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BALD EAGLE SURVIVAL AND POPULATION DYNAMICS IN ALASKA AFTER THE *EXXON VALDEZ* OIL SPILL

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Abstract: We investigated age-specific annual survival rates for 159 bald eagles (*Haliaeetus leucocephalus*) radiotagged from 1989 to 1992 in Prince William Sound (PWS), Alaska. We monitored radio-tagged eagles for ≤3 years beginning 4 months after the *Exxon Valdez* oil spill. There was no difference ($P > 0.10$) in survival rates between eagles radiotagged in oiled areas and eagles radiotagged in unoiled areas of PWS. Pooled annual survival rates were 71% for first-year eagles, 95% for subadults, and 88% for adult bald eagles. Most deaths occurred from March to May. We found no indication that survival of bald eagles radiotagged >4 months after the oil spill in PWS was directly influenced by the spill and concluded that any effect of the spill on survival occurred before eagles were radiotagged. A deterministic life table model suggests that the PWS bald eagle population has an annual finite growth rate of 2%. Given the cumulative effects of direct mortality and reduced productivity caused by the oil spill, we predicted that the bald eagle population would return to its pre-spill size by 1992.

Key words: Alaska, bald eagle, *Haliaeetus leucocephalus*, mortality, oil spill, population dynamics, Prince William Sound, radio telemetry, survival.

Prince William Sound, Alaska, provided year-round and seasonal habitat for about 5,000 bald eagles (Bowman et al. 1993). Bald eagles in PWS were associated with shoreline habitats, and nearly all nests occurred within 200 m of the beach (Hodges and Robards 1982). Extensive areas of shoreline were contaminated with oil after the oil tanker *Exxon Valdez* ran aground on 24 March 1989, spilling about 40,000 kL of crude oil into PWS. From April to August 1989, an estimated 247 bald eagles died in PWS as a result of the oil spill (Bowman et al. 1993).

Banding data for most raptors are inadequate to estimate survival rates because of small sample sizes, low rates of return, and lack of adult banding (Brownie et al. 1985). Radio-telemetry studies can provide more precise estimates of survival (White 1983, Buehler et al. 1991). Buehler et al. (1991) observed 100% survival of radio-tagged first-year eagles and 83–92% survival for older age classes in Chesapeake Bay. Wood (1992) used radio telemetry to determine minimum annual survival rates of 63, 84, and 94% for first-year, second-year, and third-year eagles in Florida, respectively.

This study was prompted by concern over the welfare of eagles exposed to areas affected by the *Exxon Valdez* oil spill. Our objectives were to determine if the spill influenced annual survival rates of bald eagles that survived the initial effects of the spill, compare survival rates among age and sex classes, and predict the time for the population to recover from the spill.

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STUDY AREA

Prince William Sound encompassed about 39,000 km² including 4,800 km of shoreline. A temperate rainforest of Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*) covered most of the convoluted coastline and islands of PWS. Summer temperatures ranged between 7 and 22 C, and winter temperatures seldom dropped below −20 C (Brower et al. 1988). Annual precipitation exceeded 400 cm in some areas of PWS (Brower et al. 1988). Prince William Sound supported salmon (*Oncorhynchus* spp.) and herring (*Clupea harengus pallasi*) fisheries, and salmon populations were augmented by hatchery operations. The nearby Copper River Delta also supported spawning runs of salmon and eulachon (*Thaleichthys pacificus*), and large numbers of waterfowl, shorebirds, gulls, and marine mammals. See Isleib and Kessel (1973) for a more detailed description of the region.

METHODS

Telemetry

We radiotagged 69 nestling bald eagles in 1989–90 when they were about 8 weeks old. We selected nestlings on the basis of their nest location in relation to shoreline oiling and accessibility of nests. We radiotagged approximately equal numbers of nestlings in western PWS, where most oil washed ashore, and eastern PWS, which was not directly oiled. We returned all nestlings to their nests immediately after tagging.

We radiotagged 79 adult and 3 3–4-year-old bald eagles between July and October 1989–91, and 6 adults and 2 3–4 year olds in January 1992. We captured approximately equal numbers of adult eagles in oiled and unoiled areas of PWS. We captured adult bald eagles in marine areas using floating, noosed fish (Cain and Hodges 1989). We captured nonbreeding adult and subadult eagles incidentally to breeding adults. We established the sex of eagles using an index based on bill depth and hallux length, where negative indices are males and positive indices are females (Bortolotti 1984). However, we considered eagles with indices between 0 and 1 unknown sex because we believe this model may misclassify large male bald eagles in Alaska as females. We promptly returned all adult and subadult eagles to the same location where they were captured.

Radio transmitters weighed 65 or 90 g and were mounted via a backpack harness made of tubular teflon ribbon. Transmitters were 1.2–1.7% of adult body mass. On nestlings, we left harnesses slack to allow for growth. Transmitters had a life expectancy of about 3 years and were equipped with mortality sensors that doubled the pulse rate after 5–7 hours without motion.

We used a Cessna 180 or 185 for aerial radiotracking using standard techniques (Gilmer et al. 1981). In PWS, we relocated birds weekly until March 1991 and then monthly until September 1992, and less frequently along the Pacific coast between Homer and Yakutat, Alaska. We also searched twice south to Seattle, Washington, and west to Kodiak Island, Alaska. Maximum reception distance of signals varied 5–150 km, depending on altitude, local topography, and individual transmitter strength.

We retrieved dead eagles as soon as possible. We froze carcasses and shipped them to the U.S. Fish and Wildlife Service, National Wildlife Health Research Center (Madison, Wis.) for necropsy. We retained selected tissues for histological examination.

Survival Rates

We estimated annual survival rates using the Kaplan-Meier procedure modified to enable staggered entry of individuals (Pollock et al. 1989). Using the log-rank Chi-square test described in Pollock et al. (1989), we tested hypotheses about differences in survival between treatment groups. These procedures enable staggered entry and censoring of individuals and do not assume constant daily survival rates. Censored birds could have included birds that emigrated from the study area, birds that lost or had weak or failed transmitters, or birds that died in the study area and were not detected. Assumptions required for the Kaplan-Meier procedure were (1) birds of each age and sex class were sampled randomly, (2) survival times were independent for individual eagles, (3) survival of eagles was not influenced by capturing or radiotagging, (4) censoring was not related to an animal's fate, and (5) newly radio-tagged animals had the same survival rates as previously radio-tagged animals (Pollock et al. 1989). We believe that these assumptions were met.

We estimated survival rates during 3 survival years: a survival year was defined as 1 September–31 August. We censored missing eagles from the risk set (those known to be alive) the week

after they were last known alive during a particular survival year. If a censored bird returned in a subsequent year, it was added when first found and contributed data to the new year's estimate, but its censored status at the end of the previous year did not change. If assumptions about random censoring are met, censoring affects only the confidence limits but not survival estimates (Pollock et al. 1989).

Although nestlings were radiotagged when about 8 weeks old, they were not considered at risk until fledging; we used 1 September as the approximate fledging date. We estimated ages of subadult eagles (1–4 yr old) by plumage characteristics following McCollough (1989). Adult eagles were those with definitive plumage and were assumed to be ≥5 years old (McCollough 1989). We defined a territorial adult eagle as one that showed obvious fidelity to a nest site. We considered adults and subadults at risk the week after radiotagging. An individual eagle could contribute to survival estimates for ≤3 survival years.

We compared survival rates among subsets of radio-tagged eagles to assess oil spill effects on their survival. We conservatively assumed that the availability and toxicity of oil to eagles was negligible by September 1991, the start of the third survival year, and therefore limited all tests of oiled versus unoiled groups to the first 2 years after the spill. We lacked knowledge about the potential exposure to oil for free-flying eagles prior to radiotagging, so we grouped eagles into oiled or unoiled groups using 3 criteria. First, we simply grouped eagles by radio-tagging region into western PWS (oiled) or eastern PWS (not oiled). Second, we grouped eagles according to shoreline oiling near the radio-tagging location. Cumulative shoreline oiling data through August 1989 were provided by the *Exxon Valdez* Oil Spill Damage Assessment Geoprocessing Group (1991). If any shoreline was oiled within a 483-m radius of the capture site, the eagle was grouped into the oiled category. This was the mean radius that included the area used intensively by nesting eagles as determined by frequent radio-telemetry relocations (Bowman et al. 1993). Finally, we grouped eagles according to their movements and potential exposure to oiled areas after radiotagging. If >50% of relocations for an eagle were in western PWS, the eagle was considered exposed to oil. We did not assume that oil exposure required direct contact with oil or rocks

coated with oil. Eagles foraging in PWS may have ingested oil residues in the tissues or on the surface of food.

We estimated survival rates for 3 age groups: first-year eagles, subadults, and adults. We compared survival rates between years for the same age classes, among age classes, between sexes (for ad only), and for territorial adults radiotagged in oiled or unoiled areas.

We used a Monte Carlo simulation with 1,000 iterations to estimate power $(1 - \beta)$ of tests to detect differences in survival rates between groups at $\alpha = 0.10$. We specified sample size and estimated survival rate for each group. For each iteration, the program generated a random number of deaths per sampling period, with the mean number of deaths dictated by the specified survival rate, and then calculated a survival rate (Kaplan-Meier) (Pollock et al. 1989) for each group and a test statistic for each comparison. We assumed that H_a was true and the proportion of times that H_o was rejected was an empirical estimate of power.

Population Dynamics

We used a deterministic life table (Grier 1980) to model bald eagle population dynamics in PWS and estimate the finite rate of population growth (λ). The model incorporated the maximum number of breeding sites (a nest territory occupied by a pair of eagles) available, the percentage of females that successfully produced young, the average number of young fledged per breeding female, age at first breeding, and survival rates for first-year and for older eagles. We estimated reproductive parameters from concurrent reproductive studies of eagles (Bowman et al. 1993). The model assumes that survival rates are constant after the first year of life. Because survival rates were not different ($P > 0.10$) for eagle age classes after the first year, we used a pooled annual survival rate of 0.90 (SE = 0.02) for eagles >1 year old.

Reproductive rates for eagles in PWS were reduced in 1989 after the oil spill (Bowman et al. 1993), but rebounded in 1990 to levels typical of bald eagle populations in other areas of coastal Alaska. Therefore, we used reproductive rates observed during a survey of 622 occupied nests in 1990 in PWS to represent baseline conditions: 57% (SE = 0.02) of females produced young and averaged 1.5 (SE = 0.03) young/successful nest.

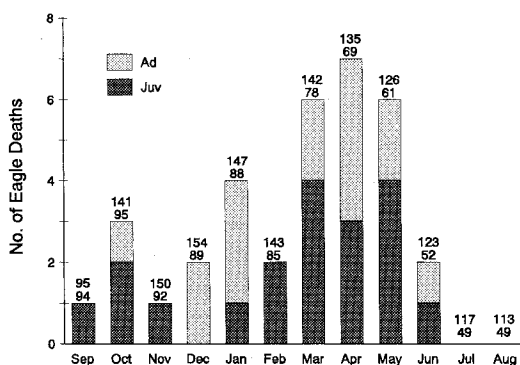We estimated the number of occupied breed-

Fig. 1. Numbers of radio-tagged adult (>5 yr old) and juvenile (<2 yr old) bald eagles dying each month in Prince William Sound, Alaska, 1989–92. Numbers above bar indicate adults (top) and juveniles (bottom) at risk.

ing sites at 1,200–1,300. We assumed this number of sites approached saturation level because nest densities (about 0.6 occupied nests/km of shoreline) and habitat were similar to southeastern Alaska where high rates of nonbreeding adults occur (Hansen and Hodges 1985). We assigned 1,300 as the maximum number of breeding sites available.

We estimated that eagles first bred at 8 years of age for 2 reasons: (1) delayed breeding is typical for healthy bald eagle populations in near-pristine environments (Hansen and Hodges 1985, Gerrard and Bortolotti 1988), and (2) we radiotagged 3 eagles as 4 year olds, but none had bred by the end of the study when they were 7 years old. We assumed that there were no unmated eagles ≥8 years old, and that eagles remained reproductively active until death. If eagles bred earlier than 8 years, population growth would be underestimated by the model; population growth would be overestimated if eagles first bred at ages >8 years, if breeding attempts do not occur every year, or if breeding declines in elderly eagles. To assess sensitivity of the model to changes in these parameters we varied values for reproductive parameters, and for survival, in repeated simulations while holding other parameters constant.

We estimated 90% confidence limits on λ using Monte Carlo simulation with 2,000 iterations. Values for each parameter in the model, excluding age at first breeding, were randomly assigned using a normal distribution with a known mean and variance. We modeled age at first breeding as a uniform distribution between 7 and 9 years.

We used a deterministic population growth model and computer software developed by Grier (1980) to predict future population levels and evaluate cumulative effects of direct mortality and impaired productivity of eagles. We portrayed 2 scenarios: 1 modeled population growth after 1989 in the absence of the oil spill (starting population of 5,242 eagles), and the second modeled growth assuming 223 fewer young were produced and 247 eagles died because of the spill in 1989 (starting population of 4,772 eagles) (Bowman et al. 1993). Because we did not know the age distribution of eagles killed in the spill, we assumed that it mirrored the age-class distribution in 1989, which was determined using a spring 1989 population index adjusted for unseen adults and the percentage of immatures in the population (see Bowman et al. 1993) and survival rates from this study. For purposes of population modeling, we assumed that the oil spill adversely affected reproduction and killed eagles only in 1989 and not afterward. This assumption is supported by studies of bald eagles in PWS (Bowman et al. 1993, White et al. 1993).

RESULTS

Survival of Radio-tagged Eagles

Thirty-four of the 159 radio-tagged eagles (15 ad, 2 subad, 17 first-yr birds) died prior to September 1992. Most mortality occurred in March, April, and May (Fig. 1). Sixteen of the 34 carcasses were recovered and suitable for necropsy (8 ad, 8 fledglings); preliminary diagnoses included emaciation (5), fractures, punctures, or lacerations (7), drowning by another eagle (1), trapping (1), and undetermined (2). We suspected ≥4 of the dead adults to have died as a result of aggressive encounters with other eagles (1 each in Jan, Apr, Jun, and Oct) on the basis of injuries sustained and an eyewitness account of the drowning incident.

Survival rates of first-year eagles were lower than for subadult ($\chi^2 = 8.212$, 1 df, $P = 0.004$) and adult ($\chi^2 = 8.565$, 1 df, $P = 0.003$) eagles (Table 1). Survival rates for subadult eagles were not different ($\chi^2 = 0.222$, 1 df, $P = 0.630$) from survival rates for adults. Survival was not different ($P > 0.10$) among years for any age class. There was no difference in survival rates between adult males and females ($\chi^2 = 0.273$, 1 df, $P = 0.600$). Estimated survival for radio-tagged eagles through the first 4 years of life

Table 1. Annual survival rates, by age group, for bald eagles radiotagged in Prince William Sound, Alaska, 1989–92.

Age class[a]	No. of indi- viduals	No. of eagle-yr[b]	Annu. survival	SE
First-yr	68	68	0.71	0.07
Subad	41	53	0.95	0.04
Ad >5 yr old	84	142	0.88	0.03
M	41	63	0.86	0.05
F	35	63	0.90	0.04

[a] Age-class periods are 1 Sep–31 Aug intervals beginning at fledging.
[b] An eagle-yr equaled 1 eagle that contributed to the survival rate for a particular year (e.g., an eagle monitored for all or part of 3 survival years contributed 3 eagle-yr).

was 61%. Given an adult survival rate of 0.88, we estimated the average life-span once eagles reach maturity (5 yr) is 19 years (Perrins and Birkhead 1983).

Oil Spill Influence on Survival

Necropsies of 16 radio-tagged eagles indicated no specific evidence of oil-caused injuries. We found no differences ($P > 0.10$) in survival rates of eagles caught in eastern versus western

PWS, or of eagles caught near versus far from oiled shorelines (Table 2).

First-year eagles that were relocated in oiled areas had lower survival than eagles that frequented unoiled areas in 1989 and had greater survival in 1990 (Table 2). These comparisons may be misleading because only 3 first-year eagles frequented oiled areas in 1989, of which 2 died. Necropsies of the 2 first-year eagles revealed no physical or histological damage indicative of hydrocarbon contamination.

All territorial adults from oiled areas survived the first year after the spill, whereas survival of eagles from unoiled areas was 0.87. There was no difference ($\chi^2 = 0.4$, 1 df, $P = 0.525$) in survival rates of territorial adults from oiled (0.86, SE = 0.08) and unoiled (0.92, SE = 0.05) areas pooled for the 2 years following the spill.

Population Dynamics

The deterministic life table model suggested a finite population growth rate of 1.02 (90% CI = 0.98–1.05) for the bald eagle population in PWS. The model was robust to changes in reproductive rates and survival rates for first-year

Table 2. Estimated annual survival rates ($\hat{S}$) for radio-tagged bald eagles in Prince William Sound (PWS), Alaska, after the *Exxon Valdez* oil spill. Comparisons are on the basis of radio-tagging region, oil near capture site, and movements after radiotagging.

Exposure criteria Age	Year	Oiled			No oil			χ^{2c}	P^d	$1 - \beta^e$
		$\hat{S}^a$	n^b	SE	$\hat{S}$	n	SE			
Tagging region[f]										
First-yr	1989	0.79	15	0.12	0.76	15	0.15	0.02	0.89	0.14
	1990	0.52	19	0.16	0.70	19	0.14	0.01	0.96	0.24
Second-yr[g]	1990	0.76	11	0.16	1.00	8	0.00	1.52	0.22	0.52
Ad[h]	1989	0.94	17	0.07	0.85	15	0.10	0.55	0.46	0.15
	1990	0.84	20	0.09	0.77	14	0.12	0.33	0.57	0.17
Oil <483 m from capture site										
First-yr	1989	0.83	13	0.11	0.73	17	0.16	0.49	0.48	0.26
	1990	0.47	15	0.17	0.74	23	0.16	0.26	0.61	0.29
Second-yr	1990	0.86	10	0.14	0.86	9	0.14	0.01	0.91	0.10
Ad	1989	0.92	13	0.08	0.88	19	0.08	0.11	0.74	0.10
	1990	0.77	14	0.13	0.84	20	0.08	0.18	0.67	0.17
Movements after radiotagging										
First-yr	1989	0.33	3	0.27	0.83	27	0.09	6.43	0.01	0.10
	1990	1.00[i]	5	NA	0.62	33	0.10	NA	NA	NA
Second-yr	1990	1.00	3	0.00	0.83	16	0.13	0.55	0.45	0.33
Ad	1989	0.92	13	0.09	0.88	19	0.08	0.11	0.74	0.10
	1990	0.80	16	0.11	0.83	18	0.09	0.01	0.93	0.10

[a] Survival rates calculated from 1 Sep to 31 Aug, using Kaplan-Meier procedure (Pollock et al. 1989).
[b] No. of eagles that contributed to the survival estimate.
[c] df = 1.
[d] Log-rank test of H_o: survival equal between oiled and no oil groups.
[e] Estimated power for test ($\alpha = 0.10$) of the above H_o, for specified differences and sample sizes as observed.
[f] West. PWS = oiled, east. PWS = no oil.
[g] Radiotagged as juv in 1989.
[h] 3 3–4 yr olds were included in adult category in 1989, 1 in 1990.
[i] Survival was 1.0 up to mid-April, when the last of the 5 birds in the subset was censored. Thereafter, survival was unknown. CI was not computed.
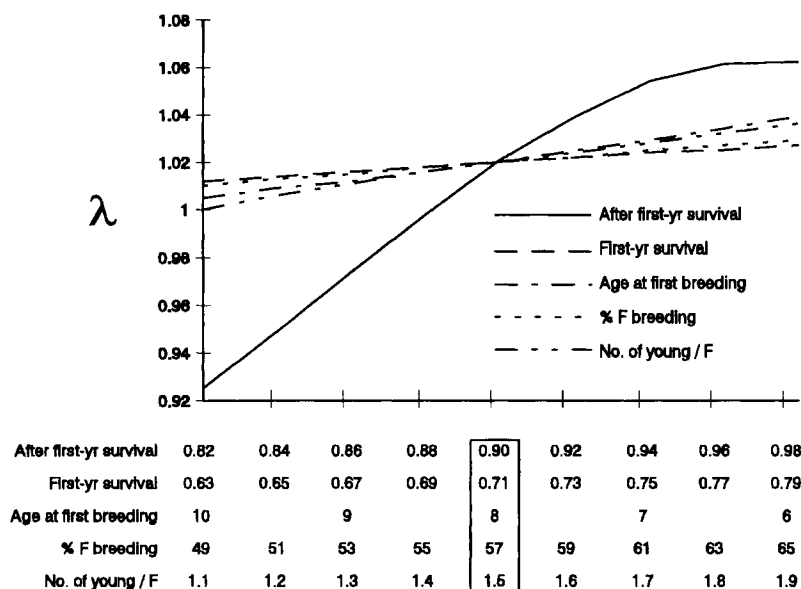
After first-yr survival	0.82	0.84	0.86	0.88	0.90	0.92	0.94	0.96	0.98
First-yr survival	0.63	0.65	0.67	0.69	0.71	0.73	0.75	0.77	0.79
Age at first breeding	10		9		8		7		6
% F breeding	49	51	53	55	57	59	61	63	65
No. of young / F	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9

Fig. 2.   Sensitivity of bald eagle population growth rate (λ) to changes in each demographic parameter, holding other parameters at their estimated values (highlighted in center column).

eagles, but sensitive to changes in survival rates for older (after first-yr) eagles (Fig. 2).

When estimated direct mortality and lost production caused by the oil spill in PWS are incorporated into the model, the injured population numbered 4,948, 5,091, and 5,234 eagles in the 3 years after the spill, respectively. We therefore predicted that, with $\lambda = 1.02$ and in the absence of an effect on survival rates, it would take about 3 years for the population to return to its estimated 1989 size of 5,242 eagles.

DISCUSSION

We believe that the eventual disappearance of about 40 radio-tagged (and not confirmed dead) eagles from our study area was due to transmitter failure or emigration, but not death. We documented transmitter failure (manifested by a drastic drop in signal strength) for 42% of territorial adults while we flew near each traditional perch. We could not document this type of transmitter failure for juveniles because they were nomadic and did not perch predictably in the same location. We censored 56% of juveniles because they disappeared for part of a survival year. Transmitter failure would account for most missing juveniles if we assume that transmitters on juveniles failed at the same rate as transmitters on territorial adults. Juveniles ranged widely and some likely emigrated as well.

Survival Rates

Because eagles were radiotagged >4 months after the oil spill occurred, we could not measure survival rates during the period when many eagles were known to have died (Bowman et al. 1993). We used 3 approaches to estimate exposure of eagles to oil because we could not ascertain each eagle's exposure to oil. Each approach had its limitations. By using only the general radio-tagging region or specific capture site to group eagles, we could not take into account potential exposure to oiled areas after the bird was radiotagged. Using radio-telemetry relocations to estimate exposure to oiled areas after tagging, we were unable to consider risk of exposure that could have occurred before we radiotagged birds. Perhaps the best comparison of survival between birds from oiled and unoiled areas is that for territorial adults. These birds remained in the oiled or unoiled areas where they were radiotagged and should have served as good indicators of oil contamination.

Sample sizes for groupings used in oiled versus unoiled comparisons were small, and power to detect biologically significant differences in survival rates between groups of eagles was poor (0.10–0.52). Biologically significant could be defined differently depending on the particular population and its status. Let us assume that a decrease of 10% in after first-year survival rate

(from 0.90 to 0.80) is biologically significant. To achieve power of 0.80 to detect a 10% difference in survival rates at $\alpha = 0.10$, we would need a sample of about 150 radio-tagged eagles in each subset, a practical impossibility. Pollock et al. (1989) recommended that 40–50 animals be radiotagged at all times to obtain good precision of survival estimates. This study illustrates, however, that much larger samples (or long-term monitoring to enable pooling across years), or lower levels of significance, may be necessary to achieve adequate power to detect small changes in survival rates.

Despite the lack of power to detect differences in survival due to the oil spill, our data do not provide evidence that the spill decreased eagle survival. Our assertion is based on (1) survival rates for most of the oiled subsets that were similar or apparently higher than survival for eagles in unoiled subsets (Table 2), (2) no evidence of oil-induced damage in tissues of radio-tagged eagles that died, and (3) 100% survival of territorial adults in the year following the oil spill.

Population Dynamics

The cumulative survival rate of bald eagles in PWS through the first 4 years of life was higher than rates reported by Brown and Amadon (1968) and Sherrod et al. (1976), but similar to survival rates for bald eagle populations in Chesapeake Bay (Buehler et al. 1991) and Maine (McCollough 1986). Survival of Maine eagles may have been abnormally high because of a winter feeding program that enhanced survival (McCollough 1986).

Chesapeake Bay and Maine populations are likely recovering from depressed levels, and there is apparently little competition for food, nest sites, or favorable winter habitat in these areas (McCollough 1986, Buehler et al. 1991). Buehler et al. (1991) predicted a finite population growth rate of 5.8–16.6%/year for the eagle population in Chesapeake Bay. The Maine bald eagle population is expected to increase 3.1–6.3%/year (McCollough 1986). The PWS population is apparently increasing at a slower rate than these populations.

Given similar survival rates among these 3 populations, only decreased productivity or increased emigration can account for the lower rate of population growth in the PWS population. We estimated that 57% of occupied nests were successful and the number of young produced per occupied nest in PWS in 1990 was 0.86 (Bowman et al. 1993); the 1986–90 average for Chesapeake Bay was 75% success for occupied nests and 1.32 young/occupied nest (Buehler et al. 1991). When reproductive rates for eagles in Chesapeake Bay are substituted into the PWS model, the resulting rate of population growth is 4.8%. This rate of growth is still higher than the rate for the PWS population and suggests that other factors, such as delayed age at first breeding, also may be responsible for lower population growth in PWS. Delayed age at first breeding would be expected for raptor populations at carrying capacity (Newton 1979). In southeastern Alaska a high rate of nonbreeding in adult eagles was found by Hansen and Hodges (1985). We observed delayed breeding (or nonbreeding) in our small sample of eagles radiotagged as subadults and monitored for 3 years.

There was also a high incidence (50% of identified causes) of adult mortality due to aggressive encounters between eagles. Densities of occupied bald eagle nests in PWS are at least as high as densities anywhere eagles breed, including southeastern Alaska (Corr 1974, Robards and Hodges 1977, Hodges 1982, Hodges and Robards 1982). Observations suggest that density dependent factors may be regulating population growth in PWS.

MANAGEMENT IMPLICATIONS

Using estimated or hypothetical survival rates and productivity data, Young (1968) and Grier (1980) illustrated that changes in survival rates have a greater effect on bald eagle populations than similar changes in reproductive rates, and suggested that rates of reproduction may be relatively inconsequential to bald eagles, a species with delayed breeding and a long life-span. Our population model supports this observation. Additional reproductive studies in PWS are required to confirm accuracy of the reproductive rates used in the model, but the model demonstrates that they will have less effect than survival rates on population growth.

Our estimates of eagle survival suggest that the population is increasing slowly and that the eagle population should have recovered from the *Exxon Valdez* oil spill by 1992. However, sampling error on survival estimates precludes a definitive conclusion about the direction and magnitude of the population trajectory. We suggest that population surveys with high precision

be conducted periodically to confirm recovery of the eagle population in PWS.

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