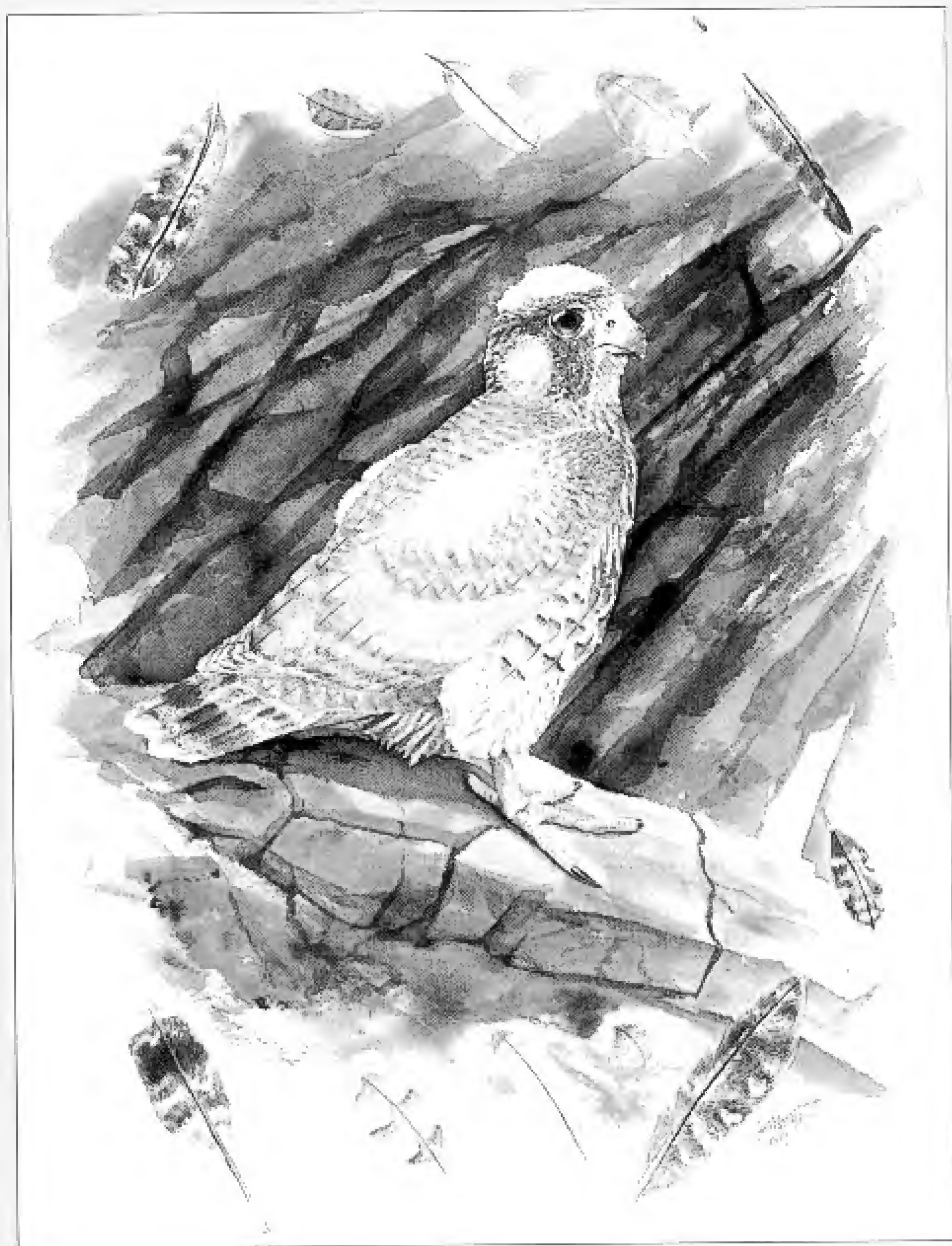


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COVER: Nestling Peregrine Falcon (*Falco peregrinus*) and selected feathers exhibiting schizochromism (see page 200). Painting by John Schmitt.

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POPULATION STATUS OF BALD EAGLES BREEDING IN WASHINGTON AT THE END OF THE 20TH CENTURY

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ABSTRACT.—From 1980–98 the population of Bald Eagles (*Haliaeetus leucocephalus*) nesting in Washington increased ($P < 0.001$) at an exponential, annual rate of 10% as adult eagles reoccupied habitat vacated during the period of widespread persecution and DDT use. Further indications of population health were linear increases in the rates of nest occupancy, productivity, and nest success. Productivity and nest success of eagles affected by contaminants along Hood Canal and the Washington side of the Columbia River estuary also increased during the study period but remained below statewide averages. By 1998, the population was widely distributed, with 89% of pairs nesting west of the Cascade crest, and 11% east of the crest. There were indications that the population stabilized from 1993–98, when statewide occupancy rates decreased ($P = 0.040$), and productivity and nest success stabilized. Modeling predicts that a statewide population of 733 breeding pairs at carrying capacity would, after 25 yr, provide an equilibrium population of 4913 eagles. Stability of the statewide population of Bald Eagles seems to be less dependent on productivity rates than on adequate numbers of replacement adults, as maintained through high survival.

KEY WORDS: *Bald Eagle, Haliaeetus leucocephalus; breeding; population status; productivity; recovery; Washington.*

Status poblacional del águila calva en reproducción en el estado de Washington a finales del siglo 20

RESUMEN.—Desde 1980–98 la población de águilas calvas (*Haliaeetus leucocephalus*) en anidación en Washington ha aumentado ($P < 0.001$) en una tasa exponencial del 10% debido a la reocupación del hábitad vacante durante el periodo de persecución directa y uso de DDT. Algunos indicadores adicionales de una población saludable fueron el incremento lineal en las tasas de ocupación de nidos, su productividad y el éxito de anidación de las águilas afectadas por los contaminantes a lo largo del canal de Hood y el costado del estuario del Río Columbia el cual también aumento durante el estudio pero que permaneció por debajo de los promedios del estado. En 1998, la población estaba ampliamente distribuida, con 89% de las parejas anidando en el oeste de Cascade Crest y 11% al este. Hubo síntomas de que la población se estabilizo desde 1993–98, cuando a nivel del estado, las tasas de ocupación disminuyeron ($P = 0.040$) y la productividad y el éxito de anidación se estabilizaron. Un modelo elaborado establece que la población a nivel del estado de 733 parejas en anidación, a su máxima capacidad de carga, después de 25 años resultaría en una población en equilibrio de 4913 águilas calvas. Finalmente, la estabilidad de la población a nivel del estado, de águilas calvas parece ser menos dependiente las tasas de productividad que de los números adecuados del reemplazo de adultos mantenidos por un alta sobre vivencia.

[Traducción de César Márquez]

For the past 25 years, the population of Bald Eagles (*Haliaeetus leucocephalus*) breeding in Washington has been extensively surveyed, researched, and managed in an effort to recover the species from

state and federal threatened status. In the 1970s, 114 nesting pairs produced a mean of 0.75 young/occupied territory (Grubb 1976). By 1985, the population had increased to 227 pairs, but productivity remained below that of other populations (McAllister et al. 1986). Surveys since the 1980s

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documented a further increase in the breeding population (Washington Department of Fish and Wildlife [WDFW], Heritage Data Base unpubl. data). The need to reevaluate the recovery status of the species prompted a review of the population (Stinson et al. 2001). Here, we report the results of that assessment for breeding eagles in Washington, including an analysis of nesting success, population numbers, and distribution. To simulate the consequences of environmental perturbations on the stability of the nesting population, we model population size and structure at carrying capacity under various vital rate regimes.

METHODS

During 1980–92, statewide Bald Eagle nest occupancy was assessed from airplane surveys conducted in early April, and productivity from helicopter surveys in early June (McAllister et al. 1986, Watson 1993). From 1993–98, biologists visited all historic nests each year during occupancy surveys, but did not conduct comprehensive productivity surveys. During that period, limited funding and volunteer efforts resulted in the documentation of nest success and productivity for a non-random sample of 28–47% of occupied territories each year. We are unaware of any overt biases in the non-random samples due to changes in survey technique (i.e., air vs. ground), distribution of sites surveyed, or changes in surveyors, that might have affected parameter estimates.

We estimated three parameters from survey information, including (1) nest occupancy—the proportion of territories with one incubating adult or two adults at the nest; (2) nest success—the proportion of occupied territories producing at least one young; and (3) productivity—the mean number of young raised to pre-fledging age (≥ 8 wk) per occupied territory. We analyzed trends of these parameters by fitting them to linear models with simple linear regression. We determined statewide trends for (1) all years from 1980–98, (2) 1993–98 only (the period of nonrandom sampling), and (3) two regional populations, the Columbia River estuary and Hood Canal, that experienced depressed productivity during the survey period (Anthony et al. 1993, WDFW Heritage Data Base unpubl. data).

Estimates of nest success in raptor populations are subject to sampling errors when pairs that fail early in the nesting season may not be discovered and counted, leading to the overestimation of productivity/occupied site (Steenhof and Kochert 1982, Steenhof 1987). Because our surveys were potentially subject to this bias, we used a second method to calculate productivity recommended by Steenhof (1987). This method calculates productivity as the product of the proportion of pairs that bred, the proportion of pairs that were successful, and the number of young/successful pair. Each parameter is estimated from a specific population subsample: proportion of breeding pairs from a preselected sample that includes only nests from the population that bred the previous year; proportion of successful pairs from all nests surveyed twice (i.e., during incubation and pre-fledging);

and young/successful pair from pairs identified in both early and late surveys. Proportion of successful pairs is not a direct computation, but is calculated with the Mayfield estimator (Mayfield 1961), which is the daily-nest-survival rate raised to the power of the length of the mean period that a nest is at risk of failing (Steenhof 1987). We used 93 d as the mean nest exposure period (McAllister et al. 1986). We did not determine trends in productivity estimated by the Steenhof method because calculations were based on combined parameter estimates that potentially biased sample variances (Steenhof 1987).

We evaluated change in distribution of nesting eagles during 1980–98 by defining five broad ecoregions; the Olympic, southwest, and Puget Sound/Islands west of the Cascade Range, and northeast, and southeast ecoregions to the east (Fig. 1). The rate of population growth in each ecoregion was calculated from the number of occupied territories documented in 1980 and 1998. We compared density of occupied nests < 2 km from marine, lake, and large river shorelines between west and east ecoregions (Washington Rivers and Marine Shoreline data base, Wildlife Resource Data Systems, WDFW).

We estimated the number of statewide breeding pairs expected at carrying capacity by fitting population growth to a logistic curve based on the number of occupied territories found each year from 1980–98. The logistic growth model is a simplistic model that assumes the population is approaching a steady density; age structure is not considered, and all individuals are assumed to have an equal chance to give birth or die (Smith 1974). Thus, the model is not subject to changing survival and mortality rates. When a population grows exponentially, a linear relationship exists between the number of offspring per parent and the sum of the densities of both generations (Morisita 1965). The slope and intercept of this regression can be used to calculate the maximum intrinsic rate of population growth and carrying capacity as detailed in Caughley (1977) and Swenson et al. (1986). We determined these two parameters independently for eastern and western Washington because of habitat differences, and summed the numbers of territories at carrying capacity for eastern and western Washington to estimate the size of the statewide breeding population at saturation. Because the logistic growth model did not address habitat limitations to the population, such as nest site availability, we assessed the reasonableness of the estimates of carrying capacity in light of visible signs of population stability (i.e., increased incidences of urban nesting and fatal encounters of territorial adults with conspecifics), and a subjective estimate of the point at which the growth would reach an asymptote. At saturation, higher nest density might result in reduced nesting success because of closer distances between adjacent nesting pairs (Anthony et al. 1994). We used logistic regression to examine the effects of nearest-neighbor distance on eagle occupancy, activity, and nest success in 1992, when the population showed signs of reaching saturation.

Beyond a certain point, the actual number of nesting pairs at carrying capacity does not affect population stability because its true indicator is age and stage structure at equilibrium (Hunt 1998). Thus, the deviation between

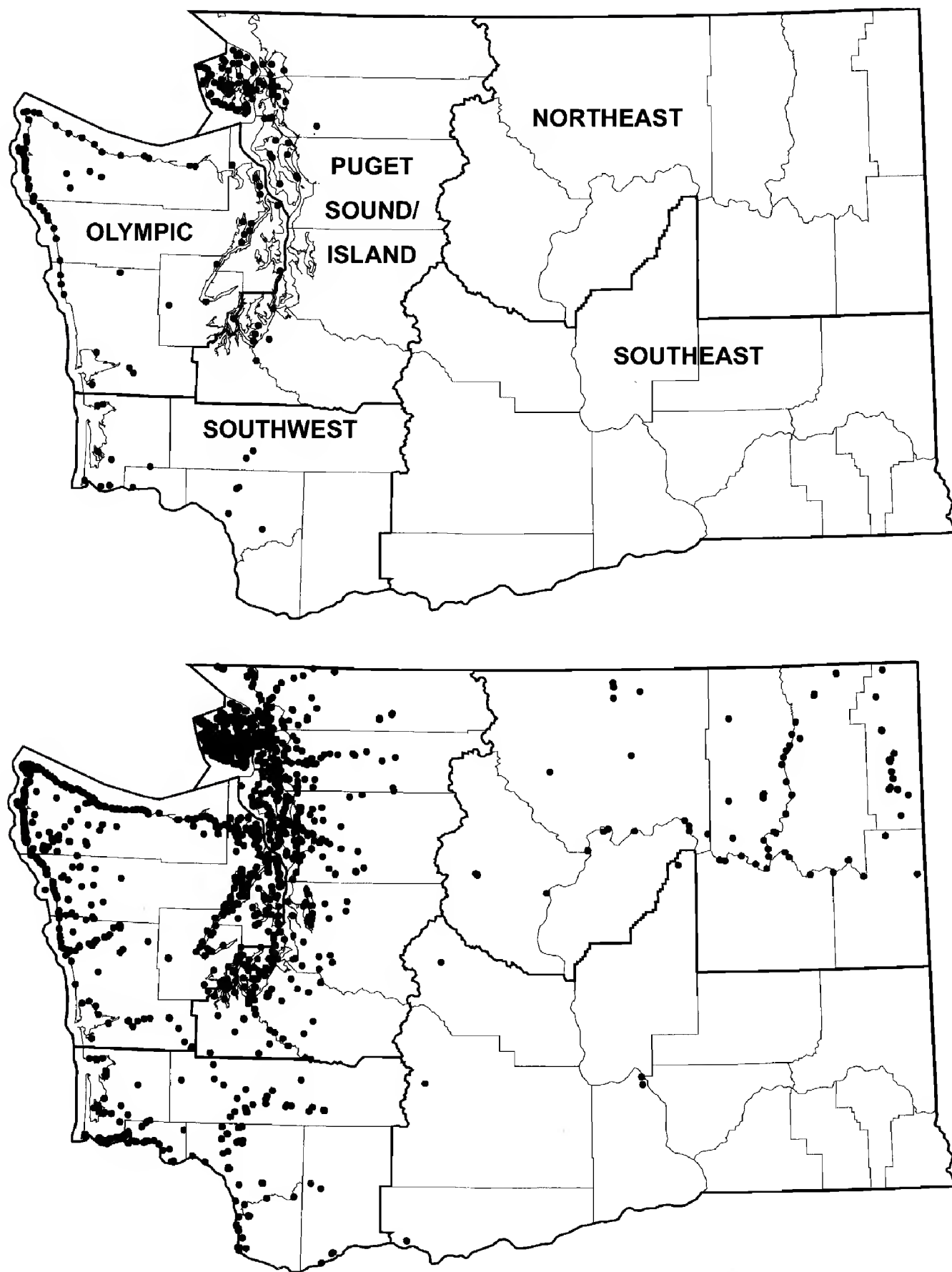


Figure 1. Distribution of Bald Eagle nests in Washington State among five ecoregions in 1980 (top) and 1998 (bottom).

future and predicted number of nesting pairs at carrying capacity was inconsequential to models of population stability. To estimate population structure and stability at carrying capacity we used a modeling approach based on Moffat's Equilibrium (Hunt 1998). Whereas traditional population modeling emphasizes density-dependent mechanisms that regulate population growth, modeling based on Moffat's Equilibrium focuses on an adaptive limit to breeding site serviceability that restricts cohort

size per unit area of landscape and consequently limits the size of the total population (Hunt 1998, Hunt and Law 2000). Causal regulation is considered modulating. Model parameters include the number of serviceable breeding locations (SBLs) at saturation (calculated from logistic modeling), age-specific survival rates, maximum longevity, and productivity. We used equations and routines from Hunt (1998) to calculate age class sizes, floater to breeder ratios, and total population size at population

Table 1. Productivity characteristics of the Bald Eagle population in Washington State from 1980–98. Standard errors are shown with summary means.

YEAR	NO. TERRITORIES SURVEYED	NO. (%) TERRITORIES OCCUPIED	PERCENT OF PAIRS BREEDING		PERCENT OF PAIRS SUCCESSFUL		NO. YOUNG/SUCCESSFUL TERRITORY	NO. YOUNG/OCCUPIED TERRITORY	
			DIRECT ^a	SAMPLE ^b	DIRECT ^a	SAMPLE ^c		DIRECT ^a	SAMPLE ^d
1980	154	105 (68)	90	94	64	52	1.40	0.90	0.68
1981	165	126 (76)	97	97	56	37	1.35	0.75	0.48
1982	189	138 (73)	88	90	55	40	1.35	0.74	0.49
1983	231	168 (73)	92	94	49	47	1.47	0.86	0.64
1984	254	206 (81)	95	96	67	58	1.44	0.95	0.80
1985	290	231 (80)	88	88	65	60	1.50	0.98	0.80
1986	301	250 (83)	94	96	73	66	1.54	1.11	0.97
1987	327	268 (82)	93	94	65	54	1.49	0.98	0.75
1988	361	309 (86)	92	93	66	56	1.50	0.98	0.78
1989	424	369 (87)	91	93	63	55	1.62	0.99	0.83
1990	477	403 (84)	93	93	70	61	1.63	1.07	0.92
1991	515	445 (86)	91	92	63	52	1.57	0.97	0.76
1992	560	468 (84)	94	94	69	61	1.47	0.99	0.85
1993	588	493 (84)	95	95	63	53	1.52	0.94	0.76
1994	636	547 (86)	93	94	70	65	1.49	1.02	0.91
1995	660	558 (85)	95	95	63	49	1.50	0.90	0.69
1996	709	594 (84)	92	93	64	56	1.41	0.93	0.73
1997	727	582 (80)	95	95	66	50	1.53	0.97	0.73
1998	841	666 (79)	91	93	74	65	1.49	1.10	0.91
Total	8409	6926 (81 ± 1)	93 ± 1	94 ± 1	65 ± 1	55 ± 2	1.49 ± 0.02	0.95 ± 0.02	0.76 ± 0.03

^a Direct measurements based on entire population.

^b Sample estimate from territories occupied the prior year (Steenhof 1987).

^c Sample estimate calculated by the Mayfield Method (Steenhof 1987) from pairs surveyed twice.

^d Steenhof (1987) estimate of productivity = (% breeding from sample) (% successful from Mayfield) (No. young/successful pair).

equilibrium based on a maximum eagle longevity of 25 yr. This was greater than the 16-yr longevity estimated for eagles from the Greater Yellowstone Ecosystem (Harmata et al. 1999), but less than the oldest documented Bald Eagle longevity record of 28 yr (Schempf 1997). Annual survival rates of adults (0.88), subadults (0.95), and juveniles (0.71), and productivity of 0.86 young/pair, were used in calculations, and were based on survival and productivity of 159 telemetered eagles and 622 occupied nests from Prince William Sound, Alaska (Bowman et al. 1995), where habitat is somewhat similar to that of coastal Washington. In any case, our interest was not so much in determining the accuracy of these statistics, but rather how changes in their values affected population stability. We modeled effects of hypothetical environmental perturbations on population size and structure by reducing the number of SBLs, the productivity rate, and age-specific survival. The barometer of population stability was the ratio between floating and breeding adults (F:B ratio), with negative ratios indicative of inadequate recruitment and population decline (Hunt 1998).

RESULTS

From 1980–98, the annual occupancy rate of Bald Eagles in Washington averaged 81% and in-

creased linearly ($r = 0.62$, $P = 0.005$; $N = 8409$ surveyed territories; Table 1); productivity averaged 0.95 young/occupied territory ($N = 6926$) and increased linearly ($r = 0.52$, $P = 0.024$); and nest success averaged 65% at occupied territories and increased linearly ($r = 0.50$, $P = 0.031$). However, for the 1993–98 sample of territories ($N = 4161$), annual occupancy rates declined by 1.3% per yr ($r = 0.83$, $P = 0.040$), and there was no trend in nest success ($P = 0.282$) or productivity ($P = 0.306$) at territories that were surveyed non-randomly ($N = 1397$). Between 1980–98 the number of Bald Eagle territories in Washington increased from 154–841 (Table 1). The number of pairs that nested each year increased logarithmically at a mean rate of 10.1% per yr ($[\log e]$ occupied territories = $4.850 + 0.101 \text{ yr}$; $r = 0.98$, $P < 0.001$).

Sample estimates of statewide eagle productivity averaged 0.19 young/yr less than direct productivity measures (Table 1). Much of this difference was

due to the Mayfield estimator for percent of successful pairs, which averaged 10% less than direct measures from the entire population. The percent of eagle pairs breeding in the preselected samples of pairs successful in the previous year averaged only 1% higher than direct measurements for the entire population from 1980–98.

Between 1980–98, the Bald Eagle population nesting on Hood Canal increased from 3–33 pairs, and the population along the Washington side of the Columbia River estuary increased from 1–24 pairs. The annual occupancy rate on Hood Canal (82%; $N = 398$ surveyed territories) was similar to the statewide rate, but lower on the Columbia River estuary (69%; $N = 328$ surveyed territories). Productivity parameters of these populations were below statewide means (Table 1). Hood Canal eagles produced 0.63 young/occupied territory ($N = 323$), with 43% of nesting attempts at occupied territories successful. Eagles along the Columbia River estuary produced 0.56 young/occupied territory ($N = 277$), and 41% of nesting attempts at occupied territories were successful. Despite the poor reproductive history of these populations, productivity increased linearly from 1980–98 on Hood Canal ($r = 0.55$, $P = 0.016$) and the Columbia River estuary ($r = 0.68$, $P = 0.001$), as did nest success (Hood Canal $r = 0.59$, $P = 0.008$; Columbia River estuary $r = 0.81$, $P < 0.001$).

A notable change in the statewide distribution of nesting Bald Eagles from 1980–98 occurred east of the crest of the Cascade Range where the number of territories increased from 0–59. Fifty-four of these territories (92%) were located in the northeast ecoregion, primarily along the upper Columbia, Spokane, and Pend Oreille rivers (Fig. 1). West of the Cascade Crest, the increase in number of nesting territories was similar among the Olympic ecoregion (380%, $N = 54$ –259), Puget Sound ecoregion (350%, $N = 90$ –405), and southwest ecoregion (292%, $N = 13$ –51). The increase in number of occupied territories was greater in southwest Washington (829%, $N = 7$ –65), than in Puget Sound (475%, $N = 61$ –351) and the Olympic ecoregion (438%, $N = 37$ –199), a difference largely due to reoccupancy of vacant nests along the Columbia River estuary. In westside ecoregions there was a progressive expansion of nesting pairs inland to major rivers and lakes along the coast and Puget Sound (Fig. 1). In 1998, the mean density of occupied Bald Eagle nests <2 km from 6416 km of forested, marine shorelines in western Wash-

ington was 1 nest/10.4 km. In eastern Washington, density was 1 nest/34.6 km along 1728 km of inland waters. We did not detect any relationship between nearest-neighbor distance and nest occupancy ($P = 0.534$), activity ($P = 0.173$), or success ($P = 0.650$) at 560 territories in 1992.

Logistic population growth modeling based on the assumption that the population was approaching a steady density, projected an ecological carrying capacity of 639 nesting pairs in western Washington, and a maximum growth rate of 9.5%. The model yielded a carrying capacity of 94 pairs in eastern Washington, and a maximum intrinsic growth rate of 16.7%. The combined total for nesting pairs (733) was used as the statewide number of SBLs, in our modeling exercise which predicted a population of 4913 eagles at Moffat's Equilibrium (25 yr after the population reaches carrying capacity). The stable population consisted of 1907 subadults and juveniles, 1540 floating adults, and 1466 breeding adults, resulting in an F:B ratio of 1.05. When other parameters were held constant, F:B ratios of the predicted population were reduced to a critical level (i.e., <0) resulting in population decline when adult survival declined 17% (0.88–0.73), or subadult survival declined 22% (0.95–0.74), or juvenile survival declined 52% (0.71–0.34), or productivity declined 52% (from 0.86–0.41 young/pair). In a hypothetical scenario where productivity and juvenile age classes were primarily impacted (e.g., nest disturbance, contaminants) the population declined when productivity rates and juvenile survival were each reduced by 31%. However, in a scenario where survival of all age classes was impacted (e.g., oil spill, prey crash) the population declined when adult survival was reduced by only 7%, subadult survival by 8%, and juvenile survival by 10%. In a scenario where the number of statewide SBLs was reduced by 50% and survival and productivity rates were maintained (e.g., habitat loss from development), the equilibrium model predicted a 50% reduction in the size of each age class and total population when the population stabilized, but the F:B ratio remained at 1.05, a condition conferring a high degree of population security.

DISCUSSION

Population Growth. Exponential population growth exhibited by the Bald Eagle population in Washington in the past 20 yr surpassed that within the contiguous United States as a whole (i.e.,

384%, $N = 1188$ – 5748 occupied territories; U.S. Fish and Wildlife Service unpubl. data). Although intense habitat management and protection of nest territories in Washington occurred during that period, including the development of 1150 eagle habitat management plans with state and private landowners (WDFW Wildlife Resource Data Systems unpubl. data), population growth was most likely a direct consequence of (1) reduced persecution that decimated the population beginning in the early 1900s (Dawson and Bowles 1909) and (2) reduced environmental levels of DDT, the insecticide that caused eggshell thinning and embryo mortality and was believed to have drastically reduced eagle populations after 1945 (Stalmaster 1987). Use of DDT was banned in 1972, eight years prior to our study. Increased rates of nest success and productivity that we documented would be expected when contaminants levels declined in eagle habitats, eagle prey, and ultimately breeding adult eagles that were also under reduced threats of direct persecution. This would be followed by increased occupancy of vacant nests at historic sites as more individuals reached maturity and the population increased. We found population increases even among contaminant-impaired eagle populations on the Columbia River estuary and Hood Canal. Although productivity remained below state-wide means for those populations, it increased significantly in the past 20 yr. At their present densities, the contribution of these regional populations to the number of nesting pairs in Washington is minor (i.e., in 1998 only 4% of nesting pairs in the state were on the Columbia River estuary, and 5% on Hood Canal), but these populations are nevertheless important as local bio-indicators of contaminant levels (Anthony et al. 1993).

Rapid repopulation of nesting habitat by Bald Eagles was in part related to the tendency of offspring to return to natal regions (Wood 1992, Driscoll et al. 1999, Harmata et al. 1999). Evidence from Montana suggests non-breeding male Bald Eagles exhibit fidelity to geographically small natal areas that are familiar to them (e.g., Greater Yellowstone Ecosystem population), whereas many females disperse more widely (Harmata et al. 1999). In Washington State, we have no data to indicate that breeding eagles from western Washington cross the Cascade Mountains and pioneer new territories in eastern Washington, although the Cascade crest is no hindrance to movement of wintering eagles (J. Watson unpubl. data). The more

rapid growth in eastern Washington compared to the west side suggests carrying capacity for nesting eagles will be reached sooner in western Washington. The density of nesting Bald Eagles in eastern Washington is presently half of that in western Washington based on available shoreline, but the amount of difference due to lower prey and nest tree densities is unknown, as is the density the east side eagle population may reach at saturation. A density of 1 nest/11 river km is reported along the upper Columbia River in southern British Columbia to the north of eastern Washington (Blood and Anweiler 1994).

Population Equilibrium. The logistic growth model, our examination of trends in nesting parameters from 1993–98, and recent occupation of eagle territories in urban areas all indicate that the population of breeding eagles in Washington is approaching saturation. Equilibrium theory predicts that as competition for the limited number of SBLs increases within a population, increased interference from floating adults for prey and nest sites should reduce productivity and survival (Haller 1996, Hunt 1998). Indeed, in Washington during the past 5 yr at least six fatal encounters between floating adults that attacked breeding adults have been documented, whereas prior to that time no similar events were reported (J. Watson unpubl. data). The linear decrease in nest occupancy, and stabilization of productivity and nest success of Bald Eagles in Washington during the 1990s are consistent with predicted modulating effects of floater pressure following population saturation (Hunt 1998), a phenomenon also documented in other Bald Eagle populations (Hansen 1987, Bowman et al. 1995). Our surveys of the subpopulation of Bald Eagles nesting in the San Juan Archipelago of northwest Washington (i.e., 90 territories) show the number of occupied territories declined by <10% in the years following a peak in 1994 (Fig. 2). This may indicate the range of population decline to be experienced throughout Washington from the density-dependent effects of floater interference. The occupancy rate of Washington Bald Eagles is unlikely to increase from present levels to high levels such as reported in Arizona (i.e., 90%, Driscoll et al. 1999), because many of the unoccupied territories have degraded habitat, excessive levels of disturbance, or may be limited by prey availability (J. Watson unpubl. data). Nevertheless, a small but increasing number of Bald Eagles in Washington demonstrated surprising tolerance to

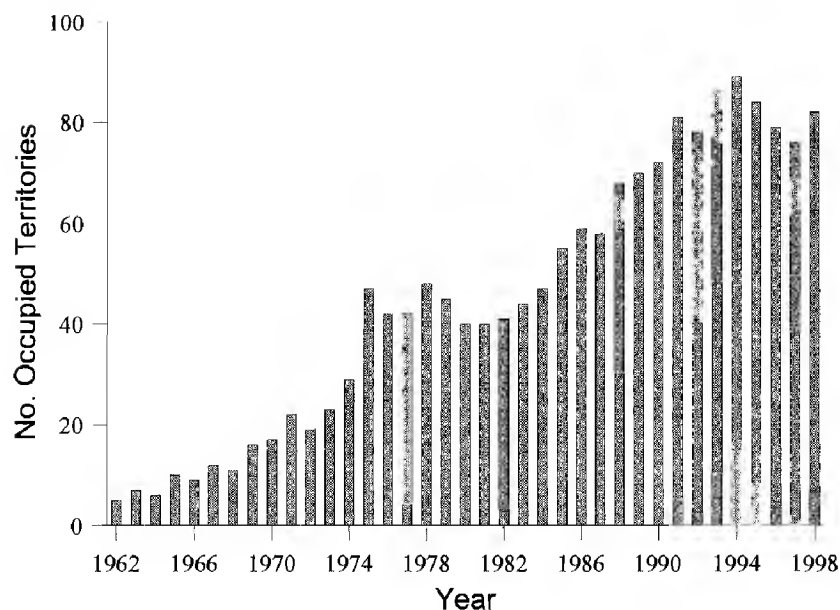


Figure 2. Growth of the Bald Eagle population in the San Juan Islands in northwest Washington. Data for 1962–79 from Nash et al. (1980), and for 1980–98 from WDFW (unpubl. data).

human activity in the 1990s (Watson et al. 1999) and established new territories in urban parks, neighborhoods, and golf courses.

The estimated productivity level of 0.95 young/occupied territory, the recent decline in nest occupancy, and stabilization of productivity and nest success rates, provide further evidence that the Washington population of nesting Bald Eagles is at saturation. However, the effects of incomplete, non-random surveys on estimates of the latter parameters is uncertain. In some cases Bald Eagle territories affected by management plans, and potentially having higher human disturbance levels, were given survey priority (S. Negri and S. Ament, pers. comm.), but productivity of such nests has not been found to be different from the general population (G. Schirato unpubl. data). Early literature suggested productivity of 0.7 young/nest was necessary for population stability (i.e., Sprunt et al. 1973). If survival is as high as reported elsewhere for juvenile and adult eagles, mean productivity of <math><1.0</math> young/nesting pair appears adequate for population stability (Buehler et al. 1991, Bowman et al. 1995, Harmata et al. 1999). Our direct estimate of statewide productivity in Washington (0.95 young/occupied territory) is within that range. Even if the sampling method more accurately reflects true productivity of Washington eagles (0.76 young/occupied territory, 20% lower than direct estimates), either survival rates are high enough to sustain such rapid population growth, or the Washington population is being supplemented substan-

tially by immigration from other populations, or both. We suspect productivity estimates from the sampling method were unrealistically low, because in Washington locations of virtually all Bald Eagle nests were well-documented and nests were highly visible from the air. This increased the likelihood of encountering adults to confirm activity even at failed nests or those where no eggs were laid, so we believe that few early nest failures were missed.

Population Stability. Predictive models based on equilibrium theory provided a prioritization of population parameters for their relevance to maintaining population stability during hypothetical environmental perturbations. While the eventual size of the Bald Eagle population in Washington will be limited by the number of SBLs, maintaining an adequate ratio of floating to breeding adults is the ultimate determinate of population stability (Hunt 1998). Ideally, the population of floating and breeding adults could be surveyed simultaneously on a periodic basis to assess population stability. In Washington, floating adults may spend up to 40% of the year in Canada and southeast Alaska from June–November (J. Watson unpubl. data). Surveys conducted in spring in Washington could allow an accounting of breeders on territories and provide an estimate of floating adults, but might be impractical because of costs. Therefore, the most important emphasis for maintaining the eagle population is to maximize survival, and prevent or ameliorate environmental factors that result in direct mortality (e.g., shooting) or indirect mortality (e.g., lead poisoning) of adults, and secondarily subadults, during their 3-yr transition to adulthood. The ratio of floating to breeding adults was least sensitive to changes in rates of productivity and juvenile survival, so these are the least important parameters to population stability. Dramatic declines in eagle productivity or juvenile survival (i.e., 50%) would have to be experienced to produce the same effects as small declines in the survival of older birds (e.g., 7–10% for adults). This corroborates Grier's (1980) conclusion that population dynamics of Bald Eagles depend more on survival than reproduction. Reproduction has more often been the parameter monitored to determine Bald Eagle status because it is a sensitive indicator of contaminant problems and it is also easier to monitor than eagle survival (Harmata et al. 1999). The equilibrium model suggests that determining a minimum number of SBLs needed to maintain population stability in Washington

should be based on what number is necessary to provide an overall reserve of nonbreeding adults adequate to buffer fluctuations in density-independent mortality factors (e.g., weather, electrocutions, oil spills). The optimum number of SBLs in Washington State, however, must be determined after consideration of aesthetic values of Bald Eagles; the public may, for example, desire to protect more territories than necessary for population stability. Current management of breeding Bald Eagles in Washington as directed by state legislation is to manage all territories equally on state and private land regardless of habitat quality. Our population model suggests the ultimate need to conserve the population is to protect the quality breeding habitats for a target number of territories, whether greater or less than the 733 projected territories, and thus ensure a stable number of breeding locations into the foreseeable future. Prioritization of existing territories for protection based on their distribution, the condition of habitat, threats to the habitat, and proximity to foraging areas is an objective of Bald Eagle recovery in Washington (Stinson et al. 2001).

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RESPONSE DISTANCE OF FERRUGINOUS PYGMY-OWLS TO BROADCASTED CONSPECIFIC CALLS

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ABSTRACT.—To assess the efficiency of broadcast surveys for Ferruginous Pygmy-Owls (*Glaucidium brasilianum*), we tested the response distance of nine, radio-tagged, adult males. We recorded vocalization and movement toward the broadcast station as separate types of responses. Response to broadcasted conspecific calls was tested for each pygmy-owl at distances from 250–700 m. Broadcasted calls elicited vocal response from all nine pygmy-owls tested at ≤ 550 m and eight of the nine pygmy-owls moved toward the broadcast station. At 600 m, eight responded vocally and seven of the nine pygmy-owls tested, moved toward the broadcast station. Of the six pygmy-owls tested at 700 m, four responded vocally and three moved toward the broadcast station. As we recorded a 100% response from a distance of ≤ 550 m, the effective coverage of areas formed by establishing survey points from 400–1400 m apart, in 100 m increments, would range from 97.7–61.7%, respectively. For these same increments, broadcast overlap would range from 54.7–0.0%, respectively. Based on response distance information, researchers may choose between different survey levels. For example, to maximize detection, researchers may develop survey protocols that canvas an area with overlapping radii and redundant sampling. Antithetically, to determine general distribution of a species over expansive areas, researchers may choose to increase survey efficiency by reducing broadcast overlap, survey effectiveness, and redundant sampling.

KEY WORDS: *Ferruginous Pygmy-Owl*; *Glaucidium brasilianum*; *broadcast survey*.

Distancia de respuesta de *Glaucidium brasilianum*, a vocalizaciones emitidas de la misma especie

RESUMEN.—Para evaluar la eficiencia de muestreos a través de difusión de llamados para *Glaucidium brasilianum*, probamos la distancia a la que respondieron nueve machos adultos con radio telemetría. Definimos vocalización y movimiento hacia la estación de difusión como dos respuestas distintas. Respuestas a llamados grabados de la misma especie se probaron a distancias de 250–700 m. Los llamados difundidos causaron respuesta vocal en los nueve tecolotitos probados a < 550 m; ocho de los nueve tecolotitos probados a 550 m respondieron con vocalización, se movieron hacia la estación de difusión. A 600 m, ocho de los nueve probados respondieron vocalmente y siete de los nueve respondieron vocalmente y se movieron hacia la estación de difusión. De seis tecolotitos probados a 700 m, cuatro respondieron vocalmente y tres se movieron hacia la estación de difusión. Ya que obtuvimos una respuesta del 100% a una distancia de 550 m, la cobertura efectiva de áreas formadas al establecer puntos de difusión de 400–1400 m, en incrementos de 100 m, cubrirían entre el 97.7–61.7%, respectivamente. Para los mismos incrementos el área de traslape de áreas de difusión efectiva cubrirían entre el 54.7–0.0%, respectivamente. Al utilizar la información de distancia de respuesta investigadores podrían escoger entre diferentes niveles de muestreo. Por ejemplo, para maximizar la detección de especies de interés, un investigador podría desarrollar protocolos que cubran toda el área con traslape de áreas de difusión y hacer muestreo redundante. Sin embargo para determinar la distribución general de una especie sobre áreas extensas, un investigador podría decidir en protocolos de muestreo que incremente la eficacia de cobertura al reducir el traslape en el área efectiva de cobertura del área de difusión

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Accurate survey methods are critical to the management and conservation of threatened and endangered species. Survey methods can provide estimates of distribution, relative abundance, habitat use, and with some species, sex ratios. These baseline data are important for evaluating the status and trends of species impacted by changing land-use practices and loss of suitable habitat. Measuring response of individuals to broadcasted conspecific calls is an important method employed for surveying avian populations (Allaire and Landrum 1975, Johnson et al. 1981, Smith et al. 1987, Stahlecker and Rawinski 1990). However, without definitive unbiased information regarding effective sampling area, broadcast surveys only provide an index of presence/absence (McLeod and Anderson 1998). The overall effectiveness of this method depends on several factors. First, responsiveness varies among species and seasonally within species (Springer 1969, McNicholl 1978). Second, terrain and other environmental factors (e.g., wind and precipitation) affects dissipation of sound waves and, thus, influences the maximum distance from which a response can be elicited (DeMaso et al. 1992) and answering calls can be heard. Third, the distance between sample points determines the degree of overlap among broadcast radii. Hence, the distance between sample points influences the potential for redundant sampling to occur, such that if the distance between sites is too small, individuals can be counted multiple times, providing overestimates of abundance or population size.

In the United States, the Ferruginous Pygmy-Owl (*Glaucidium brasilianum*) only occurs in southern Texas and southwestern Arizona. In Arizona, it is currently listed by the U.S. Fish and Wildlife Service (1997) as endangered. This species is a cavity nester that requires mature trees, including large columnar cacti for nesting, and an adequate prey base (Proudfoot and Johnson 2000). Throughout Arizona and Texas, pygmy-owl populations are fragmented by islands of suitable habitat (Oberholser 1974, Millsap and Johnson 1988, Proudfoot and Johnson 2000). The determination of population sizes and distributions are essential data for assessing population viability and the identification of critical habitat. As a case in point, information from broadcast surveys used to estimate density and distribution of pygmy-owls in Texas suggest a viable population occurs in Kenedy County (Wauer et al. 1993, Mays 1996). Information provided from these surveys was undoubtedly a key factor in the

final decision of the Service not to list the pygmy-owl as threatened in Texas (U.S. Fish and Wildlife Service 1997). These survey data were collected and interpreted without information on the territory size of this species and the distance at which pygmy-owls would respond to broadcasted conspecific calls. Hence, the frequent clustering of responses that occurred within the live oak-honey mesquite (*Quercus virginiana-Prosopis glandulosa*) forest (Wauer et al. 1993, Mays 1996) may have been the result of redundant sampling of individuals. Mays (1996) established broadcast stations 400 m apart along road transects in the initial survey and used a 400 m minimum to determine random placement of broadcast stations during her repeated survey effort. Wauer et al. (1993: 1072) used modified Emlen (1977) method and U.S. Fish and Wildlife Service Breeding Bird Survey method to conduct broadcast surveys. He provided no specific information about how the two methods were employed (e.g., distance between broadcast stations). Information obtained during a pilot study to ascertain the response distance of pygmy-owls (i.e., two radio-tagged pygmy-owls were recorded responding at 600 m from the broadcast station) prompted Mays (1996) to urge caution be used when interpreting survey data collected along transects with survey points established ≤ 400 m apart.

In January 2000, the U.S. Fish and Wildlife Service (2000) issued a standard protocol to be used for surveying areas that were proposed for future development within boundaries designated as critical habitat for pygmy-owls in Arizona. Although the protocol was based on data provided in the available literature and from information submitted by scientists and non-scientists during the public-comment period, the U.S. Fish and Wildlife Service (2000) did not support the protocol with research results or information documenting effectiveness. Hence, as was the case with Wauer et al. (1993) and Mays (1996), the survey protocol currently employed in Arizona may provide a biased measurement of pygmy-owl abundance. The objective of this paper was to provide information regarding the response distance, vocal and movement, of pygmy-owls to broadcast conspecific calls. We suggest that this information be used in the development of survey protocols that assess pygmy-owl distribution and long-term population trends accurately.

STUDY AREA AND METHODS

Research was conducted within 29 000 ha of live oak-honey mesquite forest in Kenedy County, Texas, the same forest in which Wauer et al. (1993) and Mays (1996) conducted surveys to estimate population numbers for pygmy-owls in Texas. Climate was subtropical with 68 cm and 24°C of mean annual precipitation and temperature, respectively. Elevation of the study area ranged from 5–21 m.

Nine adult male pygmy-owls (four in 1995 and five in 1996) were trapped during the nesting season (April and May; Proudfoot and Johnson 2000), fitted with transmitters, and monitored for 7–10 d prior to testing. Because spontaneous calling (bouts) of pygmy-owls are usually crepuscular (Gilman 1909, Proudfoot and Johnson 2000), testing was restricted to 30 min before and after sunset, as determined by the U.S. Naval Observatory, Washington, DC U.S.A. (http://mach.usno.navy.mil/cgi-bin/aa_rstablew.pl). Testing was not conducted when winds exceeded 24 kph or when precipitation occurred (Proudfoot and Beasom 1996).

Our testing was limited to ≤ 700 m, because when establishing the protocol for conducting call count surveys for Northern Bobwhites (*Colinus virginianus*), DeMaso et al. (1992) determined 700 m was the apex for surveyors to detect calls at 60–70 decibels (db), a similar acoustical level as produced by pygmy-owls. Two male pygmy-owls elicited by researcher's vocal mimic of the pygmy-owl's territorial call were recorded at 66–78 db (Proudfoot and Johnson 2000).

Using 3-element Yagi antennas and portable radio-receivers, two researchers tracked a radio-tagged pygmy-owl until obtaining visual contact. One researcher (R1) visually and electronically monitored the pygmy-owl while another researcher (R2) used compass bearings and pacing (Stoddard and Stoddard 1987) to establish a broadcast station at the distance desired for testing (e.g., 500 m). Researchers maintained contact via 2-way radio. If the pygmy-owl moved while R2 was locating the broadcast station, R1 relayed its new location to R2, and adjustments (repositioning of broadcast station) were made to maintain the distance desired for testing (e.g., 500 m). A portable recorder capable of producing 95–105 db at a distance of 1 m from the speaker was used by R2 to broadcast conspecific calls, recorded locally, toward the targeted individual. This equipment met output recommendations for raptor broadcast surveys (Fuller and Mosher 1987).

While at a station, broadcasting continued for 3 min, during which time any pygmy-owl movement or vocalization was recorded. The characteristic call of pygmy-owls is a simple series of interrupted single notes, hence, continued broadcast should not have hampered detectability (Proudfoot and Beasom 1996). To eliminate errors that would result from recording responses from non-targeted individuals, R1 maintained direct observation of test subjects during the initial stages of testing, radio-telemetry was used to monitor movement of radio-tagged individuals that responded during testing, and R2 located responding individuals that moved toward the broadcast station and verified identification of the test subject with radiotelemetry.

Clearly, any reduction in the distance between the broadcast station and the target individual would result

in a measurable difference in decibels received at the target's location. Thus, to test the response distance in a reasonable manner, the distance between broadcast stations should be far enough to result in a significant change in sound reception by the targeted individual. In 1995, testing began at 400 m and increased daily by 100 m increments to 700 m; each individual was tested once daily (Kennedy and Stahlecker 1993). In 1996, sampling was reversed and began at 700 m; if no response was recorded the broadcast station was moved 50 m closer and testing was continued. At each new distance interval a 5-min adjustment period (silence) was observed before broadcasting was resumed. Because we invoked a 5-min adjustment period and visually monitored each individual during testing, we were confident that the response distance recorded was the distance at which the response was elicited. This protocol (5-min of silence followed by 3-min of broadcasting) was repeated until vocal response and movement toward the broadcast station was recorded. In 1996, we selected the distance (50 m) between broadcast stations based on the time available to conduct tests. Because birds establish territories and maintain and defend areas based on energetic budgets and physical restrictions, confronting conspecifics outside territorial boundaries may be counterproductive. Hence, birds with established territories make response decisions based on assumed location of conspecific and inferred threat (Brown 1969). Therefore, the sample protocol used during 1996 may simulate natural events and behavior.

Pythagorean and Archimedes theorems were used to describe broadcast coverage based on pygmy-owl response distance information. Theoretical models were used to estimate sampling coverage with regard to effective broadcast radii and spacing of survey points (Fig. 1). For example, with an effective broadcast radius of 550 m, surveyors would essentially sample 94.8% of the rectangular area formed from multiplying the distance between survey points (600 m) by the diameter (1100 m) of the broadcast circle. With this sample effort, 34.2% broadcast overlap would occur. If survey points are established 1100 m apart, 78.5% of the described area would be sampled, with 0.0% broadcast overlap (Fig. 1).

RESULTS

In 1995, all four pygmy-owls tested at 400 and 500 m responded vocally, moved toward the broadcast station, and continued to vocalize. At 600 m, three pygmy-owls responded vocally, moved toward the broadcast station, and continued to vocalize; the fourth only responded vocally. Due to time constraints, only one pygmy-owl was tested at 700 m in 1995. It too responded vocally, moved toward the broadcast station, and continued to vocalize.

In 1996, two of five pygmy-owls tested at 700 m vocalized, moved toward the broadcast station, and continued to vocalize. A third pygmy-owl responded vocally at 700 m, moved (< 100 m) toward the broadcast station and continued to vocalize at 600 m. The fourth pygmy-owl responded vocally at 600

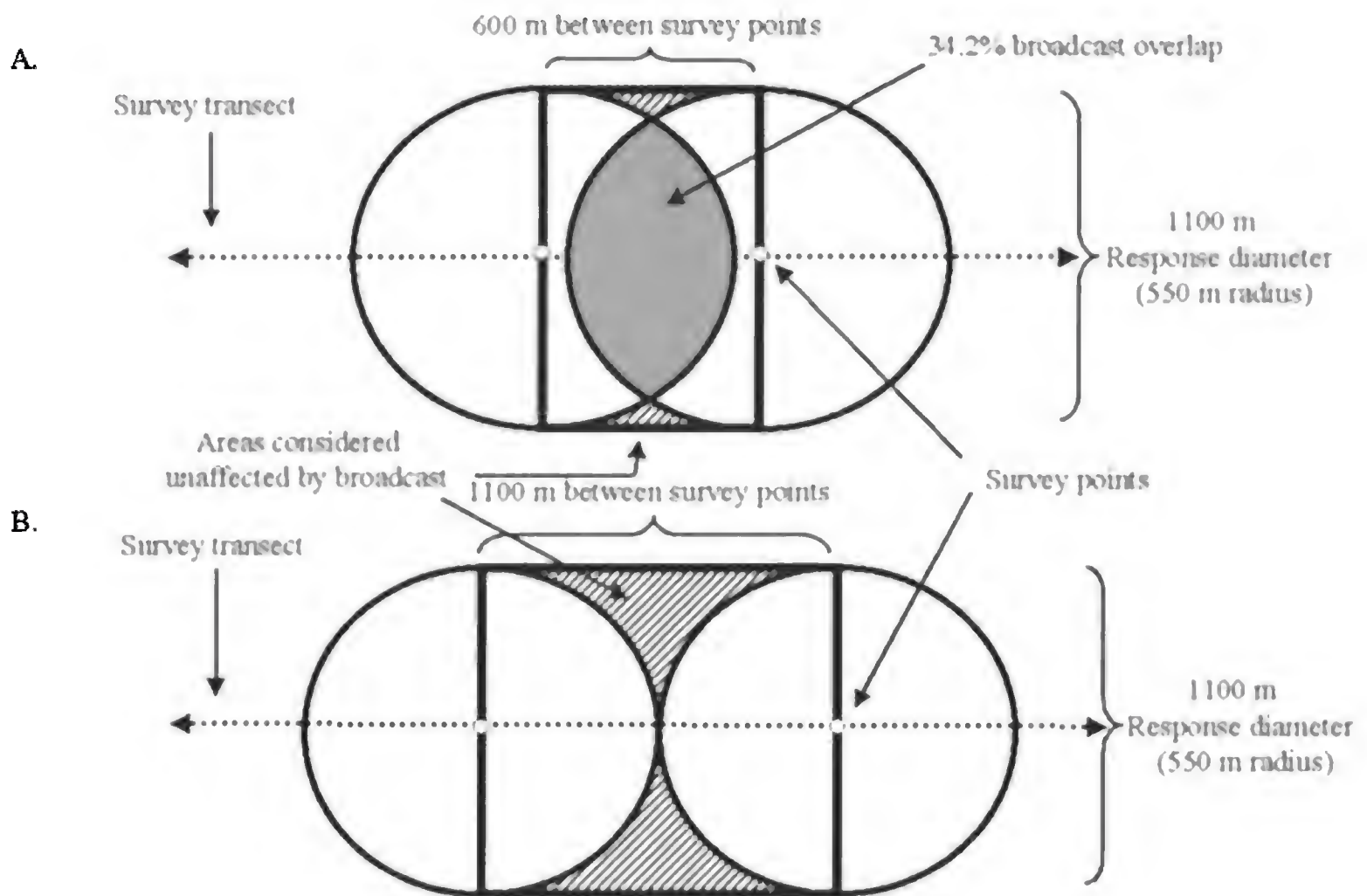


Figure 1. Schematic rendition of area surveyed along transects with broadcast points established 600 m (A) and 1100 m (B) apart, circles represent area covered with an effective broadcast radius of 550 m applied.

m and with vocalization and extensive movement at 550 m. The fifth pygmy-owl responded vocally at 550 m and with vocalization and movement at 250 m.

DISCUSSION

It is possible that repeated sampling of the same individual on the same evening during 1996 may have influenced our results. However, because we maintained constant observation of the test pygmy-owl during testing and a 5-min period of silence was employed between broadcasts, we submit that the response distance recorded was a reasonable measure of the distance at which the response was elicited (see Methods, above). In addition, because we began testing at 700 m and moved closer to the targeted individual in 50-m increments, any error from repeated sampling would result in conservative response distance estimates.

Using the distance at which 100% vocal response was recorded (550 m), the effective coverage of areas formed by establishing survey points from 400–1400 m apart would range from 97.7–61.7%, respectively; broadcast overlap would range from 54.7–0.0%, respectively (Table 1). Our sample size may be considered too small to ascribe absolute

response distance parameters. However, our data clearly show that broadcasted conspecific calls may elicit both movement toward the broadcast station and vocal response from pygmy-owls at a distance of 700 m. Consistent with Mays (1996), response distance information obtained from our study strongly suggests redundant sampling may occur along transects with survey points established ≤ 400 m apart. In addition, because several birds tested flew >500 m in response to broadcasted calls, our results question the likelihood that the mean radius of a pygmy-owl's territory is as small as Wauer et al. (1993) suggested, 297 m. Hence, Wauer et al. (1993) and Mays (1996) may have overestimated the pygmy-owl population size in Texas due to redundant detection of individuals and application of inappropriate territory size to extrapolate population estimates. Thus, biased data may have inadvertently altered the U.S. Fish and Wildlife Service's perception of a species in concern during the listing process.

Our data suggest that transects with survey points spaced from 400–600 m apart would potentially yield a high level of redundant sampling ($>30\%$ overlap). The current survey protocol authorized by

Table 1. Estimated percent coverage of rectangular area formed by multiplying observed response diameter ($2 \times$ response distance) of Ferruginous Pygmy-Owls in Texas by hypothetical distance (m) between broadcast stations. Percent overlap depicts overlap of effective hemispherical response radii. Calculations follow Pythagorean and Archimedes theorems, as simulated in Figure 1.

DISTANCE ^a	1100 m RESPONSE DIAMETER (550 m RESPONSE DISTANCE) 100% RESPONSE ^b		1200 m RESPONSE DIAMETER (600 m RESPONSE DISTANCE) 89% RESPONSE ^b		1400 m RESPONSE DIAMETER (700 m RESPONSE DISTANCE) 67% RESPONSE ^b	
	COVERAGE	OVERLAP	COVERAGE	OVERLAP	COVERAGE	OVERLAP
400	97.7	54.7	98.1	58.3	100.0	64.1
500	96.4	44.2	96.8	48.6	97.9	55.5
600	94.8	34.2	95.7	39.0	96.9	47.2
700	92.8	24.8	94.6	29.7	95.7	39.1
800	90.3	16.4	92.1	21.8	94.4	31.4
900	87.1	9.3	89.6	14.4	92.7	24.3
1000	83.6	3.3	86.7	8.0	91.4	17.0
1100	78.5	0.0	83.2	2.9	88.5	11.6
1200	72.0	0.0	78.0	0.0	85.9	6.4
1300	66.5	0.0	72.5	0.0	82.7	2.3
1400	61.7	0.0	67.3	0.0	78.7	0.0

^a Hypothetical distance between broadcast stations.

^b Response frequency based on analysis of Ferruginous Pygmy-Owl response distances in Texas.

the U.S. Fish and Wildlife Service (2000) to determine presence or absence of pygmy-owls in urban and rural areas proposed for development requires a maximum distance of 150 m and 400 m between survey points, respectively. Based on our findings, this protocol should be an extremely effective means of determining presence of pygmy-owl within areas surveyed. However, due to the excessive overlap of broadcast radii, using U.S. Fish and Wildlife Service guidelines would undoubtedly not provide accurate census data. In rural areas, the U.S. Fish and Wildlife Service authorized a maximum distance of 500 m between survey points for studies conducted to ascertain the distribution of pygmy-owls in Arizona. A distance of 800 m is allowed if bionic ears or other listing-enhancement devices are used to detect respondents. Due to tree density and background noise (rustling leaves and branches), however, 500 m is maintained as the maximum distance between survey points in riparian areas, regardless of utilization of listening aids (U.S. Fish and Wildlife Service 2000). This too should effectively sample areas surveyed for presence or absence of pygmy-owls. However, the level of overlap and, hence, high potential for redundant sampling may render this protocol inaccurate for assessing abundance and density.

The initial cost of obtaining information regarding effective broadcast radius may be substantial,

i.e., budgeting personnel and radiotelemetry equipment to conduct a response-distance study. However, the benefits of identifying the effective broadcast radius may transcend initial cost. For example, if we assume broadcast of conspecific calls will elicit 100% response from pygmy-owls at a distance of 550 m, increasing the distance between broadcast stations from 400–800 m would reduce effective broadcast coverage by 7.4%. However, it would also increase survey efficiency by 100%, and reduce overlap by 38.3%. Reducing overlapping broadcast radii would not only increase area covered, but should also reduce potential redundancies in sampling. This type of trade-off may be advantageous for surveying expansive areas with limited personnel resources. Antithetically, utilizing response-distance information, researchers may choose to canvas an area with overlapping radii to maximize detection of species of concern in areas proposed for development. To conclude, this type of research may aid species conservation by providing researchers basic information needed to develop survey protocols that maximize resource allocation with respect to survey intent and effectiveness. We suggest that the development of survey protocols should include empirical assessments of sampling effectiveness, both biologically and economically.

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POST-FLEDGING SURVIVAL AND DISPERSAL OF PEREGRINE FALCONS DURING A RESTORATION PROJECT

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ABSTRACT.—We monitored 38 juvenile Peregrine Falcons (*Falco peregrinus*) up to 3 mo immediately after their release from a hack box during 1999–2000. The restoration site was a cliff near Eagle Point Park in Dubuque, Iowa. Falcons were released in a staggered manner from mid-June until late July each summer. Older falcons remained at the site longer than at previous urban releases and interacted with the younger falcons. The four mortalities (11%) confirmed during the observation periods were discovered and reported by citizens near the release site. We used radiotelemetry, observations of color-marked birds at the hack site, and recovered mortalities to estimate weekly survival rates and dispersal patterns. We estimated weekly survival rate to be 0.988 (SE = 0.01), and our weekly resighting rate was high: 0.885 (SE = 0.03). Juveniles were observed for an average of 4.3 wk in 1999 (SD = 2.5), but only 3.4 wk in 2000 (SD = 2.3). Accordingly, weekly fidelity rates were year-specific: 0.903 (SE = 0.03) in 1999 and 0.795 (SE = 0.05) in 2000. No mortalities were attributed to Great-horned Owl (*Bubo virginianus*) predation, but substantial numbers of owls were seen in summer 2000. The presence of owls in 2000 may have contributed to the difference in fidelity rates and dispersal patterns between years.

KEY WORDS: *Peregrine Falcon; Falco peregrinus; survival; dispersal; mark-recapture model; population restoration.*

Sobre vivencia de volantones y dispersión de halcones peregrinos durante un proyecto de restauración

RESUMEN.—Monitoreamos 38 halcones peregrinos (*Falco peregrinus*) juveniles hasta 3 meses inmediatamente después de su liberación desde una “caja de suelta” durante 1999–2000. El sitio de restauración era un risco cerca la parque Punto del Águila en Dubuque, Iowa. Los halcones fueron liberados en forma escalonada desde mediados de junio hasta finales de julio de cada verano. Los halcones más adultos permanecieron por más tiempo en los sitios urbanos en los cuales interactuaron con los halcones más jóvenes. Las cuatro muertes (11%) confirmadas durante los períodos de observación fueron descubiertas y reportadas por ciudadanos cerca a los sitios de liberación. Utilizamos la telemetría y las observaciones de aves marcadas con colores en los sitios de liberación. La recolección de animales muertos fue utilizada para estimar la tasa de sobrevivencia semanal y los patrones de dispersión. Estimamos la tasa de sobrevivencia semanal en 0.988 (SE = 0.01), y una tasa de avistamientos semanal alta: 0.885 (SE = 0.03). Los juveniles fueron observados en un promedio de 4.3 por semana en 1999 (SD = 2.5), pero solo fue de 3.4 en el 2000 (SD = 2.3). En concordancia, las tasas de fidelidad semanal fueron específicas para cada año: 0.903 (SE = 0.03) en 1999 y 0.795 (SE = 0.05) en el 2000. No hubo mortalidades atribuibles a la depredación por parte de *Bubo virginianus*. Sin embargo, un número importante de búhos fue observado en el verano del 2000, lo que pudo haber contribuido a las diferencias en las tasa de fidelidad y en los patrones de dispersión entre años.

[Traducción de César Márquez]

Peregrine Falcons (*Falco peregrinus*) once nested on ledges of bluffs along Iowa rivers, but they were extirpated in the 1950s and 1960s. Recovery efforts

in urban areas have been successful, by using tops of tall city buildings as hack sites to release captive-hatched juveniles. Because of efforts in Iowa and surrounding states, the Midwestern population had grown to 67 territorial pairs by 1997, with 747 captive-produced juveniles released (Tordoff and Redig 1997). As of 2002, Peregrine Falcons are still

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listed as an endangered species in Iowa. In 1998, the Iowa Department of Natural Resources (DNR) began a program of cliff-habitat releases. During the summer of 1999, the city of Dubuque, IA became the site of Iowa's second cliff release.

Juveniles are hacked in the absence of their parents. Therefore, humans feed the juveniles with farm-raised quail carcasses, and the hatchlings interact with their siblings to learn flying and hunting skills, much as they would in the presence of their parents (Sherrod et al. 1981). Great-horned Owl (*Bubo virginianus*) predation has been a key factor in the success of previous releases (Barclay and Cade 1983, Redig and Tordoff 1988). Our project was developed to monitor the survival and dispersal of juveniles from the hack site area in the context of (1) a relatively large cliff release and (2) concerns of owl predation.

Natal dispersal is the movement from the hatch site to a breeding territory (Greenwood and Harvey 1982), and previous studies have reported on Peregrine Falcon natal dispersal (e.g., Tordoff and Redig 1997, Restani and Mattox 2000). Although natal dispersal is often used as an indicator of recruitment success, the period immediately following fledging may be the most critical to the survival of juvenile Peregrine Falcons (Barclay and Cade 1983). In this paper, we refer to this period as the "post-fledging period," and we use the dispersal of the juveniles away from the hack site as the functional end of this period.

Very little information has been published on the survival and movements of juvenile Peregrine Falcons during the post-fledging period (but see Perez and Zwank [1995] for Aplomado Falcons [*Falco femoralis*]). Our goal was to determine the initial viability of the juvenile falcons released by the Iowa DNR at the cliff hack site. To do this, we monitored movements, determined habitat use, and estimated survival during the post-fledging period, prior to migration.

METHODS

We conducted this study during June–September of 1999 and 2000 in Dubuque County, Iowa (42°30'N, 90°38'W). We placed two hack boxes at the top of a 50-m, east-facing cliff, on the Mississippi River, just below Lock and Dam 11 (Fig. 1). Birds were released in a staggered manner beginning on 22 June 1999 and 20 June 2000 (Fig. 2).

Prior to release, we banded each juvenile with a unique color- and alpha-numerically-coded leg band, in addition to the National Bird Banding lab's (USGS-BRD) anodized band. We also color-marked each juvenile on either

the wing or tail with non-toxic paint. For individuals marked on a single wing, we also marked the opposite side of the head, behind the eye, to allow observers to determine identity from any angle.

We monitored color-marked birds at the hack site from an observation point below the cliff, using a spotting scope and binoculars. Observations were taken daily, usually at 0600–0830 H, 1100–1300 H, and 1800–2030 H.

During 1999, we radio-marked five individuals with leg-mounted transmitters. Because the falcons were able to remove the leg-mounted transmitters, we switched to backpack harnesses on four individuals during 2000. We determined the location of radio-marked birds by triangulating with at least two bearings, once each day following release; bearings were taken from six fixed points surrounding the cliff site. We used Magellan GPS receivers to determine the coordinates of the fixed points, and we used LOCATE II software to estimate the actual coordinates of each bird from the bearing data (Nams 1990). We mapped the position of each bird using ArcView GIS software, version 3.2, and determined home range using the Jennrich-Turner home range algorithm (Jennrich and Turner 1969) within ArcView as an extension (Hooze and Eichenlaub 1997). Compared to other home range estimators, the Jennrich-Turner method is especially useful for determining confidence intervals of home range size and deriving the axes of groups of location coordinates (Hooze and Eichenlaub 1997).

We calculated the proportion of fatalities in our sample based on documented deaths. We calculated 95% confidence intervals (CI) for each binomial sample proportion (Burlison 1980). To facilitate comparisons with other studies, we also calculated an adjusted proportion of fatalities by considering birds that disappeared from the hack site within the first two weeks after release as dead (three of our confirmed mortalities happened in the first week and one happened in the third week). The latter method incorporates many biases and assumptions, and we suggest is a "worst case" scenario.

Just as we do not know what happened to birds that disappeared during the first 2 wk, surveys of most wildlife populations are unable to detect all animals in the intended population. To estimate the probability of surviving a given time interval robustly, it is necessary to use methods which adjust for incomplete detectability (Thompson et al. 1998). Therefore, we summarized our resighting data into weekly discrete time intervals for analysis in a Cormack-Jolly-Seber mark-recapture design (Pollock et al. 1990). This method allows the estimation of weekly survival rates (the probability of surviving one week), as well as other parameters.

We estimated demographic parameters using recovery data from dead birds and resighting data from live birds in the same estimating model (Burnham et al. 1987) in program MARK (White and Burnham 1999). In addition to the usual survival and resighting (the probability of being detected during a week given that the animal is alive) parameters, the incorporation of known deaths to the data set allowed the estimation of a fidelity parameter (ψ , the probability of remaining at the site during a given week; Burnham et al. 1987). We considered several potential models that varied by whether parameters were year-specific or pooled across the 2 yr (Table 1). Because

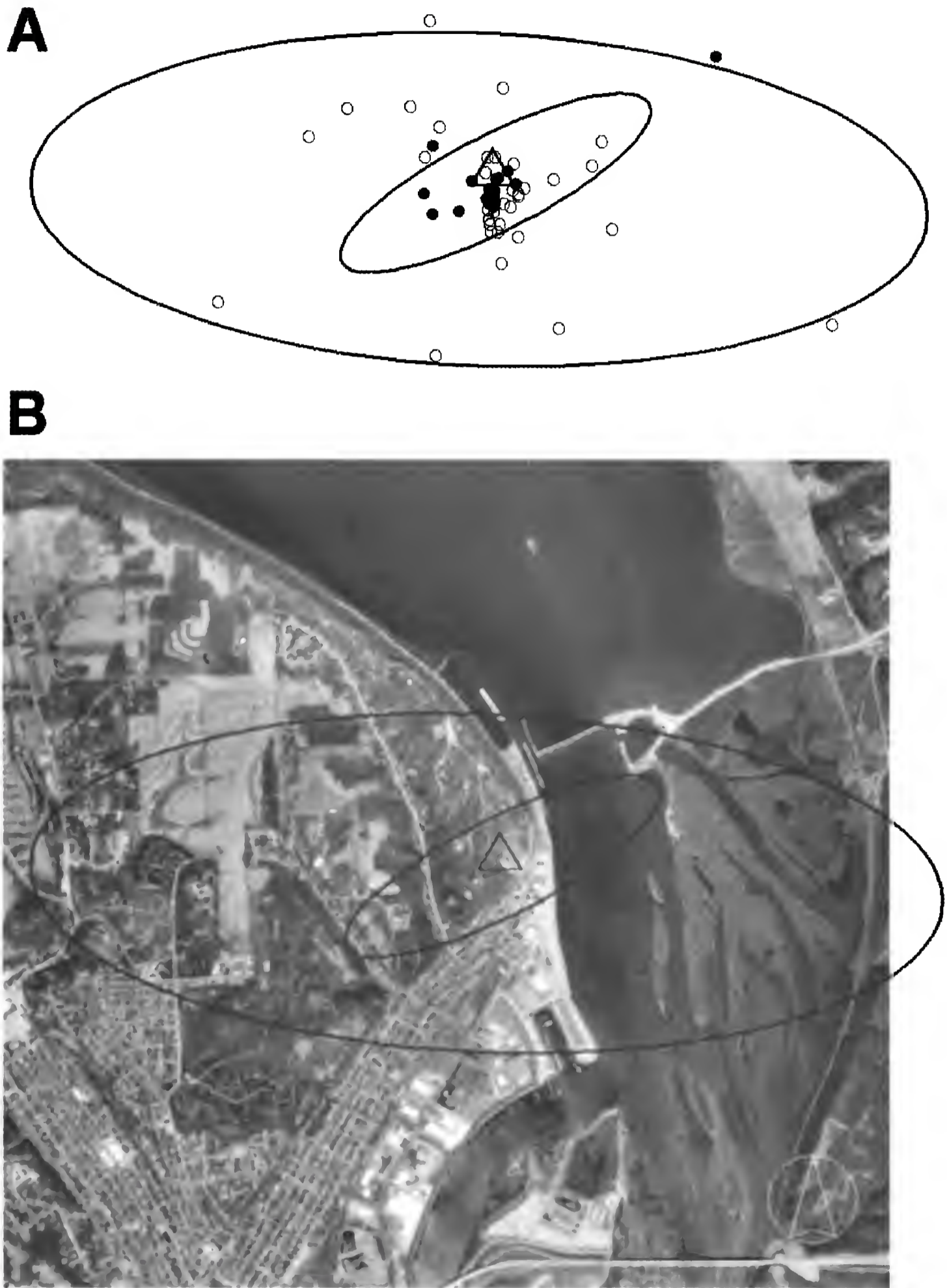


Figure 1. Ranges and locations of radio-marked, juvenile peregrine during 1999–2000 (A), immediately following release from a hack site (Δ) at Eagle Point Park in Dubuque, IA along the Mississippi River (B). Ellipsoids indicate 95% of home range as determined by the Jennrich-Turner method (Jennrich and Turner 1969); small ellipsoid is for 1999 and large ellipsoid is for 2000.

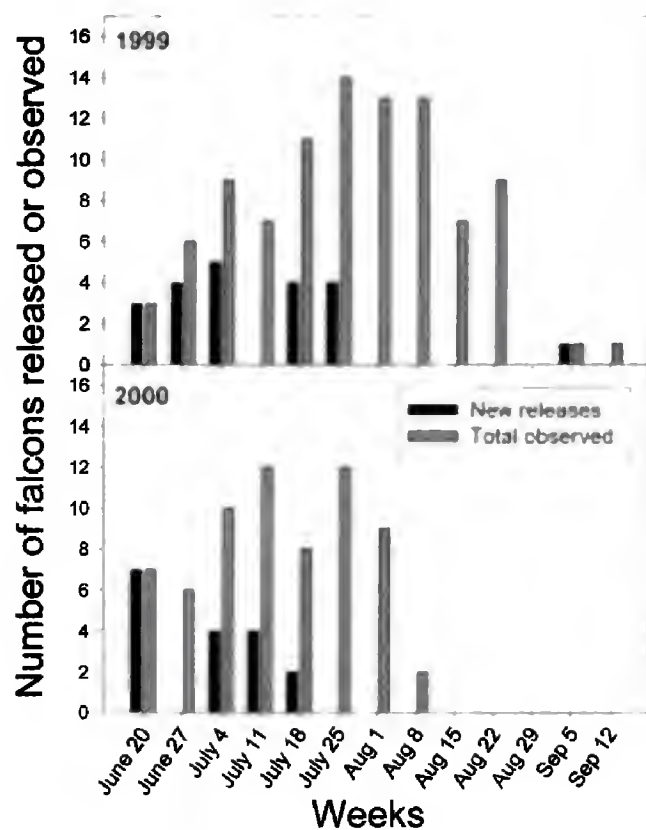


Figure 2. Numbers of juvenile Peregrine Falcons released and total numbers observed at the hack site during 1999–2000 at Eagle Point Park in Dubuque, IA.

of small cohort sizes, we used a corrected Akaike Information Criterion (AIC_c) to select the best estimating model (Burnham and Anderson 1998). We extrapolated ($\hat{S}_{10-wk} = \hat{S}_{wk}^{10}$, $\hat{S}_{annual} = \hat{S}_{wk}^{52}$) our weekly estimates to obtain 10-wk (post-fledging period) and annual survival estimates for comparison with other studies. We used the delta method for approximating variances for these extrapolated survival estimates (Weir 1990).

RESULTS

We released 38 juvenile Peregrine Falcons; 21 in 1999 and 17 in 2000 (Fig. 1). During 1999, we observed Great-horned Owls only once near the hack site, but we observed them at least eight times in 2000.

Survival. We documented four deaths (10.5%, 95% CI = ±10%) during the two post-fledging periods (1999 and 2000); by incorporating six other birds that disappeared before 2 wk the adjusted (worst-case scenario) mortality estimate was 26.0% (95% CI = ±14%). Three of 21 juveniles (14.3%, 95% CI = ±11%) were known to have died in 1999, and 1 of 17 (7.1%, 95% CI = ±7%) was found dead in 2000. The fatalities in 1999 consisted of (1) hypothermia from becoming trapped in a livestock watering tank, (2) overcome by fumes after entering a garbage dumpster recently treated with chlorine tablets, and (3) an assumed mammal predation. In the latter case, skeletal remains were found with skull intact. We used electric fencing

Table 1. Alternate models and model selection data from recoveries and resightings of juvenile Peregrine Falcons following release from a hack site in Dubuque, IA, summers 1999–2000. Each model resulted in estimates of survival (S, probability of surviving the week), resighting rate (p, probability of live birds being detected), recovery rate (r, probability of dead animals being reported), and fidelity rate (Ψ, probability of remaining at the hack site during the week). Falcons were grouped in two cohort groups by year of release; models varied by either estimating two year-specific parameters (g) or one pooled parameter (.) across years. Models were ranked by AIC values (see text), and the ΔAIC_c column shows the difference between the best model (row 1) and the alternate models' (rows 2–8) AIC values.

MODEL	ΔAIC _c	AIC WEIGHT ^a	N PARAM-ETERS
{S(.) p(.) r(.) Ψ(g)}	0.00	0.2786	5
{S(.) p(.) r(.) Ψ(.)}	1.29	0.1464	4
{S(g) p(.) r(.) Ψ(g)}	1.91	0.1074	6
{S(.) p(g) r(.) Ψ(g)}	2.16	0.0947	6
{S(.) p(.) r(g) Ψ(.)}	2.79	0.0690	5
{S(g) p(.) r(.) Ψ(.)}	3.16	0.0573	5
{S(.) p(g) r(.) Ψ(.)}	3.33	0.0528	5
{S(g) p(g) r(g) Ψ(g)}	6.31	0.0117	8

^a AIC Weight is the weight of evidence in favor of the given model being from the set of models considered. AIC Weight is a function of the model's ΔAIC_c value, compared to the other models' ΔAIC_c values (Burnham and Anderson 1998). In our data set, the best model is twice as likely (0.2786 compared to 0.1464) as the second-best model to be the best model.

immediately around the hack site to dissuade mammal scavengers, and several raccoons (*Procyon lotor*) and a red fox (*Vulpes vulpes*) were seen near the hack site. The death in 2000 resulted from an electrocution on a power pole; several falcons were seen using the utility pole as a roost prior to the mortality. No detected fatality appeared to be the result of Great-horned Owl predation. The mean distance of the four deaths from the hack site was 676 m (SD = 411).

Weekly survival ($\hat{S} = 0.988$, SE = 0.01) did not differ between years, and our weekly resighting probability (p) was also constant between years (p = 0.885, SE = 0.03; Table 1). Extrapolating the weekly survival to the entire post-fledging period resulted in a 10-wk survival rate of $\hat{S} = 0.886$ (SE = 0.07, or a 0.114 mortality rate estimate for the same 10-wk period); assuming constant survival for the first year would result in an annual survival estimate of $\hat{S} = 0.534$ (SE = 0.84).

Dispersal. We observed first-year juveniles at our hack site from 22 June–12 September in 1999 and from 20 June–9 August 2000 (Fig. 2). No juveniles from 1999 were observed in 2000, and no juveniles from either year were observed at the site in summer 2001.

Individual juvenile falcons were observed for a mean of 4.3 wk (SD = 2.5) during 1999, and for a mean of 3.4 wk (SD = 2.3) during 2000 ($F_{1,38} = 2.82$, $P = 0.10$). Weekly fidelity (ψ the probability of not dispersing from the hack site during the week) was lower in 2000 (1999: $\psi = 0.903$, SE = 0.03; 2000: $\psi = 0.795$, SE = 0.05; Table 1).

Habitat Use. We observed color-marked birds most often at the hack boxes. However, the juveniles also used the cliff face for roosting, feeding, and social interactions. In addition, we observed juveniles in trees surrounding the hack site. In 1999, 23 of the 28 (82%) “non-hack site” observations were from the cliff face; 5 of the 28 (18%) were from trees. In 2000, only 31 of 72 (43%) “non-hack site” observations were from the cliff face; 41 of the 72 (57%) were from the trees ($\chi^2 = 5.56$, $df = 1$, $P = 0.018$).

In 1999, we obtained 17 sets of useable bearings from radio-marked birds before the leg-mounted radios fell off the birds. The birds were observed picking at the leather/cotton thread attachments, and were soon able to dislodge the transmitters; otherwise, all behaviors of radio-marked birds were normal. In 2000, we obtained 40 sets of useable bearings; apparent signal bounce from the cliff walls prevented program LOCATE II from determining a precise location estimate for other sets of bearings. Birds remained closer to the cliff and hack site during 1999 than in 2000. The mean distance of radio-marked birds from the hack site was 268 m (SD = 296.2) during 1999; in 2000, the mean distance was 619 m (SD = 871, $t = 2.26$, $df = 53$, $P = 0.03$; Fig. 1). In 1999, the minimum distance from the hack site was 72 m and the maximum was 1342 m; in 2000, the minimum was 52 m and the maximum was 5329 m. The number of useful sets of bearings per bird ranged from 1–8 in 1999, and from 4–22 in 2000; the backpack harnesses in 2000 provided more useful data, although one fell off prematurely.

Movements of young peregrines were more often in an east-to-west direction (“inland” from the river), than in a north-to-south direction (along the river). The home range ellipsoid, representing 95% of their daily use, for birds in 1999 was 75 ha,

and measured 1861 m southwest-to-northeast and 512 m northwest-to-southeast; in 2000 the home range ellipsoid was 682 ha, and measured 4839 m east-to-west and 1795 m north-to-south. In both years, the ellipsoid covered Eagle Point Park, Mississippi River, islands on the river, and some urban area. At least 50% of the area was covered by the forested Park (Fig. 1).

DISCUSSION

We did not design this study to measure the effects of Great-horned Owls on juvenile Peregrine Falcons. However, the increased presence of owls at the hack site in 2000 suggests rationale for the observed changes in peregrine behavior. In 2000, when more owls were seen, peregrine juveniles had lower site fidelity, shorter mean observation periods, increased daily distance from the hack site, and greater use of more secluded perch sites (trees). However, survival did not seem to be affected by the presence of the owls. Availability of quail at the hack site may have attracted the owls, while also satisfying the owls’ dietary needs (thus, reducing predation pressure on the falcons). At the least, our study provides evidence that peregrine restoration projects can be carried out successfully in the presence of Great-horned Owls (but see Craig et al. 1988, Redig and Tordoff 1988).

The Iowa Department of Natural Resources considered this release to be successful for several reasons, including high post-fledging site fidelity and survival. Juveniles from previous urban releases in Iowa have left the hack site much earlier (ca. 2 wk) than the juveniles from Dubuque’s natural cliff site (P. Schlarbaum pers. comm.). The longer time spent at the hack site during the post-fledging season could provide for higher survival to the migratory period. Juveniles remained at hack sites in Canada for a mean of 22.9–27.7 d (Fyfe 1988) and from 4–7 wk in Sweden (Lindberg 1988), compared to our observations of 4.3 wk (30 d) in 1999 and 3.4 wk (24 d) in 2000. Fyfe (1988) also reported that birds remained longer at the site of multiple releases, similar to ours.

Radio-marked juveniles in this study had a much more limited range than juveniles or adults in other studies. For comparison, Enderson and Kirven (1983) reported long (≥ 1 km) daily movements for an adult male. Jenkins and Benn (1998) reported mean flights of 10.3–21.9 km for adult males and females, with a mean home range of 123

km². Perez and Zwank (1995) found dispersal flights of 2–16 km for juvenile Aplomado Falcons in Texas. The farthest distance we recorded a falcon from the hack site was just over 5 km, and the combined home range for our group of falcons in 2000 was just 472 ha. The continued presence of food at the hack site, in addition to forest habitat concentrated near the Mississippi River probably contributed to this observation.

Approximately 90% of all juveniles survived the 10-wk post-fledging period, and the sources of mortality were similar to those reported by Barclay and Cade (1983). Our observed mortality of 10.5% was very similar to the 10-wk mortality rate of 11.4% estimated using the mark-recapture model. The survival of these juveniles was high compared to other estimates. Burnham et al. (1988) estimated that 81% of hacked young survived three weeks in the Rocky Mountain region during 1976–85; our survival over 3 wk would be 96.4%. Barclay and Cade (1983) used estimates from other raptor species to arrive at an approximation of 75% survival during the pre-dispersal period—the same as our worst-case scenario estimate. Tordoff and Redig (1997) used resightings of Peregrine Falcons to estimate a minimum first-year survival estimate of 23%, although many survivors were probably not resighted. Our extrapolated first-year survival rate of 53% could potentially be lowered by migratory mortalities. However, the period we monitored may be the most hazardous for juveniles without parental protection (Barclay and Cade 1983); all of our documented fatalities occurred during the first 1–3 wk following fledging. Thus, annual survival could actually be higher than 53% for the birds we monitored. For comparison, Tordoff and Redig (1997) reported a survival rate of 86% for adults in the Midwest; they also determined that hacked juveniles survived at better rates than wild juveniles in the Midwest.

Juvenile survival rates are critical, because a low proportion of available individuals are recruited into the breeding population (Tordoff and Redig 1997, Restani and Mattox 2000). With 38 juveniles released in one location and high survival rates, the cliffs near Dubuque, IA on the Mississippi River have high potential to host a breeding pair in the near future.

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MORPHOLOGY, GENETICS AND THE VALUE OF VOUCHER SPECIMENS: AN EXAMPLE WITH *CATHARTES* VULTURES

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ABSTRACT.—Similarity of two of the cathartid vultures, the Greater and Lesser Yellow-headed vultures (*Cathartes melambrotus* and *C. burrovianus*) has caused field-identification problems. The primary means of distinguishing those vultures are the different flight profiles and general habitat preferences. As part of a larger study of cathartid phylogeny, we sequenced cytochrome *b* for six specimens of the two species. Sequences segregate into two groups, with two of the four Lesser Yellow-headed Vulture specimens clustering with the Greater Yellow-headed specimens. This incongruence led us to reexamine the two apparently misidentified specimens. The first bird, a specimen from the Sedgwick County Zoo, Kansas, had been acquired in 1960 and identified as a yellow-headed vulture. The name on the label of this specimen was not changed after *melambrotus* was established as a separate species in 1964. The second specimen, from Amapá, Brazil, had been identified based on observations of habitat and flight behavior. Because this voucher specimen was available for study, we were able to reexamine the specimen and corroborate the molecular identification as a Greater Yellow-headed Vulture. Without these voucher specimens, we would have misinterpreted the results from the molecular data. This is a reaffirmation of the importance of voucher specimens for accurate scientific work.

KEY WORDS: *yellow-headed vultures*; *Cathartes burrovianus*; *Cathartes melambrotus*; *voucher specimens*; *cytochrome b*; *genetics*.

Morfología, genética y el valor de los especímenes de gaveta: Un ejemplo con los buitres del género *Cathartes*

RESUMEN.—La similitud entre dos buitres del género *Cathartes*: *Cathartes melambrotus* y *C. burrovianus* ha causado problemas de identificación en campo. La diferencia principal para distinguir estas especies son el perfil de vuelo y sus preferencias de hábitat. Como parte de un estudio de filogenia de los *Cathartidae*, hicimos una secuencia del citograma *b* para seis especímenes de dos especies. Las secuencias fueron separadas en dos grupos, con dos de los cuatro especímenes de *Cathartes burrovianus* agrupados con los especímenes de *Cathartes melambrotus*. Está incongruencia permitió re-examinar a dos especímenes incorrectamente identificados. El primero, un espécimen del zoológico del Condado de Sedgwick, Kansas, fue adquirido en 1960 y fue identificado como *Cathartes burrovianus*. El nombre en el rotulo de este espécimen no fue cambiado después de que *melambrotus* fue establecido como especie aparte en 1964. El segundo espécimen, procedente de Amapá, Brasil, había sido identificado con base en observaciones de hábitat y comportamiento de vuelo. Debido a que este espécimen estaba disponible para estudio, pudimos re-examinar el ave y corroborar la identificación molecular como *Cathartes melambrotus*. Sin estos especímenes de gaveta, hubiéramos mal interpretado los resultados de los datos moleculares. Esta es una reafirmación de la importancia de los especímenes de gaveta para el trabajo científico.

[Traducción de César Márquez]

Morphologically similar avian species can be difficult to distinguish in the field. Typically, species identical in appearance are identified by where

they are located (habitat), by their song or calls (e.g., *Empidonax* flycatchers), and possibly other characters such as behavior (Zimmer et al. 2001). Here, we present genetic data and provide an example of how voucher specimens were vital in the interpretation of results and in the identification

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Table 1. Wetmore's (1964) measurements (mm) of Yellow-headed Vultures.

	LESSER YELLOW-HEADED VULTURE				GREATER YELLOW-HEADED VULTURE	
	<i>C. B. BURROVIANUS</i>		<i>C. B. URUBITINGA</i>		<i>C. MELAMBROTUS</i>	
	MALE	FEMALE	MALE	FEMALE	MALE	FEMALE
Tail length	195–225	193–230	205–238	204–236	252–275	272–285
Central rectrix width	42–49	43–49	43–50	43–51	59–70	60–67

of species of New World Vultures (*Cathartes*). This genus currently comprises three species, the Turkey Vulture (*Cathartes aura*) and the Greater and Lesser Yellow-headed vultures (*C. melambrotus* and *C. burrovianus*, respectively).

The two yellow-headed vultures are so similar that they were only recognized as separate species when Wetmore (1964) revised the genus. Wetmore designated the Greater Yellow-headed Vulture as a separate species based on differences in overall plumage color and in size. The distinctive measurements are length of the tail and the width of the central rectrices (Wetmore 1964; Table 1). The Lesser Yellow-headed Vulture was subdivided into two subspecies: *C. burrovianus burrovianus*, the smaller, northern form occurring from Mexico to Venezuela and the slightly larger *C. burrovianus urubitinga*, which occurs from Venezuela and Colombia south to northern Argentina and Uruguay.

The similarity of the Greater and Lesser Yellow-headed vultures has caused problems in identifying birds in the field (Blake 1977, de Schauensee and Phelps 1978); the slight differences in size and head color are not useful in most field circumstances. The primary means of distinguishing these two species are different flight profiles and general habitat preferences. Hilty and Brown (1986) also emphasize differences in general coloration (the Greater Yellow-headed Vulture is darker overall with less lighter coloration on the flight feathers) and in the color of primary quills (white in the lesser yellow-headed, dark in the greater yellow-headed) that can be seen from above in a flying bird. Because these plumage features are difficult to use in many field situations, the two species are likely to be most frequently identified by location and method of hunting. The Lesser Yellow-headed Vulture is considered to be a bird of marshes, flat grass, and open wetland habitats (de Schauensee 1970, Houston 1994, Hilty and Brown 1986), rarely found in forests (del Hoyo et al. 1994). In contrast,

the Greater Yellow-headed Vulture occurs in mature lowland forests and along forest edges (de Schauensee and Phelps 1978, Houston 1994, Sick 1993), rarely wandering over grassland (Hilty and Brown 1986). It hunts by flying over the forest canopy, using a sense of smell to locate carrion (del Hoyo et al. 1994, Houston 1994). Possibly because of its habitat preference, it is considered to soar at greater heights, with a steadier flight pattern, and with wings flatter than the lesser yellow-headed (Hilty and Brown 1986, Sick 1993). The potential for some overlap to occur at forest edges with flooded grassland only increases the difficulty of evaluating some sight records (see Ridgely and Greenfield 2000).

As part of a larger study of cathartid phylogeny, tissue specimens of Greater and Lesser Yellow-headed vultures were obtained from museums and from a zoo (Table 2), and a mitochondrial gene, cytochrome *b*, was amplified and sequenced for six individuals.

In this paper, we report on sequence divergence between the two species. In addition, we discuss the misidentification of two specimens, the usefulness of various methods of identifying these species, and, reiterate the importance of voucher specimens for accurate faunistic and taxonomic studies.

METHODS

Taxon Sampling. Four specimens originally identified as *C. burrovianus*, and two specimens identified as *C. melambrotus* were analyzed (Table 2).

DNA Extraction, Amplification, and Sequencing. DNA was extracted from frozen tissue samples using DNAzol (Molecular Research), according to manufacturers' instructions and then subjected to PCR reactions. Primers (Table 3) were used to amplify and sequence overlapping regions of both strands of the mitochondrial cytochrome *b* gene.

PCR reactions were run in a PTC-200 Peltier Thermal Cycler machine. Double-stranded DNA was generated in 30 μ l solutions run at 40 cycles: 20 sec at 94°C, 15 sec at 55°C, and 1 min at 72°C. The double-stranded DNA template was purified using GeneClean II (Bio 101 Inc., Vista,

Table 2. Voucher numbers and locations of the birds used in this study.

SPECIES	VOUCHER NO.	LOCATION
<i>Cathartes melambrotus</i>	LSUMNS ^a B9005	Pando Department, Bolivia
	LSUMNS B7175	Loreto Department, Peru
<i>Cathartes burrovianus</i>	KUNHM ^b 89344	Yucatan, Mexico
	KUNHM 1872	Yucatan, Mexico
	MPEG ^c CH-268	Amapá, Brazil
	SCZ ^d 4550	Sedgwick County Zoo

^a Louisiana State University Museum of Natural Sciences.

^b University of Kansas Natural History Museum.

^c Museu Paraense Emilio Goeldi.

^d Sedgwick County Zoo.

CA) and resuspended with 18 μ l of ultra pure water. Two μ l were used as template for cycle sequencing using a Prism[™] Ready Reaction DyeDeoxy[™] Terminator Cycle Sequencing Kit. Protocol for the 6 μ l reactions was: pre-heating for 1 min at 95°C, then 35 cycles at 95°C for 15 sec, 50°C for 15 sec, and 60°C for 4 min. The product was precipitated using 74 μ l of an ethanol/MgCl₂ mixture, cleaned with 95% ethanol and resuspended in 2.2 μ l of a 6 to 1 solution of formamide-EDTA. Two μ l of the sequenced product were loaded into a 6% acrylamide gel and analyzed in an ABI Model 377 DNA sequencer.

RESULTS

The cytochrome *b* sequences of these species have been deposited in Genbank (Accession numbers AF494339–AF494342). Sequences of the six specimens segregate into two groups (Table 4), with two of the four Lesser Yellow-headed Vultures (SCZ 4550 and MPEG CH-268) clustering with the Greater Yellow-headed specimens. Within each of these two groups, sequences diverge by 0.09% (one character), between the two groups divergence is 2.7% (31 characters). This level of divergence is similar to the number of sites that differ in com-

parisons of each of these species to the Turkey Vulture (*C. aura*).

The incongruence of these results led us to re-examine the two apparently misidentified Lesser Yellow-headed Vulture specimens. A tail measurement (MPEG CH-268, length 267 mm) or a measurement of the central retrix (SCZ 4550, width 62 mm) was taken. These measurements clearly fall within the range of the Greater Yellow-headed Vulture (Table 1).

DISCUSSION

The reason for the apparent misidentification of the Sedgwick County Zoo specimen (SCZ 4550) is straightforward. It had been acquired from the Cincinnati Zoo, which bought the bird in 1960, four years before Wetmore's (1964) taxonomic revision. It was identified correctly at the time as a yellow-headed vulture (*C. burrovianus*) and that identification was not changed or updated when the species within that genus were revised.

The second specimen (MPEG CH-268) illustrates the potential pitfalls associated with such similar species. In his study of the birds of the state of Amapá, Novaes (1974) cited one 1902 specimen of Lesser Yellow-headed Vulture from Cunani (02°48'N, 51°06'W, Paynter and Traylor 1991), but could not locate specimens of the Greater Yellow-headed Vulture. Despite the lack of specimens, he felt certain that Greater Yellow-headed Vultures occurred in the state at least in the forested western half.

The Amapá individual (MPEG CH-268) used in this study was collected specifically for Griffiths' on-going molecular studies, because at the time the Lesser Yellow-headed Vulture was not represented in any tissue collections. Large open-coun-

Table 3. The sequences of primers used to amplify overlapping regions of both strands of cytochrome *b*.

NAME	LOCATION ^a	SEQUENCE (5' to 3')
L14851 ^b	14851	CCTACTTAGGATCATTCGCCCT
H15149 ^c	15298	GCCCCTCAGAATGATATTTGTCCTCA
L15162 ^c	15311	CTACCATGAGGACAAATATC
H15780	15780	TAGGAATAGGATTAGTACGGAGGCAG
L15636	15636	CTAACAACCCTAGCCCTATTCTCACC
H16057	16057	CTCTGGTAACAAGACCAATG

^a Based on chicken sequence (Desjardins and Morais 1990).

^b Groth (1998).

^c Helm-Bychowski and Cracraft (1993).

Table 4. Nucleotide sites that differ in sequences of cytochrome *b* of *Cathartes melambrotus* and *C. burrovianus* and the bird from Amapá, Brazil. The Amapá specimen is identical to *C. melambrotus* except at a single site (1085).

SPECIMEN	NUCLEOTIDE SITE																
	72	117	165	195	228	249	273	285	327	396	501	534	594	603	628	648	696
<i>C. melambrotus</i>	T	C	C	G	C	G	C	T	G	T	A	G	T	A	C	C	G
Amapá specimen
<i>C. burrovianus</i>	C	T	T	A	T	A	T	C	A	C	G	A	C	G	T	T	A
	699	747	751	753	801	819	843	858	1038	1050	1062	1085	1107	1113			
<i>C. melambrotus</i>	T	C	C	T	C	A	C	C	T	T	A	C	C	A			
Amapá specimen	T	.	.			
<i>C. burrovianus</i>	C	T	T	A	T	G	T	T	C	C	G	T	*	G			

* *C. burrovianus* KUNHM 89344 has a T in position 1107.

try taxa are not often part of modern collections, and vultures may be among the most under-represented groups in modern avian collections. At the time of collection, this particular individual was perched 4 m above the ground on the edge of a gallery forest next to a seasonally-flooded grassland at Lago Cujubím (1°39'N, 50°55'W). During the previous several days, yellow-headed vultures had been seen soaring low over the adjacent open grasslands. Based on the habitat and the flight behavior and light primary shafts of a number of individuals, all had been identified as Lesser Yellow-headed Vultures. There are some fingers of primary forest in the region, but the nearest continuous forest lies some 10 km to the west. Thus, it was assumed that this bird was a Lesser Yellow-headed Vulture. The genetic data clearly refute this and reexamination of the specimen supports the reidentification as a Greater Yellow-headed Vulture. When the problem in the sequences became apparent, we reexamined the voucher specimens. Morphological measurements confirmed the information from the sequence comparisons, that this was, indeed, a Greater Yellow-headed Vulture. This specimen (MPEG CH-268) now represents the first documented record of a Greater Yellow-headed Vulture for Amapá and clearly suggests that this species ventures some distance into the wet grasslands of this region.

Debating the value of specimen-based research, of collecting, and, implicitly, of natural history collections, is becoming increasingly contentious within the scientific community. Editorials and editorial policy in leading ornithological journals appear to question various aspects of collecting (British Ornithologists' Club 2001, British Orni-

thologists' Union 1995). Papers are published based on sequence data, with little or no information about the specimens from which the sequences were derived (Ruedas et al. 2000).

That non-specimen based research has the potential to be sloppy science, with no opportunity to reexamine or verify data, has been noted (Ruedas et al. 2000). Comprehensive statements have been published about the importance of voucher specimens for accurate scientific work, and the importance of collections for science, in general (e.g., Winker et al. 1991, Remsen 1995, Winker 1996, Peterson et al. 1998, Ruedas et al. 2000). In this report, we are not attempting to add to this general review. Rather, we are verifying the necessity of voucher specimens for molecular work. Without the ability to reexamine the specimens, we might have misinterpreted the results from the molecular data used in this study.

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ORAL ADMINISTRATION OF TILETAMINE/ZOLAZEPAM FOR THE IMMOBILIZATION OF THE COMMON BUZZARD (*BUTEO BUTEO*)

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ABSTRACT.—The purpose of this study was to test the efficacy of oral administration of tiletamine/zolazepam in a bait for immobilizing Common Buzzards (*Buteo buteo*) ($N = 20$). Two different dosages and two different methods of administration were compared. A dosage of 80 mg/kg was sufficient in most birds to enable safe handling after 30–60 min, whereas the majority of animals receiving 40 mg/kg still showed defensive reflexes. Birds receiving the drug in a powder form reached the deepest stage of anaesthesia after 30 min, whereas birds receiving a solution reached this stage significantly later, but not before 60 min. When the prepared bait with 80 mg/kg powder was stored for 7 or 14 hr, respectively, effectiveness of immobilization was significantly decreased compared to bait which was administered immediately after preparation.

KEY WORDS: *Common buzzard; Buteo buteo; tiletamine, zolazepam; immobilization; oral administration; capture; Zoletil.*

Administración oral de Tiletamina/zolazepam para la inmovilización de *Buteo buteo*

RESUMEN.—El propósito de este estudio fue el de administrar oralmente tiletamina/zolazepam en un cebo para la inmovilización de *Buteo buteo* ($N = 20$). Dos dosificaciones y dos métodos diferentes de administración fueron comparados. Una dosis de 80 mg/kg fue suficiente en la mayoría de las aves para garantizar una manipulación segura después de 30–60 minutos, mientras que la mayoría de los animales que recibieron 40 mg/kg tenían reflejos para defenderse. Las aves que recibieron la droga en forma de polvo alcanzaron los estados más profundos de anestesia después de 30 minutos, mientras que las que recibieron en solución alcanzaron este estado significativamente más tarde, no antes de 60 minutos. Cuando el cebo con 80 mg/kg de polvo fue almacenado durante 7 o 14 horas respectivamente, la efectividad de la inmovilización disminuyó significativamente comparada con el cebo suministrado inmediatamente después de la preparación.

[Traducción de César Márquez]

Raptors have to be captured in a number of different situations. Birds which escape from their aviary can be dangerous to man, especially if they are imprinted on humans. To prevent such individuals from being killed, they have to be caught immediately. For therapeutic reasons, injured or young birds which are not able to migrate have to be captured. For scientific investigations, wild birds have to be immobilized to be marked, measured, transported, or fitted with a transmitter. The well-known inhalation anaesthesia for birds with isoflurane (Hochleithner 1992) cannot be used for these purposes. Also, chemical immobilization via tele-injection (Wiesner 1998) with a blow pipe or narcotic

rifle is not suitable for the capture of birds due to the possibility of producing serious injuries. Therefore, mechanical methods for capturing birds, especially nets or different types of snares, are still used. The stress for these animals is inevitable and escape attempts followed by injuries sometimes cannot be avoided. The oral administration of different narcotics with a prepared bait was tested in several avian species. Williams and Phillips (1972) tried to catch Rock Doves (*Columba livia*) using Tribromomethanol. The small safety margin of this drug resulted in a mortality rate between 2.9% and 40.6%. Alpha-chloralose, a chloral derivative of glucose (Crider and McDaniel 1967) seems to be suitable for the oral immobilization of Rock Doves (Woronecki et al. 1992, Woronecki and Dolbeer 1994, Belant and Seamans 1999), Wild Turkeys (*Meleagris gallopavo*) (Williams 1966), Marabou

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Storks (*Leptoptilos crumeniferus*) (Pomeroy and Woodford 1976), American Crows (*Corvus brachyrhynchos*) (Stouffer and Caccamise 1991), and Canada Geese (*Branta canadensis*) (Belant and Seamans 1997). Studies on the use of alpha-chloralose in raptors have not been reported. Ketamine, a dissociative anaesthetic, can be administered orally in birds and has a wide safety margin (Kösters and Jakoby 1987). This drug has been used successfully in the immobilization of raptors (Van Heerden et al. 1987) and, for instance, a Harris' Hawk (*Parabuteo unicinctus*) was immobilized with the oral administration of ketamine (Garner 1988). However, the use of ketamine alone in birds may lead to convulsions that can be prevented if it is used together with diazepam (Baronetzky-Mercier and Seidel 1995). The injection of a combination of ketamine and clonazepam, a potent benzodiazepian derivative, was shown to be effective for the immobilization of Common Buzzards (*Buteo buteo*) (Gutzwiller et al. 1984). The objectives of our study were to assess the suitability of oral administration of tiletamine-zolazepam for the immobilization of Common Buzzards, to find the optimal dosage of different preparations, and to evaluate the loss of effectiveness with storage time of the bait. Tiletamine-zolazepam is an injectable anaesthetic combination which provides rapid and smooth induction of anaesthesia (Hui Chu Lin 1996) and has been shown to be effective and safe in many species including raptors (Schobert 1987).

MATERIALS AND METHODS

Animals. We obtained 20 buzzards for this study from the raptor rehabilitation center, Fuchsenbigl, Austria. Use of animals in this study followed the Austrian law on animal experiments (§ 8 BGBl.Nr. 501/1989, GZ 68.205/83-Pr/4/96). The raptors were housed in an aviary. Most of these birds had been found injured in the wild, and 18 of the birds were unable to fly. Birds were clinically examined before and after each immobilization, and neither age, sex, or a detailed health status of the animals was known. All the animals were not fed for 24 hr prior to the application of Zoletil® to create standardized conditions concerning resorption. For the oral application of the prepared meat the birds were manually restrained. Therefore, the animals were put into a paper box and the wings were held firmly against the body. To observe the induction time the birds were transferred into a separate aviary.

Drug and Preparation of the Bait. Zoletil® (Virbac, Carros, France) is a 1:1 combination of tiletamine and zolazepam. Tiletamine is a dissociative anaesthetic with a pharmacological activity similar to ketamine (Lin et al. 1993), but is more potent (Short et al. 1989). Zolazepam is a benzodiazepine agonist and in pharmacological ac-

tivity comparable to diazepam (Loescher 1999). For handling the drug, the same precautions to avoid misuse or accidental intake by humans must be taken as for other commonly-used anaesthetics. Zoletil® comes as a freeze-dried powder suitable to adhere to different surfaces or to dissolve in solutions up to 33%. This combination is used for many domestic and exotic species (Schobert 1987). It was shown to be suitable to produce anaesthesia in buzzards via intramuscular injection (Trah 1990). Dosages for the oral administration of Zoletil® could not be found in the literature. For the oral application, the dry powder was scattered over a piece of rabbit meat or a 10% solution with sterile water was applied on the surface of the meat and allowed to dry for 20 min. For the experiments during phase three the prepared meat with Zoletil® was stored for 7 or 14 hr, respectively, at room temperature and daylight.

Study Design. *Phase one.* Test birds were randomly assigned into two groups. One group ($N = 10$) was fed meat sprinkled with 40 mg/kg of powdered Zoletil®, while the second group ($N = 10$) was fed meat covered with 40 mg/kg Zoletil® in a 10% solution of sterile water.

Phase two. Four wk later we repeated the experiment with a dosage of 80 mg/kg. The birds were again randomly assigned into one of the two groups.

Phase three. Six mo later we repeated the experiment with a dosage of 80 mg/kg powdered Zoletil®, however, a pre-administration period of 7 hr for group one and 14 hr for group two was added. Again, the birds were randomly assigned into one of the two groups.

Assessment of Depth of Anaesthesia. The depth of anaesthesia was judged clinically. We used a modified version of the scale of Gutzwiller (1984): 0 = no effect; 1 = light sedation; 2 = moderate sedation, close approach not possible; 3 = strong sedation, birds able to be handled by experienced people; 4 = superficial anaesthesia, birds able to be handled by inexperienced people; 5 = deep anaesthesia.

All birds were checked 30, 60, and 90 min after application of Zoletil® anaesthesia. If approach to and handling of the buzzards were possible (stage 3), every check included assessment of heart and respiration rate. In buzzards which had reached stage 4, the palpebral reflex, corneal reflex, head position, and neck muscle tone were tested additionally. For birds in stage 5, the reflex-monitoring system of Korbel et al. (1997) was used. Following the last check, the birds were taken out of the aviary and put into a cardboard box where they spent the night, before they were returned to their common aviary. Recovering birds were checked every 30 min in the cardboard box until they returned to stage 0. No more than two animals were immobilized at the same time.

Statistics. To test for differences in anaesthesia depth we used a non-parametric analysis of variance for repeated measurements with time course of anaesthesia as the within-subjects factor and dose as well as preparation of Zoletil® as between-subjects factors (Zar 1984). Depth of anaesthesia values were transformed to ranks for analysis. To test statistical differences in induction time between baits with different storage time we used two-tailed Mann-Whitney U -tests. Criterion for detection of statistically significant differences was $P \leq 0.05$.

Animals that died in association with the use of the

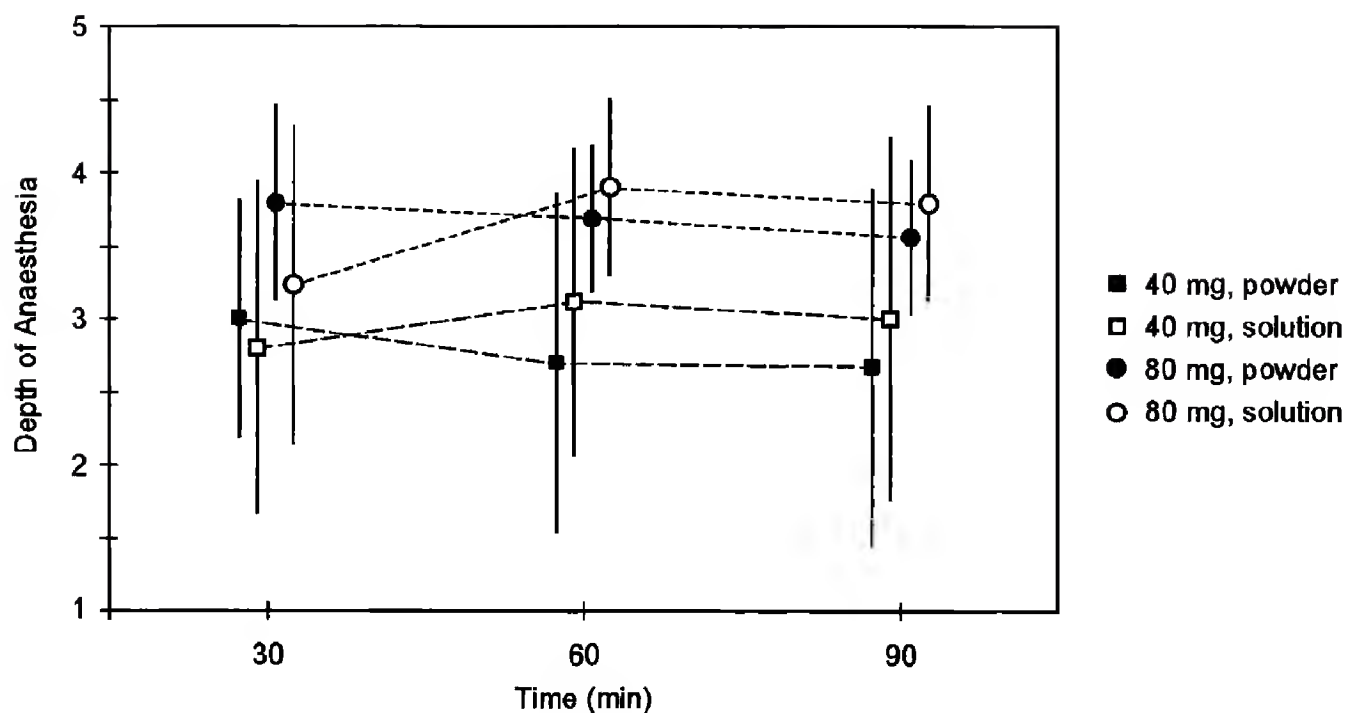


Figure 1. Depth of anaesthesia (mean \pm SE) 30, 60, and 90 min after receiving oral Zoletil[®] (0 = no effect; 1 = light sedation; 2 = moderate sedation, close approach not possible; 3 = strong sedation, birds able to be handled by experienced people; 4 = superficial anaesthesia, birds able to be handled by inexperienced people; 5 = deep anaesthesia).

drugs were necropsied following standard protocols (Rofe et al. 1996). Necropsy was carried out by the Institute of Pathology at the University of Veterinary Medicine, Vienna, Austria.

RESULTS

The 20 animals receiving 40 mg/kg had significantly lower mean values of depth of anaesthesia than the animals receiving 80 mg/kg ($P < 0.02$). Fifteen out of 18 birds receiving 80 mg/kg (83%) reached stage 4, whereas only six out of 20 (30%) birds receiving 40 mg/kg achieved that stage. However, 17 out of 20 (85%) animals of that group reached at least stage 3 (Fig. 1).

The administration form had no overall effect

on the depth of anaesthesia. However, the time course of anaesthesia depended on the preparation of oral Zoletil[®]. The groups receiving Zoletil[®] as a dry powder reached the deepest stage of anaesthesia with both dosages after 30 min, whereas the groups receiving Zoletil[®] solution did not reach this stage before 60 min (Fig. 1). This interaction between administration form and anaesthesia time course was significant ($P < 0.02$).

Storage of the drugged bait had a highly significant effect on the depth of anaesthesia (Table 1). Anaesthesia was deeper for fresh baits ($P < 0.001$) at 30, 60, and 90 min after application compared to the depth reached after administration of the

Table 1. Stage of anaesthesia (mean \pm SE) 30, 60, and 90 min after receiving oral Zoletil[®] (0 = no effect; 1 = light sedation; 2 = moderate sedation, close approach not possible; 3 = strong sedation, birds able to be handled by experienced people; 4 = superficial anaesthesia, birds able to be handled by inexperienced people; 5 = deep anaesthesia).

	N	TIME ^a	TIME AFTER APPLICATION		
			30 MIN	60 MIN	90 MIN
40 mg/kg powder	10	0	3.0 \pm 0.3	2.7 \pm 0.4	2.7 \pm 0.4
40 mg/kg solution	10	0	2.8 \pm 0.4	3.1 \pm 0.3	2.9 \pm 0.4
80 mg/kg powder	9	0	3.8 \pm 0.2	3.7 \pm 0.2	3.6 \pm 0.2
80 mg/kg solution	9	0	3.2 \pm 0.3	3.9 \pm 0.2	3.9 \pm 0.2
80 mg/kg powder	9	7	1.8 \pm 0.3	2.4 \pm 0.3	1.8 \pm 0.3
80 mg/kg powder	9	14	1.6 \pm 0.3	2.1 \pm 0.3	1.7 \pm 0.3

^a Storage time (hr).

drugged, stored bait. Mean (SE) depth of anaesthesia was highest 60 min after application with 2.4 and 2.1 for a storage time of 7 and 14 hr, respectively. Differences between 7 and 14 hr were not significant. All immobilized birds recovered completely. After 120 min of application, the depth of anaesthesia was <3 in all cases; the no effect level was reached after 5 hr in all except in four animals. These buzzards, two of them receiving 80 mg/kg Zoletil® solution and two 80 mg/kg Zoletil® in a powdered form, respectively, had an 8 hr recovery time before reaching stage 0.

Two birds died during this study. Both animals received 40 mg/kg Zoletil® as a solution. One died on the second day after the trial and one after a wk. The first bird showed massive edema of the mandibular space and intranuclear inclusions in renal tubular epithelial cells indicating a viral infection of the kidneys of an unknown origin. The second bird had an unremarkable recovery before it suddenly died seven days later. Necropsy of the bird showed severe arteriosclerosis, myocardial degeneration, cardiac insufficiency, and purulent hepatitis.

DISCUSSION

The combination of tiletamine and zolazepam has a wide safety margin and its use in birds is well documented (Schobert 1987, Blyde 1992, Hayes 1996). The depth of anaesthesia depends on the dose. Dose rates for intramuscular injection in birds range from 2 mg/kg in Common Rheas (*Rhea americana*) to 75 mg/kg in Green Herons (*Butorides virescens*) according to Schobert (1987). For buzzards, 14 mg/kg (Gray 1974) or 30 mg/kg (Trah 1990) were recommended. Giving Zoletil® orally seems to have a much wider safety margin according to our results. Therefore, there seems to be little risk for the life of non-target animals, which may accidentally feed on the bait, due to overdosing. However, in general birds were not in a stage of anaesthesia that would allow minor surgical procedures, even with a dose of 80 mg/kg. A possible explanation for this observation could be the fact that the breakdown of Zoletil® in the blood starts before the total absorption from the gastrointestinal tract has been completed.

The two deaths that occurred during the study were not a consequence of the experiment according to the post mortem findings. Any kind of anaesthesia induces a certain amount of considerable stress for each organism, which can lead to pro-

gression of preexisting diseases. This might have been the case in the first bird that died. However, 83% of the birds that received 80 mg/kg of the freshly-prepared bait, without storage time, were appropriately immobilized to allow inexperienced people to handle them safely (stage 4), whereas only 30% of the birds receiving 40 mg/kg reached that stage. Nevertheless, the dose of 40 mg/kg would be sufficient to enable handling of buzzards by experienced people, as 85% of the animals of that group reached at least stage 3.

Storage time of the drugged bait reduced the potency of the drugs. A major loss of drug effect occurred in the first 7 hr, whereas in the next 7 hr the reduction of efficacy was less. Birds which are anaesthetized using a stored bait were sedated, but could not be handled by experienced people in all cases. Therefore, the drugged bait should be replaced after several hr if it is not taken by the bird. Although baits were force fed in this study, it is likely that birds will readily accept the prepared bait in one piece as previous experiences with different raptors have shown (H. Frey unpubl. data, M. Janovsky unpubl. data).

It is interesting to note that the form of the oral drug (i.e., powder or solution) had a significant effect on the time course of the anaesthesia. The groups receiving Zoletil® in a powdered form reached the deepest stage of anaesthesia after 30 min, whereas the groups receiving Zoletil® solution did not reach this stage before 60 min (Fig. 1). The reason for this phenomenon is not yet clear, but it seems possible that the liquid drug permeates into the bait, whereas, the powder stays on the surface allowing a quicker absorption.

Pain sensation in birds is comparable to that in mammals (Gentle 1992). Therefore, surgical procedures should not be carried out if only oral Zoletil® has been administered. In addition, the widespread used cyclohexamines like ketamine or tiletamine do not produce deep enough analgesia for surgical procedures in birds if used as a monoanaesthetic (Korbel 1998, Korbel et al. 1998). Thus we do not recommend surgery in birds which are immobilized with Zoletil® only. Although most birds recovered completely after 5 hr, full recovery took 8 hr in 4 animals. In practice, drugged birds should be kept isolated at minimum of 24 hr for complete recovery because absorption and metabolization rates vary individually.

We conclude that the oral application of liquid

or powdered Zoletil® in a dosage of 80 mg/kg is an appropriate method to immobilize Common Buzzards to enable safe handling. The safety margin of the drug combination at oral administration appears to be wide enough for use in capturing Common Buzzards of unknown mass.

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HUNTING BEHAVIOR OF AND SPACE USE BY EASTERN SCREECH-OWLS DURING THE BREEDING SEASON

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ABSTRACT.—The hunting behavior of and space use by radio-tagged Eastern Screech-Owls (*Otus asio*; three pairs, one unmated male, and four fledglings) were observed in central Kentucky from March–June 1999. Screech-owls perched at a mean height of 2.2 m. The mean giving up time was 340.2 sec, while the mean time until initiating an attack was 361.4 sec. Owls attacked prey located a mean distance of 2.27 m from perches, and the mean distance between successive perches was 10.33 m. Male screech-owls perched higher in taller trees during the fledgling period, and these higher perches were apparently used when hunting insects. Male screech-owls also hunted in different areas of their ranges during different breeding periods, possibly to take advantage of temporarily abundant prey and avoid prey depletion. During the fledgling period, the hunting ranges of paired male and female screech-owls overlapped, but males and females did not hunt in areas of overlap at the same time.

KEY WORDS: *Eastern Screech-Owl; Otus asio; hunting behavior; perch time; range use; breeding season.*

Comportamiento de caza y uso de espacio de *Otus asio* durante la estación reproductiva

RESUMEN.—El comportamiento de caza y uso del espacio de *Otus asio* dotados con radio transmisores (tres parejas, un macho solitario y cuatro volantones) fue observado el centro de Kentucky desde marzo–junio 1999. Los búhos se posaron en perchas a un altura de 2.2 m. El tiempo de espera fue de 340.2 sec, mientras que el tiempo medio hasta el inicio de un ataque fue de 361.4 sec. Los búhos atacaron a presas localizadas a una distancia media de 2.27 m de las perchas y a un distancia media entre perchas sucesivas de 10.33 m. Los machos se ubicaron en las perchas más altas de árboles grandes, aparentemente estas fueron utilizadas para la caza de insectos. Los machos también cazaron en áreas distintas a sus rangos durante las diferentes etapas de su reproducción, posiblemente para aprovechar la abundancia temporal de presas y evitar agotarlas. Durante el período del crecimiento del plumaje, los rangos de caza de las parejas se traslaparon pero los machos y hembras no cazaron en áreas de traslape al mismo tiempo.

[Traducción de César Márquez]

Eastern Screech-Owls (*Otus asio*) are found throughout eastern North America and have been the subject of many studies (e.g., Belthoff et al. 1993, Sparks et al. 1994, Duguay et al. 1997). However, as with other nocturnal predators, direct observations are difficult and, as a result, little is known about the hunting behavior of screech-owls. In one study, Abbruzzese and Ritchison (1997) observed the hunting behavior of Eastern Screech-Owls in central Kentucky and found no differences between males and females during the nonbreeding season.

As the breeding season approaches, males and females in many owl species exhibit a division of labor (Reynolds and Linkhart 1987). The reproductive success of male raptors may depend on their ability to supply food for mates and offspring throughout the breeding season, beginning with courtship and continuing through the post-fledging period. Once young are able to thermoregulate and manipulate prey, females resume hunting and assist the male in provisioning nestlings and fledglings (Hovis et al. 1985, Gehlbach 1994). The changing roles of male and female screech-owls during the breeding season may influence their hunting behavior. Our objective with this study was to examine how changing food demands and the changing roles of male and female Eastern Screech-Owls might influence hunting behavior and space use during the breeding season.

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METHODS AND MATERIALS

The hunting behavior of Eastern Screech-Owls (three breeding pairs, an unpaired male, and four fledglings) was studied from 5 March–8 June 1999 at the Central Kentucky Wildlife Management Area, located 17 km southeast of Richmond, Madison County, Kentucky. Adult screech-owls were captured by checking nest boxes or by luring them into mist nets using the playback of bounce songs (Ritchison et al. 1988). Captured owls were fitted with 5–6 gm (about 3–4% of screech-owl body mass) radio-transmitters with activity switches (Wildlife Materials, Inc., Carbondale, IL) attached backpack style (Smith and Gilbert 1981). Yellow reflective tape (about 5×1 cm) was attached to the antennas of the transmitters to make it easier to locate hunting owls. Owls were allowed 1 wk to become accustomed to the transmitters before observations began. Several days prior to anticipated fledging dates, nestlings ($N = 13$ in three nests) were banded with a numbered aluminum leg band, and four (two from one nest and one each from the other two nests) were fitted with radio-transmitters.

One or two radio-tagged owls were observed each night and general locations were determined using a receiver (TR-2, Telonics Inc., Mesa, AZ) and a hand-held, two-element yagi antenna. Specific locations of owls were determined using a red-tinted flashlight. Owls were assumed to be hunting when their attention appeared to be focused on the ground (probably searching for ground-based prey) or on nearby vegetation (probably searching for birds or insects) or flying insects. When hunting, owls also exhibited frequent head movements. In addition, hunting fledglings, in contrast to non-hunting fledglings, were always perched several meters away from siblings.

Observations were made at least four times per week. Observation periods started shortly after sunset and were usually 3–4 hr in duration. We typically watched owls from a distance of 8–12 m. Adult females were difficult to observe during the nestling period because they often responded aggressively. Once females began hunting further away from nest sites (about 1 wk after young fledged), our presence seemed to have little effect on their behavior. As a result, only female hunting perches located during the fledgling period were used in analyses of hunting behavior. The breeding status of each pair was categorized as: pre-nesting, egg-laying/incubation, nestling, or fledgling.

During focal owl observations, we tape-recorded all information and perches and attack sites were marked with flagging (placed several meters away) for later analysis. Perch times were classified as either giving-up time (the owl flew to another perch without initiating an attack) or time until attack. These times were only determined when we either observed owls landing on a perch or could estimate landing time based on a change in a transmitter's pulse rate (all transmitters had activity switches).

Within 2 wk, we returned to perches and measured perch height, substrate (tree, shrub, or vine) height, the distance from the tree trunk to the perch, and the diameter at breast height (DBH) of the perch tree. We also noted tree species and measured the distance to the next perch. If successive perches were more than 50 m apart,

we determined the distance using ArcView 3.1 (Environmental Systems Research Institute, Redlands, CA).

The location of each perch was recorded in Universal Transverse Mercator (UTM) coordinates using Global Positioning System (GPS) receivers (Garmin XLS). To determine the size of the areas used by owls, we produced 100%-minimum-convex polygons (MCP) using ArcView 3.1 with the Spatial Tools and Animal Movement extensions. We also measured the distance from each perch to the nest in each territory.

Repeated measures analysis of variance was used to examine the possible effects of breeding period on the hunting behavior of male screech-owls. Small sample sizes precluded comparison of male hunting behavior to that of females and fledglings. All analyses were conducted using the Statistical Analysis System (SAS Institute 1989). All values are presented as mean \pm standard error. Individual owls are referred to by the last three digits of their (or their mate's or parents') radio-transmitter frequency.

RESULTS

Eastern Screech-Owls ($N = 4$ males, 3 females, and 4 fledglings of unknown sex) were observed for a total of 182 hr on 55 nights from 5 March–8 June 1999. Overall, these owls perched at a mean height of 2.20 ± 0.08 m ($N = 304$ perches) in trees with a mean height of 5.63 ± 0.24 m ($N = 301$ perches) and mean DBH of 0.12 ± 0.01 m ($N = 301$ perches; Table 1). Owls ($N = 11$) perched a mean distance of 0.51 ± 0.05 m ($N = 302$ perches) from the main trunk of perch trees or snags. Forty different plant species were used as perches by screech-owls, with snags (21.6%), American ash (11.1%, *Fraxinus americana*), and American sycamore (7.8%, *Platanus occidentalis*) used most frequently.

The mean giving-up time ($N = 231$ perches of 9 owls) was 340.2 ± 18.7 sec, while the mean time until initiating an attack ($N = 18$ perches of 6 owls) was 361.4 ± 32.0 sec. Owls ($N = 6$) attacked prey located a mean distance of 2.27 ± 0.28 m from perches ($N = 21$). The mean distance between successive perches was 10.33 ± 1.42 m ($N = 65$ perch changes by 9 owls).

We observed 21 attacks by screech-owls, with 11 successful and 10 unsuccessful. Eight of 18 attacks by males were successful, resulting in the capture of three birds, one small mammal, two crayfish (*Cambarus* spp.), and two insects. Attacks by females ($N = 2$) and fledglings ($N = 1$) on insects were all successful.

Although adult females and their young were only observed hunting during the fledging period, two males were observed during the pre-nesting,

Table 1. Perch characteristics and hunting behavior of male, female, and fledgling (unknown sex) Eastern Screech-Owls. Values are presented as mean \pm one standard error. Sample sizes are in parentheses.

	MALES (<i>N</i> = 4)	FEMALES (<i>N</i> = 3)	FLEDGLINGS (<i>N</i> = 4)
Perch height (m)	1.93 \pm 0.07 (258)	3.50 \pm 0.36 (22)	3.99 \pm 0.34 (24)
Tree height (m)	4.70 \pm 0.22 (255)	9.79 \pm 0.92 (22)	11.67 \pm 0.79 (24)
Distance from tree trunk (m)	0.42 \pm 0.05 (256)	0.97 \pm 0.27 (22)	0.97 \pm 0.23 (24)
Tree DBH (m)	0.10 \pm 0.01 (255)	0.19 \pm 0.03 (22)	0.23 \pm 0.05 (24)
Distance to next perch (m)	11.07 \pm 1.72 (53)	2.25 \pm 0.48 (2)	8.75 \pm 1.53 (10)
Attack distance (m)	2.32 \pm 0.30 (18)	0.75 \pm 0.25 (2)	2.5 (1)
Giving-up time (sec)	355.5 \pm 20.4 (201)	183.2 \pm 62.4 (13)	279.9 \pm 52.0 (17)
Attack time (sec)	381.3 \pm 32.3 (16)	150 (1)	255 (1)

egg-laying/incubation, nestling, and fledgling periods and a third male was observed during all but the pre-nesting period. Analysis of the hunting behavior of these males revealed that mean perch height ($F_{3,5} = 14.85$, $P = 0.006$) and tree height ($F_{3,5} = 7.99$, $P = 0.024$) varied among periods (Fig. 1), with males perching higher in taller trees during the fledgling period (Tukey's test, $P < 0.05$). We found no differences among breeding periods in other variables, including DBH of perch trees ($F_{3,5} = 2.43$, $P = 0.18$), distance of perches from the main trunk ($F_{3,5} = 3.5$, $P = 0.11$), giving-up time ($F_{3,5} = 0.71$, $P = 0.59$), and distance between successive perches ($F_{2,3} = 2.07$, $P = 0.27$). Sample sizes were too small to examine possible differenc-

es among periods in either attack time or attack distance.

For male screech-owls, the mean distance of hunting perches from nest sites did not differ ($F_{3,5} = 0.1$, $P = 0.95$) among breeding periods. Similarly, the size of male hunting ranges did not vary ($F_{3,5} = 4.13$, $P = 0.056$) among breeding periods. However, specific hunting areas used by male screech-owls did vary among breeding periods. For example, male 957 hunted primarily in the northeastern portion of his territory during the pre-nesting period, but in the southwestern section during the egg-laying/incubation period. During the nestling period, male 957 hunted primarily in the southeastern section of his territory and, during the fledgling period, again used the southwestern section (Fig. 2).

Based on a limited number of observations, the

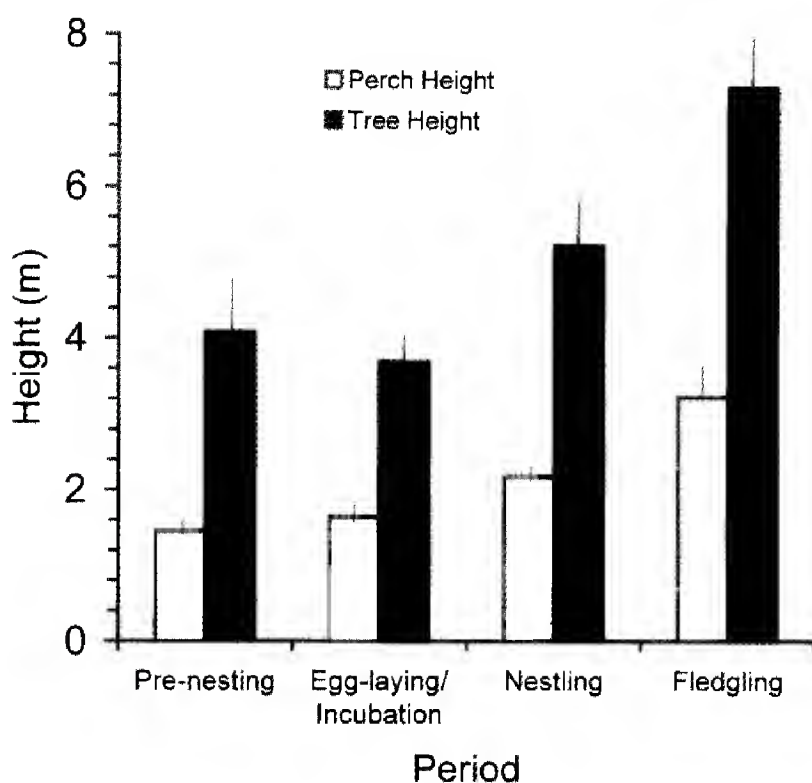


Figure 1. Variation among breeding periods in mean perch heights of and trees used by male Eastern Screech-Owls.

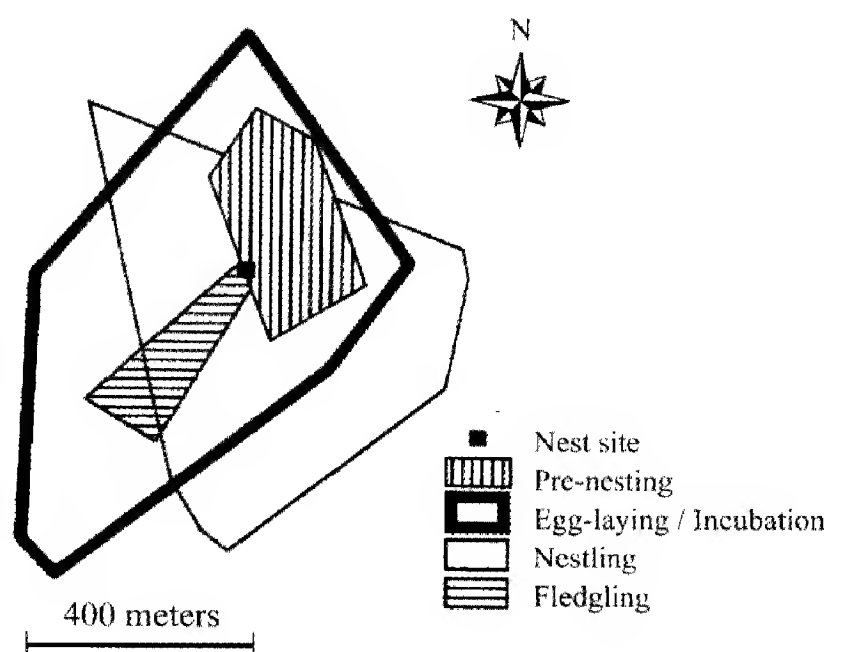


Figure 2. Variation in areas used by screech-owl male 957 during different breeding periods in central Kentucky.

sizes of hunting ranges of the three adult female screech-owls during the fledgling period were 7.0 ha (two observation periods, five perches), 9.4 ha (two observation periods, 12 perches), and 12.2 ha (five observation periods, 10 perches), respectively. During the fledgling period, the hunting range of male 957 completely encompassed that of his mate, while the ranges of males 997 and 037 included 87% and 59% of the hunting ranges of females 997 and 037, respectively. Although ranges overlapped, paired males and females were not observed hunting in the same locations on the same night.

Fledglings ($N = 4$ from three different broods) were observed during 10 observation periods (from the day of fledging through 29 d post-fledging). Young owls were first observed hunting 26 d after fledging. The first (and only observed) successful attack by a fledgling was 27 d after fledging.

DISCUSSION

Eastern Screech-Owls in our study perched at a mean height of 2.2 m. Similarly, Gehlbach (1994) found that Eastern Screech-Owls in suburban Waco, Texas, hunted from perches at a mean height of 2.6 m. However, at the same location where we conducted our study, Abbruzzese and Ritchison (1997) reported a mean perch height of 1.66 m for screech-owls during the non-breeding season. The higher mean perch height in our study may be due to seasonal differences in the types of prey being hunted and seasonal differences in vegetation height. During the non-breeding season, screech-owls in central Kentucky appeared to be hunting primarily crayfish and small mammals (Abbruzzese 1996). In contrast, screech-owls in our study often appeared to be hunting for insects, and owls may perch higher when hunting insects to increase the visual search area and to scan areas above vegetation for flying insects. For example, Village (1990) noted that Eurasian Kestrels (*Falco tinnunculus*) chose higher perches when hunting invertebrates because such perches provide larger scanning areas than low perches.

During the breeding season, vegetation is taller and thicker than during the non-breeding season and, in response, screech-owls may perch higher. Similarly, Morrison (1980) found that Loggerhead Shrikes (*Lanius ludovicianus*) selected higher perches during the summer than during the winter because such perches provide a wider field of view in taller, more dense summer vegetation.

Other nocturnal owls hunt from perches at heights similar to those used by screech-owls in our study. For example, Boreal Owls (*Aegolius funereus*) have been reported to hunt from perches at mean heights of 1.7 m (Norberg 1987) and 3.1 m (Bye et al. 1992). Lower perches may provide a better view of ground-dwelling prey in areas where low-growing cover and shrubs obscure much of the ground (Norberg 1987). In addition, low perches may allow owls to better locate prey using acoustic cues (Norberg 1987, Abbruzzese 1996).

The mean giving-up time for screech-owls in our study was 340.2 sec (or 5.7 min). Abbruzzese and Ritchison (1997) reported a mean giving-up time of 278.9 sec (4.6 min) for Eastern Screech-Owls during the non-breeding season, while Gehlbach (1994) found that screech-owls hunting primarily insects had a mean giving-up time of just 72 sec (1.2 min). Gehlbach (1994) suggested that prey abundance influences giving-up times, with owls able to assess areas of dense prey more quickly. Differences in the type of prey being hunted may also influence giving-up times. For example, Bye et al. (1992) suggested that owls need more time to search for small mammals than other prey because their movements are difficult to detect.

The mean time before initiating an attack by screech-owls in our study was 361.4 sec (or 6 min), similar in duration to the mean giving-up time (340.2 sec). During the non-breeding season, Abbruzzese and Ritchison (1997) also reported no difference between giving up and attack times for screech-owls, and suggested that predators detecting invertebrate prey do not wait long before initiating an attack because such prey are relatively easy to capture. In addition, we observed screech-owls capturing or attempting to capture flying insects on several occasions. Because flying prey can quickly move out of range, owls likely attack such prey shortly after detection. In contrast, Bye et al. (1992) suggested that Boreal Owls had longer attack, or detection, times because they waited longer before initiating attacks on small mammals. Owls attacking small mammals may wait longer because such prey are difficult to capture (Toland 1987, Bye et al. 1992, Atkinson and Cade 1993). Hayward and Hayward (1993) noted that, after detecting prey, Boreal Owls sometimes waited 10 min or more if a prey item was not in a vulnerable position.

Paired males in our study perched higher in taller trees during the fledgling period than during

the pre-nesting and egg-laying/incubation periods. This increase in perch height may have been due to a change in the type of prey being hunted. Male screech-owls appeared to hunt primarily small mammals and crayfish during the pre-nesting and egg-laying/incubation periods, but were only observed hunting insects during the fledgling period. Although this switch may have been due in part to changes in availability, other factors may have also contributed. Gehlbach (1994) suggested that male screech-owls provision females with large prey items during the pre-nesting period because females require substantial amounts of energy to produce a clutch of eggs. However, during the nestling and fledgling periods, male screech-owls may provide their young with smaller, more easily-handled prey like insects (Gehlbach 1994). McClain (1997) found that adult screech-owls delivered primarily small prey items (including beetles [Coleoptera] and moths [Lepidoptera]) to nestlings.

Male screech-owls in our study were observed hunting in different parts of their ranges during different breeding periods. One possible reason for such shifts may be to take advantage of temporarily abundant prey. For example, Gehlbach (1994) observed that screech-owls sometimes made repeated trips to and from concentrations of invertebrates such as emerging cicadas (Cicadidae) and earthworms (Lumbricidae). Village (1990) noted that raptors sometimes return to hunting sites where prey have been captured. Similarly, we sometimes observed male screech-owls make repeated visits to the edges of temporary pools and intermittent streams, possibly searching for crayfish that were sometimes abundant at such locations (pers. observ.). Abbruzzese (1996) also reported that screech-owls in central Kentucky frequently hunted for crayfish. Another reason why male screech-owls might hunt in different areas at different times is to avoid depletion of prey resources. For example, Village (1990) found that the hunting ranges of Eurasian Kestrels were not used uniformly and that kestrels were familiar with their territories and the prey resources, as shown by the use of the same hunting locations on successive days.

The hunting ranges of paired male and female Eastern Screech-Owls in our study overlapped, particularly around nest sites. However, our observations also suggest that members of a breeding pair did not use the same hunting locations on the

same nights. Other investigators have also found that paired male and female raptors forage in different areas probably to avoid competition and prey depletion (Newton 1986, Craig et al. 1988, Village 1990, Gehlbach 1994). Gehlbach (1994) observed that male and female screech-owls leaving nest sites on hunting forays usually flew in different directions. Similarly, male and female Eurasian Sparrowhawks (*Accipiter nisus*) typically hunt in different directions from the nest (Newton 1986, Selas and Rafoss 1999).

We first observed hunting by fledgling screech-owls 26 d after fledging. However, we only observed the four radio-tagged fledglings on four occasions during the period from 16–26 d post-fledging (on days 16, 19, 22, and 26). Gehlbach (1994) reported that young screech-owls attempt to capture insect prey just 9–14 d after fledging and are able to feed themselves regularly 3–4 wk after fledging.

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SHORT COMMUNICATIONS

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SCHIZOCHROMISM IN A PEREGRINE FALCON FROM ARIZONA

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KEY WORDS: *Peregrine Falcon*; *Falco peregrinus*; *albinism*; *leucism*; *schizochromism*; *cannibalism*.

Herein, we report the first record of schizochromism in the Peregrine Falcon (*Falco peregrinus*). Our example is a nestling from southern Arizona. The lack of dark brown pigment in this bird made it closely resemble the blue-gray plumage of an adult. Near fledging time, the bird was eaten by its nestmates, so this article also documents cannibalism.

Abnormal pigmentation in wild birds is unusual (but see Fitzpatrick [1980] for a species in which white patches are common), but aberrant, pale individuals have been reported for many species (Sage 1962, Ross 1973). A BIOSIS search of the literature for only a 6-yr period located over 100 titles dealing with abnormal plumage in wild birds.

Incomplete pigmentation can take many forms. True or complete albinos lack all pigment, not only in plumage but also in talons and iris. Incomplete albinos usually have patches of white feathers in otherwise normal plumage. Schizochromism is the condition in which one or more pigments are lacking, while others are expressed. Sage (1962) mentions a Green Woodpecker (*Picus viridis*) which was pure white except for its red crown. Some forms of polymorphism are probably derived from some form of albinism. A notable example is the pallid morph of the austral Peregrine Falcon (*F. p. cassini*) of southern South America (Ellis and Peres Garat 1983). In this morph, juvenile and adult birds weakly express the normal color pattern (bars, streaks, etc.), but have much less pigmentation everywhere (a condition termed leucism), especially in the areas that are lightest in normal peregrines. In pallid falcons, even the talons and bill, black and deep blue in a normal peregrine, are blond.

There are a few published accounts of albinism in the Peregrine Falcon. McGregor (1900) noted a juvenile peregrine with two white secondaries (an example of incomplete albinism) in California. A nearly pure white adult from Devonshire, England is housed at the American Museum of Natural History (specimen no. 453937). Sage (1962) included the Peregrine Falcon on the list of species for which albinism is known for the British Isles.

On 17 July 1978, we entered an eyrie in southern Arizona (32–33°N, 110–111°W) where at least three normally pigmented young had fledged about 6 wk earlier. Conspicuously scattered along the eyrie shelf and conspicuous below the eyrie were hundreds of feathers, all still in the blood (i.e., all partially grown), of a pale nestling that died at about 5 wk of age. Feathers matching this bird (and at least one peregrine talon) were also found in castings on the eyrie shelf. We retrieved as many feathers as practical (at least 374 feathers, excluding down). When these were assembled, we determined that they represented a single bird (i.e., we found 14 primaries, 9 secondaries, and 5 rectrices, none of which were duplicates). On the cover of this issue, an array of these are displayed around an illustration of this young falcon, as it would have appeared in life.

Many feathers of this bird show pale brown (buff) spots where such would occur on a normal juvenile. Other feathers have reddish tips just as for normal peregrine juveniles. These same feathers, however, are pale bluish gray, where in a normal juvenile, they would be deep chocolate brown. The extensively gray contours result in this bird more closely resembling an adult than a juvenile. Because of this resemblance we suspect that this nestling, when it reached this level of development, displayed enough of the adult sign stimuli that one of its parents responded as if it was an intruding adult and killed it. We cannot be certain that the pale bird was killed by its own family, but the presence of feathers and

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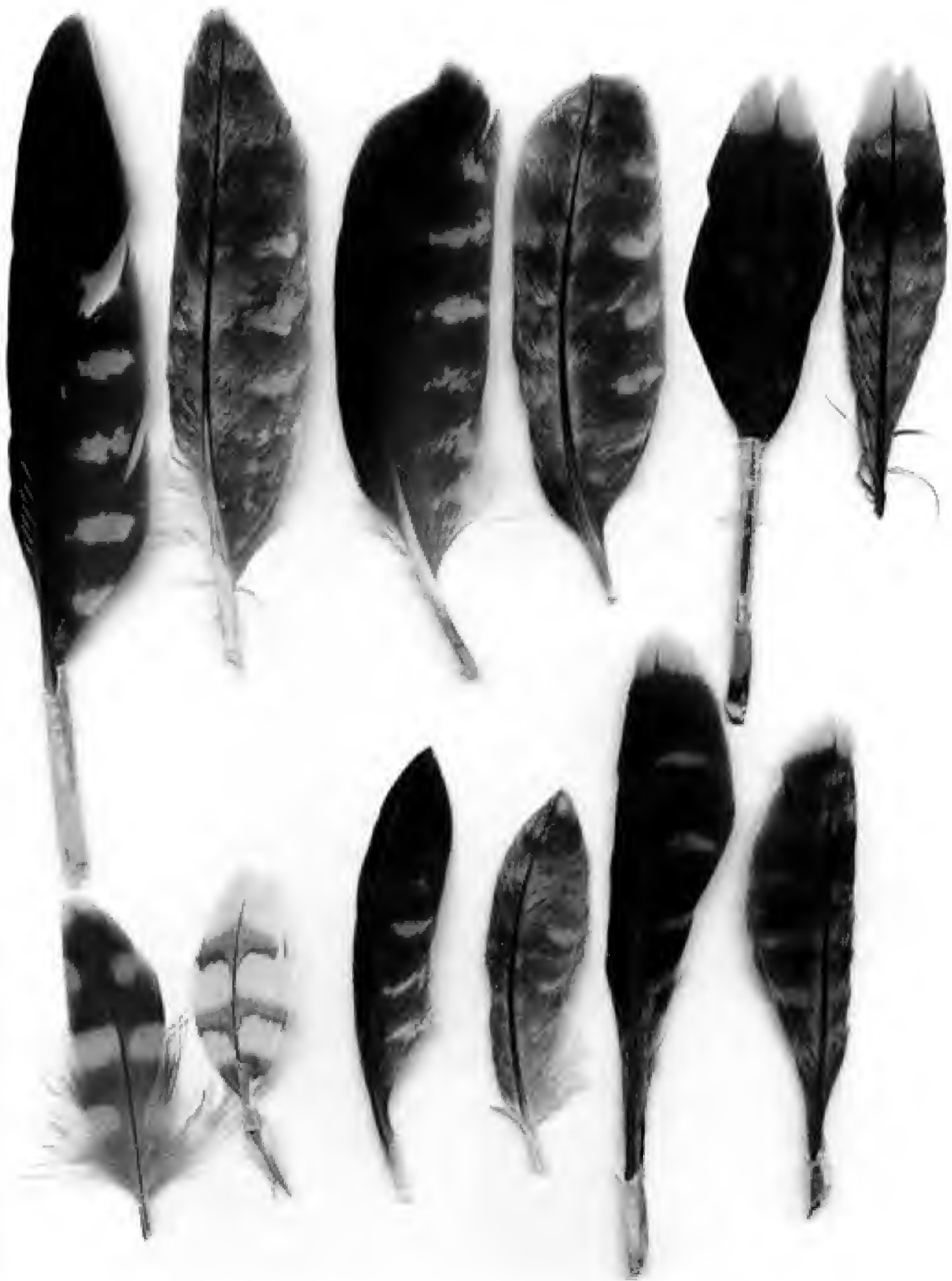


Figure 1. A comparison of feathers representing various body areas for the nestling displaying schizochromism (right feather in each pair) and two normally-pigmented nestlings (left feathers) from Arizona that died in approximately the same stage of development. Illustrated topographic regions include (left to right, top to bottom): primary flight feather, secondary flight feather, central rectrix, flank, alula, lateral rectrix. Feathers for the pale bird were normal in shape, so the shape differences between the pale and normal feathers in each pair are due to peculiarities associated with exact locus. For example, the pale secondary has a more symmetrical vane tip showing that it is from a more distal follicle than the normally-pigmented secondary. Also, the lateral rectrices are from opposite sides of the tail.

a talon in castings on the eyrie ledge demonstrate that it was eaten by the family. Also, the excellent condition of its feathers (i.e., few fault bars and none of great extent) suggest that it was healthy until the time of death.

While this individual can be called partially albino (i.e., some dark pigment is lacking) or leucistic (i.e., showing color dilution), a better term for its condition is schizochromism (literally: split coloration). Traditionally, this term has been spelled schizochroism (see Van Tyne and Berger 1976:160, Hailman and Emlen 1985), but a second "m" is obviously needed (chroma: Greek, color; chromatic: English, relating to color). Such birds have some pigments, but not others. Our specimen was normal for buff and reddish brown (Fig. 1) but lacked deep chocolate brown, the most expansive color in the plumage of normal juveniles.

Four clues lead us to conclude that the bird was eaten by falcons on the ledge. First, and most convincing, a talon and many small feathers were found in castings. Second, most of the remains were found on the eyrie ledge (a mammalian predator would likely have removed the carcass to consume it elsewhere). Third, the feathers were plucked and scattered (as is characteristic of raptorial bird kill sites), rather than chewed off and matted with blood and saliva (as is typical of mammalian kills). Finally, at least three young fledged from the eyrie: it is unlikely that the pale bird was taken and eaten by a predator of another species on the eyrie shelf without the predator killing additional nestlings. All of these facts provide evidence that the pale nestling was consumed by its own family. Cannibalism has previously been documented for five species of falcons including the peregrine (Ellis et al. 1999).

Feathers from this specimen, representing a wide range of topographic regions, were deposited in the University of Arizona ornithological collection (UA no. 17828).

RESUMEN.—Un pichón muy pálido de Halcón peregrino (*Falco peregrinus*) fue recuperado muerto en un nido al sur de Arizona. De las plumas y una garra encontrados en una egagrópila ubicada al borde de un nido al sur de Arizona, al borde del nido, concluimos que hubo cani-

balismo por parte de su propia familia. El plumaje de esta ave mostraba una ausencia del marrón oscuro, presentando un manto de color gris pálido sin diferencia al del adulto. Este es un ejemplo de esquizocromismo. Todo parece indicar que este fenómeno no había sido reportado en la bibliografía de la especie.

[Traducción de César Márquez]

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NATAL DISPERSAL OF THE CRESTED CARACARA (*CARACARA CHERIWAY*) IN FLORIDA

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KEY WORDS: *Crested Caracara*; *Caracara cheriway*; *natal dispersal*; *Florida*.

The process of dispersal has important implications for the distribution, regulation, and genetic structure of avian populations (Greenwood 1980, Greenwood and Harvey 1982, Paradis et al. 1998). Natal dispersal, movement of an individual from its site of birth to the site of first reproduction or potential reproduction, and breeding dispersal, movement of adult individuals between breeding sites (Howard 1960, Greenwood 1980), are major agents of gene flow that affect overall population relatedness and distribution. Understanding how dispersal influences population dynamics is necessary when assessing population responses to landscape change and when developing conservation plans for populations or species.

Because of the difficulty in obtaining dispersal data for large, wide-ranging species, information about dispersal in many birds is limited. This is generally true for raptors, which typically have large geographic distributions and range across wide areas. In addition, many raptors do not breed for several years post-fledging; hence, keeping track of individuals through this time period until they begin breeding is difficult for most populations.

Here, we report the first information on dispersal for the Crested Caracara (*Caracara cheriway*), a medium-sized raptor that inhabits open grasslands and pastures (Morrison 1996). In North America, extant populations of the Crested Caracara occur only in Florida, Texas, and Arizona. Despite this species' wide geographic range (Morrison 1996), many aspects of its biology remain poorly understood, perhaps because of its reputation as a pest throughout much of its Central and South American range, where it is more abundant.

Recent widespread loss of grassland and pasture habitats in Florida due to agricultural and urban expansion is perceived as a major threat to the persistence of this population of caracaras. These habitat changes and concomitant population decline (Millsap 1989, Layne 1996) led to listing of this population as federally threatened (U.S. Fish and Wildlife Service 1987). To better understand the caracara's biology and population responses to landscape change, a study of this non-migratory and iso-

lated population was initiated in 1994 (Morrison 1998, 1999, Morrison and Humphrey 2001). Study objectives included obtaining information about demographic parameters and habitat use. This paper presents information collected on dispersal of known individuals from their natal site to a breeding site within the study population.

METHODS

The study area incorporated all or parts of eight counties in the south-central peninsula and represents ca. 80% of the caracara's current breeding range in Florida (Morrison 1999). Our sample included breeding areas that were located throughout this range, but because most of Florida's caracaras currently live on privately-owned lands (Morrison and Humphrey 2001), efforts to obtain a systematic, random sample of breeding areas or to survey the entire study area were constrained by our ability to secure access from landowners.

Crested Caracaras nest primarily from December–April in Florida (Morrison 1999). We began marking nestling caracaras during the 1993–94 breeding season and continued through the subsequent three breeding seasons, 1994–95, 1995–96, and 1996–97. Nestlings were marked at 6–8 wk of age while still in the nest, or just after fledging, when they could be caught easily on the ground. Each nestling was marked with a standard numbered U S Fish and Wildlife Service aluminum band and an aluminum color band with a unique alpha-numeric code (ACRAFT, Inc., Edmonton, Alberta, Canada). The gender of marked individuals was determined by DNA analysis of blood samples taken at the time of banding (Morrison and Maltbie 1998).

To obtain demographic information for this population, we monitored nesting activity annually. We monitored 48 breeding areas during the 1994–95 breeding season and 55 breeding areas during each subsequent year through the 1999–2000 breeding season. During the 2000–01 and 2001–02 breeding seasons, we monitored activity in 15 breeding areas, a subset of the original sample; these breeding areas were located along the Kissimmee River in the core of the caracara's current range. Each year, along with collecting information on nesting success and productivity, we identified adults nesting in each breeding area by reading their leg bands with binoculars or a spotting scope. Thus, data on dispersal and recruitment were collected opportunistically, incidental to routine annual nest monitoring efforts.

We calculated dispersal distance as the straight-line distance between the nest where the individual hatched and

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Table 1. Summary of marked juvenile Crested Caracaras, observed later as breeding adults, and their dispersal distances.

YEAR	NO. NESTLINGS MARKED		KNOWN ADULTS FOUND AS BREEDERS SEX	AGE WHEN FIRST FOUND BREEDING	DISTANCE FROM NATAL AREA TO BREEDING AREA (km)
	M	F			
1994	14	16	—		
1995	23	28	F	3	19.24
			F ^a	4	19.20
1996	28	36	M ^a	3	5.84
			M	3	6.81
			M	3	3.86
			M	6	9.08
			F	3	16.97
			F	3	20.70
1997	9	6	F	4	40.72
Total	74	86	9		

^a Breeding pair.

the location where it was found as a breeding adult. We define natal dispersal as movement between birthplace and first breeding site (Greenwood 1980, Greenwood and Harvey 1982).

RESULTS AND DISCUSSION

During 1994–97, we banded 160 nestling caracaras: 74 males and 86 females in 55 different breeding areas. Since the 1998–99 breeding season, while conducting regular monitoring of nesting activity in our sample of breeding areas, we have encountered four males and five females banded as nestlings that were occupying nest sites as breeders (Table 1).

Distances between natal areas and breeding sites differed significantly ($t = -3.76$, $P = 0.02$) between male ($\bar{x} = 6.40$ km \pm 1.08 SE, range = 3.86–9.08 km, $N = 4$) and female caracaras ($\bar{x} = 23.37$ km \pm 4.38 SE, range = 16.97–40.72 km, $N = 5$). The breeding areas in which we found known males were located only two to three breeding areas away from their respective natal areas, based on a mean home-range diameter in Florida of ca. 5 km (J. Morrison unpubl. data). Despite the small sample size, these results corroborate the general pattern of female biased dispersal observed in birds (Greenwood 1980, Johnson and Gaines 1990, Paradis et al. 1998) including many raptors (Newton and Marquiss 1983, Mearns and Newton 1984, Rosenfield and Bielefeldt 1992, Millsap and Bear 1997, Wiklund 1996, Ellsworth and Belthoff 1997, Lehman et al. 2000).

Our results also supported that the age at first breeding for the Crested Caracaras was 3 yr of age, as was previously suspected (Voous 1955, Layne 1996). Caracaras attain full adult plumage during their fourth year (Layne 1996, J. Morrison pers. observ.), and throughout our study, we rarely observed a breeding caracara in Basic 1 plumage (age 2–3 yr) and never in the juvenile plumage

(age 1 yr; Wheeler and Clark 1996). While we cannot entirely rule out caracaras breeding at age 1 or 2, the rarity of breeders in Basic 1 plumage (4.7%, $N = 108$) observed in our sample of breeding areas combined with our observations of known 3-yr-old caracaras breeding support age at first breeding as 3 yr for Florida's caracaras.

Distances we report here likely constitute natal dispersal for the 3-yr-old individuals. Because we do not know age at first breeding for the two 4-yr-old females and the 6-yr-old male, we are unable to assess whether distances reported for these individuals constitute natal or breeding dispersal. In any case, the new information reported here reveals distances that female and male caracaras in Florida travel between their natal site and a breeding site.

Explanations proposed for gender-biased, natal dispersal in birds include reproductive enhancement through mate and/or resource access and inbreeding avoidance (Greenwood 1980, Newton and Marquiss 1983, Pusey 1987, Korpimäki 1988, Johnson and Gaines 1990, Daniels and Walters 2000). Inbreeding avoidance could be an important selective factor for gender-biased dispersal in this non-migratory, isolated population of caracaras. However, interpretation of the observed patterns of dispersal distances reported here is complicated because our annual monitoring efforts were focused on sites of known previous nesting, and we did not search for recruits throughout the study area, so we could have missed other pairs that may have included individuals banded as nestlings.

The low encounter rate of banded individuals during annual monitoring efforts could be due to several factors. Mortality rates could be high during the period before first breeding, but low encounter rates could also reflect the high survival and site fidelity of adults in this population (J. Morrison unpubl. data), which, combined with limited suitable nesting habitat, may restrict opportunities for recruitment. Additionally, although unlikely, there is a possibility that some juveniles dispersed outside the study area to breed in other areas within Florida or even in other parts of North or Central America. However, no reports of banded caracaras, either recoveries or resightings, from distant populations have been received thus far. Furthermore, data obtained during 1995–2000 using telemetry for 131 juvenile caracaras radio-marked as nestlings within this population suggest that juveniles remain within the species' current Florida range, at least during the first 3 yr post-fledging (J. Morrison unpubl. data). Unfortunately, no transmitters lasted long enough to provide information on recruitment of radio-marked individuals.

Conclusions about dispersal behavior derived from data collected over a finite study area can be biased due to a non-uniform probability of resighting as a function of the distance dispersed (Moore and Dolbeer 1989, Koenig et al. 1996). Because we did not search for banded individuals outside the study area, we could have missed finding other recruits. We believe, however, that our sample of breeding areas scattered throughout the study area is representative of the population and, therefore, that the observed difference in mean dispersal distances between males and females, even given the small sample

size, provides evidence for female-biased dispersal in this population of Crested Caracaras.

Continued data collection on dispersal is needed for Crested Caracaras in Florida and throughout their geographic range to understand population structure and how these raptors use habitats and respond to landscape change. Our limited observations of dispersal movements in the Florida population may not be representative of the species overall due to broad differences in land use and habitat characteristics in other parts of the caracara's geographic range. We suggest, however, that the finding of female-biased dispersal in the species probably applies throughout the range, although distances involved probably vary among populations. Although the Crested Caracara has not received much conservation attention elsewhere, recent concern for the species' status has developed in Mexico (Rivera-Rodríguez and Rodríguez-Estrella 1998) and in parts of Argentina (Goldstein 2000) because of the loss of suitable nesting habitat in those regions as a result of rapid urban growth. To better understand this species' ecology and response to landscape change throughout its geographic range, additional studies of population dynamics and movement patterns for a number of populations are essential.

RESUMEN.—Como parte de un estudio poblacional a largo plazo del Caracara de Florida (*Caracara cheriway*), durante 1998–2002, encontramos cuatro machos en reproducción y cinco hembras en el mismo estado, los cuales habían sido anillados en el nido. Nuestras observaciones sostienen que la edad de la primera reproducción en esta población ocurre a los tres años. Las distancias de dispersión de la natalidad fueron considerablemente más grandes para las hembras que para los machos; una hembra fue encontrada reproduciéndose en un lugar a 40 Km del área de nacimiento. Estas observaciones sugieren que la dispersión de las hembras en los caracaras de Florida es similar a la de otras rapaces. Los estudios de movimientos individuales y de la dinámica de poblaciones, son esenciales para entender la respuesta de los caracaras a la amplia pérdida de hábitat y a los cambios de uso de la tierra en Florida y para el desarrollo de planes de conservación eficientes para esta población aislada y residente.

[Traducción de César Márquez]

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RECENT RECORDS OF CROWNED EAGLES (*HARPYHALIAETUS CORONATUS*) FROM ARGENTINA, 1981–2000

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KEY WORDS: *Crowned Eagle*, *Harpyhaliaetus coronatus*; *new records*; *conservation*; *status*; *Argentina*.

The Crowned Eagle (*Harpyhaliaetus coronatus*) is a vulnerable species whose distribution is limited to south-central South America (Collar et al. 1992, García-Fernández et al. 1997). The species has been protected in Argentina since 1954 and is listed as a threatened species in Brazil and Paraguay. The ecology of this large eagle is poorly known. It feeds on a variety of vertebrates including snakes (e.g., *Waglerophis merremi*), birds, skunks (*Conepatus* spp.), armadillos (Dasypodidae), and weasels (Collar et al. 1992). The nest of the Crowned Eagle consists of a large platform placed in trees where one egg is laid (De la Peña 1992, Bellocq et al. 1998). Naturally low popu-

lation numbers and habitat fragmentation have been recognized as primary contributors to the eagle's current status (Collar et al. 1992). Previous studies on habitat use by this eagle identified the potential negative effects of continuing afforestation (Bellocq et al. 1998). Over 60% of the Crowned Eagle records are from Argentina, where it occurs primarily in shrublands, savannas, and semi-open woodlands (Collar et al. 1992, Bellocq et al. 1998, Gonnet and Blendinger 1998). Crowned Eagles were also reported recently in subtropical rainforests (Chébez et al. 1998, Gonnet and Blendinger 1998). Here, we report new records of Crowned Eagles and integrate them with the previous information on this species from Argentina for 1981–2000.

METHODS

Road surveys for raptors were conducted in the northwest portion of Santa Fe province, north-central Argen-

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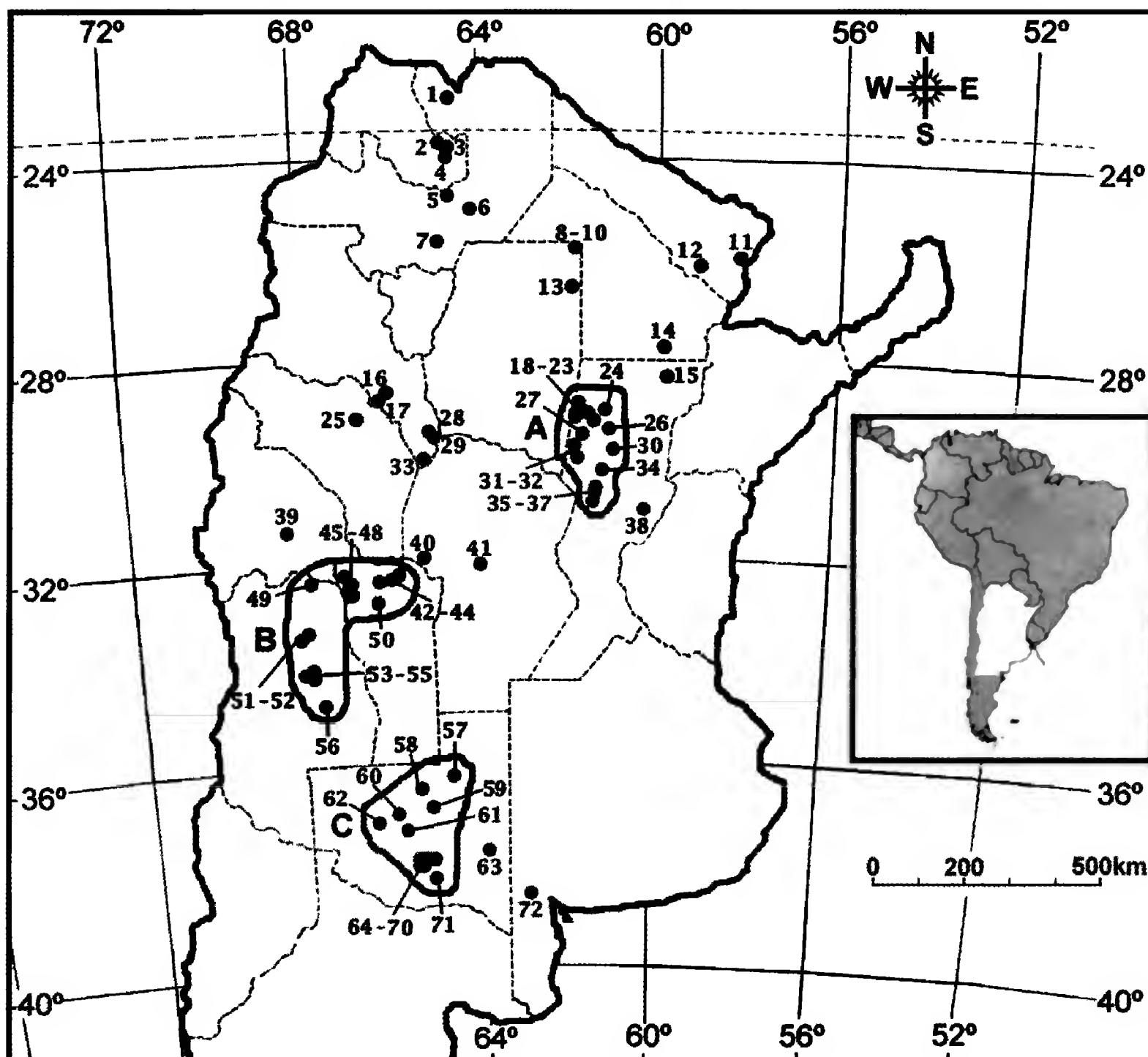


Figure 1. Observation records of Crowned Eagles (*Harpyhaliaetus coronatus*) from Argentina during 1981–2000 (see Appendix for details on records). Zones A (northwestern Santa Fe), B (northeastern Mendoza and northern San Luis), and C (central La Pampa) are areas with a high number of records.

tina (west of Zone A, Fig. 1). The survey was conducted along 210 km of road, driving north at a speed of 60–80 km/hr during 21 (from Nueva Italia to Tostado) and 22 November 2000 (north of Tostado). Raptor counts were made by one observer (who was not driving) while in transit, and occasional stops were made for identification and counting of individuals in groups.

We compiled information and identified locations for records of Crowned Eagles from Argentina for the period 1981–2000. Gonnet and Blendinger (1998) compiled most records from 1987–97. For each record, we obtained the following information (when available): location, geographic coordinates, province, date, number of individuals, and source. When not provided in the source, we obtained geographic coordinates at the Instituto Geográfico Militar (IGM) or contacted observers to

obtain complete information. Reference numbers were basically assigned for records from north to south.

RESULTS AND DISCUSSION

A total of 343 individual raptors was recorded during the road survey in 2000; these included 161 Swainson's Hawks (*Buteo swainsoni*), 114 Crested Caracaras (*Polyborus plancus*), 56 Black Vultures (*Coragyps atratus*), four Crowned Eagles (*Harpyhaliaetus coronatus*), four Black-chested Buzzard-Eagles (*Geranoaetus melanoleucus*), two Chimango Caracaras (*Milvago chimango*), one Roadside Hawk (*Buteo magnirostris*), and one Snail Kite (*Rostrhamus sociabilis*).

We observed four Crowned Eagles during the raptor

survey and one additional individual while traveling around the area. We were able to identify distinguishing plumage features of each juvenile indicating that we observed different individuals. All individuals were seen between 29–30°S and 61–62°W, where the land is used primarily for ranching.

On 20 November 2000, an adult and a juvenile eagle were observed at Hwy 2, 20.5 km north of Nueva Italia (29°47'S, 61°32'W; record number 32 in Fig. 1). The adult was perched on an electric pole and the juvenile was on the ground holding a snake (*Philodryas patagoniensis*) in its talons. The surrounding habitat had two vegetation strata, trees and grasses, where *Geoffroea decorticans* (height about 6 m) dominated the canopy covering 40–60% of the ground surface.

On 20 November 2000, a juvenile was seen perched on a power pole at Hwy 2, 37 km south of Tostado (29°26'S, 61°43'W; record number 27 in Fig. 1). We were able to approach the base of the pole without flushing this eagle. The habitat was a *Prosopis* savanna with sparse *G. decorticans*.

On 21 November 2000, we observed a juvenile perched on a *Prosopis* tree (height about 6 m), on Hwy 95, 21 km north of the intersection with Hwy 2 (29°07'S, 61°43'W; record number 21 in Fig. 1). The eagle was perched in a pasture with isolated *Prosopis* where cattle grazed. Surrounding fields were cultivated with sunflower.

On 21 November 2000, we recorded a juvenile eagle while traveling around the area. The eagle was perched on a fence pole on Hwy 13, 18.5 km south of the intersection between Hwys 13 and 98 (29°26'S, 61°03'W; record number 26 in Fig. 1). The habitat was a pasture where *Spartina* spp. was the most common grass (60–80% cover).

We compiled 72 records from Argentina for 1981–2000 (record numbers in Appendix correspond to location numbers in Fig. 1). The three zones with a high frequency of records (Fig. 1) have a semiarid climate, and each include a portion of two phytogeographic regions (following Cabrera 1971), and the ecotone between them. Zone A is located in the northwestern portion of Santa Fe province. In the northern part of this zone (Chaqueña region, de las Sabanas District), vegetation types include savannas of *Elionurus muticus* in the upper areas and *Spartina argentinensis* in the lower areas. In the southern part of zone A (El Espinal region, del Algarrobo District), close to open woodlands, *Prosopis* spp. and *G. decorticans* dominate the landscape. The area is affected by afforestation and the land is primarily used for ranching. *G. decorticans* forests occur in areas disturbed by ranching. Zone B comprises northeastern Mendoza and northern San Luis provinces. In northern San Luis (Chaqueña region, Serrano District), the original woodlands of *Schinopsis* spp. were replaced by shrublands with isolated woodlots of *Prosopis* spp. In eastern Mendoza (Monte region), the primary vegetation type is a shrubland dominated by *Larrea* spp. Other shrub species include *Monttea*

aphylla, *Bougainvillea spinosa*, *Prosopis alpataco*, and *Chiquiraga erinacea*. Zone C is located in central La Pampa province. In the eastern part of this zone (El Espinal region, del Caldén District), the natural xerophic forest dominated by *Prosopis caldenia* has been modified as a result of afforestation followed by ranching. Currently, this zone consists of shrublands (mostly *Larrea* spp.) with isolated *P. caldenia* or small woodlots. Some common accompanying shrubs are *Lycium chilense*, *Prosopis flexuosa*, *Condalia microphylla*, *C. erinacea*, and *G. decorticans*. The western part of zone C is within the Monte phytogeographic region described above.

Of the 72 records found from Argentina for 1981–2000, approximately 23%, 21%, and 19% are from zones A, B, and C, respectively (Fig. 1). Gonnet and Blendinger (1998) suggested zones A and B as potential areas to intensify research and conservation efforts. Zone A, however, has some advantages over zones B and C. First, zone A is located near the central portion of the species' range compared to other zones. Second, many of the records from zones B (40%) and C (57%) are from or around natural reserves, whereas no record from zone A is from natural reserves. Eagles are more likely to be seen in natural reserves due to increased sampling efforts and reduced habitat alteration and hunting pressure (Gonnet and Blendinger 1998). The conservation of large-bodied, low-density, upper-trophic-level species often requires suitable habitat beyond the size of existing natural reserves (Meffe and Carroll 1997). Third, our results might indicate a relatively high density of eagles in zone A, considering both the naturally low population numbers and the results of previous surveys. The available information from previous surveys (Travaini et al. 1995, Bellocq et al. 1998, Contreras and Justo 1998, Gonnet and Blendinger 1998) suggests that sighting five Crowned Eagles in 2 days of observations is unusually high.

In summary, we reported four new records of Crowned Eagles (including five individuals), compiled and provided details of records in Argentina for 1981–2000, and identified three zones with high frequency of recent records. Based on this study, we suggest the following priorities for research and conservation of the Crowned Eagle: (1) complete surveys along the Espinal phytogeographic region, (2) identify variables associated with high frequencies of eagle sightings at larger geographical scales (e.g., landscape), and (3) conduct studies on reproductive success and mortality to identify viable populations.

RESUMEN.—El águila coronada (*Harpyhaliaetus coronatus*) es una especie vulnerable y rara de ver. Se observaron cuatro ejemplares durante un relevamiento de rapaces por carretera a lo largo de 210 km, y un individuo adicional cuando se recorría el área en el centro-norte de Argentina (Santa Fe). Se compilieron los registros de Argentina para el período 1981–2000 y se identificaron tres zonas de alta frecuencia de registros recientes. La zona ubicada en el noroeste de la provincia de Santa Fe parece

ser la más apropiada para intensificar investigación y acciones de conservación, debido a su posición relativamente central dentro de los límites de distribución de la especie y su alta frecuencia de registros recientes afuera de reservas naturales.

[Traducción de los autores]

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Appendix The record number, number of sighted individuals, locality, geographic coordinates, and source of observation records of Crowned Eagles from Argentina for 1981–2000.

REC- ORD NO.	EA- GLES NO.	DATE	LOCALITY, PROVINCE ¹	GEOGRAPHIC COORDINATES	SOURCE
1	2	16–27 Jul 1989	P.N. Baritú, Salta	22°40'S, 64°37'W	Gonnet and Blendinger (1998)
2	2	30 Aug 1988	San Francisco, Jujuy	23°35'S, 64°54'W	Gonnet and Blendinger (1998)
3	1	17/18 Feb 1987	P.N. Calilehua, Jujuy	23°35'S, 64°50'W	Gonnet and Blendinger (1998)
4	1	Undated	P.N. Calilehua, Jujuy	23°40'S, 64°53'W ²	Chébez et al. (1998)
5	1	Undated	P.N. El Rey, Salta	24°42'S, 64°38'W ²	Chébez et al. (1998)
6	1	17 Nov 1989	Joaquín V. González, Salta	25°04'S, 64°38'W ²	De Lucca (1993)
7	1	June 1985	Yatasto, Salta	25°35'S, 64°56'W ²	Collar et al. (1992)
8	1	Feb 1989	R.P. Copo, Santiago del Estero	26°05'S, 62°00'W	Collar et al. (1992)
9	1	After Feb 1989	R.P. Copo, Santiago del Estero	26°05'S, 62°00'W	Collar et al. (1992)
10	1	Undated	R.P. Copo, Santiago del Estero	26°05'S, 62°00'W	Collar et al. (1992)
11	1	Dec 1987	North of Formosa City, Formosa	ca. 26°11'S, 58°11'W ²	Collar et al. (1992)
12	1	ca. 1995	Reserva Ecológica El Bagual, Formosa	ca. 26°13'S, 59°50'W ²	Di Giacomo (1996)
13	1	28 Feb 1992	Sachayoj, Santiago del Estero	26°41'S, 61°50'W ²	Salvador and Eroles (1994)
14	1	Jul 1986	Charadai, Chaco	27°38'S, 59°53'W ²	Collar et al. (1992)
15	1	Nov 1989	West of Los Amores, Santa Fe	28°03'S, 59°57'W ²	Collar et al. (1992)
16	1	Feb 1984	Nat. Hwy 38, near Catamarca City, Catamarca	28°28'S, 65°47'W ³	De Lucca (1993)
17	1	1988	Nat. Hwy 38, 30 km southwest of Catamarca City, Catamarca	28°28'S, 65°47'W	Gonnet and Blendinger (1998)
18	1	Jul 1987	Pozo Borrado, Santa Fe	ca. 28°52'S, 61°37'W	Collar et al. (1992)
19	1	Jul 1987	Antonio Pini, Santa Fe	ca. 29°07'S, 61°37'W	Collar et al. (1992)
20	1	Oct 1989	National Hwy 95, Santa Fe	ca. 29°07'S, 61°22'W	Collar et al. (1992)
21	1	21 Nov 2000	Hwy 95, Santa Fe	29°07'S, 61°43'W	This study
22	1	Sep 1988	ca. 10 km south of Antonio Pini, Santa Fe	ca. 29°07'S, 61°37'W	Collar et al. (1992)
23	1	Nov 1989	10 km east of Tostado, Santa Fe	ca. 29°14'S, 61°47'W ²	Collar et al. (1992)
24	2	11 Feb 1988	Fortin Los Pozos, Santa Fe	ca. 29°07'S, 61°07'W	Collar et al. (1992)
25	1	Jul 1986	El Cantadero, La Rioja	29°11'S, 66°44'W	Collar et al. (1992)
26	1	21 Nov 2000	Hwy 13, Santa Fe	29°26'S, 61°06'W	This study
27	1	20 Nov 2000	Hwy 2, 37 m south of Tostado, Santa Fe	29°26'S, 61°43'W	This study
28	1	Aug 1982	Recreo, Catamarca	29°16'S, 65°04'W ²	Collar et al. (1992)
29	1	Aug 1982	30 km southeast of Recreo, Catamarca	ca. 29°16'S, 65°04'W ²	Collar et al. (1992)
30	1	Oct 1989	Vera Department, Santa Fe	ca. 29°28'S, 60°13'W ²	Collar et al. (1992)
31	1	Nov 1989	NE of Montefiore, Nat. Hwy 95, Santa Fe	ca. 29°37'S, 61°52'W	Collar et al. (1992)
32	2	20 Nov 2000	Hwy 2, 20.5 km north of Tostado, Santa Fe	29°47'S, 61°32'W	This study
33	1	8 Mar 1981	Salinas Grandes, Catamarca	30°08'S, 65°25'W ²	De la Peña (1999)

Appendix. Continued.

REC- ORD NO.	EA- GLES NO.	DATE	LOCALITY, PROVINCE ¹	GEOGRAPHIC COORDINATES	SOURCE
34	1	Jan 1988	Santurce, Santa Fe	ca. 30°07'S, 61°07'W	Collar et al. (1992)
35	1	Dec 1982	Constanza, Santa Fe	ca. 30°52'S, 61°22'W	Collar et al. (1992)
36	1	Aug 1989	Virginia, Santa Fe	ca. 30°52'S, 61°22'W	Collar et al. (1992)
37	1	Jul 1988	Ataliva, Santa Fe	30°59'S, 61°27'W	Collar et al. (1992)
38	1	May 1989	25 km west of Cayastá, Santa Fe	31°12'S, 60°10'W	Collar et al. (1992)
39	1	1984	Sierra de Villicum, Albarcón, San Juan	ca. 31°14'S, 68°31'W ²	De Lucca (1992)
40	1	Jan 1982	Villa Dolores, Córdoba	31°57'S, 65°12'W ²	De Lucca (1993)
41	1	Oct 1986	Río Tercero, Córdoba	32°11'S, 65°48'W ²	De Lucca (1993)
42	1	16 Jun 1997	Nat. Hwy 20, La Unión, San Luis	32°13'S, 65°48'W	Gonnet and Blendinger (1998)
43	2	22 Feb 1991	La Higuera, San Luis	ca. 32°25'S, 65°55'W ²	De Lucca (1993)
44	1	14 Dec 1994	Nat. Hwy 20, 16 km east of Luján, San Luis	32°22'S, 65°57'W	Gonnet and Blendinger (1998)
45	2	3 Sep 1997	Nat. Hwy 20, km 430, La Tranca, San Luis	32°21'S, 67°17'W	Gonnet and Blendinger (1998)
46	1	27 Aug 1986	P.N. Sierra de las Quijadas, San Luis	ca. 32°25'S, 67°05'W ²	Nellar Romanella (1993)
47	1	1-6 Sep 1990	P.N. Sierra de las Quijadas, San Luis	32°34'S, 67°10'W ²	Gil et al. (1995)
48	1	1990	Nat. Hwy 147, 40 km northwest of San Luis city, San Luis	ca. 33°19'S, 66°21'W ²	Gil et al. (1995)
49	1	3 Apr 1997	Reserva Florística y Faunística Telteca, Mendoza	32°21'S, 68°03'W	Gonnet and Blendinger (1998)
50	2	3 Aug 1996	Nat. Hwy 7, km 798, San Luis	33°16'S, 66°21'W	Gonnet and Blendinger (1998)
51	2	6 Dec 1995	Prov. Hwy 153, km 60 and 62, Mendoza	33°51'S, 68°00'W	Gonnet and Blendinger (1998)
52	2	14 Aug 1997	Provincial Hwy 153, km 67, Mendoza	33°51'S, 68°00'W	Gonnet and Blendinger (1998)
53	3	21 May 1996	Prov. Hwy 153, km 70, Ñacuán, Mendoza	34°03'S, 67°58'W	Gonnet and Blendinger (1998)
54	2	1-8 Nov 1992	Reserva del Hombre y la Biosfera Ñacuán, Mendoza	34°03'S, 67°58'W	Gonnet and Blendinger (1998)
55	1	28 Oct 1995	Reserva del Hombre y la Biosfera Ñacuán, Mendoza	34°03'S, 67°58'W	Gonnet and Blendinger (1998)
56	1	20 Dec 1993	Nat. Hwy 143, km 170, Mendoza	35°43'S, 68°40'W	Gonnet and Blendinger (1998)
57-61		Undated	Five sites between Conhelo and El Oudre, La Pampa; 1 eagle observed at each site	36°00'S, 64°35'W to 36°58'S, 66°42'W ²	Collar et al. (1992)
62	2	2 Jan 1997	Nat. Hwy 143, surroundings of Limay Mahuida, La Pampa	37°12'S, 66°42'W ²	Kaspar et al. (1999)
63	1	10 Jun 1988	Nat. Hwy 35, km 211, La Pampa	37°38'S, 64°09'W	Delhey (1992)
64	1	18 Nov 1996	20 km east of P.N. Lihué Calel, La Pampa	ca. 37°55'S, 65°32'W ²	Belloccq et al. (1998)
65 ⁴	3	Feb 1997	12 km east of P.N. Lihué Calel, La Pampa	ca. 37°55'S, 65°32'W ²	Belloccq et al. (1998)
66	1	Oct 1988	P.N. Lihué Calel, La Pampa	37°55'S, 65°32'W ²	Collar et al. (1992)
67	1	Jun 1986	P.N. Lihué Calel, La Pampa	37°55'S, 65°32'W ²	De Lucca (1993)

Appendix. Continued.

REC- ORD No.	EA- GLES No.	DATE	LOCALITY, PROVINCE ¹	GEOGRAPHIC COORDINATES	SOURCE
68	1	1 May 1991	P.N. Lihué Calel, La Pampa	37°55'S, 65°32'W ²	De Lucca (1993)
69	1	Oct 1991	P.N. Lihué Calel, La Pampa	37°55'S, 65°32'W ²	De Lucca (1993)
70	1	21 Sep 1997	P.N. Lihué Calel, La Pampa	37°55'S, 65°32'W ²	Belloccq et al. (1998)
71	1	22 Nov 1996	40 km southeast of P.N. Lihué Calel	37°55'S, 65°32'W ²	Belloccq et al. (1998)
72		19-26 Jul 1997	Chasicó stream mouth, Buenos Aires	ca. 38°38'S, 63°06'W ²	Kaspar et al. (1999)

¹ P.N.: National Park, R.P.: Provincial Natural Reserve.² Geographic coordinates obtained by the authors.³ Coordinates provided by Rodríguez Gofii (pers. comm.).⁴ Nesting site.

NEW OBSERVATIONS OF THE PEREGRINE FALCON (*FALCO PEREGRINUS*) IN PERU

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KEY WORDS: *Peregrine Falcon*; *Falco peregrinus*; *Peru*; *South America*.

The Peregrine Falcon (*Falco peregrinus*) is a widespread terrestrial species and nests on every major landmass except Antarctica, New Zealand, and Iceland (Cade 1982). Large numbers of the subspecies *tundrius* and *anatum* of the North American arctic winter in South America (Ambrose and Riddle 1988, Cade et al. 1988, Henny et al. 1996, Fuller et al. 1998), yet most of this continent has long seemed to be virtually devoid of any nesting peregrines. With the subspecies *cassini*, the peregrine is known to be widespread only in the southern tip of South America in Chilean and Argentine Patagonia (McNutt et al. 1988).

During the last few decades, nesting peregrines have been found or suspected further north. To date in Peru, there are six to seven suspected or confirmed breeding pairs at inland locations mentioned in the literature. A pair was observed on 10 September 1937 near Yauli (Department of Huancavelica) at an elevation of 3350 m (Morrison 1939) in the Andean interior. Gochfeld (1977) reported the observations from 1972 of an adult pair near San Ramon (Department of Junin) at an elevation of about 1800 m in August and September and of a single adult male near La Merced in the same department on 6 August. Both sites are at the eastern Andean slope toward the Amazon basin. Ellis and Glinski (1980) recorded an adult pair at a probable eyrie at an elevation of ca. 1850 m in the western Andean foothills near Tacna (Department of Tacna) on 25 March 1979. From 1980–82, Schoonmaker et al. (1985) confirmed three breeding pairs at elevations of 600–1200 m in the western Andean foothills near Olmos (Department of Lambayeque). In 1979, breeding was confirmed near Guayllabamba, Ecuador in an intermontane valley at an elevation of 2400 m (Jenny et al. 1981, Hilgert 1988). Nothing is known about the breeding status of peregrines in Colombia or Venezuela (McNutt et al. 1988).

During three trips to South America in February–May 1996 and 2001, and in January 2002, I saw peregrines at nine sites in central and southern Peru. Some of these presumably represented local breeding birds and thus are notable. In addition, I summarize some recent observations collected by other ornithologists in the same areas. One aim of the article is to reiterate the suggestion by McNutt et al. (1988) that peregrines may be much

more widespread in the central and northern Andes than hitherto recorded.

METHODS

Three peregrine subspecies can be observed in South America, the locally-nesting *cassini* and the northern migrants *tundrius* and *anatum*. In the field they cannot be distinguished for sure because character distributions overlap. Here, I describe ‘typical’ birds of each subspecies and note timing and behavioral criteria that may be used to distinguish northern migrants from resident breeders.

Adult *cassini* show a striking color dimorphism. In Patagonia, a rare and very pale morph occurs which was earlier thought to represent a different species, the Pallid Falcon (“*Falco kreyenborgi*,” Ellis and Peres G. 1983). The more common morph of *cassini* is usually described as a dark subspecies with a broad moustache, and a rusty wash and strong bars on the underside (Ellis 1985, White and Boyce 1988, Ratcliffe 1993). Even in the normal *cassini* morph, however, there is considerable morphological variation across the entire range. In central Peru, Gochfeld (1977) saw an adult bird with “whitish underparts and with only a trace of barring,” which—based on the August date—could only represent *cassini*. Ellis (1985), in Argentina, also saw pale normal *cassini* birds without any colorful wash on the breast or belly. White and Boyce (1988) note that birds from central and northern Chile are generally paler overall. Some falconer birds of Peruvian origin that were shown to me in Lima had a broad moustache and were only lightly barred on a buff-rusty underside. They reminded me of the *Falco p. pelegrinoides* I had seen in Israel.

Average adult *tundrius* peregrines from the tundra are overall very pale, with often a pure white underside and sparse and thin (and sometimes almost absent) markings, a narrow moustache, and a white spot above the beak (White 1968). Typical migrant *anatum* from the North American taiga, in contrast, have broader moustaches and more extensive markings on the underside, which may have a rusty wash (White 1968).

Departure dates of wintering peregrines in South America can be estimated from telemetry data on the mean duration of migration, and from the arrival times on the taiga and tundra breeding grounds as reported in various studies in Cade et al. (1988). Northbound migration averaged 42 d (Fuller et al. 1998), so mean departure dates may be estimated at late March–early April (R.W. Nelson pers. comm.). Based on the only published account of nesting pairs in northern Peru (Schoonmaker et al. 1985), three pairs started egg-laying from mid-March to the end of May. The peregrines at Guayllabamba, Ecuador started egg-laying anytime between June–December ($N = 5$ breeding seasons; Hilgert 1988). Thus,

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migrants and breeders in Peru may overlap widely. Any peregrine seen after late March–mid April is likely a resident, while birds before that date will be either a migrant or a resident.

Sometimes peregrine pairs away from any breeding site have been observed in South America and presumably were northern migrants (e.g., observations in Peru, O. Beingolea pers. comm.). However, observations of pairs are more likely to refer to resident breeders. Also, migrants would be less likely to hunt jointly in pairs and tend to be more silent compared to resident breeders. Migrants will be especially unlikely to give the “creaking call,” which is usually heard at nesting sites when birds of opposite sexes meet (Ratcliffe 1993).

My observations in 1996 were made with 10× binoculars and were incidental to other travels. Some of the observations in 2001 and 2002 resulted from efforts to relocate birds at sites discovered years earlier or to survey areas that had appeared to be promising (especially in the Valle Sagrado). They were made with 10× binoculars and a 20–45× spotting scope. The birds in Lima were observed with the unaided eye. The minimal ranges at which I saw almost all birds ranged between 20–1000 m and the light conditions were usually excellent. Sexing and aging, where possible, was based on 22 years’ experience with several thousand individual peregrines observed in 14 countries. For each observation, I give as much detail (regarding sex, age, and plumage) as was seen.

RESULTS

Twenty-eight peregrine falcons were observed at nine locations in central and southern Peru in February–May 1996 and 2001, and in January 2002. Five birds were seen in Lima, 12 at or near sea cliffs at three coastal sites, eight at cliffs at three inland sites, and three birds were observed at inland sites away from any cliffs. Four pairs were observed at or near suitable nesting cliffs, while one pair was seen away from cliffs. Locations where I saw birds, as well as observations from the literature or from other observers, that are suspected to represent breeding birds are shown in Figure 1.

(1) Lima (12°06’S, 77°00’W): Several observations in the central parts of the city of birds that could not be aged. On 15 March 2001, two peregrines perching on buildings and hunting in Miraflores/Lima, and one peregrine with prey at the Plaza de Armas in the city center. On 9 April 2001, one peregrine hunting from the tall building beside the Sheraton hotel in the city center. On 19 May 2001, one peregrine with a freshly-caught pigeon in the city center. In addition, on 22 May 2001, I came across a stuffed adult male in a shop window in the city center. The shop clerks claimed that “somebody from the street” had given them the bird. The stuffed bird seemed unlikely to be more than a few years old. It was hardly marked on the underside, so was probably not *anatum*, but had a warm buff tinge, and thus, probably was not *tundrius* either.

(2) Lagunillas Beach, Paracas National Park, Pisco (13°54’S, 76°18’W): 3–4 February 1996, one adult pair

watched for a total of 5 hr at a sea cliff between Playa de Lagunillas and Playa de Yumaque. These birds seemed to be paired and mostly stayed together and several times also hunted jointly. Although both birds were pale beneath, the female was strongly barred on the belly and thus unlikely a *tundrius* bird. On 24 March 2001, one adult peregrine was seen at the same cliff. Ventrally it was only lightly marked on a warm, buff background. The sea cliff was 50–100 m high and, in several places, had conspicuous whitewash typical of a nesting cliff and suggestive of prolonged occupancy. On 26 March 2001, I saw one adult peregrine perching at another sea cliff of similar size ca. 2 km west of Lagunillas Beach.

(3) Laguna Grande, Paracas National Park, Pisco (14°10’S, 76°16’W): 28 March 2001, one adult pair hunted jointly over the fishing village of Laguna Grande and also perched on a hilltop, less than 1 km from the closest sea cliffs, which are well over 100 m in height in places. Both birds had strong moustaches and their rusty-buff undersides were only lightly marked, which made them unlikely to be either *tundrius* or *anatum*. On 29 March 2001, one peregrine perched in a sea cliff 1 km northwest of the village, and one adult peregrine flew off from a rocky outcrop on the east side of Mt. Wilson (ca. 2 km south of the village).

(4) Lighthouse south of Chala (15°52’S, 74°11’W): On 7 February 1996, one adult female south of the lighthouse about 2 km south of the village of Chala. A fresh plucking and a few pellets were found on top of the cliff from where I flushed the bird. On 1 April 2001, one peregrine flew by and one adult peregrine perched and also called (the “wailing call” noted by Ratcliffe 1993) on a hilltop in the same area. On 2 April 2001, one adult peregrine perched, with a white background on the underside, but heavily barred and with a broad moustache, so probably not a *tundrius* migrant. The cliffs in this area are mostly <50 m high.

(5) Cañon de Colca (15°38’S, 71°48’W): Several observations at the large (>100 m) inland cliff crossed by a tunnel between the villages of Maca and Pinchollo at an elevation of 3250 m. On 12 February 1996, one adult peregrine flew in front of the cliff. It had a broad moustache and was dark on the upperside and rusty below. On 6–7 April 2001, an adult male peregrine was observed for several hours, perched and flying along the cliff. It had a very broad moustache, almost a dark facial mask, and buffish underparts. On both days, the “creaking call” (Ratcliffe 1993) was heard, indicating the presence of a second peregrine, presumably the female, which was not seen. The cliff had extensive whitewash in several places, suggesting prolonged occupancy. It overlooks the Colca river in a rich agricultural valley and thus appears a classic peregrine haunt.

(6) City of Arequipa (16°24’S, 71°33’W): 12 April 2001, one female seen stooping in the city center (elevation 2300 m).

(7) Machu Picchu (13°08’S, 72°34’W): 17 April 2001,

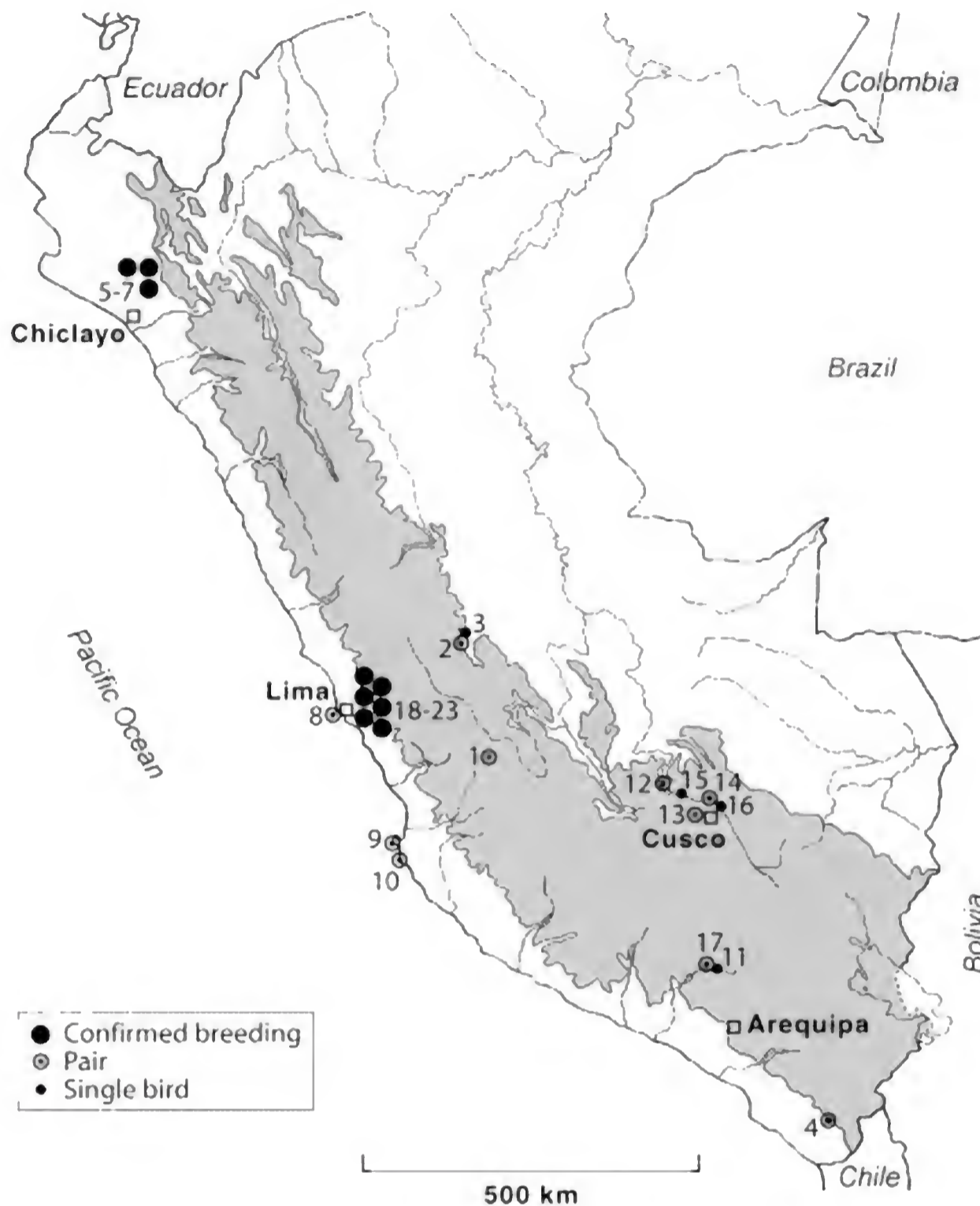


Figure 1. Distribution of the Peregrine Falcon (*Falco peregrinus*) in Peru. The locations of observations referring to known or suspected breeding pairs are shown. The dark shaded area marks Andean regions above 2000 m elevation. To its west is the coastal strip, and to its east, the Amazon forest.

Numbers mark the locations in order of their appearance in the text: 1 (Yauli; Morrison 1939), 2 (San Ramon; Gochfeld 1977), 3 (La Merced; Gochfeld 1977), 4 (Tacna; Ellis and Glinski 1980), 5-7 (Olmos; Schoonmaker et al. 1985), 8 (Lima), 9 (Lagunillas), 10 (Laguna Grande), 11 (Cañon de Colca), 12 (Machu Picchu), 13 (Cusco), 14 (Calca-Lamay), 15 (Chilca-Guayllabamba; D. Michelat, pers. comm.), 16 (Pisac; D. Michelat pers. comm.), 17 (Cabanaconde-Tapay; F. Schmitt, pers. comm.), 18-23 (Lima and surroundings; O. Beingolea, pers. comm.).

one adult pair with the tiercel attacking an intruding immature tiercel, at the east-facing cliff of Huayna Picchu, the little sugar-loaf mountain overlooking that world-famous ancient Inca city at an elevation of ca. 2400 m. The pair had broad moustaches and generally dark plumage (including strong markings below), but no pronounced buffness on the underside. They frequently gave the wailing and the creaking calls when together, and the tiercel the cacking call (Ratcliffe 1993) when chasing away the

intruder. The two adults were clearly paired, engaged in courtship flights and perched on the cliff, although not close together.

(8) Inca site of Sacsayhuaman at Cusco (13°29'S, 71°57'W): 18 April 2001, one adult pair was seen at an elevation of ca. 3400 m, soaring at this location devoid of any cliffs. They were in a "playful mood" and repeatedly made mock attacks on each other. Both birds appeared fairly dark on the underside.

(9) Valle Sagrado between Calca and Lamay (13°21'S, 71°54'W): 19 April 2001, one adult pair seen at the inland cliff at kilometer post 48 on the right side of the river valley at an elevation of ca. 3000 m. On 20 April 2001, one adult bird perched in the same cliff. On 16 January 2002, I saw one adult hunting and then perched in the cliff. This site is located in the broad valley of the Urubamba river, which has rich agricultural land at the bottom and excellent cliffs available almost continuously for many kilometers. The valley here looks like prime peregrine habitat.

(10) Valle Sagrado between Ollantaytambo and Pachar (13°18'S, 72°12'W): 18 February 1996, I caught a glimpse of a large falcon stooping east of the village of Ollantaytambo at an elevation of ca. 2800 m. On 21–22 April 2001, however, during 5 hr of observation, no peregrines were observed. On 14 January 2002, during 1 hr nothing seen either. The Urubamba valley at this site looks excellent for nesting peregrines. It is only slightly narrower than upriver at Calca, and again with abundant huge cliffs.

DISCUSSION

It is well-known that the important migrant populations of *F. p. tundrius* and *anatum* from the tundra and taiga in North America distribute themselves throughout most Latin America during the northern winter (see Fig. 1 in Henny et al. 1996 and Fuller et al. 1998). Most of my February–May observations were made at a time when the northern migrants may overlap with nesting activities of local birds. Therefore, an obvious question is whether the reported observations concern migrants or instead locally breeding birds.

Based on the plumage and behavior criteria mentioned, some birds observed at the Paracas National Park and at Chala may have been northern migrants. In the austral summer, the greater Lima area has a relatively high density of peregrines of both migrant and local origin (O. Beingolea pers. comm.). Thus, nothing can be said of the birds observed in Lima and also of that in Arequipa, because not enough details were seen.

Based on date, behavior and plumage, the pairs at Lagunillas beach, Laguna Grande, Machu Picchu, Calca, and the repeated observations in the Cañon de Colca may have been locally-breeding birds. The pair at Cusco were probably not migrants either. These observations, thus, suggest up to six new peregrine pairs in Peru. This supports the suggestion by McNutt et al. (1988) that breeding peregrines are probably more widespread in the central Andes than hitherto recorded.

Further support for this hypothesis comes from recent sightings by other observers outside the period when northern migrants are present. In August 1995, D. Michelat (pers. comm.) saw single peregrines in the Cañon de Colca as well as at two widely separated sites in the Valle Sagrado; near Pisac and downriver from Ollantaytambo, between Chilca and Guayllabamba. On 27 July

2001, F. Schmitt (pers. comm.) watched a pair in a cliff overlooking the Rio Colca on its left side, between the villages of Tapay and Cabanaconde. The pair was calling frequently and perched on a heavily white-washed ledge that looked like an eyrie. This site is some 10 km downriver from the site where I watched peregrines in that valley, so this observation presumably indicated a separate territory rather than a very distant alternate nesting cliff. According to G. Engblom (pers. comm.) the Machu Picchu pair was already confirmed in 1999. Interestingly, S. Lovis (pers. comm.) thought he heard a peregrine calling in the background of a television documentary on Machu Picchu in the early 1990s! Finally, O. Beingolea (pers. comm.) found three cliff-nesting peregrine pairs close to the city of Lima in 1993 and by 2001 had located six nesting pairs, one of which was nesting on a building in urban Lima (O. Beingolea and C. White, unpubl. data).

Collectively, these observations and the literature records mentioned above suggest the presence of at least 19–23 peregrine pairs in Peru (Fig. 1). Ten are located in the western foothills of the Andes (including Lima city), two on the coast, five to seven in the Andean highlands, and three to four on the eastern slopes of the Andes. These proportions may not be representative of the true distribution of breeding peregrines in Peru; instead they may reflect the accessibility of and ornithologists' activity in these regions, especially in the Lima area (O. Beingolea) and in northern Peru (Schoonmaker et al. 1985). Breeding pairs have been suspected at very high elevations; 3250 m in the Cañon de Colca and possibly at similar elevations near Cusco (this study) and 3300 m at Yauli (Morrison 1939).

These findings raise the usual question of whether this greater frequency of reports reflects a real population increase or just additional observation intensity. Although it may be hard to imagine that the pairs at Machu Picchu and in Lima were overlooked for a long period, at present, nothing conclusive can be said. The observations reported here greatly increase the known geographic range of the peregrine in South America. Thus, I urge others to watch out for the peregrine in the countries of the central/northern Andes.

RESUMEN.—Entre febrero y mayo 1996 y 2001 y en enero 2002, observé 28 halcones peregrinos (*Falco peregrinus*) en un total de nueve sitios en el centro y el sur del Perú. Cinco individuos fueron observados en Lima, 12 en o cerca de acantilados marítimos de tres sitios costeros, ocho en acantilados en tres sitios del interior del país. Tres individuos fueron observados en el interior del país lejos de acantilados. Aunque algunos de esos halcones podrían haber sido individuos invernantes desde Norteamérica, detalles del plumaje y del comportamiento, sugieren la presencia de parejas anidando en dos sitios costeros (al sur de Pisco) y en el interior del país (Cañon de Colca, Machu Picchu, Calca y cerca de Cusco). Ade-

más, parejas anidando, seguras o probables, recién han sido descubiertas por otros ornitólogos en siete sitios más. Estas observaciones, junto a otras anteriormente mencionadas en la literatura, sugieren al menos 19 parejas de halcones peregrinos en el Perú. Esta especie podría estar más difundida en los países de los Andes centrales de lo que ha sido observado hasta ahora.

[Traducción de Maria Teresa Chu]

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DNA POLYMORPHISMS IN BOREAL OWLS (*AEGOLIUS FUNEREUS*)

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KEY WORDS: *Boreal Owl; Tengmalm's Owl; Aegolius funereus; microsatellite; DNA polymorphisms; variability.*

Molecular analyses of polymorphic DNA-fragments are widely used in phylogenetic studies to recognize individuals, to evaluate mating strategies, and to study genetic diversity (Lawless et al. 1997, Primmer and Ellegren 1998). A limiting factor in studies that depend on species-specific variation is the number of available markers. Due to the conservational nature of DNA across species, polymorphic regions that are localized in one species will often be of great use in a number of related species. This is also the case for microsatellites, which are often localized in less conserved areas (Chambers and MacAvoy 2000). The main focus of this work was to establish DNA polymorphism in the Boreal Owl (*Aegolius funereus funereus*) that would be useful for testing paternity, inbreeding, and population genetics. Microsatellites, characterized by short, tandemly-repeated, and highly-polymorphic sequences, were chosen for the analysis. These markers have previously been used for cross-species amplification in birds (Primmer et al. 1996), and in several other

species. Although microsatellites are highly polymorphic (varying number of tandemly-repeated motifs), sequences flanking the microsatellite are still conserved enough to be present across related species, and are used for primer binding. As expected, a negative relationship between microsatellite performance and evolutionary distance has been observed (Primmer et al. 1996).

METHODS

Blood samples were collected from 44 unrelated free-ranging adult Boreal Owls (Tengmalm's Owl) nesting in Hedmark County, Norway (ca. 61°N, 11°E) in 1998. Natal as well as female breeding dispersal is extensive in the Boreal Owl, causing genetic swamping over large areas (Sonerud et al. 1988). DNA was isolated following standard protocols (Seutin et al. 1991, Krokene et al. 1996). Amplification of microsatellites in Boreal Owl (Table 1) was based on sequences obtained from the Eurasian Eagle-Owl (*Bubo bubo*; Isaksson and Tegelström 2002). Among the microsatellites used in this study Bb111 and Bb126 are GA repeats, whereas, the remaining satellites are CA repeats. Reactions were carried out in 10 µl containing 50 ng of genomic DNA, 0.5 U Taq polymerase, enclosed buffer (Perkin Elmer), 2.5 pmol of each primer and 0.2 mM of each dNTP. Genomic DNA was denatured for 3 min at 94°C prior to amplification. The polymerase chain reaction (PCR-amplification) was run for 35 cycles at 94°C (denaturation) for 15 sec, annealing for 15 sec, and elongation at 72°C for 30 sec. Annealing temperatures varied from 45°C (Bb42) to 48°C (Bb100, Bb101,

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Table 1. Primer sequences (5'–3') for amplification of Boreal Owl microsatellites. All forward primers (F = forward; R = reverse) were fluorescent labelled. Markers Bb42, Bb100, Bb101, Bb111, Bb126, Bb131, and Bb145 can be found in Isaksson and Tegelström (2002).

MARKER	PRIMER SEQUENCES
Bb20 F	GTGGTGGCACGGCTTGT
Bb20 R	TGTCAAGAGGAAGCATAAAAATACAT
Bb120 F	TAATGGTGCTGCTGGTGGGAAG
Bb120 R	CATGTGTAGGTGTGGGAGAGAA
Bb134 F	TTTCTCCACGCTTCCTTTTCATA
Bb134 R	AGAAGAATGGCTGGCAAGACTC

and Bb145) to 50°C (Bb111 and Bb134) and 52°C (Bb126). Successful amplification of Bb20, Bb120, and Bb131 was not obtained at any annealing temperatures. Microsatellites were analyzed on an ABI 373 sequencer.

RESULTS AND DISCUSSION

Of the ten primer pairs characterized in eagle-owls, seven successfully amplified DNA from Boreal Owl (Table 2). Five of these were polymorphic in Boreal Owl, where-

Table 2. Length of alleles, allele frequencies and heterozygosity among 44 unrelated Boreal Owls for seven microsatellites. Microsatellite markers Bb20, Bb120 and Bb131 did not amplify DNA successfully from Boreal Owl.

MICRO-SATELLITE MARKER	ALLELE LENGTHS	ALLELE FREQUENCIES	OBSERVED HETEROZYGOSITY
Bb42	304 bp	1.000	0
Bb100	296 bp	0.761	0.30
	298 bp	0.239	
Bb101	185 bp	0.477	0.57
	187 bp	0.034	
	189 bp	0.034	
	191 bp	0.455	
Bb111	201 bp	0.023	0.61
	203 bp	0.011	
	205 bp	0.080	
	207 bp	0.625	
	209 bp	0.136	
	211 bp	0.080	
Bb126	185 bp	0.989	0.02
	187 bp	0.011	
Bb134	144 bp	1.000	0
Bb145	242 bp	0.898	0.18
	256 bp	0.102	

as, the remaining two were monomorphic within the individuals tested in our analysis. Because Boreal Owls and eagle-owls are among the most distantly related species within the Strigidae family (Mindell et al. 1997), these microsatellites may be of potential use in most species within this family. Our findings could therefore be of great importance for the analysis of population genetics, as well as for parental testing in a wide variety of species within the Strigidae family.

RESUMEN.—Hemos utilizado los pares de indicadores con base en secuencias del gran búho euroasiático con el fin de ampliar exitosamente siete microsatélites de loci en el búho boreal (*Aegolius funereus funereus*), de los cuales cinco fueron polimorfos. El número de alelos por locus variaron entre dos a siete. La conservación de los microsatélites de loci entre el búho boreal y el gran búho euroasiático indica que las secuencias del gran búho pueden ser útiles en estudios moleculares para la mayoría de especies de la familia strigidae.

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POST-FLEDGING MOVEMENTS AND FORAGING HABITATS OF IMMATURE WHITE-TAILED SEA EAGLES IN THE NEMURO REGION, HOKKAIDO, JAPAN

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KEY WORDS: *White-tailed Sea Eagle*; *Haliaeetus albicilla*; *post-fledging movement*; *foraging habitat*; *anthropogenic food sources*; *Hokkaido, Japan*.

In Far East Asia, Hokkaido is the southernmost breeding area for White-tailed Sea Eagles (*Haliaeetus albicilla*). A small number of White-tailed Sea Eagles breed in Hokkaido and are considered resident. Many White-tailed Sea Eagles, along with Steller's Sea Eagles (*H. pelagicus*), also winter in Japan, mainly in Hokkaido.

There are some reports on natal dispersal and movements of immatures for the European populations of White-tailed Sea Eagles (e.g., Helander 1980, Saurola 1981, Król 1983, Stjernberg and Saurola 1983, Meyburg et al. 1994). These authors reported that eagles mostly do not spend their first winter near their natal areas, and some immatures migrate long distances, exceeding 1000 km. However, similar information does not exist for these eagles in Asia. The objectives of this study were to monitor the movements of immature White-tailed Sea Eagles from natal areas and to determine their foraging habitats.

STUDY AREA AND METHODS

The Nemuro region in eastern Hokkaido is mostly a volcanic plain less than 100 m in elevation. Annual mean temperature is 5.7°C and the climate is characterized by a foggy and cold summer, and by a dry and cold winter (Miyawaki 1988). The seacoast is sometimes covered by sea ice in winter. Marsh vegetation has developed around inland-bays and lakes (Miyawaki 1988). A mixed forest consisting mainly of Sakhalin fir (*Abies sachalinensis*) and birch (*Betula ermanii*) is found on the shore terrace, and a broad-leaved forest consisting primarily of Japanese alder (*Alnus japonica*) and Japanese elm (*Ulmus davidiana*) is distributed in the marshland (Miyawaki 1988). The plain was formerly covered with a broad-leaved forest dominated by oak (*Quercus mongolica*), but has been mostly converted to pasture (Takenaka and Ono 1995).

Twenty nestlings (4 wk of age) from eight nests in the Nemuro region were color-banded during 1992–95 (Fig.

1). Color bands were made of plastic, 2.5–3.0 cm high and the joint was glued with acetone. All individuals were marked with a unique combination of color bands. Nests were alphanumerically numbered (N1–N8) and nestlings were alphabetically coded (A–T). Eagles A–E hatched in 1992, F–L in 1993, M–O in 1994, and P–T in 1995.

Seven of these color-banded nestlings were also fitted with radio-transmitters. Two siblings, M and N in N4 were fitted with a 9.8 g tail-mounted transmitter in 1994. The transmitters broadcast at 144 MHz and had a battery life expectancy of 1.5 yr. An approximately 23.0 g leg-band type transmitter was attached to the tarsus of two siblings, S and T in N8, and Q in N4, P in N6, and R in N7 in 1995. Battery-life expectancy of these transmitters was 2 yr. All nestlings were returned to their nests immediately after banding and radio-tagging.

Monitoring of the nestlings in N4 were carried out by direct observations during June and July after attaching transmitters in 1994 and 1995. These observations were made at a distance of 800 m from the nest from dawn to dark at least every other day. To determine fledging day (the day of first flight) for each nestling at N4, the nest was observed every day starting on 26 June in 1994 and on 10 July in 1995. These observations continued until fledging. Radio-tagged fledglings were tracked upon departure from the natal areas. M and N were monitored at least once every three days during August, September, October, and December in 1994. Q was tracked during August and September in 1995.

In most cases, I monitored birds and nests weekly to determine departure days for the other fledglings in 1995. Departure day was defined as the day when a fledgling was gone from the natal area and moved to another location. When a fledgling could not be located on the day of its disappearance from the natal area, the departure date was recorded as the period from disappearance day to the day it was first relocated. When a fledgling disappeared from the place where it was relocated, the natal area was checked to determine if it had returned.

I made an attempt to locate color-banded immature eagles from autumn 1992 until March 1997 by opportunistic observations at likely foraging sites from Notsuke Bay to Nemuro Peninsula, including coastal areas, a lake, rivers, a bay, and a fish factory (Fig. 1), and by interviews with local residents. Regular searches were also conducted once or twice a month along a route that runs along the sea coast from Notsuke Bay to Nemuro Peninsula and

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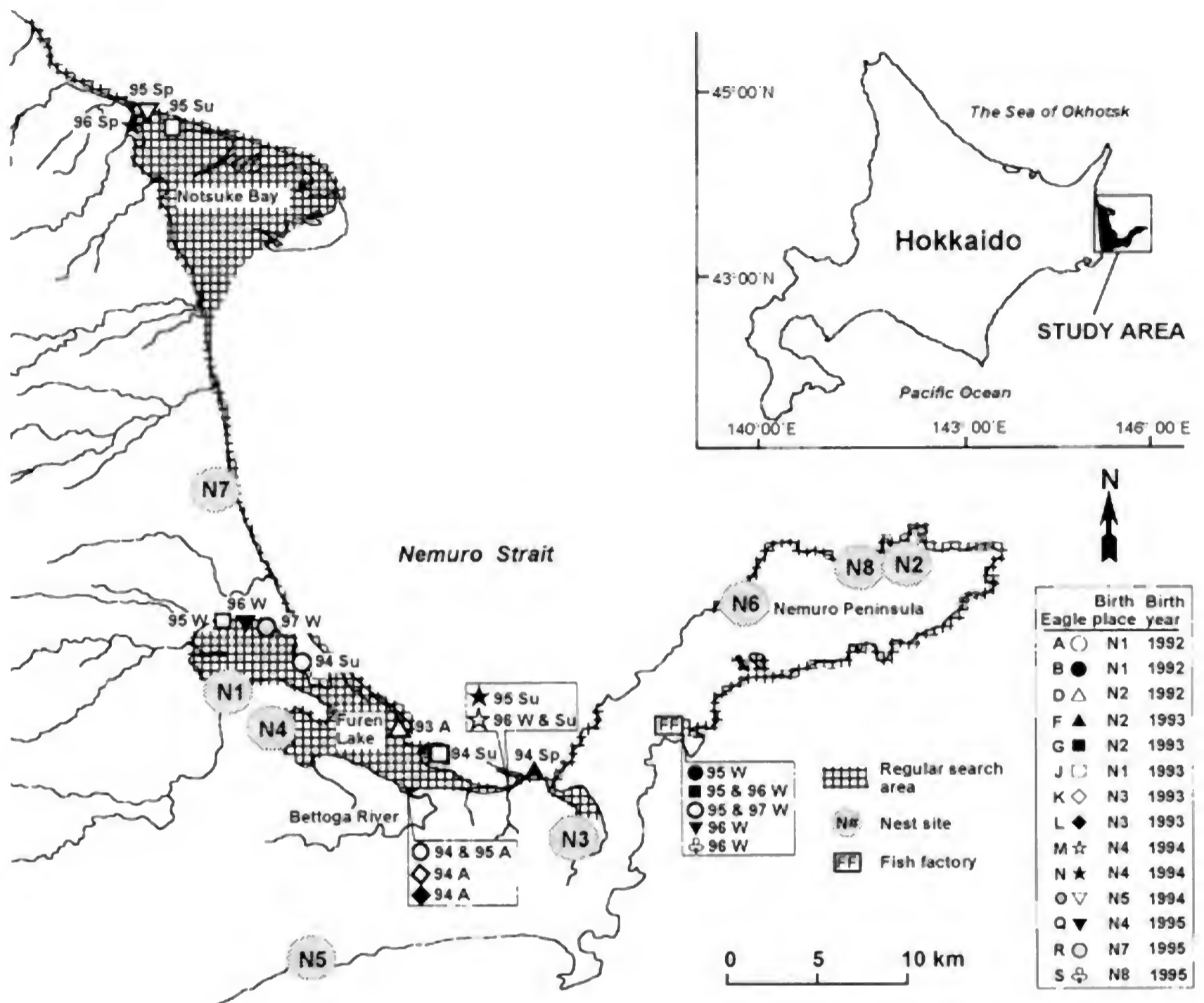


Figure 1. Study area and locations of color-banded White-tailed Sea Eagles observed during and after their first winter in the Nemuro region. Numbers beside symbols indicate year of location and letters indicate season. Seasons are defined as Sp = spring (1 April–31 May), Su = summer (1 June–31 August), A = autumn (1 September–30 November), and W = winter (1 December–30 March).

around Furen Lake from December 1994–October 1996, and in January–February 1997 (Fig. 1).

In this paper, eagles <1 yr of age are referred to as juveniles, and non-adult eagles (>1 yr and <5 yr of age) to as immatures.

RESULTS AND DISCUSSION

Twelve (60%) of 20 color-banded eagles were observed after their first winter. Six (30%) of 20 eagles were observed after their second winter, and one (6.7%) of 15 eagles color-banded in 1992–94 was observed after its third winter.

Accurate fledging dates were determined for three radio-tagged juveniles from N4. They were 28 June 1994 for M, 30 June 1994 for N, and 12 or 13 July 1995 for Q.

The dates of departures from the natal areas were determined for four juveniles. They were between 8–12 September 1994 for M and N, 9 September 1995 for Q from N4, and 13 September 1995 for P from N6. The intervals between fledging and departure were 8–11 wk. Previous studies found that, for Bald Eagles (*H. leucocephalus*), the intervals were 7 or 8 wk (Gerrard et al. 1974), 5–10 wk (McCullough 1986), 2–5 wk (Hunt et al. 1992), 4.5–10 wk (McClelland et al. 1996) and 4–11 wk (Wood et al. 1998).

Movements from the natal areas were recorded during the first autumn for six of seven radio-tagged juveniles and one color-banded juvenile (Fig. 2). The radio-transmitter of eagle R failed after 13 June 1995 and this bird was not re-sighted after 26 August 1995. Juveniles M, N,

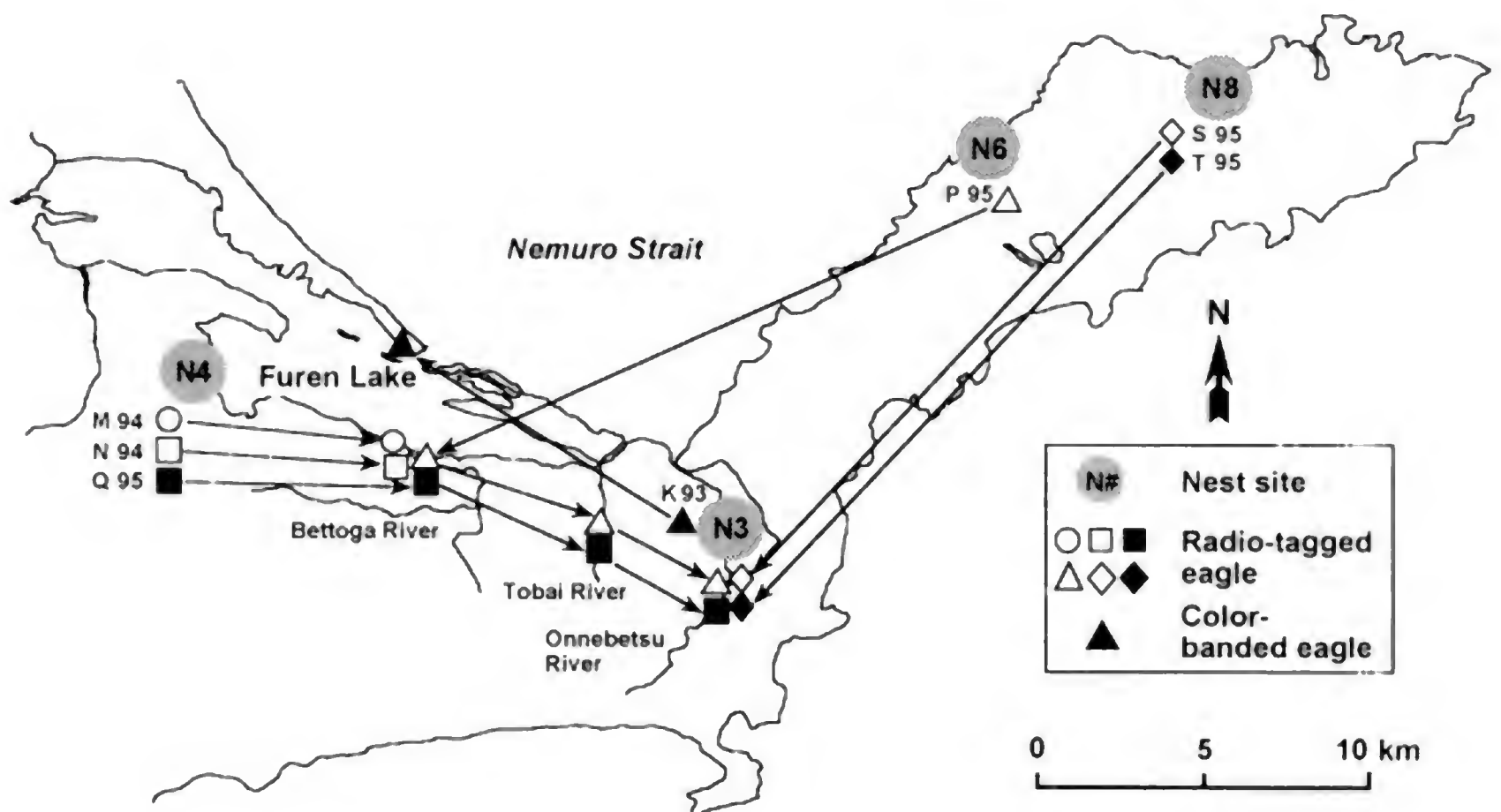


Figure 2. Enlargement of area illustrated in Figure 1. Early movements of radio-tagged and color-banded juvenile White-tailed Sea Eagles from their nests in the Nemuro region. Numbers beside symbols indicate year of hatching and letters refer to individual eagles. Arrows indicate direction of travel.

Q, and P moved from their natal areas to the Bettoga River in September (Fig. 2). Sibling juveniles S and T from N8 were occasionally detected around the natal area until 16 September 1995. They were relocated along the Onnebetsu River on 19 October 1995. Juvenile K was relocated in the first autumn at the east end of sandbar extending from northwestern part in Lake Furen. For all birds, the distance from the nest to the first location away from the nest ranged from 7.0 km to 22.0 km with no apparent preference for direction.

Pink salmon (*Oncorhynchus gorbuscha*) and chum salmon (*O. keta*) are most abundant during August–September, and October–December, respectively, in the rivers of Hokkaido (Ochiai and Tanaka 1986, Nagasawa and Torisawa 1991). Departures from the natal areas in this study seemed to correspond with the timing of salmon runs. Numerous salmon carcasses were observed on the Bettoga River between the estuary and a weir dam 4.0 km upstream from the mouth in September–October 1994. White-tailed Sea Eagles, including color-banded juveniles, and Steller's Sea Eagles were both observed on the Bettoga River feeding on salmon carcasses on the riverbanks, in shallow water and on the tidal flat at low tide. Marked juvenile White-tailed Sea Eagles were also seen on the Onnebetsu and Tobai Rivers in their first autumn (Fig. 2). Similarly, in parts of its range, the movements of post-fledging Bald Eagles are influenced by spawning salmon (Servheen and English 1979, Hodges et al. 1987). Abundant salmon carcasses are particularly

important as easily-available prey for inexperienced juveniles (McClelland et al. 1983, Stalmaster and Gessaman 1984, Restani 2000).

Fourteen of the marked eagles were found during and after the first winter in their natal areas (Fig. 1). Juveniles Q and S were observed in areas where people supplied food, including the fish factory and Furen Lake, during their first winter. At the fish factory, fish offal was dumped on the ground, and on Furen Lake and Notsuke Bay, from the end of December to the end of March fishermen discarded rough fish on the ice.

Eagles A, J, N, and Q were resighted at two or more locations (Fig. 1), eagle A most frequently (five times at three different places between its third summer and its fifth winter). Immature White-tailed Sea Eagles in Europe may disperse to areas of abundant food (Love 1983). Similarly, the movements of immature Bald Eagles are nomadic and variable (McClelland et al. 1994, Jenkins et al. 1999), and are associated with temporary concentrations of prey and carrion (Harmata et al. 1999). Based on my observations, I suggest that juvenile White-tailed Sea Eagles moved among habitats probably in response to food availability.

In spring and summer, immature eagles were found on Furen Lake, Notsuke Bay and on the coast (Fig. 1). These sites had two possible food resources: "natural food," such as fish, waterfowl (e.g., *Anas* spp., *Aythya* spp.), and seagulls; and "anthropogenic food," rough

fish discarded by commercial fishermen on the tidal flats and shores.

During autumn, color-banded eagles were mostly observed on rivers, especially on the Bettoga River (Fig. 1). The rivers in the natal area are important sources of salmon carrion in autumn, not only for first-year eagles, but also for older immatures.

During winter, most eagles were found at the fish factory and at Furen Lake when it was frozen (Fig. 1). Most White-tailed Sea Eagles and Steller's Sea Eagles that wintered in Hokkaido gathered at places where people supplied fish (Working Group for White-tailed Eagles and Steller's Sea Eagles 1996), probably because sufficient amounts of natural prey were not available (Shiraki 2001). Field observations and ring recoveries in Sweden and Finland suggested that survival, especially of first-year eagles, was improved by a supplemental winter-feeding program (Helander 1985). Survival of immature White-tailed Sea Eagles hatched in the Nemuro region also may be enhanced by anthropogenic sources of food.

RESUMEN.—Los movimientos de las áreas de natalidad y de los hábitats de forrajeo de juveniles de *Haliaeetus albicilla*, fueron examinados con telemetría y con observaciones de aves marcadas. La dispersión de las áreas de natalidad ocurrieron entre el 8–13 de septiembre, cuando los salmones (*Salmo* spp.) depositaron sus huevos y fueron abundantes en los ríos cercanos. Las águilas juveniles probablemente se movilizaron entre los hábitats en respuesta a la disponibilidad de comida. En primavera y verano, las águilas juveniles fueron observadas en áreas costeras y en lagos cercanos durante el otoño, las águilas juveniles fueron observadas principalmente en los ríos en donde se alimentaron de restos de salmón. En invierno, las águilas juveniles dependieron de restos de comida provenientes de desechos humanos, tales como peces y despojos de reses muertas. La sobrevivencia de juveniles reproducidos en la región de Nemuro, Japón puede estar reforzada por alimento de origen antrópico.

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HABITAT PREFERENCES, BREEDING SUCCESS, AND DIET OF THE BARN OWL (*TYTO ALBA*) IN ROME: URBAN VERSUS RURAL TERRITORIES

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KEY WORDS: *Barn Owl; Tyto alba; territories; habitat preferences; breeding success; feeding habits; urban habitat; Rome.*

Population density of the Barn Owl (*Tyto alba*) is dependent on available supply of small rodents in both the territory and home range (e.g., De Bruijn 1994, Taylor 1994). Moreover, habitat and nest quality are decisive factors in determining distribution, breeding success, and feeding habits of Barn Owls (De Bruijn 1994, Taylor 1994, Poprack 1996, Martinez and López 1999, Zuberogitia 2000, Baudvin and Jouaire 2001). The decrease in numbers of this owl in central Europe is probably related

to the development of new agricultural practices and loss of traditional nest sites (De Bruijn 1994). In Mediterranean Europe, owl populations seem to be more stable, likely due to a milder climate and large supply of prey (Martinez and López 1999, Zuberogitia 2000), but further information is needed.

Barn Owls commonly breed in urban areas that provide suitable nest sites (e.g., Baudvin and Jouaire 2001). The ecology of the Barn Owl is poorly known in urban habitats and no direct comparisons with neighboring habitats are available. Here, we compare data on distribution, territory characteristics, habitat preferences, breeding success, and feeding habits of Barn Owls from urban and rural areas in central Italy. Understanding

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habitat preferences of this species of special concern in relation to population parameters and feeding habits may provide useful management information for a variety of habitats, including rural and urban areas.

METHODS

The study was carried out in Rome, Italy (41°53'N, 12°28'E) from spring 1995–summer 2001; urban areas included small gardens with *Pinus pinea*, *Cupressus sempervirens*, *Cedrus* sp., and *Quercus* spp. Rural areas included open lands with grassy pastures, uncultivated fields, and small forest patches, mainly of *Q. ilex* and *Q. suber* (Ranazzi et al. 2000).

We surveyed five census plots distributed along the urban gradient that included the main habitats in the study region. Nest sites and daytime roosting sites were searched for the presence of Barn Owls. Pellets, feathers, and droppings near possible nests were considered evidence for the occupation of a site. Records of territorial screeches and calls of young were collected systematically during the entire study period and were combined with the other data to locate nests. Spacing among occupied nests was calculated for each plot by the nearest-neighbor method using data from 1997 breeding season. Regularity in nest spacing was computed for each area with the *G*-test (Ranazzi et al. 2000).

We measured percentages of (1) open lands, (2) deciduous woods, (3) conifer woods, (4) urban gardens, (5) developed areas (buildings and homes), (6) Roman ruins, and (7) waterbodies in a circular plot with a radius of 1.5 km centered in the nest site (Michelat and Giraudoux 1991) at 10 urban and 7 rural Barn Owl nests whose occupation was confirmed throughout the study period. The same variables were measured in 15 unoccupied sites randomly selected along the urban gradient in the five plots surveyed. We compare each variable measured at urban and rural territories and at occupied and random sites by Mann-Whitney *U*-tests. A sequential Bonferroni test was used to adjust the significance level to the number of comparisons using the same data set ($N = 7$).

No data on clutch size were collected to minimize disturbance of the adult owls. Moreover, many nests located in scaffolding holes of old buildings were inaccessible for inspection of eggs. Visits to nests were limited to a period when young were ca. 3–6 wk of age (De Bruijn 1994).

We studied diet by analyzing pellets collected (April–August) in 15 stable territories classified as urban or rural based on the percentage of developed areas (urban: >50% developed in the circular plot with a radius of 1.5 km centered in the nest site). Prey remains were identified using diagnostic keys and by comparison with museum specimens (e.g., Piattella et al. 1999). Differences in diet composition between urban and rural diets were tested using a χ^2 contingency table which included all the prey groups reported in Table 1. We used distance of each pellet site to the center of the city (Ranazzi et al. 2000) as a relative index of the proportion of urban areas around owl sites. Spearman rank correlations were performed between the percentage of each prey group (Table 1) and the distance to the city center.

Table 1. Percent of prey types recorded in the prey remains at Barn Owl nest and roost sites in urban areas of Rome, central Italy.

	URBAN SITES <i>N</i> = 7	RURAL SITES <i>N</i> = 8
Invertebrates	0.69	1.18
Anura	0.00	0.05
Reptilia	0.13	0.00
Columbidae	0.25	0.10
<i>Sturnus vulgaris</i>	1.01	0.00
<i>Passer</i> spp.	4.53	0.88
Fringillidae	0.94	0.25
Other		
Passeriformes	2.96	1.97
AVES total	9.69	3.19
<i>Suncus estruscus</i>	1.89	3.14
<i>Crocidura</i> spp.	2.14	3.98
<i>Talpa</i> sp.	0.00	0.15
INSECTIVORA total	4.03	7.27
Chiroptera	1.01	1.77
<i>Muscardinus avellanarius</i>	0.00	3.34
<i>Microtus savii</i>	59.18	41.52
<i>Apodemus</i> spp.	5.72	22.36
<i>Rattus</i> spp.	2.83	1.23
<i>Mus domesticus</i>	16.67	15.23
Other mammalia	0.06	2.85
RODENTIA total	84.47	86.54
Total prey (<i>N</i>)	1590	2035

RESULTS

Based on the distribution of 31 Barn Owl territories, density was generally higher than those recorded in central Europe (Table 2). Mean nest spacing ranged from 1.8 km–3.0 km. The *G*-test (0.78) indicated a substantial regularity in nest distribution. Mean density in the sub-areas surveyed ranged from 8–21 territories/km².

Open lands contributed half the available area of the census plots in rural territories; this decreased in urban territories. Wooded and developed areas made up the remaining part of rural and urban territories, respectively (Fig. 1). The percentages of both deciduous and conifer woods, as well as of urban gardens showed significant differences between urban and rural territories (deciduous woods: $U = 0$, $P = 0.0006$; conifer woods: $U = 0$, $P < 0.001$; urban gardens: $U = 6.5$, $P = 0.005$). Occupied territories contained a significantly higher proportion of open lands than random plots (Table 3), but a lower proportion of vegetable gardens and developed areas.

Out of 14 breeding attempts, 2 failed (14.3%), 1 produced one fledgling (6.7%), 7 produced two fledglings (50.0%), 3 produced three fledglings (14.3%), and 1 produced four fledglings (6.7%). The mean number of

Table 2. Breeding density of Barn Owls from selected European studies.

STUDY REGION	CENSUS PERIOD	CENSUS AREA (km ²)	MEAN DENSITY (TERRITORIES/100 km ²)	SOURCE
Poland, Krakow	1991–95	6289	0.8	Bartmanska et al. 2000
Czech Rep., Olomouc	1983–95	1451	1.3	Poprack 1996
South Poland	1984–88	1640	1.4	De Bruijn 1994
West Germany	1960–72	841	1.7	De Bruijn 1994
Netherlands, Liemers	1967–84	250	2.4	De Bruijn 1994
Southwest Scotland	1981–85	2200	3.2	De Bruijn 1994
East Germany	1968–74	1000	3.3	De Bruijn 1994
Netherlands, Achterhoek	1967–84	250	5.3	De Bruijn 1994
Germany, Bergenhusen	1974–79	100	10.0	De Bruijn 1994
Italy, Rome	1995–2001	241	12.8	This study

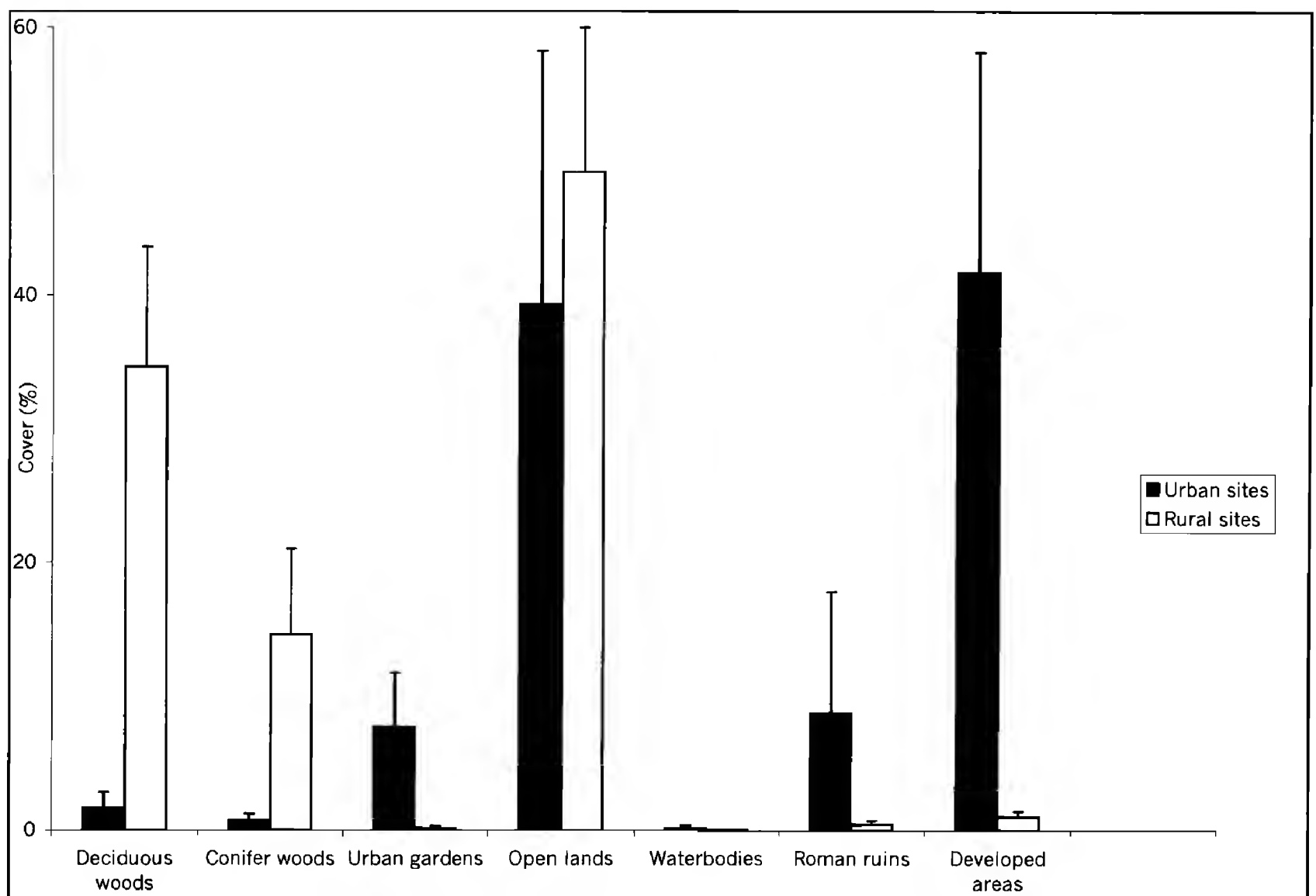


Figure 1. Percentage of different habitat cover types found within a 1.5 km diameter plot centered on Barn Owl nests in urban ($N = 10$) and rural ($N = 7$) areas in Rome. Error bars represent the SD around the mean percentage of each land cover variable.

Table 3. Percentages of seven land cover variables (mean \pm SD) in seventeen occupied territories and fifteen random sites in Rome, central Italy.

VARIABLE	OCCUPIED TERRITORIES	RANDOM SITES	P-LEVEL
Deciduous woods	15.2 \pm 20.1	1.5 \pm 2.3	0.02
Conifer woods	6.5 \pm 10.6	0.7 \pm 1.3	0.02
Urban gardens	4.6 \pm 7.1	15.9 \pm 8.8	0.001*
Open lands	43.4 \pm 32.1	2.0 \pm 2.4	<0.0001*
Waterbodies	0.1 \pm 0.3	1.3 \pm 3.5	0.83
Roman ruins	5.3 \pm 14.3	1.2 \pm 2.1	0.35
Developed areas	25.0 \pm 32.2	77.5 \pm 11.1	0.0001*

* $P < 0.05$ based on Mann-Whitney U -test after Bonferroni correction—see methods for details.

fledglings was 2.0 (SD = 1.2) per breeding pair and was low compared to data collected in central Europe (Table 4). Urban owls reared more fledglings than rural ones (2.3 vs. 1.7 fledglings per breeding pair, $N = 7$ breeding attempts per each habitat), but this difference was not significant ($U = 10$, $P = 0.20$).

Based on 3625 prey analyzed, rodents (Rodentia) and shrews (Insectivora) represented 94% of total prey in Barn Owl diets from rural sites and 88% in urban sites (Table 1). Birds, especially sparrows, increased in urban areas. Differences in diet composition between urban and rural sites were significant ($\chi^2 = 436.74$, $P < 0.001$, $df = 17$). The percent numbers of *Crocidura* shrews ($r_s = 0.58$, $P = 0.02$, $N = 15$), *Muscardinus* dormices ($r_s = 0.52$, $P = 0.05$, $N = 15$), and *Apodemus* mice ($r_s = 0.64$, $P = 0.01$, $N = 15$) increased with the distance to the city center. The percent numbers of both *Microtus* voles ($r_s = -0.66$, $P = 0.007$, $N = 15$) and rats ($r_s = -0.73$, $P = 0.002$, $N = 15$) strongly decreased with the distance to the city center.

DISCUSSION

The close nest spacing in our study area is probably due to high availability of nest sites (De Bruijn 1994,

Baudvin and Jouaire 2001). In urban areas, ruins, towers, and old farmhouses provided a surplus of nest cavities. Barn Owls primarily defend their nest sites rather than a breeding territory around them; thus, feeding areas overlap extensively and are dynamic depending on nest supply and prey densities (Taylor 1994). Stable weather typical of the Mediterranean basin may further account for high population levels in Rome, compared to more forested rural areas. In both urban and rural sites, open lands represented the primary foraging habitat found within Barn Owl nesting areas (De Bruijn 1994) and the proportion of open habitats was significantly less at random sites (Table 3). Ruins and gardens were abundant in urban territories providing more foraging areas for owls at these sites compared with rural areas.

The breeding success, although based on a limited sample, was lower than those recorded in central Europe. The abundance of rodents has been reported to strongly influence the reproduction of Barn Owls (Taylor 1994). Owls in areas with generally drier climates probably have lower prey densities (e.g., Herrera and Hiraldo 1976) compared to populations from central Europe, and a reduction in the availability of rodents seems plausible to explain the low breeding rate in Rome. The switch to-

Table 4. Breeding success of Barn Owls from selected European study areas.

STUDY REGION	DURATION OF STUDY (YEARS)	MEAN FLEDGLINGS PER PAIR	SOURCE
Czech Rep., Olomouc	12	5.0	Poprack 1996
Slovakia	4	4.5	Sarossy 2000
East Germany	7	4.3	Taylor 1994
France, Burgundy	25	4.0	Baudvin and Jouaire 2001
Germany, Saarland	5	3.9	Poprack 1996
Southwest Scotland	13	3.2	Taylor 1994
Holland	6	3.1	De Bruijn 1994
Spain, Vizcaya	6	2.2	Zubergoitia 2000
Italy, Rome	5	2.0	This study

ward synantropic rodents and birds in the diet at urban sites reflects the reduction of prey diversity that occurs in such areas and indicates the variation in the availability of small mammal species along the urban gradient (Piatella et al. 1999). The predation on abundant rat populations may explain locally high breeding success in urban areas compared to neighboring farmlands (Martinez and López 1999).

RESUMEN.—Recolectamos datos (1995–2001) sobre la distribución, características del territorio, preferencias de hábitats, tasas de reproducción y hábitos alimenticios de la lechuza de campanario (*Tyto alba*), en reproducción en zonas rurales y urbanas de Roma, Italia. La distancia entre nidos osciló entre 1.8 km. A 3.0 km. Los territorios urbanos incluyeron más espacio abierto que los rurales. Los territorios de las lechuzas contenían un mayor porcentaje de áreas boscosas (21.7%) que los sitios escogidos al azar (2.2%). El número medio de volantones producidos por pareja en reproducción (2.0 ± 1.2) fue inferior a aquellos registrados para el centro de Europa. Los roedores representaron el 94% de las presas en áreas rurales, pero solo el 88% en sitios urbanos. Los roedores del género *Microtus* y las ratas, dominaron la dieta de los sitios urbanos, mientras que los del género *Apodemus* y *Muscardinus* fueron depredados en sitios rurales. Un clima estable y el aprovisionamiento de nidos, probablemente contribuyeron a los altos niveles de población en Roma. Una reducción general de pequeños mamíferos en las áreas secas del Mediterráneo puede explicar el bajo éxito reproductivo comparado con las poblaciones del centro de Europa.

[Traducción de César Márquez]

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INCIDENCE OF NATURALLY-HEALED FRACTURES IN THE PECTORAL BONES OF NORTH AMERICAN ACCIPITERS

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KEY WORDS: Accipiter; Sharp-shinned Hawk; *Accipiter striatus*; Cooper's Hawk; *Accipiter cooperii*; Northern Goshawk; *Accipiter gentilis*; fractures; North America; injuries.

The theory of skeletal strength proposed by Alexander (1981, 1984) suggests that optimal bone strength evolved to minimize risk of fracture. Alexander theorized that a balance between risk of bone failure and energy cost of growth and utilization of heavier skeletons should be achieved through natural selection. He further suggested that fracture incidence should increase as the cost of injury falls. Studies of fractures in wild populations are potentially useful in testing this theory. Few such studies have been published.

In this study, we focused on healed fractures to the pectoral girdle in three species of *Accipiter*. Members of this genus are specialized forest predators that primarily use a series of perch and scan periods from the concealment of tree branches and bushes before chasing prey (Bent 1937). The Sharp-shinned Hawk (*Accipiter striatus*), primarily a small-bird predator, Cooper's Hawk (*A. cooperii*), and Northern Goshawk (*A. gentilis*), both more generalized small-mammal and bird predators, hunt in this manner (Reynolds and Meslow 1984). This type of hunting behavior may lead to collisions with branches and other such obstacles. Therefore, evidence of fractures should be present in these species, given that the cost of such injury is low enough to enable individuals to recover.

METHODS

Skeletal specimens ($N = 339$) of accipiters were visually inspected for evidence of healed fractures to the pectoral girdle (furcula, scapula, and coracoid). The pectoral girdle was chosen due to its importance to flight and its potentially higher susceptibility to breakage from head-on impact. Healed fractures were identified by the presence of bone callus associated with fracture lines. Species examined were *A. cooperii* ($N = 115$: 37 males, 64 females, 14 unknown sex), *A. gentilis* ($N = 52$: 26 males, 22 females, 4 unknown sex) and *A. striatus* ($N = 172$: 65 males, 90 females, 17 unknown sex). Birds that were known to have been captive for any period of time were not included.

Specimens were collected between 1921 and 1998. The frequencies of birds with healed fractures collected from each decade were compared to determine if rates increased with time. Due to smaller sampler sizes it was not possible to compare frequencies from the 1920s, 1940s, and 1960s

RESULTS

Evidence of healed fractures was found in 18.6% (63 hawks with 67 fractures) of the individuals ($N = 339$). There was no significant difference among the species ($\chi^2 = 1.78$, $df = 2$, $P > 0.25$; Table 1). There was also no significant difference between the combined sexes (sexes were combined to increase sample size) ($\chi^2 = 0.96$, $df = 1$, $P > 0.25$; Table 1). The majority of fractures occurred in the furcula (16%, $N = 54$), 12 fractures were found in the scapula (4%); and only one healed fracture was found in the coracoid (0.3%). Fractures occurred in varying locations in both the furcula and scapula. Fractures at the center of the furcula (i.e., point of fusion of the clavicles) were common as well as fractures near or at the midpoint of each clavicle shaft. Fractures of the scapula were most often at either the extreme distal end or near the center. Scapular fractures sometimes healed at an angle (e.g., one scapula healed at the midpoint of the bone at an angle of about 45°). There was no significant difference between the frequencies of healed fractures in birds collected during the 1930s, 1950s, 1970s, 1980s, and 1990s ($\chi^2 = 2.01$, $df = 4$, $P > 0.50$).

DISCUSSION

The incidence of healed pectoral fractures confirms that impact injuries occur in accipiters and that some are able to recover. The cause of fractures could not be determined from museum specimens. There were no biases in fracture incidences with respect to species and sex (Table 1). We suggest that both natural and anthropogenic causes were involved in fractures. If anthropogenic causes were more significant than natural causes, than fracture incidence would be expected to increase through time due to increased urbanization. However, this was not the case.

Raptor rehabilitation efforts have increased since 1960 (T. French pers. comm.). This could affect healed fracture incidence for birds collected in subsequent years. Our data showed no significant change in the fracture

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Table 1. Incidence of healed pectoral-bone fractures in each of the three North American *Accipiter* species.

	NUMBER OF FRACTURE FRACTURES INCIDENCE	
Cooper's Hawk ($N = 115$)	26	23%
Northern Goshawk ($N = 52$)	10	19%
Sharp-shinned Hawk ($N = 172$)	27	16%
Total male ($N = 128$)	29	23%
Total female ($N = 176$)	31	18%
Unknown sex ($N = 35$)	3	9%

rates for years before and after 1960, suggesting that rehabilitation is not a factor.

Healed fracture incidence represents only birds that survive injury. Therefore, the incidence of impact injury to the pectoral girdle in accipiters may be higher than that found in this study, especially in urban habitats. Boal and Mannan (1999) found that collision with man-made objects were responsible for 69.8 percent of mortalities of Cooper's Hawks in an urban environment. Many of the specimens used for this study were collected during migration, making it impossible to know if they frequented rural or urbanized habitats during the breeding and winter seasons. The fact that fracture incidence did not increase through time with increased urbanization suggests that either anthropogenic causes of fracture are not more significant than natural causes or that birds are less likely to survive collision with man-made structures. Further studies of fracture incidence among urban versus rural birds are needed. Also, the scarcity of healed coracoid fractures could be the result of less frequent injury to this bone, or birds are less likely to survive coracoid injury. Peregrine Falcons (*Falco peregrinus*) with broken coracoids are unable to fly well enough to catch prey and, therefore, are unable to recover (T. French pers. observ.).

Previous studies have focused on healed fractures in the long bones of wild birds. Brandwood et al. (1986) examined wild-caught individuals from three avian families for incidence of long-bone fractures. They found a fracture incidence of 0.4% for total bones examined in anatids, 0.4% and 0.5% in two larid samples and 0.2% in columbids. It was theorized from these data that most birds would be either unlikely to suffer fractures or would not survive them (Brandwood et al. 1986). Accipiters represent an exception to this hypothesis as do members of the genus *Gyps*. In a small sample of White-backed Vultures (*G. africanus*) and Rueppell's Griffons (*G. rueppellii*), Houston (1993) found a 20% incidence of healed ulnar fracture. Unlike accipiters, these vultures do not engage in high-risk behavior. Houston theorized that the high fracture incidence in *Gyps* is due to skeletal fragility related to extreme weight reduction.

Considering the importance of flight to accipitrine

hunting style, it is significant that these birds are able to recover from a pectoral-bone fracture. This suggests the cost of such an injury is sufficiently low as to balance the risk of collision inherent in the behavior of this genus. In addition, the susceptibility to fracture of the furcula and scapula in accipiters may be compensated by the lower energy cost of a light skeleton.

RESUMEN.—Especímenes de Museo ($N = 339$) de *Accipiter striatus*, *A. cooperii* y *Accipiter gentilis* fueron examinados debido a la evidencia de fracturas soldadas naturalmente en el hueso pectoral. La incidencia general de las fracturas fue del 19%. No hubo diferencia en la frecuencia de fracturas entre las tres especies o entre sexos. La mayoría de las fracturas ocurrieron en la "furcula." La frecuencia de las fracturas soldadas sugieren que las heridas causadas por impacto son comunes en los accipiters, y que algunos individuos son capaces de recuperarse.

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PLASMA CHEMISTRY REFERENCE VALUES IN FREE-LIVING BONELLI'S EAGLE (*HIERAAETUS FASCIATUS*) NESTLINGS

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Few studies dealing with plasma biochemistry and physiology in wild birds can be found in the scientific literature. Most research papers deal with domestic or captive birds kept in zoos, rehabilitation facilities, or research centers (Lewandoski et al. 1986, Redig 1991, Ferrer 1993, Dobado-Berrios et al. 1998). The knowledge of normal reference values in plasma for wild species is very important to a wide range of multidisciplinary subjects. Veterinarians need to have this information in order to better diagnose the condition of wild birds entering into rehabilitation centers or zoos (Lepoutre et al. 1983, Cooper et al. 1986). Also, information gathered from hematological research is of a great importance for ecologists because such data may provide insights into the health of individuals being studied. Body condition that is related to other ecological factors such as survival, fecundity, or habitat quality, could be estimated by levels of urea, uric acid, and other blood parameters (Cherel et al. 1987, Ferrer et al. 1987, García-Rodríguez et al. 1987, Robin et al. 1987).

It is important to know normal blood parameter reference values for endangered species involved in a reintroduction or restoration program in order to better understand the physiological status of the released birds. Normal reference values in blood chemical constituents are known only for 5% of bird species which have been studied mostly in captive situations (Ferrer 1993).

Although most of the information available comes from captive birds, it might be expected that a captive condition might affect hematological values (Bell and Freeman, 1971, Miglirioni et al. 1973, Wolf et al. 1985, Sturkie 1986, Ferrer et al. 1987, García-Rodríguez et al. 1987). Factors such as age or sex influence the total variation found in plasma enzymes, proteins, metabolites, and other organic molecules. However, presently these factors are poorly understood due to difficulty of gathering information on different age-classes in wild species. Other factors affecting values of chemical components in plasma are circadian rhythm (García-Rodríguez et al. 1987), seasonal changes (Wolf et al. 1985), or plasma storing methods (Bustamante and Traviani 1993).

The Bonelli's Eagle (*Hieraaetus fasciatus*) is an endan-

gered species that has suffered a rapid population decline in most areas of Europe including Spain (Cugnasse 1984, Palma et al. 1984, Hallmann 1985, Arroyo et al. 1990). In this article, we present normal chemical plasma values found in a free-living endangered population of nestlings of this bird of prey. Data from 21 biochemical substances (including metabolites, total protein, inorganic ions, and enzyme activities) and differences found between sexes in this age-class are reported. In addition, we have examined the differences found between free-living and captive birds of this long-lived raptor.

METHODS

We have studied a breeding population of Bonelli's Eagles in the province of Cádiz that is located in southern Spain (5°32'W, 36°41'N). We collected blood from both nestlings of a free-living population of south Spain and from captive young Bonelli's Eagles. The diet of eagles in our region included a preponderance of rabbits (*Oryctolagus cuniculus*), and Red-legged Partridge (*Alectoris rufa*) (Gil-Sánchez et al. 1994, Ontiveros and Pleguezuelos 2000). Birds kept in captivity were fed ad libitum with partridge and rabbit. Blood collected from free-living nestlings was taken when individuals were between 47–53-d-old, about 10 d before fledging. One of us climbed or descended to several nests each year to band and measure young; at the same time 2 ml of blood was extracted from the brachial vein of the wing. To minimize circadian variations of blood parameters, we extracted all blood samples between 1100–1500 H. CST blood was collected in lithium-heparin tubes and the plasma was separated by centrifugation (10 min: 907.2 × g). Cellular fraction and plasma samples were immediately frozen (–80°C). Analyses were carried out 4 mo later with a Hitachi 747 multichannel automatic analyzer (Tokyo, Japan) with the reagents recommended by Boehringer-Mannheim (Darmstadt, F.R.G.). Plasma was analyzed (abbreviations and methods indicated in parentheses) for amylase (AMY; maltoheptaose reaction), cholesterol (CHOL; cholesterol esterase), creatinine (CREA; Kinetic Jaffé reaction), creatinine kinase (CK; optimal standard method DGKC), glucose (GLUC; hexokinase method), aspartate aminotransferase (AST; DGKC technique), alanine aminotransferase (ALT; DGKC technique), total protein (TP; biuret reaction), triglycerides (TRIG; enzymatic method), urea (UREA; urease method, uric acid (UA, uricase method), alkaline phosphatase (AP; paranitrophenylphosphate method), colinesterase (CHE), L-lactate dehydrogenase (LDH; SFBC technique), bilirubin (BILIR; DPD method), calcium (Ca; cresolphthalein complexone reaction), phosphorus (iP; molybdenum blue reaction),

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sodium (Na; Indirect potentiation, E. Selective), potassium (K; Indirect potentiation, E. Selective), magnesium (Mg; blue xilidil reaction).

The cellular fraction of the blood sample was used to sex all eagles. For this analysis, primers 2945F, cfR and 3224R were used following Ellegren (1996). The total number of eagles sampled included 28 nestlings, 14 females and 14 males, belonging to a free-living population and two young males and three young females that were captive birds.

We used the Student's *t*-test to check for differences in the means of blood parameters between sexes and between captive and free-living birds. Non-parametric tests were employed for those variables not meeting either normality or homoscedasticity assumptions (Siegel and Castellan 1988). When data were not normally distributed, they were \log_e transformed (Sokal and Rohlf 1995). After this transformation only two blood parameters AP and K, did not exhibit a normal distribution. The assumption of homoscedasticity was met for those parameters showing a normal-like distribution. Levene's *F* test was used to test for homoscedasticity. All tests were two-tailed and statistical significance was set at $P < 0.05$. Means are given with \pm SD. Range for all variables are also provided. In some cases due to the small plasma volume of some samples, it was not possible to do analyses of all chemical parameters, thus, the sample size is not the same for all blood components. Analyses were performed with the Statistica Ax 99 package (Statistica, 1996. Version 5. Statsoft, Inc.).

RESULTS

Normal reference plasma chemistry values for nestling Bonelli's Eagles are shown in Table 1. We found differences in parameters between male and female free-living nestlings in two of 21 plasma parameters measured. Males showed higher glucose levels in blood and lower AST activity than females (Table 1). We found that urea ($t = 3.78$, $df = 32$, $P < 0.001$), uric acid ($t = 3.21$, $df = 33$, $P < 0.001$), alkaline phosphatase ($t = 3.52$, $df = 32$, $P < 0.001$) and creatinine kinase ($t = 2.5$, $df = 32$, $P < 0.05$) values were lower than these measured in captive eagles. However, glucose ($t = -3.89$, $df = 33$, $P < 0.001$) was higher in captive compared with free-living birds.

DISCUSSION

Our data showed that there were differences in plasma glucose levels between sexes in nestling Bonelli's Eagles, males showing higher levels than females (Table 1). Polo (1995) failed to find any difference in this parameter between sexes in eight avian orders including Falconiformes. However, Polo (1995) only examined birds that were in captivity. Levels of glucose in plasma have been correlated with metabolic rate (Umminger 1977). Birds that showed a high metabolic rate, because of high activity such as flying with fast flapping, would also have higher levels of glucose in plasma. Bonelli's Eagle exhibits a strong sexual size dimorphism, with females being much larger than males, which implies that male and female nestlings of this species would have different metabolic

rates due to different growth patterns. Consequently, each sex may be exposed to a different energy demand during the nestling period. For example, female nestlings gain an estimated 7.3 grams more than males each day during the nestling period, which lasts on average between 59 (Minguez et al. 2001) and 63 d (Real et al. 1998). Therefore, males might have higher glucose levels because they have a lower growth energy demand than females. The larger females might channel more glucose into tissue formation than smaller males. Recently, Casado et al. (2002) also found that male nestling Booted Eagles (*Hieraaetus pennatus*) had higher glucose than females, another raptor with strong sexual size dimorphism. However, González and Hiraldo (1991) studying free-living nestling Marsh Harriers (*Circus aeruginosus*) (also shows sexual size dimorphism), found the reverse tendency with female nestlings having higher glucose levels than males. A possible explanation is that glucose level in nestlings could indicate the quantity and quality of food received by each individual during the growing period. Parents might be able to allocate food in an asymmetric way within brood favoring either sex depending on environmental condition or food availability; for example, the larger or older sibling may receive more food in years of scarcity (Mock et al. 1987).

As most of the knowledge in normal plasma reference values comes from captive birds, we compared blood chemistry values from our free-living nestling sample with values gathered from five juvenile captive eagles. These birds were young eagles in their first year; therefore, they were a few months older (less than a year different) than eagles sampled in nature. For this reason, age may have affected the differences found between these two groups. Among the differences we found were those related to nitrogen residues. Captive birds had lower UREA and UA than free-living ones. These chemical constituents have been associated with physical condition in birds of prey. An increase in plasma levels of these nitrogen residues has been predicted when birds are subjected to a food stress situation (Ferrer 1994, Alonso-Alvarez and Ferrer 2001). Therefore, lower levels in UREA and UA in captive birds indicate that these eagles were in better body condition than free-living birds.

In this study we also found that captive eagles showed higher plasma glucose levels than free-living ones, which might be in accordance to what has been reported in other birds (Lewandoski et al. 1986, Casado et al. 2002). Captive individuals also showed lower CK activity than free-living ones. This enzyme mediates in muscle contraction and is related with physical activity. Thus, it seems reasonable that eagles in captivity would show lower CK activity than free-living eagles. The difference found on AP activity between captive and free-living birds might be attributed to an age effect rather than to a captive condition since it is well known that the activity of this enzyme decreased with aged in birds of prey. Concretely, this enzyme is related to the ossification of frontal

Table 1. Blood chemistry values for male and female nestling Bonelli's Eagles from Cádiz, Spain.

VARIABLE	(SI UNIT)	MALES				FEMALES				df	P ^a
		MEAN ± SD	RANGE	N	MEAN ± SD	RANGE	N	t			
GLU	(mmol L ⁻¹)	15.74 ± 2.09	(12.06-18.50)	14	14.03 ± 1.39	(11.3-15.7)	14	2.53	26	0.017*	
UREA	(mmol L ⁻¹)	2.27 ± 0.56	(1.33-3.17)	13	2.52 ± 0.98	(0.92-4.33)	14	-0.84	25	0.40	
UA	(μmol L ⁻¹)	684.2 ± 279.6	(434.3-1362.5)	14	827.0 ± 307.6	(321.3-1362.5)	14	-1.51	26	0.14	
CREAT	(μmol L ⁻¹)	24.8 ± 4.43	(17.7-35.4)	14	25.6 ± 3.54	(21.2-32.7)	14	-0.44	26	0.66	
PT	(g/l)	29.8 ± 1.5	(26-32)	10	30.1 ± 2.8	(27-36)	10	-0.36	18	0.71	
AP	(UI L ⁻¹)	2148 ± 696	(509-3160)	13	2280 ± 327	(1696-2759)	14	-0.63 (1)	25	0.52 ^b	
CHE	(UI L ⁻¹)	1191 ± 272	(961-1860)	9	1140 ± 228	(789-1571)	10	0.45	18	0.65	
AMY	(UI L ⁻¹)	1148 ± 303	(784-1776)	9	942 ± 194	(726-1310)	12	1.90	19	0.07	
CK	(UI L ⁻¹)	3859 ± 883	(2116-5034)	13	3853 ± 918	(1880-5352)	14	0.01	25	0.98	
LDH	(UI L ⁻¹)	1647 ± 160	(1489-1903)	6	1828 ± 355	(1199-2417)	8	-1.15	12	0.27	
COL	(mmol L ⁻¹)	4.35 ± 0.60	(3.06-5.54)	16	4.53 ± 0.73	(3.29-5.80)	14	-0.65	26	0.51	
TG	(mg/dl)	71.3 ± 35.1	(28-166)	13	81.7 ± 36.6	(36-161)	14	-0.93	25	0.35	
MG	(mmol L ⁻¹)	0.77 ± 0.04	(0.54-0.70)	6	0.72 ± 0.07	(0.54-0.79)	8	-1.67	12	0.12	
NA	(mmol L ⁻¹)	66.09 ± 1.27	(65-67.7)	5	65.95 ± 1.81	(62.2-68.1)	8	0.13	11	0.89	
K	(mmol L ⁻¹)	22.3 ± 11.9	(0.89-28.1)	5	21.2 ± 11.7	(1.15-28.6)	8	0.21 (1)	11	0.82 ^b	
CA	(mmol L ⁻¹)	2.15 ± 0.25	(1.63-2.38)	6	2.20 ± 0.23	(1.80-2.45)	8	-0.25	12	0.80	
P	(mmol L ⁻¹)	1.55 ± 0.23	(1.2-1.94)	7	1.49 ± 0.19	(1.10-1.68)	8	0.54	13	0.59	
BILIR	(μmol L ⁻¹)	1.19 ± 0.17	(0.85-1.70)	6	1.02 ± 0.51	(0.51-2.22)	8	0.44	12	0.66	
AST	(UI L ⁻¹)	171.6 ± 18.2	(153-202)	6	203.2 ± 31.5	(139-241)	8	2.17	12	0.04*	
ALT	(UI L ⁻¹)	11.5 ± 2.2	(10-16)	6	13.3 ± 3.8	(9-19)	8	-1.04	12	0.31	
GGT	(UI L ⁻¹)	14.5 ± 14.5	(7-53)	9	15.1 ± 10.0	(5-39)	12	-0.41	19	0.67	

^aProbability that means are equal calculated with a Student's *t*-test.

^bProbability that distributions are equal calculated with a Mann-Whitney *U*-test.

* Significantly different, *P* < 0.05.

bones that take place throughout the immature to adult age period (Dobado-Berrios and Ferrer 1997, Viñuela et al. 1991).

RESUMEN.—Este artículo presenta información sobre los valores de referencia normales de parámetros bioquímicos presente en plasma sanguíneo de pollos estudiados en libertad de Águila perdicera (*Hieraaetus fasciatus*). Investigamos diferencias en los parámetros sanguíneos entre sexos. Los machos muestran unos niveles más altos de glucosa y una actividad enzimática de Alanino-amino transferasa (AST) mayor que las hembras. Los pollos fueron comparados con jóvenes águilas mantenidas en cautividad. Las águilas cautivas tuvieron valores más bajos de urea, ácido úrico, fosfatasa alcalina, creatinina quinasa y valores más altos de glucosa que los pollos marcados en libertad.

[Traducción de los autores]

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LETTERS

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THE FOX KESTREL (*FALCO ALOPEX*) HOVERS

Although typically considered related and similar to other kestrels (*Falco* spp.), the little-known, Sahel-endemic Fox Kestrel (*Falco alopes*) seems to exhibit some unusual morphological traits and behaviors. Reports have been particularly contradictory concerning the hovering ability of this species. For example, “Is misnamed ‘kestrel’ as has few kestrel-like habits. Normally, in open country perches on trees, and catches prey by short swoop from perch to ground; does not hover. In general behaviour more a long-winged, long-tailed falcon than kestrel.” (Brown et al. 1982, *The birds of Africa*, Vol. 1, Academic Press, London, U.K.), or “. . . long narrow pointed wings and, for falcon, exceptionally long graduated tail giving atypical kestrel shape, more like long-tailed hobby” and “Rarely, if ever, hovers” (Ferguson-Lees and Christie 2001, *Raptors of the world*, A&C Black, London, U.K.). Another description is “Long, broad wings and graduated tail suggest buoyant flight and good hovering ability” (del Hoyo, J., A. Elliott, and J. Sargatal [Eds.]. 1994, *Handbook of the birds of the world*. Vol. 2. Lynx Edicions, Barcelona, Spain). A buoyant, slow flight mode was previously hypothesized by Cade (1982, *The falcons of the world*, Cornell Univ. Press, Ithaca, NY, U.S.A.), who added that “certainly more observations are needed before one can be sure about all of its hunting and flying characteristics.”

Hovering occurs in distantly-related raptors, including in some Falconidae and some Accipitridae, certainly as a result of convergent evolution. However, hovering has been used as a behavioral trait indicating phylogeny (Boyce and White 1987, Pages 1–21 in D.M. Bird and R. Bowman [Eds.], *The ancestral kestrel*, *J. Raptor Res.* Rep. No. 6). Specifically, hovering is thought to be a derived trait. Thus, Boyce and White (1987) suggested that the Fox Kestrel was a primitive kestrel based on the reported absence of hovering.

On 16–17 August 2001, during a visit to the Mandara Mountains, Cameroon, I observed a pair of these falcons hunting over pastureland near the village of Roumsiki (1100 masl). During my observations, performed in late afternoon and early morning, the sky was clear and the falcons’ activity seemed stimulated by a light breeze on the grassy slopes. The kestrels were actively soaring and gliding. The latter flight behavior seemed much slower and steadier than that of a Eurasian Kestrel (*Falco tinnunculus*). The flight of the Fox Kestrels gave the appearance of miniature Lammergeiers (*Gypaetus barbatus*). After hanging in the wind at 5–10 m above the ground (stationing), the falcons were able to hold their position with minimal, compensative movements of the wings and tail. Occasionally, the falcons beat their wings obviously (hovering). Once prey was located, the kestrels descended at an angle, slowly and continuously (with no diving or descent hesitations). The birds did not stay on the ground, but the prey (probably orthopterans) was consumed in the air.

I videotaped flight sequences of Fox Kestrels for later comparison with Eurasian Kestrels under similar conditions as far as weather, terrain, and prey involved (orthopterans) are concerned. A 5-min hunting sequence of one Fox Kestrel gave the following results: 4 stationings in the air, 2 with and 2 without hovering; 6 hovering bouts, with a mean of 2.2 ± 1.2 (SE) wing beats/bout; and 4 descents to the ground. Filming of a Eurasian Kestrel in Italy during a 5-min period revealed the following results: 11 stationing bouts, 8 with and 3 without hovering; 15 hovering bouts, with a mean of 7.3 ± 9.3 (SE) wing beats/bout; and 2 descents to the ground. Analysis of direct flights revealed similar wing beat frequencies for both Fox and Eurasian kestrels (near 6/sec), and the positions of wings and tails during soaring and gliding were also similar.

Although Africa may well have been an important site for kestrel radiation, the suggestion that the Fox Kestrel might be the most primitive of the typical kestrels (Boyce and White 1987) seems unlikely. Provided that hovering has some phylogenetic value, finding it infrequent in a species may suggest either the incipient or the reduced trait. Coupling the variation of hovering among falcon species with that of unrelated traits may solve this problem of evolutionary direction as well as help clarify the relationships of falcons. In adulthood, the Fox Kestrel has an unusually pale eye similar to the Greater Kestrel (*Falco rupicoloides*), its probable closest relative (Olsen et al. 1989, *Emu* 89:193–203). From photographs (Kemp and Kemp 1998, *Sasol birds of prey of Africa and its islands*, New Holland, London, U.K.); it seems that the juvenile Fox Kestrel has paler eyes (different than in adults) than the juvenile Greater Kestrel. The latter has decidedly dark eyes like most falcons and all the Old World falconets (often considered primitive falcons) at any age. For this reason the Greater Kestrel, with more *tinnunculus*-like proportions and flight behavior (more frequent hovering), might be a transitional form from the Eurasian to the Fox Kestrel rather than the reverse. At first sight, the extremely developed wings and tail of the Fox Kestrel may suggest the Eleonora’s Falcon

(*Falco eleonora*), also a candidate for an ancestral falcon (Olsen et al. 1989). However, the wings and tail of the Fox Kestrel have a narrower base, as if resulting from a distal enlargement of gracile structures of *tinnunculus*-like ancestors. Thus, the Fox Kestrel may be less an atypical kestrel than usually assumed. Also, I suggest that the infrequent hovering of Fox Kestrels can be explained. This kestrel may have departed from more *tinnunculus*-like birds through specialization to inexpensive, slow flight for hunting small, scattered, and not very mobile prey in dry savannahs. The suggested resemblance, size apart, with the Lammergeier may represent convergent evolution toward the ability to remain on the wing for long periods in order to hunt broken terrain.

I thank J.C. Bednarz, W.S. Clark, and S.K. Sherrod for useful suggestions.—**Tiziano Londei, Dipartimento di Biologia, Università degli Studi, Via Celoria 26, 20133 Milano, Italy; e-mail address: londeit@tin.it**

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PROBABLE BREEDING OF SHORT-EARED OWLS IN SOUTHERN WEST VIRGINIA

During spring and summer of 2001, we observed adult and juvenile Short-eared Owls (*Asio flammeus*) frequenting grassland habitats of three reclaimed mine sites in Logan, Fayette, Kanawha, and Boone counties in southern West Virginia. This species has been previously reported as an uncommon migrant or winter visitant in West Virginia (Hall 1983, West Virginia birds, Special Publication Carnegie Museum of Natural History No. 7, Pittsburgh, PA U.S.A.); however, there are no breeding or nesting records in the state (Buckelew and Hall 1994, The West Virginia breeding bird atlas. Univ. of Pittsburgh Press, Pittsburgh, PA U.S.A., Holt and Leasure 1993, in A. Poole and F. Gill [Eds], The birds of North America No. 62, The Academy of Natural Sciences, Philadelphia, PA U.S.A.). Our sightings lend support to the idea that Short-eared Owls are opportunistic and will colonize areas when the conditions are suitable.

Eight different adult female and three different adult male Short-eared Owls were observed multiple times on or near reclaimed grassland areas. They were identified as different individuals based on their repeated occurrence in specific areas of each mine. Sex determination was based on observed plumage pattern differences (Sibley 2000, Alfred A. Knopf, Inc. New York, NY U.S.A.). Observation dates ranged from 14 March–13 July 2001, with sightings occurring between 0545 and 1130 H, and 1630 and 2000 H EST. The owls were observed in a suite of behavioral contexts. Most were observed flying low over grassland habitat actively foraging. One individual male was observed in an acrobatic aerial display with a male Northern Harrier (*Circus cyaneus*). A very vocal female was observed on the ground consuming an unidentified mammalian prey item. Several individuals were observed perched on large rocks.

Two juvenile Short-eared Owls were observed flying low over grassland areas on two separate mine sites in Logan and Boone counties on 11 June 2001 and 19 June 2001, respectively. On each occasion, juveniles were closely accompanied by an adult female. In one case, the juvenile was following a female that was hunting and capturing prey. Juvenile plumage patterns were similar to the adults with more black on the facial disks and tawny feather tips. The time of year that the juveniles were present and the lack of suitable habitat elsewhere in this region, strongly suggests that these individuals fledged from nests on the mine sites. Typical breeding dates for this species range from mid-April to June in most years (Mikkola 1983, *Br. Birds* 65:453–460).

Short-eared Owls prefer to forage and nest in open habitats such as old fields, hay meadows, pastures, prairies, dunes, and marshes (Johnsgard 1988, Smithsonian Institution Press, Washington, DC U.S.A.). Mountaintop mining valley fill (MTMVF) practices in West Virginia convert large areas of mature hardwood forest to early successional habitats consisting of low- to medium-height grassland plant communities. The three reclaimed MTMVF mine sites on which owls occurred included expansive networks of contoured grassland habitat (ca. 1600–2000 ha at each site) that ranged in age from 5–19 yr old. Reclaimed sites are dominated by a mixture of grasses and forbs (native and non-native) with scattered shrub/seedlings of autumn olive (*Elaeagnus umbellata*), black locust (*Robinia pseudoacacia*), and European black alder (*Alnus glutinosa*). These areas support dense small mammal populations that include white-footed mice (*Peromyscus leucopus*), deer mice (*Peromyscus maniculatus*), southern bog lemmings (*Synaptomys cooperi*), and meadow voles (*Microtus pennsylvanicus*) (Chamblin 2002, M.S. thesis, West Virginia University). These species along with an abundant grassland bird assemblage dominated by Grasshopper Sparrows (*Ammodramus savannarum*), Eastern Meadowlarks (*Sturnella magna*), Horned Larks (*Eremophila alpestris*), and Killdeer (*Charadrius vociferus*) apparently provide an adequate prey base for Short-eared Owls on these sites.

The North American breeding distribution of Short-eared Owls ranges from western Alaska east through Canada to Newfoundland, south to central California, and east across the north-central states to New Jersey (Johnsgard 1988). Holt and Leasure (1993) indicate that this species occurs year-round north and west of West Virginia while northern breeding populations are migratory. Our observations and accounts from others suggest that this species may be expanding its range along the southern edge of the previously reported North American breeding range. Several accounts have confirmed Short-eared Owl presence and breeding on coastal grassland habitats in Virginia, Maryland, and North Carolina (Ilf 2001, *N. Am. Birds* 55:284–287). Besides West Virginia, Short-eared Owl breeding also has been documented on reclaimed mine sites in Kentucky (Stamm and Clay 1989, *Kentucky Warbler* 65:75–76); however, breeding populations appear to be restricted to a few larger reclaimed areas (Palmer-Ball et al. 1990, *Kentucky Warbler* 66:73–80). This species exhibits some degree of nomadism with fairly long-distance movements by juveniles and adults (Clark 1975, *Wildl. Monogr.* 47:1–67, Cramp 1985, Oxford Univ. Press, Oxford, UK, Mikkola 1983, *Br. Birds* 65:453–460). Such behavior undoubtedly contributes to the ability of Short-eared Owls to find and colonize the newly-created grassland habitats in eastern states, allowing an expansion of the breeding range. This range expansion may be temporary, however, after succession renders these sites unsuitable for Short-eared Owls.—**Frank K. Ammer and Petra Bohall Wood, West Virginia Cooperative Fish and Wildlife Research Unit, BRD/USGS, and Division of Forestry, West Virginia University, P.O. Box 6125, Morgantown, WV 26506 U.S.A.; e-mail address: fammer@wvu.edu**

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ENDANGERED EGYPTIAN VULTURE (*NEOPHRON PERCNOPTERUS*) ENTANGLED IN A POWER LINE GROUND-WIRE STABILIZER

Avian mortality is one of the highest environmental costs of power lines all around the world. Research has widely demonstrated the killing of thousands of birds in some regions, and power-line mortality has contributed to declines in some populations of rare species (see review in Ferrer, M. and G.F.E. Janss 1999, *Birds and power lines*. Ed Quercus, Madrid). Mortality on power lines is traditionally associated with two types of accidents: electrocution and collision (Janss, G.F.E. 2000, *Biol. Conserv.* 95:353–359). Electrocution occurs when the bird touches two wires or, more frequently, a wire and the grounded metallic pylon; in addition, collisions with overhead wires usually take place when visibility is low (at night or in foggy weather) and species involved are usually flocking birds, such as ducks or gulls (Hass, D. 1980, *Ecol. of Birds* 2:117–157; Avian Power Line Interaction Committee [APLIC] 1996, *Suggested practices for raptor protection on power lines: the state of the art 1996*, Edison Electric Institute and Raptor Research Foundation, Washington, DC U.S.A.; Ferrer et al. 1991, *J. Field Ornithol.* 62:181–190).

Here we describe a new type of accident in power lines, entanglement in power line ground-wire stabilizer. This has been suggested before: see Olendorff et al. 1981, *Suggested practices for raptor protection on power lines: the state of the art 1981*, *J. Raptor Res. Rep.* 4:1–111. We observed this type of entanglement in a 66 kw transmission line, property of Empresa Nacional de Electricidad, Sociedad Anónima (ENDESA), crossing the island of Fuerteventura (Canary archipelago, Spain). On 10 November 2000, at dusk, we found a subadult Egyptian Vulture (*Neophron percnopterus*) with its right talon hooked up on a ground-wire stabilizer placed on one side of the power pole (Fig. 1). This individual probably perched on the stabilizer, as it is frequently observed among roosting individuals (see below). It could have caught its right talon in the lower hook-shaped structure, preventing escape. We rescued the bird the next morning; it was exhausted but still alive. Its ankle joint was seriously damaged. Consequently, it was necessary to amputate its talon. The bird was a 3-yr-old female. It had been captured using a cannon net in September 2000 as part of a population monitoring research program. The metallic ring on its right tarsus probably exacerbated the damage, as it hung from the stabilizer.

Egyptian Vultures in Fuerteventura usually roost along the 30 km on this power line year round; up to 125 individuals have been observed at one time with a maximum of 13 birds/pylon; ca. 96% of the total population on the island (Donázar et al. 2002, *Biol. Conserv.* 107:89–97). Electrocutions and collisions have been reported on the island, affecting Egyptian Vultures and other endemic and endangered avian species (Lorenzo, J.A. 1995, *Ecología* 9:403–



Figure. 1. The immature female Egyptian Vulture entangled in the stabilizer.

407; Lorenzo et al. 1997, *Vieraea* 26:1–10). Hooking in stabilizers was never observed before, although vultures often roost in these structures (36.9% of the individuals roosting in pylons perch on stabilizers, $N = 384$, unpubl. data). It cannot be discarded, however, that some injured birds may escape after entangling. In fact, during 2001 we have observed four free-ranging individuals with fractured legs; another bird was missing a leg. Survival probabilities of these individuals would be consequently reduced. As the use of leg paddle traps is unknown on the island it seems reasonable to examine the role that entanglement may have in the occurrence of leg injuries. Finally, the Canarian population of this species is endemic to the archipelago (*N. p. majorensis*; Donázar, et al. 2002, *J. Raptor Res.* 36:17–23) and is extremely endangered (26 breeding pairs in 2001, Donázar et al. 2002). Casualties on power lines has caused the mortality of 14% of the extant Canarian Egyptian Vultures (16 cases of electrocution, 1 case of collision, and 1 case of entanglement) and represent an important risk to this population. This problem should also be considered in the design of power lines potentially used by large roosting birds in other regions of the world.

We would like to thank the Consejería de Medio Ambiente del Cabildo Insular de Fuerteventura and the Project REN 2000-1556 GLO that funded this research. We thank the staff of UNELCO-ENDESA for the assistance during the rescue of the bird. We also wish to thank Juan J. Negro, José A. Donázar, Fernando Hiraldo, Miguel Ferrer, and Robert M. Lehman for reviewing early drafts of this letter.—**Laura Gangoso and César J. Palacios, Department of Applied Biology, Estación Biológica de Doñana, C.S.I.C., Pabellón del Perú, Avda M^a Luisa s/n, 41013 Sevilla, Spain; e-mail address: laurag@ebd.csic.es**

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BARRED FOREST-FALCON (*MICRASTUR RUFICOLLIS*) PREDATION ON A HUMMINGBIRD

Hummingbirds are widely regarded as having few predators away from the nest. However, incidental attacks upon hummingbirds by a number of bird species have been reported and may exert at least a moderate selective pressure. Wright (1962, *Auk* 79:112) reported a Baltimore Oriole (*Icterus galbula*) killing a Ruby-throated Hummingbird (*Ar-*

chlochus colubris). In addition, Brown-crested (*Myiarchus tyrannulus*) (Snider 1971, *Am. Birds* 25:780–784; Gamboa 1977, *Auk* 94:157–158) and Gray flycatchers (*Empidonax wrightii*) (Seutin and Apanius 1995, *Wilson Bull.* 107:565–567) have been observed to prey on hummingbirds. A Greater Roadrunner (*Geococcyx californianus*) was observed catching hummingbirds at a feeder in Arizona (Spofford 1976, *Condor* 78:142). In contrast, it appears only a few raptors prey on hummingbirds. This might be because hummingbirds offer such small energy rewards for a large predator (Seutin and Apanius 1995). Nevertheless, Merlins (*Falco columbarius*) have been observed chasing and catching hummingbirds successfully (Sprat 1927, *Condor* 29:71–72; Lowery 1938, *Auk* 55:280; Mayr 1966, *Auk* 83:664), and both Mayr (1966) and Balgooyen (1976, *Univ. Calif. Publ. Zool.* 103:1–83) observed American Kestrels (*Falco sparverius*) catching hummingbirds in the air. Also, Peeters (1963, *Wilson Bull.* 75:274) observed a Sharp-shinned Hawk (*Accipiter striatus*) catch an Anna's Hummingbird (*Calypte anna*).

Only two small raptors, Bat Falcons (*Falco ruficularis*) and Tiny Hawks (*Accipiter superciliosus*), take large numbers of hummingbirds (Beebe 1950, *Zoologica* 35:69–86; Stiles 1978, *Auk* 95:550–553). Beebe (1950) estimated that 16% of a Bat Falcon's diet consisted of hummingbirds, but he did not believe that these falcons had developed a specific hunting technique to catch hummingbirds. In contrast, Stiles (1978) suggested that Tiny Hawks are hummingbird specialists that employ three different techniques to catch hummingbirds. These tactics include still-hunting, waiting in ambush by a hummingbird's territorial perch, and flying rapidly between several territorial hummingbird perches. We report here a capture of a hummingbird by a Barred Forest-Falcon (*Micrastur ruficollis*). This forest-falcon used a tactic not reported before.

The capture occurred at Loma Linda Botanical Gardens (00°01.62'S, 078°40.55'W) at ca. 2065 m elevation along the Old Nono-Mindo Road about 6 km west of village of Tandayapa in northwest Ecuador. The gardens comprise 30 ha and include abandoned pasture, secondary forest, and primary cloud forest. The capture occurred in part of abandoned cattle pasture where the first author maintains 30 hummingbird feeders, which daily attract 12–15 hummingbird species. The most common species at the Loma Linda feeders are: Green (*Colibri thalassinus*) and Sparkling violet-ears (*C. coruscans*), Western (*Chlorostilbon mellisugus*) and Andean emeralds (*Amazilia franciae*), Booted Racket-tails (*Ocreatus underwoodii*), White-bellied (*Acestrura mulsanti*) and Purple-throated woodstars (*Philodice mitchellii*), and Buff-tailed Coronets (*Boissonneaua flavescens*).

Most of the feeders at Loma Linda have an attendant (aggressive) hummingbird that keeps other hummingbirds away from their feeder. Although different hummingbird species guard feeders, a hierarchy of possession is evident. Western Emeralds and Booted Racket-tails guard their feeders, but rarely attempt to evict larger hummingbirds; the two woodstar species do not display territorial behavior. The most aggressive and successful at defending feeders are Sparkling Violet-ears, which is the largest and most common species at Loma Linda. Typically, Sparkling Violet-ears chase other hummingbirds for 3–10 m, before returning to a favored perch. In mid-June 2000, a Barred Forest-Falcon flew into a nearby tree at the edge of the abandoned pasture about 15 m from one of the guarded feeders and landed about 15–18 m up in the tree. The Barred Forest-Falcon stayed in the tree for 5–10 min, watching the hummingbirds before attacking one of the birds guarding a feeder. Rather than giving chase to the Sparkling Violet-ear, the raptor flew straight to the perch used by the hummingbird. As the violet-ear returned to its perch after chasing away another hummingbird from its feeder, the falcon intercepted and captured the hummingbird as it landed on its perch. The Barred Forest-Falcon then flew into an inga tree (*Inga edulis*), plucked out several of the hummingbird's breast feathers, and fed. The hummingbird did not die immediately, but continued to flutter. After several minutes the forest-falcon flew off with the remains of the hummingbird.

The attack by the forest-falcon suggests the bird anticipated the return of the hummingbird to its favored perch. Moreover, the kind of territoriality displayed by Sparkling Violet-ears and their habit of remaining perched in exposed, prominent locations adjacent to each feeder seems to make them especially vulnerable to interception. This intercept strategy appears nearly identical to the ambush strategy employed by Tiny Hawks (Stiles 1978), the only difference is that the Barred Forest-Falcon did not move in close (2–3 m) to the perch (ambush strategy), but rather began its attack 20 m away. The behavior of the Sparkling Violet-ears suggests that all species of territorial hummingbirds might be susceptible to this type of attack, especially where large concentrations of feeders and hummingbirds occur.

We thank Georgia Southern University for funding travel for Mark Welford and C. Ray Chandler for his comments on a first draft.—**Tony Nunnery, Lomo Linda, Tandayapa Valley, Pichincha Province, Ecuador, and Mark R. Welford (corresponding author), Dept. of Geology and Geography, Georgia Southern University, Statesboro, GA 30460-8149 U.S.A.; e-mail address: mwelfgeog@gsvms2.cc.gasou.edu**

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BOOK REVIEWS

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Raptors of the World. By James Ferguson-Lees and David A. Christie. 2001. Houghton Mifflin, Boston, MA. 992 pp., 4 tables, 60 figures, 112 color plates, numerous range maps. ISBN 0-618-12762-3. Cloth, \$60.00—For any birdwatcher enjoying a successful trip to a new country, few events are more frustrating than glimpsing an unknown raptor as it flies swiftly out of sight, never to be seen again nor identified with certainty. Diurnal raptors, whether perched or flying, are often difficult birds to identify, even for experts. Fortunately, many regional field guides serve as excellent resources for raptor identification, yet no one has dared assemble a guide to all of the globe's diurnal birds of prey. *Raptors of the World* aspires to accomplish this ambitious goal.

This enormous field guide is organized into a list of species, several chapters of natural history and identification information, color plates, detailed species descriptions, a bibliography, and an index. Following the species list, a brief Introduction informs the reader that 313 species within four orders—Ciconiiformes (New World vultures), Accipitriformes, Falconiformes, and Sagittariiformes—are recognized and treated within the text, departing from the standard treatment that places all diurnal raptors in the order Falconiformes. The next chapter, aptly titled *Using This Book*, describes the general content and format of the color plates, distribution maps (in three colors distinguishing migratory pathways and seasonal ranges), and condensed caption texts that accompany the plates and maps. This chapter also contains a brief overview of topics included in each of the species accounts.

Several chapters devoted to general identification cover bird topography, morphological measurements, and sex and age differences. The three pages of line drawings within the Raptor Topography chapter are well done and useful. Reversed sexual size dimorphism is treated in some detail and then related to identification, as are wingspan and total length measurements in the next chapter.

Unlike traditional field guides on raptors, the book also includes lengthy chapters on migration, molt, and anatomy.

A brief treatment of taxonomy and nomenclature completes the final two introductory chapters. The authors acknowledge that not all changes in taxonomy stemming from recent DNA-DNA hybridization studies could be incorporated into the book because of time constraints related to publication. Thus, their classification remains conventional aside from the aforementioned division of the Falconiformes into four orders. Use of English names generally follows that proposed by the British Ornithologists' Union's Records Committee, but North American buteos are still labeled "hawks" rather than "buzzards," and "vulture" refers to taxonomically unrelated Old and New World species. In sum, the introductory materials encompass 79 pages.

The most important parts of any field guide are the color plates, range maps, and species accounts. *Raptors of the World* contains an astounding 112 color plates that show 2115 individual birds, more than half of which are depicted in flight. The three artists—Kim Franklin, David Mead, and Philip Burton—illustrated adult and juvenile plumages of perched and flying individuals of each species. Most plates depict three species, but some wide-ranging species with variable plumage command several pages of artwork.

The first three color plates categorize raptors by general size (large, medium, small) and distribution (New and/or Old World) to help the unacquainted birder narrow the bewildering array of species down to genus. Plates of species follow, each identically structured. Preceding the English name is a number corresponding to the list of species at the beginning of the book. Next are the scientific name and a page number referencing the species account in the main text followed by data on total length, wingspan, and tail length in centimeters (with the midpoint of the range in inches). Size of the male in proportion to the female is given as a percentage. Opposite each species plate is a distribution map and condensed text describing overall appearance, flight characteristics, and aspects of general biology that may aid in

identification. A list of similar-looking species is referenced by plate number.

Extensive identification and natural history information appear within the Systematics section, a chapter that spans a colossal 622 pages. At the top of each species account are repeated the English and scientific names and plate numbers. A large map in varying shades of gray accompanies the text and usually provides the same information illustrated in the color maps. Distribution, behavior (e.g., migratory, breeding), habitat, food, and worldwide population are reviewed. Estimated size of the global population is categorized by numbers (1–7) that represent orders of magnitude (e.g., 5 = 10 001–100 000 individuals). The largest subsection describes field characteristics, which are grouped by age, sex, geography, and size for both flying and perched birds. Characteristics helping to separate similar-looking species are also furnished. A list of references, abbreviated by author and year, completes each species description.

The authors and artists completed a formidable project and succeeded in producing an attractive and useful book. However, preparing such a tome means it will be out-of-date and incomplete the moment it is published, mostly because of publication deadlines rather than the authors' inattention to recent advances. Given that the primary purpose of the book is to serve as a *field* guide, I found most of the information in the opening chapters on natural history unrelated to identification and therefore unnecessary. Much of it was also dated, despite the 55-page bibliography. For example, the authors discuss reversed sexual size dimorphism at length, but an important paper on phylogenetic effects is not mentioned. The ability of raptors to see within the ultraviolet range also eludes discussion, and the role of ultraviolet plumage characteristics is stated as unknown, but a wealth of literature chronicles these aspects of natural history.

I studied carefully the plates and text of all North American species, the group with which I am most familiar, and found several significant errors. The most noticeable and troublesome shortcomings concerned the accuracy of the distribution maps, which were produced by relying on previous books. For example, maps for Mississippi Kite (*Ictinia mississippiensis*), Common Black-Hawk (*Buteogallus anthracinus*), Crested Caracara (*Caracara plancus*), and Peregrine Falcon (*Falco peregrinus*) either were poorly done or did not indicate

large areas where these species occur. Although the authors ended literature reviews for most species in the mid-1990s, they should have examined some contemporary sources (e.g., the *Birds of North America* series) to update and verify distributions. A few of the plates also contained errors. For example, age-specific plumages of the Bald Eagle (*Haliaeetus leucocephalus*), leg color of the Turkey Vulture (*Cathartes aura*), and color of the tarsus feathers of the Golden Eagle (*Aquila chrysaetos*) were incorrect. These errors were unsettling because North American raptors are well known compared with species inhabiting remote parts of the globe. The style of the three artists also differed significantly, which, in my opinion, detracted from the consistency in plate presentation.

Despite these errors, *Raptors of the World* deserves a place in the libraries of globetrotting birders who have a deep interest in and appreciation for raptors, simply because it provides so much information in an accessible format. The sheer bulk of this "field guide," however, means that it will rest on a coffee table or bookshelf far more often than inside a backpack. Ornithologists who venture abroad will also find it a handy reference when used in combination with regional field guides. Finally, public and academic libraries certainly should include this book in their ornithological collections to complement texts that focus on raptor natural history.—**Marco Restani, Department of Biological Sciences, St. Cloud State University, St. Cloud, MN 56301 U.S.A.**

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The Spanish Imperial Eagle. By Miguel Ferrer. 2001. Lynx Edicions, Barcelona, Spain. 224 pp., 36 tables, 58 figures, numerous black-and-white photographs. ISBN 84-87334-34-2. Cloth, \$28.00—If you are interested in the behavior, population ecology, or conservation of raptors, then you must become familiar with this book. It is a quick and informative (although sometimes tedious) read that introduces you to one of the rarest raptors in the

world. The chapters cover taxonomy, biometry and physiology; distribution and status; feeding; reproduction; the dependence period; dispersal; mortality; population dynamics; population genetics; and conservation. The author draws on his 20 years of experience with *Aquila adalberti* in Doñana National Park to develop these topics and put them into a larger ornithological perspective. His stated objectives are to (1) present all up-to-date information on the species and (2) provide a thought-provoking basis for the conservation of Spanish Imperial Eagles.

Ferrer accomplishes his first objective with a wonderful mix of photographs, line drawings, 36 data tables, and 57 graphs and maps. Methods and statistical analyses are summarized to aid your own interpretation of the results. As an indication of the types of analyses reported, consider these findings that I found most important: (1) blood chemistry (especially urea, a possible indicator of condition) varies with age, hatching date, and many aspects of behavior; (2) using latex gloves while handling eagle chicks reduces staphylococcus infection; (3) apparent increases in reproductive success with breeder age are due to territory quality, not some inherent property of the pair; (4) territory intrusion by immatures correlates positively with territory quality; (5) most of population renewal is due to a few very productive pairs; (6) fledglings in poor nutritional condition remain in their natal territories longer and eventually dispersed shorter distances than fledglings in good condition; (7) wind direction correlates positively with dispersal direction; (8) occasional observations of subadults breeding, that we often dismiss as anecdotal, can in fact represent important population-stabilizing mechanisms (as numbers decline, age of reproduction declines, which increases population growth, which reduces breeding by subadults); and (9) conservation efforts that reduce juvenile and adult mortality are more likely to benefit the population than those aimed at augmenting reproduction.

The strongest part of the book reflects Ferrer's primary expertise in the dispersal and development of independence by juvenile eagles. The 50 pages devoted to these topics are strong and well argued. I particularly liked the conceptualization of these two poorly understood phases of most birds' lives. Dependence was divided into an early stage (fledgling to development of soaring flight) affected by the physical condition of the chick and

a later stage (soaring flight to independence) primarily affected by the physical condition of the parent. Likewise, juvenile dispersal was divided temporally into phases of "local dispersal," "first departure from the natal population," "exploration," "temporary settlement," and "return to the natal population." Such detailed investigation of mobile, wide-ranging birds with clear links between behavioral changes and important mechanisms is especially noteworthy.

I was troubled by the shallow depth of treatment other topics received. There was little quantitative assessment of habitat use or habitat needs of the species. Certainly this should have been a cornerstone of investigation for such a rare species. No rigorous assessment of range contraction was provided (maps are presented but are not tied to habitat changes, habitat quality, configuration, etc.). Despite a rather strong dependence hinted at between rabbits and eagles, no quantitative measures of prey were provided, and no attempts were made to link eagle population dynamics with those of their prey. The behavior of eagles is described, but there are no quantitative presentations of time budgets or relative importance of foraging styles. We are not given a complete picture of how this eagle spends a typical day. Sociality is not mentioned. Home-range dynamics and use of space by breeders are dealt with only superficially. The chapter on population genetics is only five pages long.

The certainty of some findings is also presented a bit too strongly for my taste. For example, Ferrer claims that males and females can be distinguished "with certainty" by morphology, yet he shows that the sexes overlap in all physical characteristics. He also asserts that urea concentrations in the blood indicate the nutritional state of individual birds, but he does not discuss the potential problems with drawing this conclusion. I am not an expert in this area and found the correlations between urea and behavior exciting. However, upon discussing this with several more-knowledgeable colleagues, I found that although urea concentration does reflect protein catabolism and/or degradation that can come from the individual or from its diet, it is tricky with a carnivorous bird to confidently identify the source of variation in the values. Ferrer should have discussed these uncertainties more openly.

The biggest failure of this book is that Ferrer makes no attempt to put the work on Spanish Im-

perial Eagles into the broader context of behavior, ecology, and endangered species conservation. The literature cited is badly dated and heavily skewed toward raptors. Few articles (other than the author's own) beyond 1990 are cited. This may not bother the ardent raptor biologist, but the importance of this story for avian conservation and ecology in general will be lessened by this shortcoming.

Lynx Edicions has attractively packaged this book and done a splendid job reproducing the figures and photographs. However, they have done a poor job in proofreading and finalizing the text. On average, one typo occurs on every page, as do misalignments that confuse some of the tables. Many of the errors stem from translation into English, for which the author and publishers have my sympathy. However, a quick proofing by someone proficient in English would have cleaned up 90% of these issues.

Despite these drawbacks, Ferrer has provided a thought-provoking basis for conservation of Spanish Imperial Eagles. However, I am not convinced that he has provided an action-provoking basis, and in my opinion that is what is really needed. He

has armed those interested in raptor conservation with relevant biological information and shown clearly that simply reducing the risk of electrocution will likely benefit the species. However, he has not given us any insights into the Spanish political system nor any indication of the likelihood that the Spanish people will embrace the changes needed to save this species. We all know that biology is only one side of the conservation equation; social, economic, and political considerations will always be important. As biologists, we must understand all of these dimensions to effectively enter into action-provoking discussions with policy makers, managers, and planners.

In summary, this is a classic case study of an imperiled raptor. Those interested in large raptors, especially eagles, will find it required reading. Those interested in conservation will do well to study the last chapter. Behavioral ecologists and population biologists will find important data to relate to their own studies. It belongs on library shelves, but not on the shelf of every ornithologist.—**John M. Marzluff, College of Forest Resources, University of Washington, Seattle, WA 98195 U.S.A.**

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The Raptor Research Foundation, Inc. 2003 annual meeting will be held on 2–6 September 2003 in Fairbanks, Alaska. For information about the meeting see the following website: <http://www.alaskabird.org> or contact Nancy DeWitt (birds@alaskabird.org).

Persons interested in predatory birds are invited to join The Raptor Research Foundation, Inc. Send requests for information concerning membership, subscriptions, special publications, or change of address to OSNA, P.O. Box 1897, Lawrence, KS 66044-8897, U.S.A.

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RAPTOR RESEARCH FOUNDATION, INC., AWARDS

Lifetime Achievement Awards

The **Tom Cade Award** recognizes an individual who has made significant advances in the area of captive propagation and reintroduction of raptors. Nomination packets can be submitted at any time. Contact: **Brian Walton**, Predatory Bird Research Group, Long Marine Laboratory, University of California, Santa Cruz, CA 95064 U.S.A.; tel. 408-459-2466; e-mail: walton@cats.ucsc.edu.

The **Fran and Frederick Hamerstrom Award** recognizes an individual who has contributed significantly to the understanding of raptor ecology and natural history. Nomination packets can be submitted at any time. Contact: **Dr. Clint Boal**, Texas Cooperative Fish and Wildlife Research Unit, BRD/USGS, Texas Tech University, 15th Street & Boston, Ag Science Bldg., Room 218, Lubbock TX 79409-2120 U.S.A.; tel. (806) 742-2851; e-mail: cboal@ttacs.ttu.edu.

Student Recognition and Travel Assistance Awards

The **James R. Koplín Travel Award** is given to a student who is the senior author and presenter of a paper or poster to be presented at the RRF annual meeting for which travel funds are requested. Contact: **Dr. Patricia A. Hall**, 5937 E. Abbey Rd. Flagstaff, AZ 86004 U.S.A.; tel. 520-526-6222; e-mail: pah@spruce.for.nau.edu. Application Deadline: due date for meeting abstract.

The **William C. Andersen Memorial Award** is given to the students who are senior authors and presenters of the best student oral and poster presentation at the annual RRF meeting. Contact: **Laurie Goodrich**, Hawk Mountain Sanctuary, 1700 Hawk Mountain Road, Kempton, PA 19529 U.S.A.; tel. 610-756-6961; email: goodrich@hawkmountain.org. Application Deadline: due date for meeting abstract; no special application is needed.

Grants

For each of the following grants, complete applications must be submitted to the contact person indicated by **15 February**. Recipients will be notified by 15 April.

The **Dean Amadon Grant** for \$200–400 is designed to assist persons working in the area of distribution and systematics (taxonomy) of raptors. Contact: **Dr. Carole Griffiths**, 251 Martling Ave., Tarrytown, NY 10591 U.S.A.; tel. 914-631-2911; e-mail: cgriff@liu.edu.

The **Stephen R. Tully Memorial Grant** for \$500 is given to support research, management, and conservation of raptors, especially to students and amateurs with limited access to alternative funding. Contact: **Dr. Kim Titus**, Alaska Department of Fish and Game, Division of Wildlife Conservation, P.O. Box 240020, Douglas, AK 99824 U.S.A.; e-mail: kimt@fishgame.state.ak.us.

The **Leslie Brown Memorial Grant** for up to \$1,000 to support research and/or dissemination of information on birds of prey, especially to proposals concerning African raptors. Contact: **Dr. Jeffrey L. Lincer**, 9251 Golondrina Dr., La Mesa, CA 91941 U.S.A.; e-mail: jefflincer@tns.net.