the values the user input on the “Model Parameter” spread sheet to predict population growth under the conditions that the user specified. The program displays the results of each year in each simulation on the “Stochastic Output” spreadsheet and the program summarizes that output on the “Stochastic Summaries” spreadsheet. The program automatically redirects the view from one spreadsheet to the next as the calculations are completed. For each simulation the program displays the calculated abundance at each time step, the number of eggs taken, the number of chicks taken, and the growth rate for that time step.

In the program, maximum egg take can be modeled as a specific number (ie: 294 eggs) or as a percentage of all eggs laid in the Missouri River (ie: 42%). If modeling egg take as a specific number, on the “Model Parameters” spreadsheet use the click button on the left that says “RUN MODEL with max egg take as a specific number.” If modeling egg take as a percentage of eggs, use the click button on the right that says “RUN MODEL with max egg take as a percentage.” This click button accesses a similar but different macro that appears as “Module 3” in the macro list.

The model can also be run while setting any and all standard deviation values to zero. Setting the SD to zero will hold those values constant at the average level, potentially allowing the user to examine the effect of a specific population parameter on overall population growth and abundance.

On the summaries spread sheet the program displays the calculated average abundance at each time step, the standard error upper bound on abundance, the standard error lower bound on abundance, the average egg take at each time step, the average chick take at each time step, the proportion of simulation that took the maximum number of eggs in each at time step, the proportion of simulations that took the maximum number of chicks at each time step, the cumulative probability of extinction at each time step, the population growth rate at each time step, the 95% upper and lower bound at each time step, the over all average population growth rate, and the average adult survival for that set of simulations. We chose to present one standard deviation above and below the average abundance, rather than the 95% confidence interval because the 95% confidence intervals for the simulated data are generally very large and difficult to interpret. A

single standard deviation represents approximately a 65% confidence interval, meaning that 65% of the time the population trajectory will fall within the upper and lower bound.

There are also three graphs generated on the “Stochastic Summaries” spreadsheet, one displaying the population abundance (+/- 1 standard deviation confidence interval), one displaying the average growth rate, and one displaying the cumulative probability of extinction. The data created in both “Stochastic Output” and the “Stochastic Summaries” spreadsheets are erased each time the “run model” button on the “Model Parameters” spreadsheet is pressed. We recommend copying and pasting any data or graphs the user wants to save on either of these two output spreadsheets into a separate excel file.

The fourth spreadsheet in the program is titled “Statistical Distribution.” In this spreadsheet the values displayed are the products of embedded calculations based on input values from the “Model Parameters” spreadsheet. There are beta distribution calculations and Log-normal distribution calculations. These values are used by the macro in conjunction with random numbers (generated by the macro) to randomly select parameter estimates for the population for each simulation and at each time step based on the mean and standard deviation values input by the user. Do not alter the values or the formulas in any cell in columns B, C, D, H or I. These columns have been locked to prevent accidental alteration of these calculations. The statistical distributions spreadsheet also contains the values for the sampling variance on Adult, juvenile and chick survival, as well as on alkali wetland fecundity. These values are estimated from USACE and USFWS data or were derived from published values. These values can be manipulated but could dramatically alter the performance of the model.

The parameter inputs can be manipulated to reflect any management scenarios that could be applied to the Great Plains populations. For example, if the user wants to explore the potential effect of reducing mortality of chicks during the pre-fledging phase in the river habitats, the user could increase the mean chick survival value in cell “C16” on the “Model Parameters” spread sheet. Or if the user wanted to explore the effect of increasing Egg Take allowances, the user could increase the average percentage of eggs taken value in cell “C21” on the “Model parameters” spreadsheet. The user can also affect the variability in the system by manipulating the standard deviation values for each parameter in column D on the “Model Parameters” spreadsheet. Please do not change the variance values in column E, as they are embedded formulas that automatically recalculate when the standard deviation values are altered.

We recommend caution when manipulating parameter input values in the model. Because of limitations in Microsoft Excel’s random number generator properties, we recommend that the number of repeated simulations should not exceed 5000 and that the number of years in the simulations should not exceed 50. We also recommend caution when inputting parameter estimates at extremely high or low levels for survival parameters. Inputting an adult survival value greater than 0.91 often returns an error message stating “Run Time Error 1004: Unable to get the BetaINV property of the worksheet function class.” We believe this occurs because MS-Excel does not have the statistical ability in its beta distribution function to deal with the extremely large or small alpha and beta parameters that are created by a high (or low) mean survival values. Alpha and beta parameters are values that describe that shape of the sampling distribution for the survival parameters. Beta distributions are statistical distribution that are bounded

between zero and one, and this distribution is frequently used in wildlife population modeling (Caswell 2001).³

Infrequently, the macro will fail to execute, even when all input parameters values are at reasonable levels. In these cases an error message will appear that says “Run Time Error 1004: Unable to get the BetaINV property of the worksheet function class,” or “Run Time Error 1004: Unable to get the NormINV property of the worksheet function class.” This occurs because occasionally the VBA random number generator, which is supposed to generate a random number greater than zero but less than one, will generate a zero or negative value for one of the 12 random numbers required to execute the model. The only solution to this problem is to click the “End” button on the error message box, and then try re-running the model. If this error message appears more than twice in a row it probably means that one of the input parameters is outside the acceptable range for the program to work (as discussed above); adjust the parameter values and try running the program again.

If you encounter further problems with this program that are not addressed in this users guide, or if you have questions on using this program and population model, please contact Conor McGowan: cpm4h9@mizzou.edu or cpm4h9@hotmail.com with “Piping Plover model” in the subject line of the email. Happy Modeling!

³ For more detail on beta distributions, their statistical properties and their application please see H. Caswell. 2001. Matrix Population Models: Construction, analysis and interpretation. Sinauer Associates Inc, Massachusetts, USA

Chapter Five

Evaluating the operating definition of the phrase “best available science” in Section 7 consultations.

Evaluating the operating definition of the phrase “best available science” in Section 7 consultations.

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Running head: Best Science for incidental take

Abstract --- Endangered species legislation calls on the US Fish and Wildlife Service (USFWS) to use the best science available when evaluating endangered species management decisions such as listing, delisting, and incidental take. Information that is used to evaluate endangered species management decisions could potentially have a major impact on the outcome of that management. We evaluated the literature cited from biological opinions prepared by the USFWS for Section 7 Consultations regarding piping plover (*Charadrius melodus*) incidental take issues. We found high reliance on unpublished resources (~ 63% of resources) and low reliance on peer-reviewed published literature (~ 16% of resources). On average, only 8% of available published articles were consulted as part of the consultations that we reviewed. We calculated a diversity index for each biological opinion using count data for in-text citations to determine if the opinions were relying many or just a few resources in the decision making process. Mean diversity for opinions that we examined was 88% (Highest possible = 100%). We specifically examined the use of available population models within each opinion to study how the USFWS has dealt with competing conclusions from multiple data sources in the consultation process. We found that even when all available population models were cited and consulted, little if any reasoning was provided for choosing to rely on the conclusions from one population model over another. This might leave the USFWS open to criticism regarding their process for determining what of the available science is “best.”

Endangered species legislation prohibits the killing, harming, harassment, or destruction of habitat of a protected species, however, many endangered species protection laws provide for exceptions to those rules (McGowan and Ryan 2007 (Chapter 2 of this dissertation)). For example, in the United States when a federal agency; like the Forest Service or the Army Corps of Engineers, plans to undertake any action that might affect a federally-listed endangered or threatened species, the agency must apply for an incidental take permit from the U.S. Fish and Wildlife Service (hereafter USFWS) (USFWS 1998). Section 10 of the US Endangered Species Act (ESA) states “The Secretary may permit...any taking otherwise prohibited by section 9(a)(1)(B) if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity,” or if “the taking will not appreciably reduce the likelihood of the survival and recovery of the species in the wild” (US-ESA of 1973. U.S. Code 16, section 668d).

Section 7 of the ESA dictates that other government agencies must consult with the USFWS to determine what affect any action might have on an endangered species. The USFWS is charged with using the “best science” available to determine if the proposed actions would “jeopardize” the species and can grant or deny the permit depending on its assessment (U.S. Fish and Wildlife Service 1998). Jeopardizing a species is defined in legislation and government documents as “appreciable reducing the probability of survival in the wild” (USFWS 1998).

In 1994, the Department of the Interior released a policy on information standards for carrying out the ESA to define the concept of best science available (FR, July 1, 1994 (Vol. 59), pg. 34271). The Federal Register states:

“The services receive and use information on the biology, ecology, distribution, abundance, status and trends of a species from a wide variety of sources as part of their responsibility to implement the act. Some of this information is anecdotal, some of it is oral and some of it is found in written documents. These documents include status surveys, biological assessments and other unpublished material (“gray material”) from State natural resource agencies and natural heritage programs, Tribal governments, other Federal agencies, consulting firms, contractors, and individuals associated with professional organizations and higher educational institutions. The Services also used published articles from juried professional journals. The reliability of the information contained in these sources can be as variable as the sources themselves. As part of their routine activities, service biologists are required to gather, review, and evaluate information from these sources prior to undertaking listing, recovery, consultation and permitting actions.”

In the same document, USFWS biologists are instructed “To the extent consistent with sections 4, 7, and 10 of the ESA, and to the extent consistent with the use of the best scientific and commercial data available, use primary and original sources of data as the basis for recommendations...” In the 1998, the Service published Section 7 consultation guidelines for endangered and threatened species managers. The manual lays out the procedures to apply for and assess an incidental take permit. However, the manual does not set standards or criteria for the “best science” or the “jeopardy” phrases. The above quotations represent the only criteria regarding the standards for the “best available

science” and they do not set a hierarchy for what types of scientific data are better than others. This can result in widely differential interpretation of terms and could potentially be a shortfall of the incidental take permitting process.

The Section 7 consultation process is quite straight forward. If, during the planning stages for any action a public agency determines that their actions might impact a listed species, the agency request the initiation of a “formal consultation” (USFWS 1998). To analyze an incidental take permitting issue the USFWS writes and issues a Biological Opinion (USFWS 1998). The Services has 90 days to assess the situation and formulate a draft biological opinion and the action agency has a further 45 days to review that draft (USFWS 1998). The opinion is a document that consists of a review of relevant literature and data to assess the status of the species in question and to assess the likely affects of the proposed actions (USFWS 1998). The opinions conclude with an “incidental take statement” that articulates what levels of take are permissible and under what conditions take can occur (USFWS 1998).

Conroy et al. (2006) and Beier et al. (2006) recently showed that the science used in the consultation process to write the biological opinions can greatly affect the outcome of Section 7 consultations. In a critique of the science used to manage and conserve the Florida panther (*Puma concolor coryi*) in south Florida, those authors documented repeated over-reliance on a set of panther habitat-use analyses that were methodologically flawed (Beier et al 2006, Conroy et al. 2006). Inferences from these flawed habitat analyses has created major problems for panther populations as construction projects in potential panther habitats were permitted under the Section 7 consultation process (Gross 2005). These analyses of Florida panther biological opinions

showed that utilizing a diversity of sources of information may be more important for successful consultations than relying on a single peer reviewed article in a highly respected journal (Gross 2005, Beier et al. 2006, Conroy et al. 2006).

Habitat conservation planning is the avenue through which private citizens request permits to affect listed species (Beatley 1991, Aengst et al. 1997). Several recent analyses have shown that many habitat conservation plans lack quantitative and demographic analyses (Smallwood 2000, Harding et al. 2001, Peterson et al. 2004). Harding et al. (2001) and Peterson et al. (2004) concluded that habitat conservation plans often will not meet their conservation goals, in part because of a lack of quantitative and demographic input.

The types and diversity of resources and the information accessed to evaluate the status of a species and predict the consequences of incidental take could affect the results of incidental take permitting. With this research, we examined Biological Opinions for piping plovers (*Charadrius melodus*) throughout their range to evaluate the types and diversity of resources used in the Section 7 consultation process. Piping plovers are small migratory shorebirds that breed along the Atlantic coast, throughout the northern Great Plains and in low numbers in the Great Lakes region (Haig and Elliot-Smith 2004, Haig et al. 2005). This species often conflicts with water management, beach renourishment, and human recreational activities because of its ocean beach and riverine habitat needs (Sidle et al. 1991, Haig and Elliot-Smith 2004). We document the types of resources cited in the literature and in the text (e.g.: peer reviewed, personal communications) and we calculated a diversity index for in-text citations to determine if the opinions relied on many or just a few information sources. Our goal is to describe what the operating

definition is for the “best available science” in the incidental take permitting process. We also closely examined the use of quantitative population models in each of the opinions. There have been several attempts to model piping plover populations each with varying methodologies and results for the USFWS to evaluate and chose among. Examining how the USFWS uses each of the different models might provide insight into how the determination of “best available science” is made. The piping plover is a representative species for this evaluation because its ecology and population dynamics have been extensively studied. Furthermore, it has been listed under the Endangered Species Act for over 20 years and there have been numerous biological opinions filed on many different federal projects throughout the species’ range (Sidle et al. 1991).

METHODS

We acquired multiple biological opinions regarding piping plovers by searching electronic resources via the University of Missouri library and on the internet. We also made direct requests to the USFWS to provide completed biological opinions that we could not access via the internet. Our sample represents those opinions that we could access. Almost all of the opinions that we used for this study were done for more than one species. We only examined the portions of the opinions that related to the general system and planned action, as well as the piping plover-specific sections.

We categorized each reference listed in the literature cited section of the opinion as: 1) peer reviewed, 2) graduate thesis or dissertation, 3) book, book chapter, or conference proceedings, 4) unpublished reports or data, and 5) personal communications. We created this categorization and believe it represents a descending order of rigor and accountability in the available resources, with peer reviewed citations having the most

prior scrutiny from the scientific community and personal communications having the least. We present the percentages that each type of resource comprised of the total number of referenced resources. We then counted the number of times that each reference was cited in the text of the biological opinions and incidental take statements. We again present the percentage that each type of resource comprised of the total number of in-text references.

We used those same in-text citation data to calculate a diversity index for in-text citations. This was intended to evaluate whether the opinion was relying on information from just a few resources rather than multiple resources. We used the commonly applied Simpson's index of diversity, which is a measure of species richness and evenness derived from count data (Simpson 1949, Tucker 2005). With this measure of diversity, the closer the calculated value is to one, the more diverse the sample is. We used the following calculation to measure the evenness of in-text citations in each biological opinion:

$$D = 1 - \{(\sum n(n-1)) / (N(N-1))\}$$

Where n is the number of times a specific document was referenced in the text of the opinion and N is the total number of in-text citations in the opinion.

Central questions with regard to the operating definition of best available science are not only what is used, but also how it is used and what is not used. We first used a course measure to determine what resources were not utilized. We generated a list of available peer-reviewed, piping plover publications by searching the "Wildlife Ecology Studies Worldwide" electronic database using the search terms "Piping and Plover" and restricting our search to journal articles and monographs. This created a list of

publications on piping plovers through 2007. We decided that any article published prior to six months before the issuance of the biological opinion was available to be used by the authors of that opinion. We then compared the number of used peer-reviewed published resources to the total available resources. We acknowledge that not all piping plover articles are relevant to all consultations. For example the Loegering et al. (1995) paper on ghost crab (*Ocypode quadrata*) predation of piping plover chicks has no relevance to a consultation in the Missouri River where ghost crabs do not exist. Thus, with this course measure we would never expect a full 100% match between used and available. However, our approach does provide a relative index as to what extent available peer-reviewed published resources are being used in Section 7 consultations.

To more closely examine these two important questions, we focused our efforts on the use of quantitative demographic analyses and population models for piping plovers in the Biological Opinions. Harding et al. (2001) and Peterson et al. (2004) concluded that the success of habitat conservation planning efforts might be limited by a lack of quantitative analyses and examining the use of population models in biological opinions could lead to similar conclusions. We identified all population models for piping plovers for both the Atlantic coast breeding population and the Great Plains breeding population. We first examined whether the available models and their predictions were used in each of the opinions. We then assessed whether the USFWS gave priority to any of the population models when more than one was available and how that prioritization was determined. This assessment required qualitative evaluation of the language within the biological opinions.

RESULTS

We acquired and examined 17 Biological Opinions for piping plovers from throughout their range dating from 1990 to 2007. We acquired opinions spanning 17 years and from throughout their geographic range. The ranged in scope from evaluating the potential impacts of municipal firework displays to celebrate a national holiday, to water flow management and ecological restoration of a major river system in the Great Plains. The ranges in length from six to nearly 400 pages, and number of cite resources ranged from three to 150. Our sample was representative of the range of size and scope of issues that Section 7 Consultations have addressed for piping plovers. Three of the 17 opinions were excluded from our study. Two because the opinion concluded that plovers would not be affected by the proposed actions and the documents did not contain any citations on piping plovers and a third opinion was a “supplement” to a previously issued opinion and the document did not contain any citations.

In the documents there was a high reliance on unpublished reports (mean = 62.7% of used resources) or data (Table 1). Peer reviewed resources comprised mean of 16.5% of cited resources and graduate theses comprised just over 9% of resources on average (Table 1). There were three opinions in our sample that contained no references to peer reviewed materials. Books were approximately 5% of cited resources and personal communications comprised 6.8% of used resources (Table 1). In the text of the opinions there was again a high reliance on unpublished materials (mean = 63.1% of all the in-text citations) (Table 2). Peer reviewed materials comprised a mean of 15.4% of in-text citations and theses comprised 12.3% (Table 2). Books totaled 3.1% of in-text citations and personal communications were 5.9% of all citations on average (table 2).

Table 1: Number, and type of citations from 14 biological opinions for section 7 consultations on Piping Plover

Opinion No.	year	No. Citations	% Peer Rev.	% Theses	% Books	% Unpub.	% Pers. Comm.	% Used ^a
1	1990	72	25.00	9.72	2.78	51.39	8.33	27.69
2	1996	5	0.00	0.00	0.00	100.00	0.00	0.00
3	1997	14	21.43	28.57	7.14	42.86	0.00	1.40
4	2000	145	24.14	8.28	15.17	44.83	7.59	21.34
5	2001	17	17.65	5.88	5.88	70.59	0.00	1.74
6	2003	9	11.11	0.00	11.11	55.56	22.22	0.53
7	2003	40	22.50	25.00	5.00	40.00	7.50	4.79
8	2003	79	18.99	13.92	2.53	56.96	8.86	7.98
9	2004	23	13.04	8.70	0.00	69.57	8.70	1.54
10	2005	3	0.00	0.00	0.00	100.00	0.00	0.00
11	2005	11	0.00	0.00	0.00	90.91	9.09	0.00
12	2005	57	19.30	10.53	1.75	49.12	17.54	5.61
13	2006	117	24.79	10.26	2.56	59.83	1.71	14.36
14	2006	150	32.67	5.33	12.67	46.00	3.33	24.26
Average (S. D.)			16.47 (10.36)	9.01 (8.8)	4.76 (5.05)	62.69 (20.75)	6.78 (6.75)	7.94 (9.84)

^a The used percent of peer reviewed published resource that were available at the time the opinion was written.

Table 2: Number, type and evenness of in-text citations from 14 biological opinions for section 7 consultations on Piping Plover

Opinion No.	Year	No. in-text citations	% Peer Rev.	% Theses	% Books	% Unpub.	% Pers. Comm.	Diversity ^a
1	1990	154	18.18	16.23	1.30	59.09	5.19	0.96
2	1996	7	0.00	0.00	0.00	100.00	0.00	0.73
3	1997	47	17.02	17.02	4.26	61.70	0.00	0.88
4	2000	386	23.32	27.72	8.03	34.97	5.96	0.99
5	2001	30	23.33	3.33	3.33	70.00	0.00	0.91
6	2003	24	4.17	0.00	4.17	66.67	25.00	0.74
7	2003	102	15.69	43.14	2.94	32.35	5.88	0.96
8	2003	240	21.67	26.25	1.67	43.33	7.08	0.97
9	2004	39	28.21	7.69	0.00	58.97	5.13	0.91
10	2005	6	0.00	0.00	0.00	100.00	0.00	0.61
11	2005	29	0.00	0.00	0.00	96.55	3.45	0.73
12	2005	147	10.88	5.44	0.68	57.14	23.13	0.95
13	2006	277	24.91	20.22	1.44	52.71	0.72	0.98
14	2006	353	28.05	4.53	16.15	49.86	1.42	0.98
Average (S. D.)			15.38 (10.55)	12.25 (13.29)	3.14 (4.38)	63.09 (22.16)	5.93 (8.11)	0.88 (0.12)

^a Measure of the evenness of intext citations calculated using the Simpson's index of diversity (Simpson 1949)

The mean diversity index value across all 14 opinions was 0.88 (range = 0.61 to 0.99) (Table 2), meaning that the in-text citations of the opinions were 88% diverse. Diversity increased as the over all number of references increased on a natural log scale ($p < 0.0001$, $R^2 = 0.84$). As a point of reference we obtained and calculated the same diversity index for a Florida panther biological opinion issued in 2002. This opinion was written in the time frame that Beier et al. (2006) and Gross (2005) showed an over reliance by the USFWS on one or two flawed studies to make incidental take decisions. This opinion had a diversity index value of 0.95.

On average the opinions used only 7.9% of the available published articles on piping plover biology. Use of available published articles ranged from 0% used to 24.1%. Of the 14 opinions that we reviewed for this study, 11 opinions had one or more published population models for piping plovers available for consideration at the time of consultation. Ten of those 11 opinions referenced or used information from the population models in the text of the opinion to inform the conclusion and the jeopardy decision.

The use of population models and the results of the models varied across opinions, time and geographic range. For the Atlantic Coast breeding population, the 1996 Recovery Plan (USFWS 1996) included an attempt to model the population and generate target fledge ratios (the number of chicks successfully produced per breeding pair). The opinions that we reviewed relating to incidental take permits for the Atlantic Coast population all discussed those target fledge ratios and how the proposed actions under investigation would affect achievement of fledge ratio goals. Several East Coast opinions concluded that the proposed actions were acceptable as long as average fledge

ratios met the prescribed recovery plan targets. Plissner and Haig (2000) developed a spatially explicit meta-population model for Piping Plovers as a species that subdivided the population into sub-populations with varying rates of productivity survival and dispersal amongst the sub-populations. This model and its results were seldom used or referenced in the Atlantic Coast biological opinions that we reviewed.

In the Great Plains region, there have been three published population models since 1993 (Ryan et al 1993, Plissner and Haig 2000, and Larson et al 2002). These models vary in mathematical structure and complexity. Ryan et al. (1993) presented a relatively simple, stage structured, stochastic, matrix model. The opinions frequently noted Ryan et al.'s (1993) conclusion that if adult survival was held constant at 66% and immature survival were held at 60%, fledge ratios of 1.13 chicks produced per breeding pair were required to maintain a stable population size. This analysis made several key assumptions about annual survival that at the time were unable to be verified with empirical data (Ryan et al 1993). In 2000, Plissner and Haig published the aforementioned spatially explicit meta-population model. The Opinions noted that Plissner and Haig (2000) concluded that 2.0 fledglings per breeding pair would be required to maintain a stable population of piping plovers in the Great Plains. Finally, Larson et al (2002) developed and published a habitat based population model for piping plovers in the Great Plains that separated the population into two habitat types but did not require any assumptions about the spatial dynamics of the population.

We reviewed three opinions from the Great Plains that were produced after one or more of these models were published. Two of those three discussed the implications of the results from these models in the text of the opinions. However, when multiple

models were available that presented differing views and conclusions about plover population dynamics, the opinions relied more heavily on the model that resulted in the lowest required fledge ratios to maintain population stability. In one notable case where all three Great Plains population models were published well in advance of the opinion, the opinion discussed all three but used the Ryan et al. (1993) 1.13 fledgling per pair conclusion to guide management decisions. The text gave no reason for disregarding the other, more recent and mathematically complex population models. We acknowledge that there are numerous methodological criticisms to be made about each of these population models and there are certainly valid reasons for choosing one model as better than the others for any given incidental take situation. However, no criticisms were leveled and no reasons supporting the choice of Ryan et al. (1993) over the others were given. Without discussion of the relative merits of the different models the USFWS is vulnerable to criticisms that the Ryan et al. (1993) model was chosen because it requires the lowest fledge ratios which are easier to obtain and require less management and fewer restrictions.

The third opinion that referenced none of the published population models chose instead to rely on information on population modeling and viability provided in a personal communication by Mike Larson and Mark Ryan (co-authors on the Larson et al. (2002) population model) even though all three models were published at least one year prior to the issuance of the opinion. The communication referenced apparently provided information on population modeling exercises directly related to the incidental take question at hand, and may in fact be the best available science because of its immediate relevance. However, the opinion did not reference or consult prior and published

population models. It again did not discuss the reasons for considering information from an unpublished personal communication as “best” relative other peer reviewed and published information.

DISCUSSION

Our evaluation shows a disproportionate reliance on unpublished materials in the Section 7 Consultation process for piping plovers. There are some potential benefits to using unpublished resources. Data on the local status of plover populations and on recent breeding success for example may only be available in unpublished annual reports from the USFWS. These reports might allow for up to date, site-specific information to be included in the consultation process. However, these data sources are not reviewed by the general scientific community. Without peer review, methodological flaws or faulty inferences are less likely to be exposed, criticized and revised (Armstrong 1997). We also documented that few of the available peer reviewed published articles are being referenced by the authors of the biological opinions that we reviewed. Because the biological opinions themselves are not peer reviewed (Smallwood et al. 1999), use of well vetted, peer-review literature might be even more important.

Alternatively, Beier et al. (2006) and Conroy et al. (2006) demonstrated that peer reviewed articles are not without error and may not be the best available science. We conclude from their work and that of Gross (2005), that it is important to utilize a variety of resources. Our analysis of the diversity of the in-text citations is at best inconclusive. Eighty-eight percent diverse seems to be high level of diversity, however when compared to the one Florida panther Opinion (95% diverse) that was written in the time frame that Beier et al. (2006), Conroy et al. (2006) and Gross (2005) identified as problematic, 88%

may be low. It might be better to evaluate diversity on a per author basis rather than a per article basis. The Florida panther opinion that we examined had 46% of all the in-text citations referencing 14 different reports or articles by one lead author. In fact almost 23% of all the in-text citations came from two reports by the same author and 24% of the articles in the literature cited were by one lead author. This suggests a heavy reliance on the science and conclusions of one individual. Though some of the piping plover opinions we reviewed had as many as 45% of the in-text references attributed to a single lead author in the literature cited, those resources were agency papers and products attributed to the USFWS, not single independent authors. Papers and reports attributed to an agency can have numerous different authors for each document and over time.

Our examination of the use of quantitative population models in these biological opinions revealed an interesting pattern. The Federal Register states “As part of their routine activities, service biologists are required to gather, review, and evaluate information from these sources prior to undertaking listing, recovery, consultation and permitting actions.” (FR, July 1, 1994 (Vol. 59), pg. 34271). In most of the biological opinions that we examined, information was gathered from a variety of sources and there was moderate to high diversity of reliance on those varied resources. However, when closely examining the use and evaluation of published population models that have different structure, results and conclusions, we found little or no “evaluation” of the differing models and little or no explanation for placing higher value on one model over any other. We again acknowledge that each of the population models are open to criticism and may vary in usefulness to each of the incidental take decisions and biological opinions that we examined, but we find it troublesome that often no

justification was given for choosing the conclusions and management recommendations of one model over another.

This work represents one of the first attempts to examine and evaluate the application of the “best science available” concept into real world endangered species management. We conclude that in the biological opinions we examined, managers are using a diverse array of information resources to evaluate the potential effects of incidental take. However, consulting a variety of resources does not automatically lead to using the “best science available.” It is clear that service biologist made some evaluative judgments and determined which resources were the “best” science for specific consultations. What is not clear, in the case of quantitative population modeling at least, is how those judgments were made and what the basis is for those judgments.

From this study we conclude with three recommendations for use in future consultations and incidental take management:

- Use caution when relying on unpublished resources. They can provide locally usefully and timely information however they are typically not peer reviewed and may lack methodological and scientific rigor.
- It is important for the authors of biological opinions to explicitly justify their reasons for choosing one scientific study as better than some other, similar study with differing results and conclusions. Not justifying these choices leaves the concept of “best available science” open to criticism.
- The service should consider a peer review process for biological opinions. These documents and the decision therein can have substantial impacts on the recovery of a listed species. Therefore opening the consultations to

outside, independent, and objective review will improve the process and help ensure that the “best science” is truly best.

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VITA

Conor McGowan, was born in Long Island, NY in 1978. His childhood years were spent in Long Island and in London, England, where his family resided from 1988 to 1992. He attended Garden City High School and after graduation he went to Wake Forest University in Winston-Salem North Carolina where he majored in Biology. After one year off from school pursuing field-based avian conservation jobs in Oregon, Hawaii, and coastal North Carolina, he started a master's degree in Zoology at NC State University. He studied the breeding ecology of American Oystercatchers in coastal North Carolina. Upon completion of his thesis, he came to the University of Missouri to pursue a Ph.D. in Fisheries and Wildlife Sciences. Conor has studied birds in the Galapagos Islands, Hawaiian rain-forest, old-growth forests of the Pacific Northwest, barrier islands of North Carolina's coast, and the prairies and rivers of the northern Great Plains. He has published four papers from his master's degree research, one field note on piping plover nest site selection and is looking forward to publishing this dissertation.

Conor married his best friend Cate Calhoun in August of 2003. They are proud parents of Eva (born November 8th 2006), to whom this dissertation is dedicated.