(ISSN 0892-1016)

THE JOURNAL OF RAPTOR RESEARCH

VOLUME 32 SEPTEMBER 1998 NUMBER 3

CONTENTS

BREEDING DISTRIBUTION AND NEST-SITE HABITAT OF NORTHERN GOSHAWKS IN WISCONSIN. Robert N. Rosenfield, John Bielefeldt, Dale R. Trexel and Thomas C.J. Doolittle.. 189

SOLITARY AND SOCIAL HUNTING IN PALE CHANTING GOSHAWK (*MELIERAX CANORUS*) FAMILIES: WHY USE BOTH STRATEGIES? Gerard Malan.......... 195

FOREST MANAGEMENT EFFECTS ON NESTING HABITAT SELECTED BY EURASIAN BLACK VULTURES (*AEGYPIUS MONACHUS*) IN CENTRAL SPAIN. Juan A. Fargallo, Guillermo Blanco and Eduardo Soto-Largo.......... 202

SELECTION OF SETTLEMENT AREAS BY JUVENILE BONELLI'S EAGLE IN CATALONIA. Santi Mañosa, Joan Real and Jordi Codina.......... 208

WINTER FORAGING ECOLOGY OF BALD EAGLES ON A REGULATED RIVER IN SOUTHWEST IDAHO. Gregory S. Kaltenecker, Karen Steenhof, Marc J. Bechard and James C. Munger.......... 215

URBAN, SUBURBAN AND RURAL RED-TAILED HAWK NESTING HABITAT AND POPULATIONS IN SOUTHEAST WISCONSIN. William E. Stout, Raymond K. Anderson and Joseph M. Papp.......... 221

HIGHWAY MORTALITY OF BARN OWLS IN NORTHEASTERN FRANCE. Sylvie Massemin and Thierry Zorn.......... 229

THE EFFECT OF BURROW SITE USE ON THE REPRODUCTIVE SUCCESS OF A PARTIALLY MIGRATORY POPULATION OF WESTERN BURROWING OWLS (*SPEOTYTO CUNICULARIA HYPUGAEA*). Eugene S. Botelho and Patricia C. Arrowood.......... 233

BREEDING-SEASON FOOD HABITS OF BURROWING OWLS (*ATHENE CUNICULARIA*) IN SOUTHWESTERN DOMINICAN REPUBLIC. James W. Wiley.......... 241

SHORT COMMUNICATIONS

RATES OF OPEN-FIELD FORAGING BY THE MISSISSIPPI KITE (*ICTINIA MISSISSIPPIENSIS*). E. William Wischusen.......... 246

EVALUATION OF NECK-MOUNTED RADIO TRANSMITTERS FOR USE WITH JUVENILE OSPREYS. Lauren N. Gilson.......... 247

ORGANOCHLORINES AND MERCURY IN PEREGRINE FALCON EGGS FROM WESTERN NORTH CAROLINA. Tom Augspurger and Allen Boynton.......... 251

IMPORTANCE OF BIRDS AND POTENTIAL BIAS IN FOOD HABIT STUDIES OF MONTAGU'S HARRIERS (*CIRCUS PYGARGUS*) IN SOUTHEASTERN SPAIN. José A. Sánchez-Zapata and José F. Calvo.......... 254

PREY BROUGHT TO RED-SHOULDERED HAWK NESTS IN THE GEORGIA PIEDMONT. Doug L. Howell and Brian R. Chapman.......... 257

PEREGRINE FALCONS (*FALCO PEREGRINUS*) NEST IN A QUARRY AND ON HIGHWAY CUTBANKS IN ALASKA. Robert J. Ritchie, Terry J. Doyle and John M. Wright.......... 261

LICE (PHTHIRAPTERA: AMBLYCERA, ISCHNOCERA) OF RAPTORS IN HUNGARIAN ZOOS AND REHABILITATION CENTERS. Szabolcs Solt.......... 264

LETTERS.......... 267

THE JOURNAL OF RAPTOR RESEARCH

A QUARTERLY PUBLICATION OF THE RAPTOR RESEARCH FOUNDATION, INC.

VOL. 32 SEPT 1998 NO. 3

J. Raptor Res. 32(3):189–194
© 1998 The Raptor Research Foundation, Inc.

BREEDING DISTRIBUTION AND NEST-SITE HABITAT OF NORTHERN GOSHAWKS IN WISCONSIN

ROBERT N. ROSENFIELD
Department of Biology, University of Wisconsin, Stevens Point, WI 54481 U.S.A.

JOHN BIELEFELDT
Park Planning, Racine County Public Works Division, Sturtevant, WI 53177 U.S.A.

DALE R. TREXEL
Department of Ecology, Evolution, and Behavior, 100 Ecology Building, 1987 Upper Buford Circle, University of Minnesota, St. Paul, MN 55108 U.S.A.

THOMAS C. J. DOOLITTLE
Bad River Band of Lake Superior Chippewa, P.O. Box 39, Odanah, WI 54861 U.S.A.

ABSTRACT.—We found Northern Goshawks (*Accipiter gentilis*) nesting widely throughout the northern two-thirds of Wisconsin during 1996–97, with no evidence of range contraction as might be expected as one index of changing status if the state's breeding population were declining. During 1977–97, habitat was sampled on 0.04 ha circular plots at 37 goshawk nests, of which 78% were in deciduous trees, especially trembling aspen (*Populus tremuloides*). Mean nest-tree height, mean nest-tree diameter-at-breast-height (dbh), and mean tree density were 25 m, 41 cm, and 423 stems/ha, respectively. A comparison of these and 20 other habitat features at nest sites found by unbiased vs. potentially-biased methods failed to detect statistically significant differences between these two data sets. Goshawks nested in a broad array of forest types, including pine plantations and forest fragments in agriculturally-dominated landscapes.

KEY WORDS: *Northern Goshawk*; Accipiter gentilis; *nesting distribution*; *breeding range*; *nest-site habitat.*

Distribucion del habitat de anidación de *Accipiter gentilis* en Wisconsin

RESUMEN.—Encontramos a *Accipiter gentilis* anidando a travès de los dos tercios del norte de Wisconsin durante 1996–97, sin evidencias de que su rango de distribución disminuya como se podría esperar de su declinación en su población reproductiva. Durante 1977–97, se hicieron muestras de habitat en parcelas circulares de 0.04 ha en 37 nidos de azor de los cuales 78% se encontraban en árboles caducifolios (*Populus tremuloides*). La media de altura, la media de diámetro a la altura del pecho y la media de la densidad de los árboles fue de 25m, 41 cm y 423 troncos/ha respectivamente. Una comparación de estas y otras 20 características de habitat en los sitios de los nidos encontrados entre métodos sin sesgos y potencialmente sesgados no reportó diferencias estadísticamente significativas entre estos dos grupos de datos. Los azores anidaron en una amplia variedad de tipos de bosques incluyendo plantaciones de pinos y fragmentos de bosques en paisajes agrícolas.

[Traducción de César Márquez]

There is concern that populations of the Northern Goshawk (*Accipiter gentilis,* hereafter goshawk) may be declining in North America because of human-induced habitat alterations (Braun et al. 1996, Kennedy 1997, Squires and Reynolds 1997). There is a consequent need for baseline information on key population attributes that may, or may not, yield evidence of population declines (Block et al. 1994). Kennedy (1997) investigated the possibility of goshawk population declines in North America, in part through a literature review of its breeding distribution for evidence of range contractions. Because the bulk of studies she examined were geographically limited, she was unable to provide information from mid-continental regions. Moreover, the majority of studies on nest-site habitat in North America have focused on goshawks in the western U.S., where most investigations have used biased searching techniques to locate nests (Squires and Reynolds 1997). Apfelbaum and Seelbach (1983) have reported nest tree species at 22 goshawk nests in the midwestern U.S., but their tally included Pennsylvania as a supposedly midwestern state and did not associate nest-tree data with specific locales at state or sub-state levels. As with distributional data mentioned above, published information on nest-site habitat for the goshawk in mid-continental North America thus remains very limited.

Here, we show that the goshawk currently (1996–97) has a wide breeding distribution in Wisconsin with no sign of range contraction and describe nest-site habitat for 37 nests sampled from 1977–97, including a comparison of habitat features at goshawk nests found by unbiased vs. potentially biased methods.

STUDY AREA AND METHODS

We found goshawk nests in Wisconsin from 1996–97 by using three methods. First, we searched four quadrats, each about 3885 ha in size, that were objectively established (i.e., without past or present knowledge of forest seral stages or use of these sites by goshawks) within predominately wooded habitats in the northern third of Wisconsin. Second, we searched historic goshawk nesting areas and, third, we obtained nest-site information from nonproject personnel including staff of the Wisconsin Department of Natural Resources (WDNR) and other agencies, falconers, and others.

Nest-site habitat data were collected in 0.04 ha circular plots (11.3 m radius) centered on the nest tree, using procedures described by James and Shugart (1970) as modified by Titus and Mosher (1981). We sampled habitat at four occupied nests (i.e., eggs laid), one in each of the four quadrats during 1996–97, and at 33 occupied nests elsewhere in Wisconsin during 1977–97 (Fig. 1). Of these 37 geographically separate goshawk nest sites, 23 (62%) were sampled in the year of breeding, while four and seven other nest sites were sampled one or two years, respectively, following nesting. Habitat at three nest sites unaltered by logging or other human activities was sampled five to seven years after discovery of nests. Table 1 describes vegetation and physical measurements obtained at each nest site.

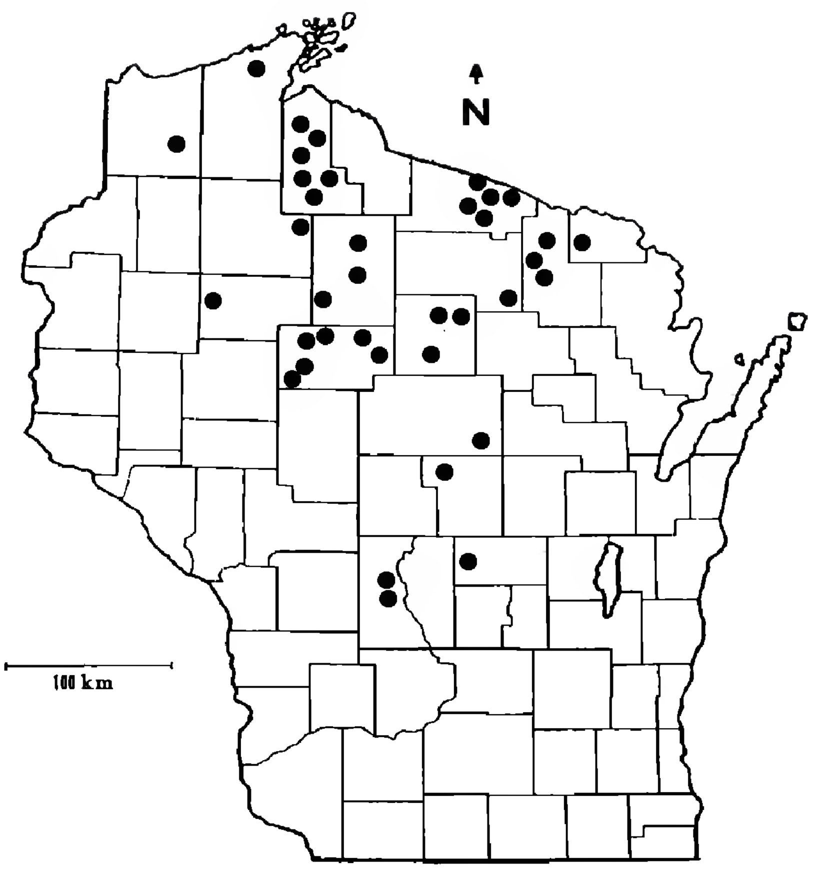

Figure 1. The distribution of Northern Goshawk nests (1977–97) in Wisconsin at which nest-site habitat data were collected.

In addition, we arbitrarily divided the nest-site habitat sample into those nests that we regarded as found by unbiased means ($N = 21$) and those found by potentially biased means ($N = 16$). We compared values of habitat features in these two categories on the premise that nests found by potentially biased searching techniques may not be representative of nest-site habitats used by goshawks (Siders and Kennedy 1996, Squires and Reynolds 1997). Nests found by unbiased means were characterized as those found on the quadrats ($N = 4$) or detected during activities other than searching for goshawks, such as songbird inventories, botanical surveys, recreational hiking, and other incidental discoveries, excluding cruising forests for timber ($N = 17$). Nests were classified as being found by potentially biased methods when searches for goshawk nests were conducted in habitat presumed suitable for nesting, such as mature, late seral northern hardwood-conifer forests in Wisconsin ($N = 9$) or when nests were found during timber cruising ($N = 7$), an activity that may not equally represent all potential nesting habitats (Hayward and Escano 1989). The majority of habitat variables did not exhibit normal distributions in Lilliefors Tests so nonparametric Mann-Whitney U tests were used to assess the potential significance of differences between

Table 1. Vegetational and physical features measured at Northern Goshawk nest sites in Wisconsin.

FEATURE	DESCRIPTION
Dist. to Water	Distance to nearest permanent water source (pacing or USGS 7.5′ quadrangle)
Dist. to Opening	Distance to nearest forest opening ≥5 ha in size (pacing or USGS 7.5′ quadrangle)
Nest Tree Height	Height of nest tree (Haga altimeter)
Nest Tree dbh	Diameter at breast height of nest tree
Nest Height	Height of nest (meter tape or Haga altimeter)
Nest Percent	(Nest height/Tree height) (100)
Degree Slope	Maximum slope by altimeter or clinometer
Tall Shrub Index	Index of tall shrubs <3 cm dbh and ≥shoulder height[a]
Low Shrub Index	Index of low shrubs <3 cm dbh between knee and shoulder height[a]
Under. Density	Number of understory trees ≥9 cm dbh per ha
Canopy Height	Mean height of five canopy trees in study plot (Haga altimeter)
Total Canopy	Percent of area over study plot occluded by overstory foliage[b]
Decid. Canopy	Percent of area over plot (not of total canopy) occluded by deciduous overstory foliage[b]
Conif. Canopy	Percent of area over plot occluded by evergreen overstory foliage[b]
Total Understory	Percent of area over plot occluded by understory foliage[b]
Decid. Understory	Percent of area over plot occluded by deciduous understory foliage[b]
Conif. Understory	Percent of area over plot occluded by coniferous understory foliage[b]
Total Ground	Percent of ground in plot covered by ground-layer foliage[b]
Decid. Ground	Percent of ground in plot covered by ground-layer deciduous foliage[b]
Conif. Ground	Percent of ground in plot covered by ground-layer coniferous foliage[b]
Tree Density	Number of canopy trees ≥9 cm dbh per ha
Basal Area	Basal area in m^2/ha of canopy trees
Mean dbh	Mean dbh of canopy trees in study plot

[a] Sum of stems on 4 plot radii.

[b] 40 ocular tube readings.

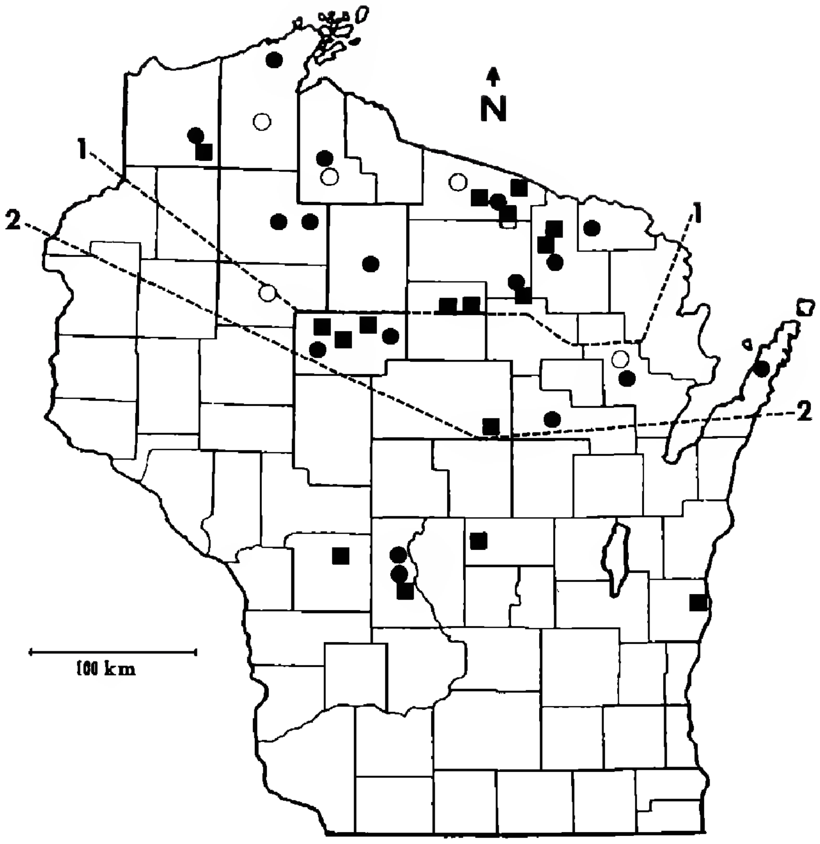

Figure 2. Known past and present distributions and 1996–97 nest-site locations of the Northern Goshawk in Wisconsin. 1 = resident range (Gromme 1963); 2 = resident range (Robbins 1991). Unshaded circles = counties with nests before 1958; shaded circles = 1996 nest locations; squares = 1997 nest locations.

these two nest-site categories. Because of the number of multiple univariate comparisons, we calculated that an alpha of 0.002 was the appropriate level of significance for statistical inference (Sokal and Rohlf 1981). Statistical analyses were performed on SYSTAT (Wilkinson 1992)

RESULTS AND DISCUSSION

In 1996–97, goshawk nests ($N = 34$) were widely distributed in the northern two-thirds of Wisconsin (Fig. 2). The historic breeding distribution of the goshawk in Wisconsin in the mid-1800s, before timber harvests became pervasive in the northern part of the state, is unknown. The first known nests in the state were found in the 1890s (Robbins 1991). Through 1958, only about 12 nests had been documented, all in five northern counties. As late as 1964 (Scott 1964), the northern half of the state held only 15% of the in-state members of the Wisconsin Society for Ornithology. During the same time period, no goshawk nests were reported in the southern half of the state in counties that held the large majority of society members engaged in ornithological activity (Scott 1964). Gromme (1963) probably based his map of the resident range of the goshawk (Fig. 2) on the above nests plus summer sight records. Robbins (1991) pictured a somewhat larger resident range for the goshawk. We interpret their maps of resident range as a presumed but perhaps imperfectly documented breeding range. Squires and Reynolds (1997) explicitly interpreted Robbins' map as breeding range in the state. Several additional sources portray a similar resident or breeding distribution in northern Wisconsin (Palmer 1988, Johnsgard 1990, Rosenfield et al. 1991).

The current nesting range of the goshawk in Wisconsin is much larger than previously indicated by maps of resident and breeding ranges. Therefore, there is no current evidence for a contraction of the goshawk's nesting range in the state, as might be expected as one index of changing status if the state's breeding population were declining (Kennedy 1997). S. Postupalsky (pers. comm.) has suggested that the 1996–97 nesting range may actually indicate that there has been an expansion in the breeding distribution, as has occurred in Michigan (Postupalsky 1991) and possibly other eastern states (Kennedy 1997).

The southern distributional limit mapped by Robbins (1991) corresponds approximately with the present limit of extensively forested lands in northern Wisconsin (Wisconsin Department of Natural Resources 1995). Many of the nests found in 1996–97 south of Robbins' line were located in woodlands in the predominately agricultural landscape that characterizes much of the southern half of the state.

We tallied the nest tree species used by goshawks at 37 nests in 1977–97, a sample including 25 nests from 1996–97 and 12 nests from prior years (Table 2). Of these nests, 29 (78%) were built in deciduous trees. Of these, 12 (41%) were in trembling aspens (*Populus tremuloides*). Goshawks nested in a broad array of other deciduous trees at ≤17% of the nest sites. Nest trees occurred in woodlands at both early (e.g., trembling aspen and white birch [*Betula papyrifera*]) and late seral stages (e.g., sugar maple [*A. saccharum*] and eastern hemlock [*Tsuga canadensis*]). Forest stands used for nesting included such varied habitats as pine plantations, upland maple (*Acer* spp.) and maple-oak (*Quercus* spp.) woodlands, black ash (*Fraxinus nigra*) swamps, and aspen monotypes, as well as forest fragments in southern Wisconsin. It is not surprising that the goshawk appears to use diverse woodland habitats for nesting in Wisconsin, given its wide breeding distribution over two-thirds of the state and the breadth of its nesting habitats throughout North America (Braun et al. 1996, Squires and Reynolds 1997). Elsewhere in the western Great Lakes region, trembling aspen was also used for nesting in a majority (10 of 14) of recent (1994–96) goshawk nest sites in Minnesota (Martell and Dick 1996). In Michigan, aspens (*P. grandidentata* and *P. tremuloides*) were again used more frequently than other tree species by breeding goshawks (S. Postupalsky pers. comm.).

Table 2. Tree species used for nesting (%) by Northern Goshawks in Wisconsin, 1977–97.

TREE SPECIES	NO. NEST TREES (N = 37)
Trembling aspen (*Populus tremuloides*)	12 (32)
Sugar maple (*Acer saccharum*)	5 (14)
Yellow birch (*Betula alleghaniensis*)	5 (14)
White pine (*Pinus strobus*)	5 (14)
Eastern hemlock (*Tsuga canadensis*)	3 (8)
Northern red oak (*Quercus rubra*)	2 (5)
Black ash (*Fraxinus nigra*)	2 (5)
Red maple (*Acer rubrum*)	1 (3)
White birch (*Betula papyrifera*)	1 (3)
Basswood (*Tilia americana*)	1 (3)

Of the 37 goshawk sites sampled, four were in pine plantations. Nests were built in white pine (*Pinus strobus*, $N = 3$) and a trembling aspen ($N = 1$) within the plantation. These four plantation nests, all found by unbiased means, were located in northeasternmost ($N = 1$) and southcentral ($N = 3$) parts of the state. At least three of these four plantation nests fledged young. S. Postupalsky (pers. comm.) also reports that pine plantations have recently been used as nest sites by successfully breeding goshawks in Michigan. Squires and Reynolds' (1997) review of nest-site habitats used by goshawks in North America did not report pine plantations as occupied nesting habitat.

Our comparison of habitat features at goshawk nest sites found by unbiased vs. potentially biased means failed to detect statistically significant differences ($P > 0.002$) between these two data sets for any of the 23 analyzed features with the sample sizes available (Table 3). The overall statistical similarity between habitat features at goshawk nests found by unbiased vs. potentially biased methods thus appeared to uphold the utility of a pooled sample as a descriptor of goshawk nest-site habitat in Wisconsin, a sample derived from a wide geo-

Table 3. Northern Goshawk habitat features at nests found by unbiased methods, potentially biased methods, and pooled methods. Data are reported as mean values ±SE (95% confidence interval). P = exact probability value of test between unbiased vs. biased categories.

FEATURE	UNBIASED $N = 21$	BIASED $N = 16$	POOLED $N = 37$	P
Dist. to Water (m)	193.8 ± 43.3	241.1 ± 75.9[a]	213.5 ± 40.0[b] (132.3–294.7)	0.75
Dist. to Opening (m)	184.0 ± 43.0	133.5 ± 21.0[a]	163.3 ± 26.6[b] (109.3–217.1)	0.89
Nest Tree Height (m)	23.7 ± 0.6	25.7 ± 0.8	24.6 ± 0.5 (23.6–25.6)	0.05
Nest Tree dbh (cm)	35.7 ± 2.4	47.4 ± 3.3	40.8 ± 2.2 (36.4–45.2)	0.01
Nest Height (m)	14.1 ± 0.4	15.5 ± 0.6	14.7 ± 0.4 (14.0–15.4)	0.12
Nest Percent (%)	59.9 ± 1.8	60.3 ± 2.1	60.1 ± 1.3 (57.4–62.8)	0.85
Degree Slope (°)	6.2 ± 1.6	6.2 ± 1.6	6.2 ± 1.2 (3.9–8.6)	0.79
Tall Shrub Index	14.4 ± 4.1	33.4 ± 10.2	22.6 ± 5.1 (12.2–33.0)	0.21
Low Shrub Index	46.9 ± 8.1	49.6 ± 10.7	48.0 ± 6.4 (35.1–61.0)	0.84
Under. Density (trees/ha)	283.3 ± 42.7	268.8 ± 42.5	277.0 ± 30.0 (216.1–338.0)	0.78
Canopy Height (m)	23.9 ± 0.7	25.5 ± 0.7	24.6 ± 0.5 (23.6–25.6)	0.10
Total Canopy (%)	78.7 ± 4.4	85.5 ± 3.1	81.6 ± 2.9 (75.8–87.4)	0.20
Decid. Canopy (%)	61.5 ± 7.5	72.3 ± 7.7	66.2 ± 5.4 (55.3–77.1)	0.12
Conif. Canopy (%)	17.2 ± 6.5	13.1 ± 6.8	15.4 ± 4.7 (6.0–24.9)	1.00
Total Understory (%)	45.4 ± 6.0	48.1 ± 4.8	46.6 ± 3.9 (38.6–54.6)	0.74
Decid. Understory (%)	40.3 ± 6.2	43.1 ± 5.0	41.5 ± 4.1 (33.2–49.8)	0.65
Conif. Understory (%)	5.1 ± 3.3	5.0 ± 2.6	5.1 ± 2.1 (0.7–9.4)	0.94
Total Ground (%)	58.4 ± 4.3	43.9 ± 5.0	52.1 ± 3.4 (45.2–59.1)	0.03
Decid. Ground (%)	56.4 ± 4.3	42.0 ± 5.3	50.2 ± 3.5 (43.1–57.3)	0.04
Conif. Ground (%)	2.0 ± 1.1	1.9 ± 1.1	2.0 ± 0.8 (0.4–3.5)	0.75
Tree Density (trees/ha)	458.3 ± 63.4	376.6 ± 42.3	423.0 ± 40.4 (341.0–505.0)	0.69
Basal Area (m^2/ha)	28.2 ± 3.7	33.0 ± 3.2	30.3 ± 2.5 (25.2–35.4)	0.16
Mean dbh (cm)	27.0 ± 1.4	32.9 ± 1.8	29.4 ± 1.2 (27.0–31.9)	0.03

[a] $N = 15$ due to missing data.

[b] $N = 36$ due to missing data.

graphic area of the state. For this pooled sample, mean nest-tree height was 25 m and mean nest-tree dbh was 41 cm (Table 3). Mean canopy height (25 m) was identical to mean nest-tree height, but mean tree dbh within 0.04-ha sample plots surrounding nest trees was substantially less than mean nest-tree dbh (29 vs. 41 cm). Mean canopy closure was 82%. Squires and Reynolds (1997) have suggested that such a high degree of canopy closure is one of the most uniform aspects of habitat at goshawk nest sites in North America.

ACKNOWLEDGMENTS

We gratefully acknowledge the support and cooperation of the many individuals who made this study possible, especially the assistance provided by S. Adams, J. Axtell, A. Belleman, T. Booms, C. Hutler, B. Isenring, J. Mills, D. Noble, L. Petersen, B. Pierce, and B. Smith. This manuscript was greatly improved by the comments of D. Andersen, W. Gould, M. Martell, J. Marzluff, and S. Postupalsky. We also thank the staff of the Necedah National Wildlife Refuge of the U.S. Fish and Wildlife Service, and the Chequamegon National Forest of the U.S. Forest Service for their help. We thank The Nature Conservancy and the Wisconsin Department of Natural Resources for providing some nest locations. This project was funded in part by the Bureau of Endangered Resources and the Bureau of Forestry in the Wisconsin Department of Natural Resources, and the Wisconsin Falconers Association The Personnel Development Committee at University of Wisconsin at Stevens Point also provided research funds and sabbatical support to the senior author.

LITERATURE CITED

APFELBAUM, S.I. AND P. SEELBACH. 1983. Nest tree, habitat selection, and productivity of seven North American raptor species based on the Cornell University nest record card program. *Raptor Res.* 17:97–113.

BLOCK, W.M., M.L. MORRISON AND M.H. REISER. 1994 The Northern Goshawk: ecology and management. *Stud. Avian Biol.* 16:1–2.

BRAUN, C.E., J.H. ENDERSON, Y.B. LINHART, C.D. MARTI AND M.R. FULLER. 1996. Northern Goshawk and forest management in the southwestern United States. Wildl. Soc., Tech. Rev. 96-2.

GROMME, O.J. 1963. Birds of Wisconsin. Univ. Wisconsin Press, Madison, WI U.S.A.

HAYWARD, G.D. AND R.E. ESCANO. 1989. Goshawk nest-site characteristics in western Montana and northern Idaho. *Condor* 91:476–479.

JAMES, C.F. AND H.H. SHUGART. 1970. A quantitative method of habitat description. *Audubon Field Notes* 24:727–736.

JOHNSGARD, P.A. 1990. Hawks, eagles, and falcons of North America. Smithsonian Inst. Press, Washington, DC U.S.A.

KENNEDY, P.L. 1997. The Northern Goshawk (*Accipiter gentilis atricapillus*): is there evidence of a population decline? *J. Raptor Res.* 31:95–106.

MARTELL, M. AND T. DICK. 1996. Nesting habitat characteristics of the Northern Goshawk (*Accipiter gentilis*) in Minnesota. Final Rep., Minnesota Dep. Nat. Res., Nongame Wildl. Program, Project No. 9407382. St. Paul, MN U.S.A.

PALMER, R.S. 1988. Handbook of North American birds, Volume 4. Yale Univ. Press, New Haven, CT U.S.A.

POSTUPALSKY, S. 1991. Northern Goshawk *Accipiter gentilis*. Page 168 *in* R. Brewer, G.A. McPeek and R.J. Adams, Jr. [EDS.] The atlas of breeding birds of Michigan. Mich. State Univ., East Lansing, MI U.S.A.

ROBBINS, S.D., JR. 1991. Wisconsin birdlife, population and distribution, past and present. Univ. Wis. Press, Madison, WI U.S.A.

ROSENFIELD, R.N., J. BIELEFELDT, R.K. ANDERSON AND J.M. PAPP. 1991. Status reports: accipiters. Pages 42–49 *in* B.E. Pendleton and D.L. Krahe [EDS.], Proceedings midwest raptor management symposium and workshop. Natl. Wildl. Fed. Sci. Tech. Ser. 15, Washington DC U.S.A.

SCOTT, W.E. 1964. Quarter-century history of the society. *Pass. Pigeon* 26:28–62.

SIDERS, M.S. AND P.L. KENNEDY. 1994. Nesting of accipiter hawks: is body size a consistent predictor of nest habitat characteristics? *Stud. Avian Biol.* 16:92–96.

SOKAL, R.R. AND F.J. ROHLF. 1981. Biometry. 2nd Ed. W.H. Freeman and Co., San Francisco, CA U.S.A.

SQUIRES, J.R. AND R.T. REYNOLDS. 1997. Northern Goshawk (*Accipiter gentilis*). *In* A. Poole and F. Gill [EDS.], The birds of North America, No. 298. The Academy of Natural Sciences, Philadelphia, PA U.S.A.

TITUS, K. AND J.A. MOSHER. 1981. Nest-site habitat selected by woodland hawks in the central Appalachians. *Auk* 98:270–281.

WISCONSIN DEPARTMENT OF NATURAL RESOURCES. 1995. Wisconsin's biodiversity as a management issue. A report to the department of natural resources managers. Wis. Dep. Nat. Res., Madison, WI U.S.A.

WILKINSON, L. 1992. SYSTAT: The system for statistics. SYSTAT Inc. Evanston, IL U.S.A.

Received 29 December 1997; accepted 17 May 1998

J. Raptor Res. 32(3):195–201
© 1998 The Raptor Research Foundation, Inc.

SOLITARY AND SOCIAL HUNTING IN PALE CHANTING GOSHAWK (*MELIERAX CANORUS*) FAMILIES: WHY USE BOTH STRATEGIES?

Gerard Malan[1]
Percy FitzPatrick Institute, University of Cape Town, Rondebosch, 7700 South Africa

Abstract.—I observed Pale Chanting Goshawks (*Melierax canorus*) using solitary and social hunting strategies. Most goshawks hunted predominantly alone, but if an individual was unable to flush and catch a cornered rodent from a shrub, other family members joined in a social hunt. Goshawks perched near or on the tops of shrubs and repeatedly struck at rodents until they were caught. Other family members did not pursue the goshawk that caught prey, even if it did not make the initial hunt. During social hunts, there was no evidence of a dominance hierarchy in families when they were not hunting. I found hunting success of individual goshawks to be low (11–12%) for both solitary and social hunts. Only large rodents were caught during social hunts, whereas smaller vertebrates (lizards and birds), and invertebrates, were caught during solitary hunts. It appeared that dominant breeders did not kleptoparasitize or dominate subordinate family members during social hunts to maximize their individual hunting success. Juveniles were significantly less successful than adults in capturing rodent prey, but may have increased their foraging efficiency and survival by participating in social hunts. Dominant Pale Chanting Goshawks that allowed offspring to partake in social hunts may, therefore, behave selfishly to increase their inclusive fitness.

Key Words: *Pale Chanting Goshawk;* Melierax canorus; *social hunting; juvenile survival; prey size; energy intake.*

Caza individual y social de *Melierax canorus*: por que utilizar ambas estrategias?

Resumen.—Observe a *Melierax canorus* utilizar estrategias de caza individual y social. La mayoría de los azores cazan principalmente en forma individual, pero si un individuo no es capaz de capturar a un roedor acorralado en un rastrojo, otros miembros de la familia se pueden unir en una cacería social. Los miembros restantes de la familia no persiguen al azor que ha capturado la presa. Durante la cacería social, no hubo evidencia de dominancia jerárquica la cual existe cuando no estan cazando. Encontré que el éxito individual de caza fué menor al 11–12% en ambas modalidades individual y social. Los grandes roedores fueron capturados sólo en cacerías sociales, mientras que los vertebrados mas pequenos (lagartijas y aves), así como tambien los invertebrados fueron capturados durante la caza individual. Sugiero que los reproductores dominantes no practican el kleptoparasitismo o dominan a miembros subordinados de la familia durante la caza social con el fin de maximizar el éxito de la caza individual. Los juveniles fueron menos exitosos que los adultos en capturar roedores pero pudieron haber aumentado su eficiencia de forrajeo y sobreviviencia al participar en la caza social. Los dominantes *Melierax canorus* que permitieron a sus hijos participar en la caza social pudieron haber actuado en forma autosuficiente con el fin de aumentar su vigor.

[Traducción de César Márquez]

Predators can use various hunting strategies to increase their individual foraging success. They can hunt alone or in association with related or unrelated conspecifics, or even with heterospecifics (Packer and Ruttan 1988, Ellis et al. 1993). In such hunting associations, they can pursue strategies ranging from active participation, where all individuals participate fully and benefit from social hunts, to kleptoparasitism (Hector 1986, Scheel and Packer 1991, Heinsohn and Packer 1995, Steele and Hockey 1995). Predators may adopt one or more of these strategies if their individual hunting success is low or if prey is large and difficult to catch (Packer and Ruttan 1988). The optimal combination of strategies should

---

[1] Present address: Department of Zoology, University of Durban-Westville, PBX54001, Durban 4000, South Africa.

maximize their net energy return (Hansen 1986, Bednarz 1988).

The Pale Chanting Goshawk (*Melierax canorus*) is a large, common raptor that inhabits the arid regions of southern Africa. In one study in the Little Karoo, South Africa, Pale Chanting Goshawks were found to live in family groups consisting of a breeding unit of either a polyandrous trio (a pair plus an additional cobreeding male) or a monogamous pair, with or without nonbreeders (up to two) and juveniles (up to four) (Malan et al. 1996). Cobreeders participated fully in reproductive activities, including copulations, but nonbreeders were actively excluded from the nesting area during the breeding season. Whereas polyandrous trios were recorded only in broken veld, delayed dispersal by nonbreeders and juveniles was the norm in all vegetation types. A dominance hierarchy existed in families with the female breeder on top followed by the male breeder and cobreeder and then the nonbreeders and juveniles (Malan and Jenkins 1996). Although the Pale Chanting Goshawk is a generalist feeder, relatively large rodent prey (45–124 g, *Otomys unisulcatus, Parotomys brantsii, and Rhabdomys pumilio*) that forage near vegetation or in the open make up most of the biomass in its diet (Malan and Crowe 1996). Other prey taxa include a range of other vertebrates as well as invertebrates. Pale Chanting Goshawks are obligate perch hunters and hunt from natural (trees or shrubs) or artificial (fence posts and telephone poles) perches from which they gently swoop to the ground (Malan and Crowe 1997).

This study tests the hypothesis that Pale Chanting Goshawks use solitary and social hunting to maximize their individual hunting success in capturing large and difficult to catch rodent prey. I observed the methods used by Pale Chanting Goshawks during social hunts, as well as the size of the prey caught during solitary and social hunts. Secondly, the solitary and social hunting strategies of large families ($\bar{x}$ = 5.5 goshawks) in one habitat were compared with small families ($\bar{x}$ = 3.4) in another habitat. Thirdly, I compared the hunting tactics of juvenile Pale Chanting Goshawks with those of adults as well as foraging fledglings, still dependent on their parents for food.

STUDY AREA AND METHODS

The 146 km^2 study area was located near Calitzdorp (Little Karoo, 33°32'S, 21°48'E) in South Africa. It receives an average annual rainfall of 20 cm and the topography is generally flat. It is utilized for extensive Ostrich (*Struthio camelus*) farming.

Two semi-arid vegetation types occurred in the study area, broken veld (Karroid Broken Veld vegetation type, Acocks 1988) in the north and dwarf shrubland (Succulent Karoo) in the south. Broken veld consisted of small trees and shrubs (1–3 m high) scattered in a matrix of low shrubs. Dwarf shrubland consisted of a sparse layer of prostrate succulents and herbs. In dwarf shrubland, Pale Chanting Goshawks only occupied areas with a high availability of perches (mostly fenceposts) whereas broken veld with its abundant trees and shrubs was probably saturated with Pale Chanting Goshawk families (Malan 1995).

I defined hunts as flights by goshawks from perches to attack prey on the ground or in the air. During each hunt, I aged the participating goshawk(s) as follows: adults, juveniles or goshawks in immature plumage, and fledglings or offspring still fed by their parents for up to 80 d after leaving nests (Malan 1995). For adults, hunting data of breeders and nonbreeders were combined. Three hunt outcomes were recognized: successful hunts or hunts that ended when goshawks landed on the ground and caught prey, unsuccessful hunts or hunts that ended when goshawks landed on the ground but failed to catch prey, and abandoned hunts or hunts that ended when goshawks flew down from perches and, upon reaching the point of impact, briefly hovered about 1 m above potential prey, then flew off without the prey. Hunting of termites was not analyzed because they were not chased (Malan and Crowe 1996).

Using instantaneous sampling (Lehner 1979), I followed a focal Pale Chanting Goshawk by vehicle and recorded aspects of its hunting behavior every 60 sec. During each observation period, the focal goshawk was followed from 60–300 min and, when it was out of sight, the observation period was terminated. The hunting behavior of all other family members within 100 m of the focal goshawk was also recorded. The study was conducted from February 1988–March 1989, but the hunting behavior of mated adults was only studied in the nonbreeding and prelaying (from first copulation until egg-laying) periods. The hunting behavior of goshawks was also recorded during casual observations during the summer breeding seasons of 1989–95. Solitary and social hunts were recorded during 64 observation periods (total observation time = 11 139 min, $\bar{x}$ = 174 min, SD = 67 min), and solitary hunts during an additional 17 observation periods (2074 min; $\bar{x}$ = 123, SD = 54 min). The hunting behavior of 15 adults were studied for 57 observation periods (9398 min), five juveniles for 16 observation periods (2651 min), and two fledglings for eight observation periods (1164 min). Capture rates per hour were calculated for each observation period and compared between observations periods for single and social hunts, goshawk age classes, and hunt outcomes.

A solitary hunt is defined as only the focal Pale Chanting Goshawk hunting. A social hunt involved either the focal goshawk hunting and being joined on the ground by family members, or the focal goshawk joining family members in a hunt. I termed these hunts "social" because family members hunted together in a nonaggressive and cooperative manner. A social hunt was successful

Table 1. A comparison of solitary and social striking rates (per hour) by adult, juvenile, and feldgling Pale Chantmg Goshawks. During successful hunts, prey was caught. During unsuccessful hunts, goshawks landed on the ground but failed to catch prey. During abandoned hunts, goshawks briefly hovered about 1 m above potential prey but flew off without prey.

	ADULTS	JUVENILES	FLEDGLINGS	KRUSKAL-WALLIS	df
Solitary hunts					
Successful	0.15 ± 0.24[1]	0.07 ± 0.13	0.04 ± 0.10	2.66 ns	2
Unsuccessful	1.00 ± 1.05	1.74 ± 1.02	1.35 ± 1.45	8.49*	2
Abandoned	0.12 ± 0.24	0.21 ± 0.26	0.11 ± 0.31	4.24 ns	2
All solitary hunts	1.26 ± 1.13	2.02 ± 1.21	1.49 ± 1.69	6.99*	2
Social hunts					
Successful	0.04 ± 0.11	0.01 ± 0.06	0.00	1.51 ns	2
Unsuccessful	0.08 ± 0.19	0.02 ± 0.07	0.00	3.24 ns	2
Abandoned	0.05 ± 0.13	0.00	0.00	3.90 ns	2
All social hunts	0.17 ± 0.28	0.03 ± 0.09	0.00	6.01*	2

[1] = mean ± 1 SD.
* = $P < 0.05$.

if any of the participating Pale Chanting Goshawks caught prey. Due to my small sample size, only social hunts involving two goshawks were analyzed. In all social hunts analyzed, only adult Pale Chanting Goshawks joined the focal adult or juvenile.

Pale Chanting Goshawk families are strictly territorial and unrelated conspecifics were not tolerated within territories (Malan and Jenkins 1996). Pale Chanting Goshawks thus always hunted in association with family members. This association was compared between the significantly larger polyandrous families in broken veld ($\bar{x}$ = 5.5 goshawks) and smaller monogamous families in dwarf shrubland ($\bar{x}$ = 3.4; Malan 1995). The presence of family members within a 100 m radius of the focal animal was compared between three families each from broken veld and dwarf shrubland. For each observation period, I calculated the proportion of time spent alone or in close proximity with one or more family members, either adults, juveniles, or fledglings. Data from 21 focal individuals were analyzed for 63 observation periods(10 055 min; $\bar{x}$ = 160, SD = 66 min) and arcsine transformed to improve normality (Zar 1984).

RESULTS

Prey was attacked on the ground in 99% (N = 397) of all hunts. When prey was pursued on the ground, it was chased actively, very often with wings aloft and flapping. If vertebrate prey, such as an otomyinid rodent, was cornered under a shrub and a family member joined the focal goshawk on the ground, the Pale Chanting Goshawks would surround the shrub and/or perch on top. Individuals would then repeatedly strike at the rodent by jumping into the shrub (flush-and-ambush strategy; Bednarz 1988). In four hunts, all unsuccessful, a Pale Chanting Goshawk attacked a bird from a perch and actively chased the bird in horizontal flapping flight.

The frequency of successful solitary hunts by adults in dwarf shrubland (14%; N = 86) was not significantly different from the frequency of successful solitary hunts in broken veld (10%, N = 107, Log-likelihood Ratio with Yates correction: G_c = 0.31, $P > 0.50$). The frequency of successful social hunts by adults also did not differ significantly between broken veld (25%, N = 8) and dwarf shrubland (21%, N = 14; G_c = 0.12, $P > 0.70$). The rates per hour that adults participated in successful, unsuccessful, or abandoned solitary hunts did not differ significantly (t-test, $P > 0.05$,) between large and small families. Likewise, the rates per hour that adults participated in successful, unsuccessful, or abandoned social hunts did not differ significantly (t-test, $P > 0.05$,) between large and small families. My sample size prevented a comparison between the hunting rates of successful, unsuccessful, or abandoned hunts for both juveniles and fledglings from large and small families.

When the solitary and social striking rates of all age classes were considered, the ratio of solitary to social hunts by adults was significantly less than for juveniles and fledglings (Table 1). Juveniles (4%, N = 100) and fledglings (4%, N = 25) were equally successful in solitary hunts, but were significantly less successful than adults (11%, N = 193; G_c =

Table 2. Percent time per observation period Pale Chanting Goshawks hunted within 100 m of other family members in large ($\bar{x}$ = 5.5 goshawks) and small families ($\bar{x}$ = 3.4).

NUMBER PRESENT	SMALL FAMILIES $n = 3$	LARGE FAMILIES $n = 3$	F	df
Zero	84 ± 44	85 ± 54	0.01 ns	1, 46
One	15 ± 44	11 ± 38	0.36 ns	1, 46
Two	0 ± 4	1 ± 4	2.62 ns	1, 46
Three	0 ± 0	1 ± 6	4.23*	1, 46
Four	0 ± 0	0 ± 3	2.04 ns	1, 46

* = $P < 0.05$.

4.23, $P < 0.05$). Juveniles engaged in social hunts (2%, N = 102) significantly less often than adults (10%, N = 215, G_c = 6.75, $P < 0.01$), whereas fledglings did not participate in social hunts at all. During social hunts, once prey was caught, goshawks flew off with the item and they were not pursued by the remaining family members. Adults caught prey during social hunts in 23% (N = 22) of hunts. The adult that initiated the social hunt caught the prey in 20% (N = 10) of instances, whereas the focal adult that joined the hunt caught the prey in 27% (N = 12) of instances (G_c = 0.05, $P > 0.75$). The frequency of success of the solitary hunts by adults (11%, N = 193) did not differ significantly from their success in social hunts (23%, N = 22, G_c = 1.40, $P > 0.10$). Juveniles participated in two social hunts, joining the hunt in both instances, and in one of these hunts, the juvenile was successful. During casual observations, juveniles that participated in social hunts caught rodent prey in four instances.

During all successful hunts only vertebrates (rodents, lizards and birds) were caught. Hunts for arthropods were probably so quick, and in the lower vegetation layer, that they were not seen. During solitary hunts, fledglings caught one lizard (*Sauria* spp.), juveniles caught three lizards and one bird, and adults caught 18 rodents (86%), two lizards and one bird. During social hunts, only rodents were caught with adults capturing five rodents and one juvenile catching one rodent.

Table 3. Percent time per observation period adult, juvenile, and fledgling Pale Chanting Goshawks spent within 100 m radius of other family members.

NUMBER PRESENT	ADULTS	JUVENILES	FLEDGLINGS	F	df
Zero	85 ± 41	99 ± 13	99 ± 16	7.30**	2, 69
One	13 ± 38	1 ± 13	1 ± 16	7.07**	2, 69
Two	0 ± 16	0	0	2.18 ns	2, 69
Three	0 ± 10	0	0	0.97 ns	2, 69
Four	0 ± 6	0	0	0.49 ns	2, 69

** = $P < 0.01$.

When the association of family members between large and small families was investigated, adults of large families spent significantly more time in close proximity (<100 m) to three family members than did adults from small families (Table 2). The proportion of time spent alone ($\bar{x}$ = 99, SD = 13%) or in close proximity to one family member ($\bar{x}$ = 1%, SD = 13%, ANOVA, all $P > 0.05$) did not differ significantly between juveniles of large and small families. Adults, compared to juveniles and fledglings, spent significantly less time alone and significantly more time in close proximity to one family member (Table 3).

DISCUSSION

Despite the potential advantages associated with hunting in groups, such as an increase in individual hunting success and energy return (Bednarz 1988), Pale Chanting Goshawks still predominantly hunt alone. The average hunting success of individual adult goshawks in social hunts was only 11.5%, half of the 23% success of social hunts in which two goshawks participated. Nevertheless, it was similar to the 11% hunting success of individuals in solitary hunts.

Why would Pale Chanting Goshawks follow two hunting strategies that contribute the same to an individual's hunting success? For social hunting to be a viable option, the individual benefits of this hunting strategy must equal or exceed that of hunting singly (Hansen 1986). First, such benefit could only result if prey captured in family pursuits is, on average, larger than that caught in solitary pursuits (Steele and Hockey 1995). Pale Chanting Goshawks preyed mostly on relatively large otomyinid rodents (mean body mass = 124 g), as well as the smaller *Rhabdomys pumilio* (mean body mass = 45 g) (Malan and Crowe 1996). In broken veld and dwarf shrubland, these rodents contributed 87% or 22 682 g of biomass and 68% or 249 individuals to the vertebrate diet. Pale Chanting Goshawks also preyed on smaller mammals, small birds, hatchling tortoises, small snakes and lizards, as well as sunspiders, harvester termites, grasshoppers and beetles (Malan and Crowe 1996). The average mass of rodents caught in the two vegetation types was 90 ± 40 (±1 SD) g, birds 70 ± 34 g, and reptiles 12 ± 9 g (Malan and Crowe 1996). Thus, because only rodents were captured in social hunts, the average size of prey captured in this way was indeed larger than those caught in solitary hunts. In terms of hunting socially, it was also the prey biomass obtained during these hunts, and not only the relative success or hunting technique used, that was important to each individual. Although the hunting success in solitary and social hunts was equal, the energy returns from hunting large animals in social hunts may have surpassed the returns from hunting smaller prey in solitary hunts.

A second reason why Pale Chanting Goshawks may use two hunting strategies is that their individual hunting success in catching vertebrate prey in solitary hunts is low. Solitary adult hunting success of 11% was substantially lower than the mean of 59% (range = 31–72%) for 11 raptor species that hunt ground-dwelling prey (Toland 1986). It is even lower than the 19–33% success ($\bar{x}$ = 27%) for raptors that hunt other birds in the air, a technique generally thought to be less successful than searching for prey on the ground (Toland 1986). The low success of Pale Chanting Goshawks highlights the difficulty they experience in catching vertebrate prey in a shrub-rich substrate. Since solitary hunters have low success of catching large vertebrate prey, Pale Chanting Goshawks may adopt a social hunting strategy to supplement their solitary hunting and thus increase their overall hunting success.

In spite of the apparent benefits of hunting socially, adult Pale Chanting Goshawks did not habitually hunt together. Adults spent only 15% of time within 100 m of family members, compared with 71% of the time that Harris' Hawks (*Parabuteo unicinctus*) spent within 50 m of group members (Bednarz 1988). Pale Chanting Goshawks could, however, visually monitor each other's movements by perching on the highest available perch. The flapping wing motions during a pursuit may act as a signal to other family members that a hunt is in progress. If the prey animal was cornered, a solitary Pale Chanting Goshawk probably cannot act as a hunter and a beater, and would thus fail in its solitary attack strategy. If the hunting goshawk could attract family members, however, it would have some chance of obtaining the prey. The hunting behavior of the initiator and the goshawks that subsequently join hunts, therefore, appear to be selfish. If individuals behaved selfishly during social hunts, then why did dominant Pale Chanting Goshawks not attempt to increase their hunting success by kleptoparasitizing subordinate family members and why was there no aggressive behavior observed between family members during a social hunt? Pale Chanting Goshawks do kleptoparasitize Booted Eagles (*Hieraaetus pennatus*) with rodent prey (unpubl. data) and at nesting sites, subordinate cobreeding Pale Chanting Goshawks do transfer prey to the dominant female and male breeder, but not vice versa (Malan and Jenkins 1996). Packer and Ruttan (1988) predicted that if single prey items are hunted, but not shared amongst participants, group members will always cooperate fully in hunts. Dominant Pale Chanting Goshawk breeders may not kleptoparasitize subordinate members because the initial benefit (suckers payoff, Axelrod and Hamilton 1981) of hunting prey not to be shared may result in defection by subordinates, with a subsequent decrease in the rate of social hunts. Likewise, if dominant breeders exert their dominance on subordinate members upon arrival at cornered prey, prey may escape and again no benefits can be gained by the goshawks participating. By displaying no obvious aggression towards each other during social hunts, not perceived to be a common trait among raptors (Faaborg and Bednarz 1990), each family member may increase its individual hunting success.

Given the increase in hunting success when combining social and solitary hunting, one would expect not only the hunting success per Pale Chanting Goshawk in bigger families to be greater, but also individuals from bigger families would be expected to spend more time hunting socially. Surprisingly few differences were found between the hunting strategies of large and small families (5.5 vs. 3.4 goshawks). Emlen (1994) suggested that the benefits of social activities such as social hunting may be secondarily derived after families formed because goshawks were constrained through a factor such as a lack of territorial space, from dispersing, and breeding in pairs. Even if the benefits of social hunting are secondarily derived, I suggest that hunting in families may hold fitness benefits for participants.

First, the participation by rapacious juveniles in social hunts may hold additional benefits associated with hunting relatively large prey that are not shared by family members (Stacey and Ligon

1987). Prior to independence, young raptors experience a high mortality rate (Newton 1995), partly because of their low foraging efficiency (Heinsohn 1991). If they are raised in a social family and delay dispersal from that family, the benefits of philopatry may include participation in social hunts (Heinsohn et al. 1988). Juveniles were involved in social hunts, albeit at a very low rate (0.03/hr). Furthermore, they were only able to catch lizards (estimated mass 10 g; Malan and Crowe 1996) in solitary hunts, but caught rodents in social hunts. The benefits of hunting relatively large rodent prey may increase their foraging efficiency and survival during the critical 12–16 mo of their life. If this was the case, it is difficult to explain why these juveniles engaged in social hunts significantly less often than adults did. The fact that juveniles only occupied a segment of the territory in close proximity to the nesting site (unpubl. data), may have made them less able to detect family members hunting in other segments of the territory.

Second, breeders may also gain fitness benefits from hunting with their offspring. The success from hunting socially, measured in terms of survival fitness, may be higher than if determined directly from the hunting success of individuals in the family (Packer and Ruttan 1988, Koenig and Mumme 1990). The act of allowing other family members to partake in social hunts may thus benefit the individual that cornered the prey indirectly, as the loss in direct fitness is compensated by a gain in indirect fitness. Individual Pale Chanting Goshawks that allowed other family members to partake in social hunts may be therefore behaving selfishly to increase their inclusive fitness.

ACKNOWLEDGMENTS

This research was supported in part by grants from the Foundation for Research Development (M.Sc. bursary), the Frank M. Chapman Memorial Fund (American Museum of Natural History), the Bob Blundell Memorial Scholarship, the Leslie Brown Memorial Grant, the University of Cape Town Equipment and Research Committee, and the Percy FitzPatrick Institute. I thank Grant Benn, Tim Crowe, Phil Hockey and David Jacobs for valuable comments of an earlier draft. I also thank the landowners at Calitzdorp for access to their properties.

LITERATURE CITED

ACOCKS, J.P.H. 1988. Veld types of South Africa. *Mem. Bot. Surv. S. Afr.* No. 57.

AXELROD, R. AND W.D. HAMILTON. 1981. The evolution of cooperation. *Science* 211:1390–1396.

BEDNARZ, J.C. 1988. Cooperative hunting in the Harris' Hawk (*Parabuteo unicinctus*). *Science* 239:1525–1527.

ELLIS, D.H., J.C. BEDNARZ, D.G. SMITH AND S.P. FLEMMING. 1993. Social foraging classes in raptorial birds. *BioScience* 43:14–20.

EMLEN, S.T. 1994. Benefits, constrains and the evolution of the family. *Trends Ecol. Evol.* 9:282–284.

FAABORG, J. AND J.C. BEDNARZ. 1990. Galapagos and Harris' Hawks: divergent causes of sociality in two raptors. Pages 359–383 *in* P.B. Stacey and W.D. Koenig [EDS. Cooperative breeding in birds. Cambridge Univ. Press, Cambridge, U.K.

HANSEN, A.J. 1986. Fighting behavior in Bald Eagles: a test of game theory. *Ecology* 67:787–797.

HECTOR, D.P. 1986. Cooperative hunting and its relationship to foraging success and prey size in an avian predator. *Ethology* 73:247–257.

HEINSOHN, R.G. 1991. Slow learning of foraging skills and extended parental care in cooperative breeding White-winged Choughs. *Am. Nat.* 137:874–881.

———, A. COCKBURN AND R.B. CUNNINGHAM. 1988. Foraging, delayed maturation, and advantages of cooperative breeding in White-winged Choughs, *Corcorax melanorhmphos*. *Ethology* 77:177–186.

——— AND C. PACKER. 1995. Complex cooperative strategies in group-territorial African lions. *Science* 269: 1260–1262.

KOENIG, W.D. AND R.L. MUMME. 1990. Population ecology of the cooperative breeding Acorn Woodpecker. Princeton Univ. Press, Princeton, NJ U.S.A.

LEHNER, P.N. 1979. Handbook of ethological methods. Garland STMP Press, New York, NY U.S.A.

MALAN, G. 1995. Cooperative breeding and delayed dispersal in the Pale Chanting Goshawk *Melierax canorus*. Ph.D. dissertation, University of Cape Town, Rondebosch, South Africa.

——— AND T.M. CROWE. 1996. The diet and conservation of monogamous and polyandrous Pale Chanting Goshawks in the Little Karoo, South Africa. *S. Afr. J. Wildl. Res.* 26:1–10.

——— AND A.R. JENKINS. 1996. Territory and nest defense in the Pale Chanting Goshawks: do the cobreeders help? *S. Afr. J. Zool.* 31:170–176.

——— AND T.M. CROWE. 1997. Perch availability and prey visibility: factors that may constitute habitat quality in the Pale Chanting Goshawk. *S. Afr. J. Zool.* 32:14–20.

———, T.M. CROWE, R. BIGGS AND J.J. HERHOLDT. 1997. The social system of the Pale Chanting Goshawk *Melierax canorus*: monogamy versus polyandry and delayed dispersal. *Ibis* 139:313–321.

NEWTON, I. 1995. The contribution of some recent research on birds to ecological understanding. *J. Animal Ecol.* 64: 675–696.

PACKER, C. AND L. RUTTAN. 1988. The evolution of cooperative hunting. *Am. Nat.* 132:159–198.

SCHEEL, D. AND C. PACKER. 1991. Group hunting behavior of lions: a search for cooperation. *Anim. Behav.* 41: 697–709.

STACEY, P.B. AND J.D. LIGON. 1987. Territory quality and dispersal options in the Acorn Woodpecker, and a challenge to the habitat-saturation model of cooperative breeding. *Am. Nat.* 130:654–676.

STEELE, W.K. AND P.A.R. HOCKEY. 1995. Factors influencing rate and success of intraspecific kleptoparasitism among Kelp Gulls (*Larus dominicanus*). *Auk* 112:847–859.

TOLAND, B. 1986. Hunting success of some Missouri raptors. *Wilson Bull.* 98:116–125.

ZAR, J.H. 1984. Biostatistical analysis. Prentice-Hall, Inc., Princeton, NJ U.S.A.

Received 20 May 1996; accepted 9 May 1998

BUTEO BOOKS

The following Birds of North America Species Accounts are available through Buteo Books, 3130 Laurel Road, Shipman, VA 22971. TOLL-FREE ORDERING: 1-800-722-2460; FAX: (804) 263-4842.

Barn Owl (1). Carl D. Marti. 1992. 16 pp.
Boreal Owl (63). G.D. Hayward and P.H. Hayward. 1993. 20 pp.
Broad-winged Hawk. (218). L.J. Goodrich, S.C. Crocoll and S.E. Senner. 1996. 28 pp.
Burrowing Owl (61). E.A. Haug, B.A. Millsap and M.S. Martell. 1993. 20 pp.
Common Black-hawk (122). Jay H. Schnell. 1994. 20 pp.
Cooper's Hawk (75). R.N. Rosenfield and J. Bielefeldt. 1993. 24 pp.
Crested Caracara (249). Joan L. Morrison. 1996. 28 pp.
Eastern Screech-owl (165). Frederick R. Gehlbach. 1995. 24 pp.
Ferruginous Hawk (172). Marc J. Bechard and Josef K. Schmutz. 1995. 20 pp.
Flammulated Owl (93). D. Archibald McCallum. 1994. 24 pp.
Great Gray Owl (41). Evelyn L. Bull and James R. Duncan. 1993. 16 pp.
Gyrfalcon (114). Nancy J. Clum and Tom J. Cade. 1994. 28 pp.
Harris' Hawk (146). James C. Bednarz. 1995. 24 pp.
Long-eared Owl (133). J.S. Marks, D.L. Evans and D.W. Holt. 1994. 24 pp.
Merlin (44). N.S. Sodhi, L. Oliphant, P. James and I. Warkentin. 1993. 20 pp.
Northern Saw-whet Owl (42). Richard J. Cannings. 1993. 20 pp.
Northern Goshawk (298). John R. Squires and Richard T. Reynolds. 1997. 32 pp.
Northern Harrier (210). R. Bruce MacWhirter and Keith L. Bildstein. 1996. 32 pp.
Red-shouldered Hawk (107). Scott T. Crocoll. 1994. 20 pp.
Red-tailed Hawk (52). C.R. Preston and R.D. Beane. 1993. 24 pp.
Short-eared Owl (62). D.W. Holt and S.M. Leasure. 1993. 24 pp.
Snail Kite (171). P.W. Sykes, Jr., J.A. Rodgers, Jr. and R.E. Bennetts. 1995. 32 pp.
Snowy Owl (10). David F. Parmelee. 1992. 20 pp.
Spotted Owl (179). R.J. Gutiérrez, A.B. Franklin and W.S. Lahaye. 1995. 28 pp.
Swainson's Hawk (265). A. Sidney England, Marc J. Bechard and C. Stuart Houston. 1997. 28 pp.
Swallow-tailed Kite (138). Kenneth D. Meyer. 1995. 24 pp.
White-tailed Hawk (30). C. Craig Farquhar. 1992. 20 pp.
White-tailed Kite (178). Jeffrey R. Dunk. 1995. 16 pp.

J. Raptor Res. 32(3):202–207
© 1998 The Raptor Research Foundation, Inc.

FOREST MANAGEMENT EFFECTS ON NESTING HABITAT SELECTED BY EURASIAN BLACK VULTURES (*AEGYPIUS MONACHUS*) IN CENTRAL SPAIN

JUAN A. FARGALLO
Departamento de Ecología Evolutiva, Museo Nacional de Ciencias Naturales (CSIC), J. Gutiérrez Abascal 2, E-28006 Madrid, Spain

GUILLERMO BLANCO
Departamento de Biología Animal, Universidad de Alcalá de Henares, 28871 Madrid, Spain

EDUARDO SOTO-LARGO
Pza. Mariano de Cavia 1, 28007 Madrid, Spain

ABSTRACT.—We studied two recently established colonies of Eurasian Black Vultures (*Aegypius monachus*). One was located in an abandoned maritime pine *(Pinus pinaster)* plantation formerly used for resin production. The other colony was in a Scots pine (*P. silvestris*) plantation currently used for timber production. The vultures used nest sites with mature trees in forest openings and on steep slopes in the medium-upper portions of mountains. These openings had few roads. Differences in tree density, distance to nearest neighbor's nests, and tolerance to high road density were observed between the nest sites used by the two colonies. Sylvicultural practices in either forests explained differences in nest-site selection between the colonies.

KEY WORDS: *Eurasian Black Vulture;* Aegypius monachus; *forest management; nest-site habitat.*

Efecto del manejo del bosque sobre la selección del hábitat de nidificación del Buitre Negro (*Aegypius monachus*) en España central

RESUMEN.—Se ha estudiado el efecto del manejo del bosque sobre los requerimientos del hábitat de nidificación del Buitre Negro (*Aegypius monachus*) en dos colonias de reciente ocupación. Las colonias se establecieron en dos plantaciones de pino autóctono del Sistema Central español. Ambos bosques se encuentran sometidos a distinto tipo de explotación. En uno de ellos (*Pinus silvestris*) se realiza una explotación maderera y el otro (*Pinus pinaster*) se encuentra en la actualidad en desuso tras haber estado dedicado a la extracción de resina. En ambos bosques los buitres seleccionaron lugares de nidificación con árboles maduros para colocar los nidos, en áreas más desprovistas de vegetación, con una mayor pendiente y situados en el tercio medio-superior de las montañas. En estas áreas también se encontró un menor número de caminos. Se observaron diferencias en la densidad de árboles, en la distancia al nido ocupado más próximo y en la tolerancia a los caminos entre las dos colonias. El diferente uso del bosque explica estas diferencias.

[Traducción Autores]

The Eurasian Black Vulture (*Aegypius monachus*) is a widespread Palearctic species that has suffered a marked decline in Europe in the last century, disappearing from 15 countries (Bijleveld 1974, Tucker and Heath 1994) and experiencing a marked reduction in range and abundance in Greece, Russia, Spain, Ukraine, and also likely in Turkey (Blanco and González 1992, Tucker and Heath 1994). About 200 pairs remained in Spain from 1966–72 (Bernis 1966, Hiraldo 1974). Nevertheless, in the last 20 yr, the population has increased to an estimated 774 pairs in 1990 (González 1990) and 900–1000 pairs in 1992 accounting for between 67–90% of the breeding European population (González in Tucker and Heath 1994). Few studies have focused on the analysis of habitat requirements of this species with only cursory mention of some colonies (Hiraldo 1974, Torres et al. 1980, Bermejo 1990, Donázar 1993).

Loss and alteration of habitat are the most serious threats for raptor populations (Bijleveld 1974, Tucker and Heath 1994). Their large territories, home ranges, and historical persecution by humans have made populations of forest breeding raptors sensitive to forest management and habitat change (Fuller 1996, Niemi and Hanowski 1997). Indeed, declines in Eurasian Black Vulture populations had been previously attributed to alterations of nesting habitats (Hiraldo 1974, Bermejo 1990, Blanco and González 1992, Donázar 1993). The spread of human settlements and intensive agriculture in lowlands and pastures in highlands have caused forest fragmentation and loss of forested areas. Further, intensive forestry has created mosaics of mature stands mixed with successional stands of various ages (e.g., Moorman and Chapman 1996, Niemi and Hanowski 1997).

The Eurasian Black Vulture is closely associated with forests during the breeding season (7–8 mo) and afterwards because it uses trees as roosts and shelters (Bernis 1966, Hiraldo and Donázar 1989). Understanding the interaction between forest management practices and habitat selection by the Eurasian Black Vulture is crucial in planning conservation measures for this threatened species. Furthermore, recent colonization of some areas enables us to determine its habitat requirements because individuals select optimal vacant nesting sites, unlike older colonizations where individuals have been forced to occupy lower quality nesting sites (Brown 1969, Kadmon 1993, Ferrer and Donázar 1996).

STUDY AREA AND METHODS

Two recently established colonies of Eurasian Black Vultures were studied in two pine forests of the Spanish Central Range (Sierra de Guadarrama, 40°30′N). The first colony (colony A) was located in a young maritime pine (*Pinus pinaster*) plantation (forest A) in the province of Madrid (mean elevation = 1150 m, range = 930–1387). This pine species is native of the Spanish Central Range (Mallada 1892). The old pine forest was felled during the Spanish Civil War (1936–39) and the post-war period, and only a few old pines remained. After the war, the area was reforested with a new plantation of pines, making the new forest a mixture of mostly trees about 60-yr old interspersed with few mature trees. Resin extraction was the main use of this forest. The first pair of Eurasian Black Vultures became established in this area in 1985 and, by 1994, the colony consisted of 11 pairs. The second colony (colony B) was in a Scots pine (*Pinus sylvestris*) plantation (forest B) in the province of Segovia (mean elevation = 1600 m, range = 1315–2196). Scots pine is also native of the Spanish Central Range (Galera and Martín 1990). Nowadays, wood exploitation for timber is the main use of this forest. In past years, trees were felled in large areas and newly reforested. Presently, this forest is intensively managed and only individual mature trees are selectively cut leaving some protected areas for natural regeneration. These areas are located in the upper third of the mountains. Vultures became established in this forest in 1989, and, by 1994, the colony consisted of seven pairs. The two colonies are 32 km apart.

Table 1. Variables used to characterize Eurasian Black Vulture nesting and random plots. Variables were measured from a point in the center of the plot.

DT—Stem diameter tree at breast height in cm
HT—Height of tree in m
DNN—Distance of nearest nest of same species in m
SCH—Height of scrubland in plot in m
DTT—Mean distance to three nearest trees in m
PCC—Percent canopy cover in plot
PUC—Percent understory cover in plot
PSC—Percent of scrubland in plot
PPL—Percent of pastureland in plot
PRL—Percent of rockyland in plot
NT1—Number of trees in plot with a stem diameter <15 cm
NT2—Number of trees in plot with a stem diameter 16–30 cm
NT3—Number of trees in plot with a stem diameter 31–50 cm
NT4—Number of trees in plot with a stem diameter >51 cm
NOT—Total number of overstory trees in plot
ATT—Elevation of plots in meters
TII—Topographic irregularity index (number of 20 m contours cut by four 500 m lines from nest tree in the four cardinal compass directions)
NPM—Nest position on mountain by mountain division (1 = upper third, 2 = medium third, and 3 = lower third)
DNV—Distance to nearest village in m
MOR—Meters of paved and unpaved roads within plots with a radius of 500 m from nest trees

We used procedures suggested by Titus and Mosher (1981), Andrew and Mosher (1982), Bosakowski et al. (1992), Donázar et al. (1993), and Moorman and Chapman (1996) to study habitat selection by these vultures. Nest sites were located in the spring of 1994 and plotted on a 1:25 000 topographic map. Characteristics of nest sites were measured in the field during the winter of 1994–95. A nest site was defined as the nest tree plus a circular plot of 25 m radius around it (Titus and Mosher 1981). Vegetation cover within these 25-m radius plots was visually estimated from the center of each subplot (Prodon 1976, Rubio and Carrascal 1994). We also noted the number of trees in different diameter categories and the heights of nest trees. Measurements on other habitat variables (Table 1) studied were obtained from the topographic map.

Random sampling was used to estimate available nest-

Table 2. Sample means ± SD of nest sites and random plots in each colony and forest. ANOVA F values are given for differences between random and nest site plots, between nest sites of each colony, and between random plots of each forest.

	Colony A				Colony B			
	Nest Sites (16)[a]	Random (31)	F	*P*	Nest Sites (13)	Random (37)	F	*P*
Var.			(df = 1, 44)				(df = 1, 47)	
DT	57.6 ± 21.0	32.8 ± 10.7	27.5	0.00	62.9 ± 13.5	51.4 ± 12.7	7.6	0.00
HT	13.3 ± 1.7	10.0 ± 2.4	25.1	0.00	13.8 ± 2.2	15.2 ± 3.5	1.1	0.31
DNN	379 ± 205	582 ± 347	5.1	0.02	1362 ± 930	731 ± 478	1.9	0.17
SCH	1.6 ± 0.3	1.8 ± 0.6	1.2	0.28	1.3 ± 1.0	1.8 ± 0.9	2.4	0.26
DTT	8.9 ± 5.6	4.0 ± 2.7	14.1	0.00	7.6 ± 3.1	5.4 ± 5.3	4.3	0.04
PCC	42.0 ± 20.7	64.3 ± 24.5	7.1	0.01	40.6 ± 23.8	62.5 ± 31.4	3.4	0.07
PUC	22.4 ± 19.4	5.5 ± 10.6	20.1	0.00	10.6 ± 12.6	11.9 ± 14.9	0.0	0.90
PSC	73.1 ± 26.7	68.7 ± 28.9	0.5	0.48	35.6 ± 26.4	43.1 ± 29.4	0.2	0.60
PPL	1.3 ± 5.0	19.0 ± 25.6	8.0	0.00	17.8 ± 18.4	24.4 ± 24.7	0.1	0.72
PRL	25.6 ± 24.1	12.3 ± 16.4	3.3	0.07	21.1 ± 25.3	7.6 ± 13.2	2.1	0.15
NT1	12.8 ± 22.6	24.6 ± 24.3	1.5	0.23	20.2 ± 13.6	38.6 ± 56.5	0.0	0.84
NT2	7.8 ± 4.6	23.5 ± 25.9	11.3	0.00	21.6 ± 17.2	25.8 ± 17.4	0.6	0.44
NT3	2.8 ± 2.4	7.8 ± 8.3	6.8	0.01	17.3 ± 10.5	31.4 ± 22.8	2.5	0.12
NT4	0.7 ± 0.9	0.4 ± 0.8	1.1	0.29	11.0 ± 9.9	11.0 ± 10.3	0.1	0.71
NOT	24.3 ± 24.0	57.0 ± 45.5	10.4	0.00	70.1 ± 24.0	107 ± 66	1.1	0.29
ATT	1158 ± 43.7	1135 ± 93.6	1.3	0.26	1761 ± 46.0	1591 ± 42	15.9	0.00
TII	20.6 ± 3.3	14.7 ± 4.7	17.5	0.00	19.3 ± 3.0	15.9 ± 4.8	3.6	0.06
NPM	1.3 ± 0.4	1.8 ± 0.8	6.2	0.01	1.4 ± 0.5	2.1 ± 0.9	6.3	0.02
DNV	1521 ± 542	1774 ± 715	0.7	0.40	4075 ± 869	4317 ± 1393	0.1	0.78
MOR	235 ± 217	501 ± 333	3.8	0.05	133 ± 216	408 ± 425	7.0	0.01

[a] Sample size.

ing habitat. In each colony, random points were plotted on the map with a numbered grid (Titus and Mosher 1981, Hubert 1993). Random points were always included within forested areas, excluding those habitats in which vultures did not nest (e.g., young pine plantations). Once random points were located, the nearest tree was randomly selected as the center of the plot.

To compare nearest-neighbor distances, only nests occupied during the 1994 breeding season were considered. As in other colonies (Bernis 1966, Hiraldo 1983), some pairs did not breed during the year of our study, but the vultures remained close to their nests during the breeding season. For this reason, we considered these nest sites as occupied in order to determine the distance between neighboring nests sites.

We considered all the nests found in the winter of 1994–95, even though some were likely old nesting sites of the same vulture pairs. An analysis of variance (STATISTICA 1993) using planned comparisons was designed to analyze for differences between nests and random plots, and to analyze for differences between habitats. This approach allowed us to include all nest sites in the analysis, avoiding potential pseudoreplication that may have resulted from including nest sites of the same birds. Variables deviating from normality were logarithmic transformed, and percentages were arcsin-transformed.

Results

All the nests found in the two colonies were in pines. In both colonies, we detected a tendency to nest in trees with the greatest diameter (DT, Table 2). Nest trees also were farther from neighboring trees (DTT) and, consequently, had lower canopy covers (PCC). These results indicated a tendency for Eurasian Black Vultures to nest in openings. Percentage and height of scrubland (PSC and SCH, respectively) did not affect nest-site selection.

In terms of nest-site topography, vultures frequently selected areas located in the middle-upper third of the mountains (NPM), with steep slopes (TII, Table 2). Both colonies occurred at high elevations (>1000 m). The minimum elevation (ATT) for a nest was 1090 m (colony A) and the maximum was 1880 m (colony B)

In relation to human disturbance, Eurasian Black Vultures nested in areas with fewer meters of road (MOR) in the surrounding area but the proximity of human habitation (DNV) did not appear

Table 2. Extended.

Diff. A–B		Diff. A–B	
Nests		Random	
F	*P*	F	*P*
(df = 1, 27)		(df = 1.66)	
0.0	0.84	41.3	0.00
0.2	0.69	46.8	0.00
5.9	0.03	1.0	0.32
1.6	0.23	0.0	0.91
0.4	0.54	1.1	0.31
0.8	0.40	0.0	0.98
3.6	0.07	5.5	0.02
8.6	0.00	15.5	0.00
15.6	0.00	3.1	0.09
0.7	0.18	1.5	0.23
1.4	0.25	1.0	0.32
1.9	0.18	0.1	0.72
12.5	0.00	46.9	0.00
19.2	0.00	78.0	0.00
6.3	0.02	13.2	0.00
136.2	0.00	253.3	0.00
0.1	0.78	1.2	0.27
2.3	0.14	1.2	0.29
39.5	0.00	88.1	0.00
5.9	0.03	2.0	0.17

to affect nest-site selection. The closest nest was 800 m from a village.

Forest B had a higher tree density (NT3, NT4, and NOT) with taller (HT) and thicker (DT) trees than those in forest A (Table 2). Forest B also had a smaller percentage of scrubland (PSC), higher elevation, and a greater distance from the closest village.

Nest trees used by vultures in colony A were taller than in random plots but, in colony B, they were not. Furthermore, in colony A, nest sites had a higher proportion of understory cover (PUC) with lower overstory tree density (NOT). There was also a trend to nest in sites with a higher percentage of rocky land (PRL). Vultures in colony B nested in areas with tree densities similar to those of random plots and with higher densities than those of colony A. This was due to the different structural characteristics of the two plantations, plantation B having a higher average tree density. Also, in colony A but not in B, the percentage of pastureland (PPL) was smaller in nest plots than in random plots.

We found a mean distance of 637 m between adjacent nest sites (DNN) of different pairs of vultures in the two colonies. The minimum distance was 150 m (colony A) and the maximum was 2325 m (colony B). The distance between neighboring nests was significantly shorter in colony A (379 m) than in B (1362 m).

Between colony differences in the nest-site selection in regard to DNV were due to the differences in the distance between both plantations, colony A being significantly closer to a village. MOR was similar in both forests but nest plots in colony A had significantly more meters of road than those in colony B.

Discussion

Management practices resulted in a modification of the pine forests we studied including alteration of structural diversity, sizes of trees, and species composition. These effects were more pronounced when forests were used for timber production than when they were used for resin collection. However, the current synthetic elaboration of glue has provoked a decreasing demand of resin in home and international markets causing the end of traditional management for resin extraction (Gil et al. 1990). Furthermore, the damage caused to the trees by the stem cuts results in a low quality of wood timber, which is hardly marketable (Gil et al. 1990). In this management type, the size of tree is less important and tree felling does not occur.

Eurasian Black Vultures, like other large vultures such as the Lappet-faced Vulture (*Torgos tracheliotus*), White-headed Vulture (*Trigonoceps occipitalis*), and White-backed Vulture (*Gyps africanus*), build their nests in the tops of trees (Houston 1974, Mundy 1982, Hustler and Howells 1988). Eurasian Black Vultures build large nests with a diameter of nearly 160 cm and 93 cm in height (Torres et al. 1980). These large nests frequently fall to the ground because of their weight (Bernis 1966). Most nest trees used in both colonies were older than 60 yr, indicating that vultures prefer mature trees in which to build their nests. This preference is probably due to the fact that large trees are necessary to hold their nests. Because there were only a few mature trees in colony A and these were distributed homogeneously on the upper and middle third of the mountain, their preference for older trees restricted the distribution of vultures in forest

A. In colony B, vultures used old trees restricted to the middle-upper third of the mountains where the forest had been protected from timber harvesting. There, trees located next to the crest of mountains are more exposed to hard climatic conditions and were shorter than expected in relation to the diameter of the stem.

Habitat around nests was characterized by open areas with pronounced slopes. Hiraldo and Donázar (1989) found Eurasian Black Vultures breeding in steep areas increase the amount of time available for searching for food and vultures breeding in plains are more dependent on thermals for flying (Hiraldo and Donázar 1989). Open areas with steep slopes may also provide greater visibility of predators, favor an easy nest access, and limit human disturbance (Titus and Mosher, 1981, Speiser and Bosakowski 1988, Donázar et al. 1993, Moorman and Chapman 1996). In colony B, vultures had fewer options to select nest sites because mature trees were only in protected areas. The larger percentage of pastureland in the random plots in colony A was due to the fact that a higher proportion of habitat was devoted to cattle grazing on the lower and middle thirds of the mountains.

The mean distance to the nearest-neighbor nest in colony A (379 m) was within the range (175–521 m) described by Hiraldo (1974), and significantly less than distances between random plots indicating a tendency for vultures to aggregate in this forest. For colony B this distance is much greater and did not differ from random plots. In forest B, vultures are forced to nest in protected areas while the apparently more natural conditions in forest A allowed them to select optimal nesting sites.

Human disturbance has been suggested as the main factor limiting Eurasian Black Vulture productivity (Garzón 1973, Hiraldo 1977, Bermejo 1990). Vultures in our study seemed to avoid human activity by nesting in areas with less paved and unpaved roads. There were no significant differences in road density between both forests, however, nesting areas in colony A contained a higher number of roads. This higher tolerance to roads was associated with the large number of old and abandoned roads left after resin extraction ended in the forest. The distance to the nearest village did not seem to be a factor that influenced nest-site selection, probably because the village was far enough away to be tolerated by this species.

This study demonstrates differences in habitat selection of Eurasian Black Vultures between areas subjected to different forestry practices. In forest A, where resin collection was carried out until 1985, traditional management afforded a relatively natural structure of the forest with few modifications in size class distribution and tree density. After the end of sylvicultural management, rapid occupation by vultures and homogeneous nest distribution was possible in this forest. In contrast, sylvicultural management for pine harvesting in forest B resulted in a greater modification of the size class distribution and tree density. Such forest management has forced the vultures to nest in protected areas located in the upper third of the mountains where there are mature trees resulting in a less homogeneous distribution of this vulture colony.

ACKNOWLEDGMENTS

We thank to L.M. Carrascal and J. Moreno for useful comments on earlier drafts. C. Cornelius, J. Torres, and F. Jaksić improved the final manuscript. Fernando Gómez, Mario Herrera and Ana Dimas participated in the field work. We also thank Stephie Michalsky for improving the English.

LITERATURE CITED

ANDREW, J.M. AND J.A. MOSHER. 1982. Bald Eagle nest-site selection and nesting habitat in Maryland. *J. Wildl Manage.* 46:383–390.

BERMEJO, C. 1990. Situación actual del Buitre Negro (*Aegypius monachus*) en la provincia de Avila. I Congreso Internacional sobre Aves Carroñeras, Cuenca. Aedenat-CODA. ICONA, Madrid, Spain.

BERNIS, F. 1966. El Buitre Negro (*Aegypius monachus*) en Iberia. *Ardeola* 12:45–99.

BIJLEVELD, M. 1974. Birds of prey in Europe. Macmillan Press Ltd, London and Basingstore, U.K.

BLANCO, J.C. AND J.L. GONZÁLEZ. 1992. Libro rojo de los vertebrados de España. MAPA-ICONA, Madrid, Spain.

BOSAKOWSKI, T., D.G. SMITH AND R. SPEISER. 1992. Niche overlap of two sympatric-nesting hawks *Accipiter* spp. in the New Jersey-New York Highlands. *Ecography* 15· 358–372.

BROWN, J.L. 1969. Territorial behavior and population regulation in birds. *Wilson. Bull.* 81:293–329.

DONÁZAR, J.A. 1993. Los buitres ibéricos: biología y conservación. J.M. Reyero [ED.]. Madrid, Spain.

———, F. HIRALDO AND J. BUSTAMANTE. 1993. Factors influencing nest-site selection, breeding density and breeding success in the Bearded Vulture (*Gypeatus barbatus*). *J. Appl. Ecol.* 30:504–514.

FERRER, M. AND J.A. DONÁZAR. 1996. Density-dependent fecundity by habitat heterogeneity in an increasing population of Spanish Imperial Eagles. *Ecology* 77:69–74.

FULLER, M.R. 1996. Forest raptor population trends in

North America. Pages 167–208 *in* R.M. DeGraaf and R.I. Miller [EDS.], Conservation of faunal diversity in forested landscapes. Chapman-Hall, London, U.K.

GALERA, R.M. AND S. MARTÍN. 1990. Regiones de procedencia de *Pinus sylvestris* L. en España: su uso en las repoblaciones. *Ecología* (special issue) 1:527–540.

GARZÓN, J. 1973. Contribución al estudio del estatus, alimentación y protección de las falconiformes de España central. *Ardeola* 19:280–330.

GIL, L., J. GORDO, R. ALÍA, G. CATALÁN AND J.A. PARDOS. 1990. *Pinus pinaster* Aitón en el paisaje vegetal de la Península Ibérica. *Ecología* (special issue) 1:469–496.

GONZÁLEZ, L.M. 1990. Censo de las poblaciones reproductoras de Aguila Imperial y Buitre Negro en España. *Quercus* 58:16–22.

HIRALDO, F. 1974. Colonias de cría y censo de los Buitres Negros (*Aegypius monachus*) en España. *Naturalia Hispanica* 2.

———. 1977. El Buitre Negro (*Aegypius monachus*) en la Península Ibérica. Ph.D. dissertation. Universidad de Sevilla, Sevilla, Spain.

———. 1983. Breeding biology of the Cinereous Vulture. Pages 193–203 *in* S.R. Wilbur and J.A. Jackson [EDS.], Vulture biology and management. Univ. California Press, Berkeley, CA U.S.A.

——— AND J.A. DONÁZAR. 1989. Foraging time in the Cinereous Vulture *Aegypius monachus*: seasonal and local variations and influence of weather. *Bird Study* 37:128–132.

HOUSTON, D.C. 1974. Breeding of the White-backed and Rüppell's Griffon vultures, *Gyps africanus* and *G. ruppellii*. *Ibis* 118:14–40.

HUBERT, C. 1993. Nest-site habitat selected by Common Buzzard (*Buteo buteo*) in southeastern France. *J. Raptor Res.* 27:102–105.

HUSTLER, K. AND W.W. HOWELLS. 1988. Breeding biology of the Whitehead Vulture in Hwange National Park, Zimbabwe. *Ostrich* 59:21–24.

KADMON, R. 1993. Population dynamics consequences of habitat heterogeneity: an experimental study. *Ecology* 74:826–825.

MALLADA, L. 1892. Catálogo general de las especies fósiles encontradas en España. Boletín de la Comisión del Mapa Geológico de España 18:1–253.

MOORMAN, C.E. AND B.R. CHAPMAN. 1996. Nest-site selection of Red-shouldered and Red-tailed hawks in a managed forest. *Wilson Bull.* 108:357–368.

MUNDY, J.P. 1982. The comparative biology of southern African vultures. Vulture Study Group, Johannesburg, South Africa.

NIEMI, G.J. AND J.M. HANOWSKI. 1997. Concluding remarks on raptor responses to forest management. a Holarctic perspective. *J. Raptor Res.* 31:191–196.

PRODON, R. 1976. Le substrate, facteur écologique de la vie en eau courante: observations et experiénces sur les larves de *Microptera testacea* et *Cordulogaster annulatus*. Ph.D. dissertation, Univ. Lyon, Lyon, France.

RUBIO, J.L. AND L.M. CARRASCAL. 1994. Habitat selection and conservation of an endemic Spanish Lizard *Algyroides marchi* (Reptilia. Lacertidae). *Biol. Conserv.* 70 245–250.

SPEISER, R.T. AND T. BOSAKOWSKI. 1988. Nest-site preferences of Red-tailed Hawks in the Highlands of southeastern New York and northern New Jersey. *J. Field Ornithol.* 59:361–368.

STATISTICA. 1993. Zoologiska Institute. Uppsala Univ, Uppsala, Sweden.

TITUS, K. AND J.A. MOSHER. 1981. Nest-site habitat selected by woodland hawks in the central Appalachians. *Auk* 98:270–281.

TORRES, J.A., P. JORDANO AND J. VILLASANTE. 1980. Estructura y dinámica temporal de una colonia de Buitre Negro, *Aegypius monachus*, en Sierra Morena central (Cordoba). *Bol. Est. Cent. Ecol.* 9:67–71.

TUCKER, G.M. AND M.F. HEATH, EDS. 1994. Birds in Europe: their conservation status. BirdLife International, Cambridge, U.K.

Received 12 October 1997; accepted 15 May 1998

J Raptor Res. 32(3):208–214
© 1998 The Raptor Research Foundation, Inc.

SELECTION OF SETTLEMENT AREAS BY JUVENILE BONELLI'S EAGLE IN CATALONIA

Santi Mañosa, Joan Real and Jordi Codina
Departament de Biologia Animal, Facultat de Biologia, Universitat de Barcelona, Avinguda Diagonal, 645, 08028 Barcelona, Catalonia, Spain

Abstract.—We observed Bonelli's Eagles (*Hieraaetus fasciatus*) outside their breeding range in Catalonia (northeastern Spain) to identify the main dispersal areas of juvenile eagles in the Central Catalan Basin. Autumn counts were conducted to determine the size of the nonbreeding population in the main dispersal area and to analyze factors leading to the selection of settlement areas by juveniles. The permanent eagle population in the area was estimated at about 18 >1-yr-old eagles for an average density of 0.90 eagles/100 km^2. Number of eagles in the area <1-yr old fluctuated seasonally between 17–94 eagles with the number reaching a maximum in early autumn followed by a rapid decline during winter. Juvenile settlement areas appeared to be selected based on gamebird and rabbit abundance, rather than on landscape variables. As a consequence, game management measures appeared to be key in the conservation of Bonelli's Eagles.

Key Words: *Bonelli's Eagle;* Hieraaetus fasciatus; *population density; conservation; habitat selection; juvenile dispersal; Spain.*

Selección de zonas de asentamiento juvenil por el águila perdicera en Cataluña

Resumen.—Describimos la localización y características ambientales de la principal zona de dispersión juvenil de las águilas perdiceras *Hieraaetus fasciatus* en Cataluña (España). Se realizaron conteos otoñales en coche con objeto de evaluar el tamaño de la población de águilas no reproductoras que utilizan esta zona, y se analizan los factores que conducen a la selección de las áreas de asentamiento dentro de esta región. La población de águilas dispersantes está formada por una población permanente de unos 18 ejemplares de más de 1 año de edad, a una densidad de 0.90 aves/100 km^2 , y una población estacionalmente fluctuante de 17–94 aves de menos de un año de edad, máxima a principios de otoño y que disminuye rápidamente a lo largo del invierno. Las águilas seleccionan sus áreas de asentamiento en función de la abundancia de perdiz y conejo, mientras que las características del paisaje aparecen como un factor mucho menos importante. La gestión correcta de la caza menor constituye una pieza clave para la supervivencia de las águilas perdiceras jóvenes en sus áreas de asentamiento temporal.
[Traducción Autores]

In long-lived birds of prey, the acquisition of sexual maturity is delayed for several years which means that nonbreeding individuals, whose behavior and ecology may differ from that of breeding birds, form an important fraction of the total population (Newton 1979, Ferrer 1993a, Omland and Hoffman 1996, Bustamante et al. 1997). In some species, nonadult birds (those not in definitive plumage) tend to settle in areas outside the breeding range in juvenile dispersal or temporary settlement areas where they stay for variable periods before moving to another settlement area or joining breeding populations (González et al. 1989, Ferrer 1993b, 1993c). Lack of appropriate settlement areas or reduction in their habitat quality may result in a reduction in nonadult survival and have serious effects on population stability. Full understanding of the habitat requirements of long-lived raptors during the nonadult stages of their lives is, therefore, necessary to design appropiate conservation measures for threatened or declining populations.

The Bonelli's Eagle (*Hieraaetus fasciatus*) is such a long-lived raptor. It does not mature sexually until 2–4 yr of age (Newton 1979, del Hoyo et al. 1994, Real and Mañosa 1997) and it is an Endangered Species whose populations have declined across Europe (Rocamora 1994, Real and Mañosa 1997). The current Spanish breeding population of 675–751 pairs represents about 80% of the European population (Arroyo 1991, Real et al. 1994, Real et al. 1996). The nonbreeding population is

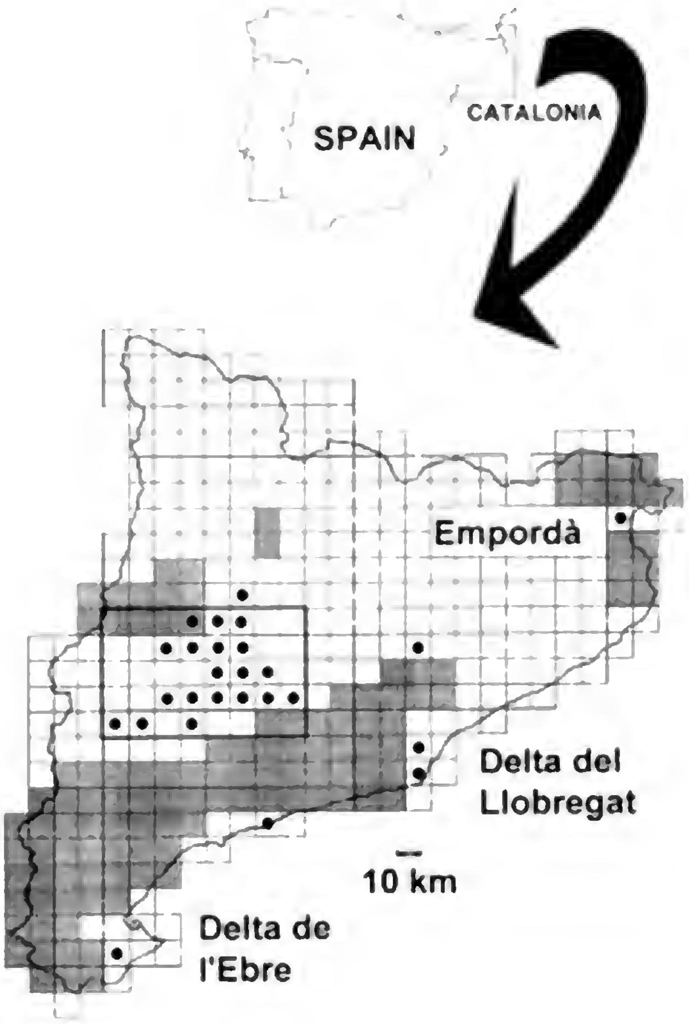

Figure 1. Breeding range (shaded area) and dispersal areas (dotted squares) of Bonelli's Eagles in Catalonia (northeastern Spain). The area depicted in Fig. 2 is outlined.

estimated at approximately 700 eagles (Real and Mañosa 1997). These nonbreeders are commonly seen in areas unsuitable for nesting (Cramp and Simmons 1980, Cugnasse and Cramm 1990, Real et al. 1990, Cheylan et al. 1996), but little is known concerning the exact locations and physical characteristics of these settlement areas (González et al. 1989, Ferrer 1990, Cugnasse and Cramm 1990, Real and Mañosa 1992, Real et al. 1994). The aims of this study were to describe the locations and general characteristics of juvenile dispersal areas of Bonelli's Eagles in Catalonia (northeastern Spain), to estimate the size, age structure, and temporal dynamics of the nonbreeding population in these areas, and to describe factors leading to the selection of these settlement areas by nonadult Bonelli's Eagles.

Study Area and Methods

In Catalonia, the Bonelli's Eagle breeds along the coastal and pre-pyrenean mountain ranges (Fig. 1). A total of 70–80 pairs, or about 10% of the European breeding population, breeds in this area (Real and Mañosa 1992). From 1980–94, we compiled observations from as many ornithologists as possible on Bonelli's Eagles outside their breeding range. Results were plotted on 10 × 10 km Universal Transverse Mercator (UTM) squares showing the presence or absence of nonbreeding eagles in each square of the grid (Fig. 1).

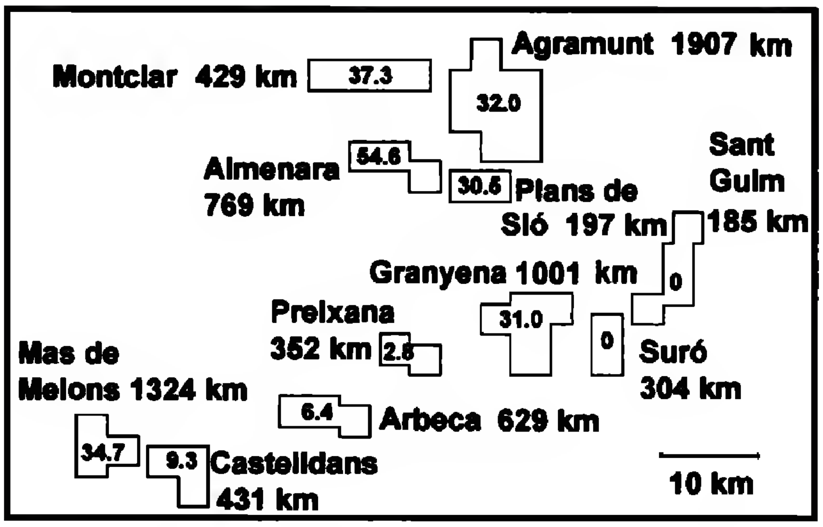

Figure 2. Areas surveyed during autumn eagle counts. The number of km driven and the linear index of abundance of eagles/1000 km are given for each sector.

We also conducted counts of Bonelli's Eagles in the autumn and winter in central Catalonia in the main nonbreeding area where eagles were more seen. Counts were conducted for 3 yr from August–March 1991–94. They were made to identify the areas used by eagles and to estimate the number of eagles in the region. We selected 11 sectors (Fig. 2), each one exhibiting uniform landscape in terms of topography, land use, and vegetation cover and patterns. Sectors were selected in order to represent all types of landscape patterns available to eagles. Each sector was driven by car once a week at a speed not exceeding 40 km/hr. Because of the large area to be covered on each sampling day, counts were conducted continuously from sunrise to sunset. In order to obtain an equal sample for each sector at all times of the day, the sampling sequence for sectors was changed each day, but travel constraints prevented perfect randomization. To analyze potential biases introduced by different sampling schedules between sectors, counts were grouped into four time intervals: early morning (before 0700 H), morning (0700–1200 H), afternoon (1200–1700 H), and evening (after 1700 H).

Transects were driven by teams of two observers with each team including at least one trained observer and a driver. Each team counted eagles in all of the sectors so bias among observers was reduced. Each time a sector was sampled, time of day, number of km driven, number of Bonelli's Eagles, and number of prey species (Red-legged Partridge [*Alectoris rufa*] and European rabbits [*Oryctolagus cuniculus*]) were recorded. When possible, eagles were classified as juvenile (<1 yr), immature (1–2 yr), subadult (2–3 yr), or adult (>3 yr) according to plumage criteria (Parellada 1986). Although some practice is needed to sex Bonelli's Eagles in the field, the most probable gender of the eagles was determined based on size and color, females being larger and darker than males (Parellada 1986). When possible, eagles were classified as having full or empty crops. From these data, linear indices of abundance for partridges, rabbits, and each age class of eagles were computed for each sector.

Distance sampling (Buckland et al. 1993) was used to estimate the abundance and density of eagles within the sectors where they were observed. Each eagle observation was plotted on a 1:50 000 map and the perpendicular distance to the transect was measured. An overall detection function was fitted to these data using the DISTANCE software (Laake et al. 1993). Because most contacts (93%) involved single birds, the model was fit by considering each eagle as a single object. To obtain a more robust estimate of the detection function, observations were truncated at 1.0 km after visual inspection of the distance histogram (Buckland et al. 1993). The detection function allowed the estimation of the Effective Strip Width for the counts and an overall density estimate was computed from sector density estimates weighted by the area of each sector.

To determine habitat selection, each sector was divided on 3 × 3 km UTM squares (900 ha or 9 km^2 each) on which landscape variables (Table 1) were measured from a CORINE Land Cover 1:250 000 habitat digital database processed by means of Arc/Info software, and 1:50 000 military maps. The landscape of every sector was then described using the same variables averaged over each sector. The area variables were measured on the 900 ha unit squares from each sector and could be considered as proportions, so they were better analysed by converting them to log-ratios (Table 1) to make them independent from one another (Robertson et al. 1993). Spearman rank correlation coefficients between eagle abundance indices and partridge indices, rabbit indices, habitat variables, and land cover log-ratios were computed to show the cause of spatial variation in eagle abundance between sectors.

RESULTS

Location and General Characteristics of the Juvenile Dispersal Areas. We compiled 81 independent observations of Bonelli's Eagles outside their usual breeding range in Catalonia. Some occurred along the coast, in wetland areas such as Aiguamolls de l'Empordà, Delta del Llobregat, and Delta de l'Ebre, but most observations came from the Central Catalan Basin. Overall, 14 records were reported in coastal areas (2.8 records per 100 km^2) and 67 in inland areas (3.19 records per 100 km^2) (Fig. 1). The main juvenile dispersal area in Catalonia was a belt of extensive dry farmland, bounded by the Pyrenees, the litoral mountains, and the irrigated lands of central Catalonia. This is a nearly flat region, lacking cliffs and large forests, and is mainly devoted to cereal crops and to a lesser extent olive, almond and vineyards.

Origin, Sex-ratio, Age-ratio and Abundance of Birds. Within the 11 sectors, 355 autumn counts were conducted along 7528 km. Eagles were observed on only 322 counts from nine sectors over 7039 km. In these counts, Bonelli's Eagles were observed on 196 occasions totalling 211 birds. One eagle was observed on 183 (93%), two on 11 (6%), and three on two (1%) occasions. Over 25% of eagles had wing tags that had been attached in 1989–93 while they were nestlings in the nearby Catalan breeding population. Forty (27%) of the juvenile, six (18%) of the immature, and four (28%) of the subadult eagles had wing tags. No eagles tagged outside Catalonia (24 eagles in southeastern France and 60 eagles in Murcia-Alacant, southeastern Spain) were seen in our survey.

Table 1. Names, definition, units, and source for habitat variables recorded in juvenile Bonelli's Eagle settlement areas.

NAME	DESCRIPTION, UNITS AND SOURCE
ROAD	Paved roads (km/9 km^2); CORINE
TRAC	Unpaved roads and tracks (km/9 km^2); CORINE
URBA	Urban, residential, and other developed areas (ha/9 km^2); CORINE
CROP	Arable land (ha/9 km^2); CORINE
BUSH	Low bush and dry grassland (ha/9 km^2); CORINE
WOOD	Woodland (ha/9 km^2) (m^2); CORINE
MIXE	Land occupied by agriculture with areas of natural vegetation (ha/9 km^2); CORINE
ALTI	Average elevation (m) (Minimum elevation + Maximum elevation)/2; 1:50 000 map
RELI	Relief index (total number of 20 m isolines crossed by the diagonals of the 3 × 3 km square [*N*]); 1:50 000 map
IKARUFA	Number of partridges/1000 km; Linear counts
IKACUNI	Number of rabbits/1000 km; Linear counts
IKAFASC	Number of Bonelli's Eagles/1000 km; Linear counts
LR-WC	Log-ratio woodland-crop: ln(wood/(crop+1)+1)
LR-BC	Log-ratio bush-crop: ln(bush/(crop+1)+1)
LR-MC	Log-ratio mixed-crop: ln(mixed/(crop+1)+1)
LR-BW	Log-ratio bush-wood: ln(bush/(wood+1)+1)
LR-MW	Log-ratio mixed-wood: ln(mixed/(wood+1)+1)
LR-MB	Log-ratio woodland-crop: ln(mixed/(bush+1)+1)

For eagles whose sexes were determined (24%), 32 were male (63%) and 19 female (37%; Binomial test, $P = 0.093$). For wing-tagged birds of known sex, six were male and three female.

Table 2. Summary statistics for the variables recorded in sectors where Bonelli's Eagles were detected (N = 9) or not detected (N = 2). Mean ± SD (minimum-maximum)

	EAGLES PRESENT (N = 9)	EAGLES ABSENT (N = 2)
ROAD	2.1 ± 0.7 (0.9–3.3)	2.3 ± 2.5 (0.5–4.1)
TRAC	23.3 ± 6.1 (11.4–34)	26.3 ± 1.0 (25.6–27.1)
URBA	2.3 ± 2.3 (0–6.5)	2.0 ± 1.5 (0.9–3.0)
CROP	640 ± 195 (274–898)	618 ± 151 (510–725)
BUSH	49 ± 70 (0–223)	7 ± 7 (2–11)
WOOD	23 ± 28 (0–67)	45 ± 2 (43–46)
MIXE	186 ± 182 (213–567)	229 ± 162 (114–343)
ALTI	402 ± 65 (307–544)	676 ± 18 (663–689)
RELI	35 ± 10 (21–48)	50 ± 10 (43–56)
IKARUFA	2582 ± 1900 (182–5470)	1115 ± 362 (859–1372)
IKACUNI	78 ± 90 (8–289)	39 ± 1 (38–39)
IKAFASC	26 ± 17 (3–55)	—

No significant difference in the linear index of abundance of eagles was found between periods when the six sectors visited in every period were included (Kruskall-Wallis test, x^2 = 0.05, df = 2, P = 0.98), or when the 10 sectors visited in both the last two periods were compared (Kruskall-Wallis test, x^2 = 0.006, df = 1, P = 0.94). There was also no significant difference between the linear indices of abundance of nonjuvenile (immatures, subadults, and adults) eagles between periods (Kruskall-Wallis test, x^2 = 1.377, df = 2, P = 0.50 or x^2 = 0.006, df = 1, P = 0.94). Therefore, periods were considered together.

Of the 198 individuals we aged, 145 (73%) were juvenile, 33 (17%) immature, 14 (7%) subadult, and six (3%) adult eagles. Two (1%) were only identified as >1 yr of age. The observation rate for nonjuveniles remained fairly stable around 0.75 eagles/100 km during autumn and winter, while the observation rate of yearlings reached a maximum of 3.2 eagles/100 km in early autumn and declined to 0.72 eagles/100 km in December–March (Fig. 4). As a consequence, the age ratio of nonjuveniles to juveniles increased from 0.37 in August to 1.00 after November. The seasonal variation on eagle abundance correlated to a similar variation in Red-legged Partridge abundance (r_s = 0.73; P = 0.015, N = 10).

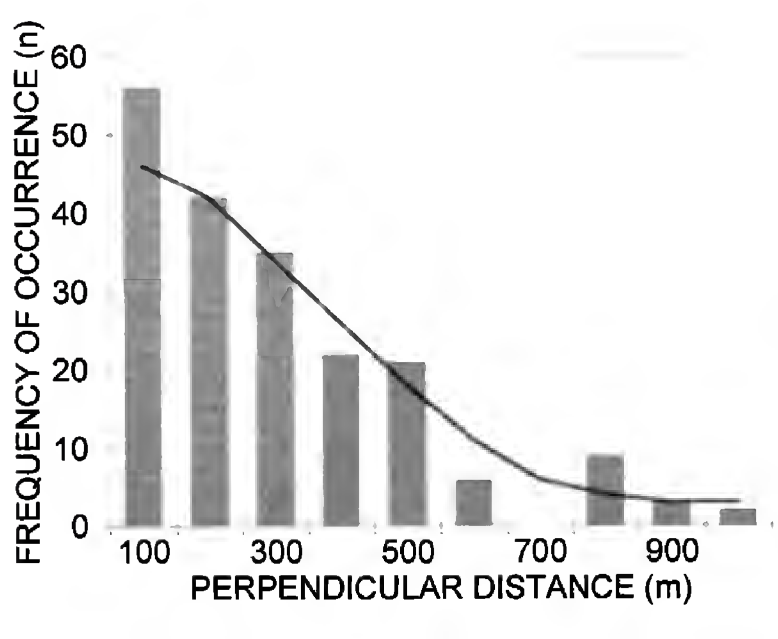

Figure 3. The number of eagles observed in relation to the perpendicular distance to the transect line. The solid line shows the shape of the adjusted detection model.

The age ratio also varied between sectors (0.91 in Mas de Melons, N = 42 eagles; 0.67 in Almenara, N = 40 eagles; 0.45 in Montclar, N = 16 eagles; 0.16 in Granyena, N = 29 eagles; and 0.07 in Agramunt, N = 59 eagles), but was not significantly related to eagle abundance (r_s = 0.68; P = 0.205, N = 5) or prey abundance (r_s = −0.50; P = 391, N = 5). The linear index of abundance of young eagles was positively correlated with the linear index of abundance of mature eagles (r_s = 0.63; P = 0.04, N = 11).

A uniform function with two cosine adjustments was found to be the best model to fit the perpendicular distance data (Fig. 3). After truncation at 1.0 km, only 196 birds remained in the analysis, and the Effective Strip Width was estimated at 416 m (95% C.I. = 370–469). For the sectors where eagles were observed, we obtained an overall encounter rate of 2.7 eagles/100 km (95% C.I.= 1.9–3.8), and an average density estimate of 3.2 eagles/

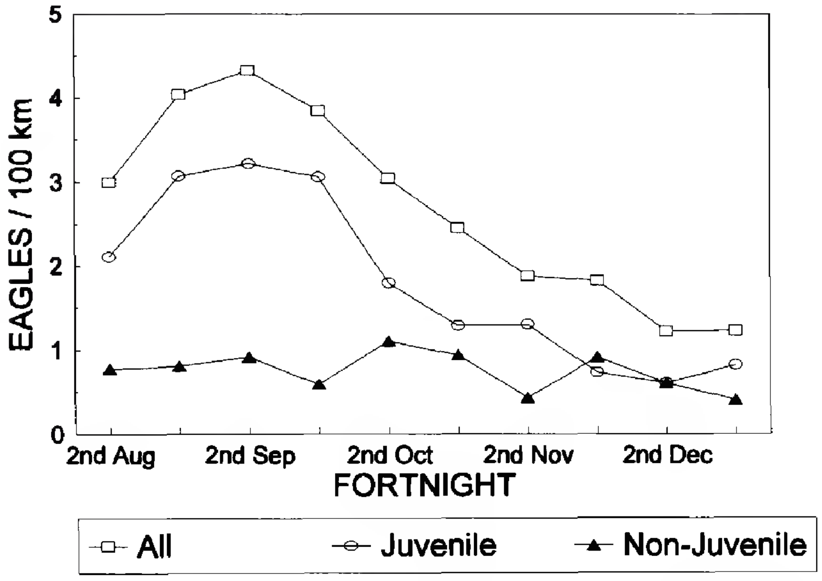

Figure 4. The number of Bonelli's Eagles in the central Catalonia dispersal area according to the time of the year expressed as the number of individuals seen/100 km of transect.

100 km^2 (95% C.I.= 2.3–4.5). Maximum eagle density was attained in late September (5.2 eagles/100 km^2), and minimum density in January–March (1.5 eagles/100 km^2). The number of mature eagles remained fairly stable at about 0.90 eagles/100 km^2, while the number of young showed large variation from late September (3.9 eagles/100 km^2) to late winter (0.73 eagles/100 km^2, Fig. 4). If these density estimates are extrapolated to the entire potential dispersal area in central Catalonia (dotted area in Fig. 1 = 2000 km^2), an estimate of 18 nonjuvenile eagles and 17–94 yearlings would be obtained for the entire area.

Selection of Settlement Areas, Eagle Behavior and Habitat Use. Although the sequence of sampling was changed on each sampling day, differences in the average sampling time between sectors were detected (Kruskall-Wallis test, $P < 0.000$). However, a two-way ANOVA of the square-root transformed $(x + \frac{3}{8})^{1/2}$ index of eagle abundance (Zar 1984) showed no significant relationship between eagle abundance indices and time of day ($F_{3,323} = 0.759$; $P = 0.518$) and a significant effect of sector ($F_{10,323} = 5.537$; $P < 0.000$), so differences in eagle abundance between sectors were not the result of a different sampling pattern (Fig. 2). A large variation was found in the habitat characteristics of the sectors where eagles were observed (Table 2). The linear index of abundance of Bonelli's Eagles was positively correlated with the linear index of abundance of Red-legged Partridge ($r_s = 0.74$; $P = 0.01$, $N = 11$), and to the log-ratio of bush-woodland ($r_s = 0.73$; $P = 0.01$, $N = 11$). If only the nine sectors where eagles were observed were considered, the linear index of abundance of Bonelli's Eagles was found to be positively correlated with the Red-legged Partridge linear index of abundance ($r_s = 0.72$; $P = 0.03$, $N = 9$), the rabbit linear index of abundance ($r_s = 0.77$; $P = 0.02$, $N = 9$), the log-ratio mixed-crop ($r_s = 0.87$; $P = 0.002$, $N = 9$), and the log-ratio bush-woodland ($r_s = 0.71$; $P = 0.031$, $N = 9$).

Within the settlement areas, eagles were observed feeding on Red-legged Partridge on one occasion, twice on European rabbits, and once on Quarry Guineafowl (*Numida meleagris*). Unsuccessful attacks were observed once on feral pigeon (*Columba livia*), once on Wood Pigeon (*Columba palumbus*), once on pigeon (*Columba sp.*), twice on Red-legged Partridges, and once on an unidentified bird. Full crops were observed in 43% of the eagles whose crop contents could be determined ($N = 120$, 57% of all eagles). This was probably an overestimate since empty crops were probably recorded as unnoticed more than full crops. For those sectors in which crop contents were estimated for >5 eagles, those with a higher linear index of eagle abundance had a larger proportion of eagles with full crops ($r_s = 0.98$; $P = 0.02$, $N = 4$). In some sectors, eagles were more often observed perching (Plans de Sió 100%, Almenara 69%, Mas de Melons 50%) than in others (Montclar 25%, Granyena 29%, Agramunt 31%), but this was not related to eagle abundance ($r_s = -0.20$; $P = 0.70$, $N = 6$). Of 97 perches observed, trees were the most frequently used (52%) followed by large rocks (19%), the ground (18%), large transport power poles (6%), buildings (4%), and small distribution power poles (1%).

Discussion

Eagle abundance was related to partridge and rabbit abundance in the dispersal area. In sectors with larger eagle concentrations, eagles with full crops were more frequently encountered and these areas had more bush, dry grassland or cropland mixed with natural habitats relative to woodland or homogeneous farmland. These open habitats were likely the most suitable for eagles because of their prey abundance and because their open habiat maximized eagle foraging success (Bohall and Collopy 1984, Janes 1985, Preston 1990). Therefore, given the large variation in habitat pattern between sectors where eagles were observed, we concluded that nonadult Bonelli's Eagles selected settlement areas mainly on food availability rather than their topographic and landscape patterns. A similar habitat selection strategy has been described in other species of birds, in which food is a proximate factor for habitat selection (Hildén 1965, Hutto 1985, González et al. 1989, Ferrer 1990, Gerrard et al. 1990, Heredia et al. 1991, Bustamante et al. 1997).

Although yearlings form the bulk of the nonbreeding Bonelli's Eagle population in central Catalonia, older birds form the stable fraction of the population. Yearlings arrive in the region in late summer, following independence (Real et al. 1989), and probably occupy only those areas left available by older birds which die, move to other areas, or are recruited into the breeding population. In these dispersal areas, young eagles find sufficient food and probably avoid competition with breeders. The seasonal decline of the dispersal population after October probably results from a combination of factors including further innate dispersal (Horn 1983), competitive exclusion (Waser 1985), juvenile mortality (Real et al. 1996, Real and Mañosa 1997), or a decline in prey availability (Newton 1979, Ferrer 1993b, Brodeur et al. 1996) which was supported by the fact that the seasonal decline of eagle abundance matched that of Red-legged Partridges. A continuous turnover of eagles with other dispersal areas, temporary returns to natal areas, or movements to nearby breeding areas may also occur as has been described in the Spanish Imperial Eagle (Ferrer 1993c).

According to our wing-tagging information, the Catalan breeding population is the main source of Bonelli's Eagles that come to the dispersal area in central Catalonia. A higher number of males were observed in the area than females. This suggested that there may be female-biased juvenile dispersal in this species, as is commonly found in raptors (Clarke et al. 1997) or, alternatively, a sex-biased mortality (Ferrer and Hiraldo 1992). Given the size and demographic parameters (Real and Mañosa 1997) of the breeding population in Catalonia, every year 55 independent juveniles, 22 immatures, and 9 subadults enter this eagle population. The dispersal area in central Catalonia can only give permanent refuge to about 35 of these birds, so many must disperse farther away. Although the mechanisms of dispersal and recruitment in Bonelli's Eagles are only partially understood, in our opinion increasing the carrying capacity of dispersal areas adjacent to breeding areas would contribute to reduced juvenile dispersal, decrease preadult mortality, and improve recruitment rates in nearby breeding areas (González et al. 1989), which would be particularly helpful to stop the decline of isolated subpopulations or those found on the edge of the species range. Sensible game management is an essential tool to achieve this objective. However, reduced dispersal will only be advantageous if the main mortality factors for eagles in the area (Real et al. 1996, Real and Mañosa 1997) are also eliminated.

Acknowledgments

We are indebted to Rodrigo del Amo, Joan Manel Baqués, Dani Díaz, Joan Estrada, Daniel González, Ferran Gonzàlez-Prat, Glòria Laviga, Montse López, Marc Noguera, Vittorio Pedrocchi and Montserrat Tortosa for contributing to field work. We also thank Dr. J. Loman, Dr. H. Källander, Dr. G.R. Bortolotti and Dr. M. McGrady for their useful comments on earlier versions of the manuscript. Josep Garriga, Cap del Servei d'Agents Rurals del Departament d'Agricultura, Ramaderia i Pesca de la Generalitat de Catalunya, provided access to the CORINE database, and Emili Ponsa helped in the extraction of data from it. We especially thank Fundació Miquel Torres for continous financial support of Bonelli's Eagle conservation in Catalonia. We also thank Caixa de Sabadell for financial support of the fieldwork on which this paper is based. The Fundació Bosch i Gimpera cooperated in the administration of part of the project. The first author received a postdoctoral grant from Comissió Interdepartamental de Recerca i Innovació Tecnològica (CIRIT) de la Generalitat de Catalunya (BPOST-9316).

Literature Cited

Arroyo, B. 1991. Resultados del censo nacional de águila perdicera. *Quercus* 70:17.

Bohall, P.G. and M.W. Colloy. 1984. Seasonal abundance, habitat use, and perch sites of four raptor species in north-central Florida. *J. Field Ornithol.* 55:181–189.

Brodeur, S., R. Décarie, D.M. Bird and M. Fuller. 1996. Complete migration cycle of Golden Eagles breeding in northern Quebec. *Condor* 98:293–299.

Buckland, S.T., D.R. Anderson, K.P. Burnham and J.L Laake. 1993. Distance sampling. Chapman and Hall, London, U.K.

Bustamante, J., J.A. Donázar, F. Hiraldo, O. Ceballos and A. Travaini. 1997. Differential habitat selection by immature and adult Grey Eagle-Buzzards *Geranoaetus melanoleucus*. *Ibis* 139:322–330.

Cheylan, G., A. Ravairol, J.M. Cugnasse, J.M. Billet and C. Joulot. 1996. Dispersion des aigles de Bonelli *Hieraaetus fasciatus* juvéniles bagués en France. *Alauda* 64:413–419.

Clarke, A.L., B.E. Sæther and E. Røskaft. 1997. Sex biases in avian dispersal: a reappraisal. *Oikos* 79:429–438.

CUGNASSE, J.M. AND P. CRAMM. 1990. L'erratisme de l'aigle de Bonelli *Hieraaetus fasciatus* en France. *Alauda* 58:59–66.

CRAIG, E.H., T.H. CRAIG AND L.R. POWERS. 1986. Habitat use by wintering Golden Eagles and Rough-legged Hawks in southern Idaho. *J. Raptor Res.* 20:69–71.

CRAMP, S. AND K.E.L. SIMMONS [EDS.]. 1980. The birds of the western Palearctic. Vol. 2. Hawks to Bustards. Oxford Univ. Press, Oxford, U.K.

DEL HOYO, J., A. ELLIOT AND J. SARGATAL [EDS.]. 1994. Handbook of the birds of the world. Vol. 2. New World Vultures to Guineafowl. Lynx Edicions, Barcelona, Spain.

FERRER, M. 1990. Dispersión juvenil de la población de águilas imperiales del Parque Nacional de Doñana. Ph.D. dissertation, Universidad de Sevilla, Sevilla, Spain.

———. 1993a. El águila imperial. Eurofauna, Madrid, Spain.

———. 1993b. Reduction in hunting success and settlement strategies in young Spanish Imperial Eagles. *Anim. Behav.* 45:406–408.

———. 1993c. Juvenile dispersal behavior and natal philopatry of a long-lived raptor, the Spanish Imperial Eagle *Aquila adalberti. Ibis* 135:132–138.

——— AND F. HIRALDO. 1992. Man-induced sex-biased mortality in the Spanish Imperial Eagle. Biol. Conserv. 60:57–60.

GERRARD, J.M., G.R. BORTOLOTTI, E.H. DZUS, P.N. GERRARD AND D.W.A. WHITEFIELD. 1990. Boat census of Bald Eagles during the breeding season. *Wilson Bull.* 102:720–726.

GONZÁLEZ, L.M., B. HEREDIA, J.L. GONZÁLEZ AND J.C. ALONSO. 1989. Juvenile dispersal of Spanish Imperial Eagles. *J. Field Ornithol.* 60:369–379.

HEREDIA, B., J.C. ALONSO AND F. HIRALDO. 1991. Space and habitat use by Red Kites *Milvus milvus* during winter in the Guadalquivir marshes: a comparison between resident and wintering populations. *Ibis* 133: 374–381.

HILDÉN, O. 1965. Habitat selection in birds: a review. *Ann. Zool. Fenn.* 2:53–75.

HORN, S.H. 1983. Some theories about dispersal. Pages 55–62 *in* R.S. Swingland and P.J. Greenwood [EDS.], The ecology of animal movement. Oxford Univ. Press, Oxford, U.K.

HUTTO, R.L. 1985. Habitat selection by non-breeding migratory land birds. Pages 455–476 *in* M.L. Cody [ED.], Habitat selection in birds. Academic Press, Orlando, FL U.S.A.

JANES, S.W. 1985. Habitat selection in raptorial birds. Pages 159–188 *in* M.L. Cody [ED.], Habitat selection in birds. Academic Press, Orlando, FL U.S.A.

LAAKE, J.L., S.T. BUCKLAND, D.R. ANDERSON AND K.P. BURNHAM. 1993. Distance user's guide, Version 2.0. Colorado Coop. Fish Wildl. Res. Unit, Colorado State Univ., Fort Collins, CO U.S.A.

NEWTON, I. 1979. Population ecology of raptors. T. & A.D. Poyser, Berkhamsted, U.K.

OMLAND, K.S. AND S.W. HOFFMAN. 1996. Seasonal, diet, and spatial dispersion patterns of Golden Eagles migration in southern Montana. *Condor* 98:633–636.

PARELLADA, X. 1986. Variació del plomatge i identificació de l'àliga cuabarrada *Hieraaetus fasciatus fasciatus.* Pages 87–96 *in* Rapinyaires Mediterranis, II. Centre de Recerca i Protecció de Rapinyaires, Barcelona, Spain.

PRESTON, C.R. 1990. Distribution of raptor foraging in relation to prey biomass and habitat structure. *Condor* 92:107–112.

REAL, J. AND S. MAÑOSA. 1992. La Conservació de l'àliga perdiguera a Catalunya. Univ. Barcelona, Barcelona, Spain.

——— AND ———. 1997. Demography and conservation of western European Bonelli's Eagle *Hieraaetus fasciatus* populations. *Biol. Conserv.* 79:59–66.

———, S. MAÑOSA, J. BERTRAN, P. ROMERO, A. SOROLLA, J. FERNÀNDEZ AND L. BALAGUER. 1989. El procés d'independització en els joves d'àliga perdiguera, *Hieraaetus fasciatus,* al Parc Natural de Sant Llorenç del Munt i l'Obac. Pages 43–47 *in* I Trobada d'estudiosos de Sant Llorenç del Munt i l'Obac. Diputació de Barcelona, Servei de Parcs Naturals, Barcelona, Spain.

———, S. MAÑOSA, R. DEL AMO, J.A. SÁNCHEZ ZAPATA, M.A. SÁNCHEZ, D. CARMONA AND J.E. MARTÍNEZ. 1990 La regresión del águila perdicera: una cuestión de demografía. *Quercus* 70:6–12.

———, S. MAÑOSA AND J. CODINA. 1996. Estatus, demografía y conservación del águila perdicera (*Hieraaetus fasciatus*) en el Mediterráneo. Pages 83–89 *in* J. Muntaner and J. Mayol [EDS.], Biología y conservación de las rapaces mediterráneas, 1994. Monografías, No. 4 SEO, Madrid, Spain.

——— S. MAÑOSA, J. CODINA AND R. DEL AMO. 1994. Estado de varias poblaciones de águila perdicera en Europa occidental durante 1993. *Quercus* 98:8–11.

ROBERTSON, P.A., M.I.A. WOODBURN, W. NEUTEL AND C.E BEALEY. 1993. Effects of land use on breeding pheasant density. *J. Appl. Ecol.* 30:465–477.

ROCAMORA, G. 1994. Bonelli's Eagle *Hieraaetus fasciatus.* Pages 184-185 *in* G.M. Tucker and M.F. Heath [EDS.], Birds in Europe, their conservation status. Birdlife Conserv. Ser. No. 3. BirdLife International, Cambridge, U.K.

WASER, P.M. 1985. Does competition drive dispersal? *Ecology* 66:1171–1175.

ZAR, J.H. 1984. Biostatistical analysis. Prentice Hall, Princeton, NJ U.S.A.

Received 6 June 1997; accepted 20 May 1998

J. Raptor Res. 32(3):215–220
© 1998 The Raptor Research Foundation, Inc.

WINTER FORAGING ECOLOGY OF BALD EAGLES ON A REGULATED RIVER IN SOUTHWEST IDAHO

GREGORY S. KALTENECKER
Department of Biology, Raptor Research Center, Boise State University, Boise, ID 83725 U.S.A.

KAREN STEENHOF
USGS Forest and Rangeland Ecosystem Science Center, Snake River Field Station, 970 Lusk St., Boise, ID 83706 U.S.A

MARC J. BECHARD AND JAMES C. MUNGER
Department of Biology, Raptor Research Center, Boise State University, Boise, ID 83725 U.S.A.

ABSTRACT.—We studied Bald Eagle foraging ecology on the South Fork Boise River, Idaho, during the winters of 1990–92. We compared habitat variables at 29 foraging sites, 94 perch sites, and 131 random sites. Habitat variables included river habitat (pool, riffle, run), distance to the nearest change in river habitat, distance to nearest available perch, number and species of surrounding perches, and average river depth and flow. Eagles foraged more at pools than expected, and closer (<15 m) to changes in river habitat than expected. Where eagles foraged at riffles, those riffles were slower than riffles where they perched or riffles that were available at random. Where eagles foraged at runs, those runs were shallower than runs at either perch or random sites. Eagles perched less at riffles and more at sites where trees were available than expected. Changes in river habitat represent habitat edges where river depth and flow change, making fish more vulnerable to eagle predation. Fish are more susceptible to predation at shallower river depths and slower flows. Slower river flows may be related to decreased surface turbulence, which also increases vulnerability of fish to aerial predation.

KEY WORDS: *Bald Eagle;* Haliaeetus leucocephalus; *wintering ecology; foraging ecology; dams; rivers; Idaho.*

Ecología del forrajeo de invierno de águilas Calvas en un rio regulado del suroeste de Idaho

RESUMEN.—Estudiamos la ecología de forrajeo de águilas Calvas en el Río South Fork Boise en Idaho, durante los inviernos de 1990–1992. Comparamos las variables de habitat en 29 sitios de forrajeo, 93 perchas y 131 sitios al azar. Las variables de habitat incluyeron habitats del río (pozos, escorrentias, otros), la distancia al cambio de habitat mas cercano del río, la distancia mas cercana a una percha disponible, el número y especies de perchas alrededor y el promedio de profundidad y escorrentia. Las águilas forrajearon mas en los pozos de lo esperado, y mas cerca (15 m) a los cambios de habitat en el río de lo esperado. En los sitios poco profundos en donde las águilas forrajearon, estos fueron mas lentos que aquellos en donde las águilas utilizaron perchas disponibles al azar. En los sitios en donde las águilas se percharon en escorrentias, estas fueron menos profundas que las de las perchas o sitios al azar. Las águilas utilizaron menos perchas en sitios de escorrentias y mas en sitios en donde los árboles estaban mas disponibles de lo esperado. Los cambios de habitat en el río estaban representados por las orillas en donde la profundidad y el flujo variaban, haciendo a los peces mas vulnerables a la depredación de las águilas. Los peces son mas susceptibles a la depredación en los niveles menos profundos y en escorrentias mas lentas los cuales pueden estar relacionados con la disminución de la turbulencia en la superficie, lo que aumenta la vulnerabilidad de los peces a la depredación aerea.

[Traducción de César Márquez]

The winter diets of Bald Eagles (*Haliaeetus leucocephalus*) differ depending on locale, habitat, weather conditions, and prey availability, but fish are selected most often when available (Stalmaster 1987). Bald Eagles may concentrate during winter near dams where open water and fish are readily available (Steenhof et al. 1980). Dams can keep downstream areas from freezing and can provide a reliable source of fish that have been killed or stunned while passing through dam turbines (Steenhof 1978, Brown et al. 1989). In rivers, benthic-feeding fish are the most commonly taken

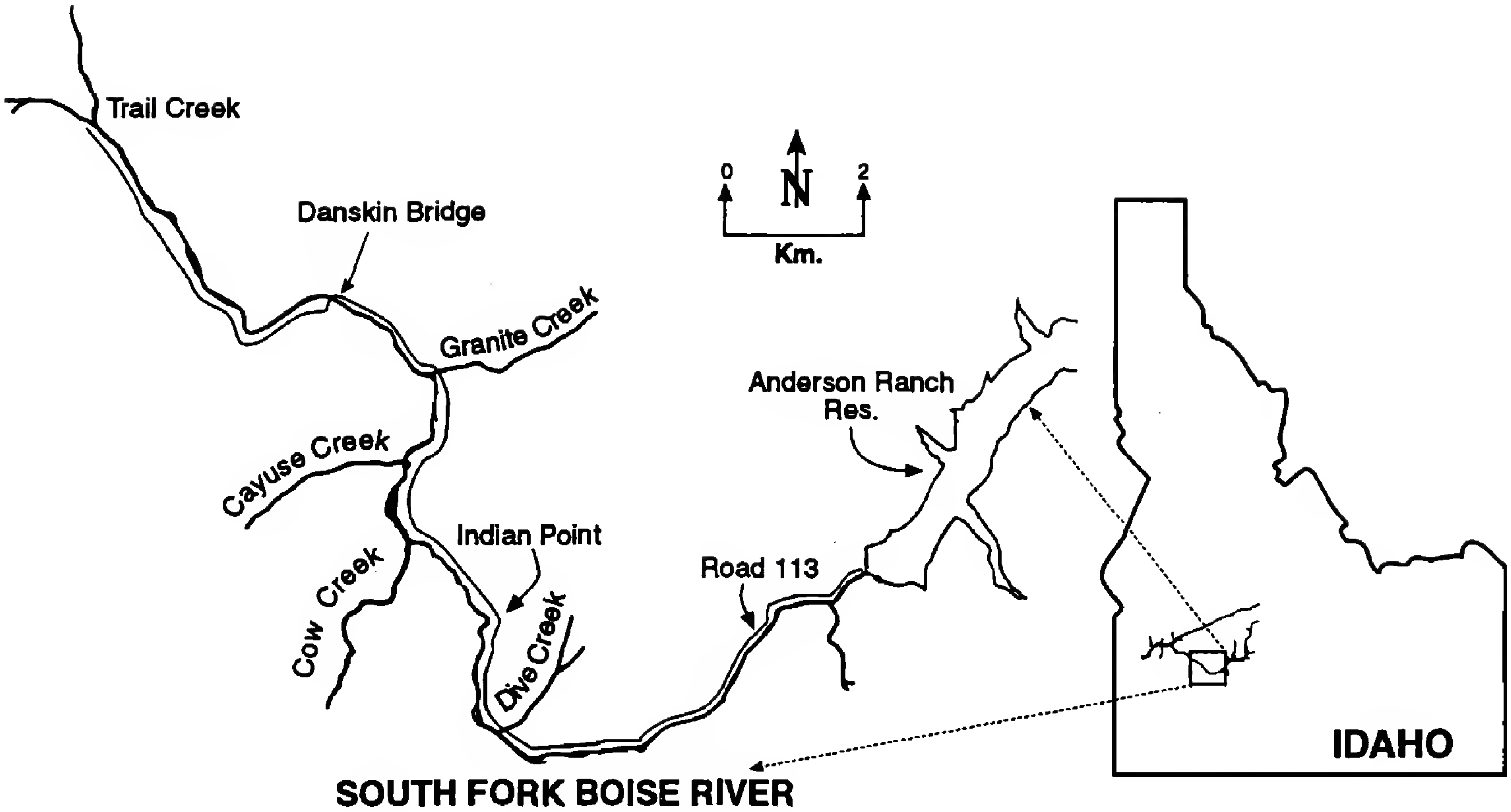

Figure 1. South Fork Boise River, Idaho, studied during winters 1990–92; Anderson Ranch Dam to Trail Creek.

prey (Dunstan and Harper 1975, McEwan and Hirth 1980, Todd et al. 1982, Haywood and Ohmart 1986, Hunt et al. 1992), but eagles may also take rainbow trout (*Oncorhynchus mykiss*) if available (Brown et al. 1989, Spahr 1990, Brown 1993). The Boise River, a tributary of the Snake River, is a major drainage containing free-flowing and regulated river reaches and three reservoirs. Mammal carrion and fish are the main prey of Bald Eagles wintering in the Boise River System (Kaltenecker and Bechard 1995, Kaltenecker 1997). We studied foraging ecology of Bald Eagles on the South Fork Boise River during the winters of 1990–92, and present results which identify and describe foraging and perching habitat.

STUDY AREA AND METHODS

The South Fork Boise River flows from the Sawtooth Mountain Range in southwestern Idaho and drains an area of approximately 1568 km^2 (Gebhards 1964). Anderson Ranch Dam, a U.S. Bureau of Reclamation power-generating and irrigation facility, is 19 km downstream from the town of Pine. Our study area included approximately 20 km of river located between Anderson Ranch Dam and Trail Creek and was easily accessible by vehicle along U.S. Forest Service road #113 (Fig. 1). Both the river and Bald Eagles perching along it could be seen from our observation points on the road. The South Fork flowed through a steep-sided valley dominated by shrub-steppe vegetation consisting of sagebrush (*Artemisia tridentata*), bitterbrush (*Purshia tridentata*), native perennial grasses (*Poa secunda, Pseudoregnaria spicata, Aristida longiseta*), and exotic annuals cheatgrass (*Bromus tectorum*) and medusahead rye (*Taeniatherum caput-medusae*). Cottonwood/willow riparian vegetation (*Populus trichocarpa, Salix* spp., *Betula* spp., *Alnus* spp.) dominated the river bottom and other riparian areas. Some mixed-conifer stands (*Pinus ponderosa, Pseudotsuga menziesii*) were present on north-facing slopes. Elevations ranged from 1100–1220 m, and temperature extremes varied from −30°C to 16°C during December–March. River flows were regulated by Anderson Ranch Dam and were maintained at the standard winter minimum flow (approx. 91 m^3/s) throughout both winters. Drought conditions prevailed during both years of the study.

We conducted a total of 224 hr of foraging observations on 28 d between 15 December–1 March (12 days during the first winter, and 16 days during the second winter). Observations were conducted by one person from a vehicle using 8 × 30 binoculars and 45× spotting scope. We began observing at dawn, and continued throughout the day until all eagles left the river or returned to night roosts. Observation points were selected so that perched or flying eagles and the river were in full view between 150–500 m away from the observer. We recorded foraging activity as successful or unsuccessful attempts at fish prey. A foraging site was defined as the exact point in the river where a foraging attempt was made. Foraging attempts were initiated either from the wing or nearby perch locations. We identified fish species taken by eagles from observation of prey captures or feeding, or by analysis of prey remains collected from feeding sites immediately after eagles departed. Remains used to identify fish species included scales, opercular bones, and mandibles. During observations, we also recorded all perches used by eagles within 75 m of the river. Perch sites were defined as any tree, cliff, or rock outcrop where we observed eagles perching. Perches from which prey strikes were initiated were included in the sample of perch sites.

Once a foraging site had been identified, we returned to it during late February or March of the same winter and measured surrounding habitat. Because river flows were regulated at a constant level throughout both winters, we assumed that surrounding habitat did not change significantly between observation of prey captures and measurement of habitat. We used a line-transect method modified from Bovee (1982) and Platts et al. (1983) to measure physical habitat parameters associated with each foraging site. At each foraging site, we recorded predominant river habitat (three categories: pool, riffle, run), distance to nearest change in river habitat, and distance to the nearest perch. Furthermore, a transect was established across the river perpendicular to flow, and river depth, stream flow, and bottom substrate were recorded at five equidistant points (verticals) along the transect. At each vertical, we measured depth and flow using a Price AA flowmeter. At each end of the transect, we recorded the number of surrounding perches and predominant species of tree within a 75 m arc. At all eagle perch sites located within 75 m of the river, we measured surrounding habitat as described above for foraging sites. Lastly, we selected an additional sample of sites by converting random numbers into distances (m) downstream from Anderson Ranch Dam. After locating random sites using a metric tape, we measured surrounding habitat similar to foraging and perch sites.

Habitat variables associated with foraging, perch, and random sites were analyzed using logistic regression (LOGIST procedure, SAS 1990), which determines the effects of several different independent variables on a single dependent variable (Harrell 1986, Trexler and Travis 1993). The dependent variable in our analyses was site type (three categories: foraging, perch, or random). We conducted three separate logistic regression analyses, comparing foraging to random sites, foraging to perch sites, and perch to random sites. We used stepwise logistic regression, with the significance level to enter the model and to remain in the model set at 0.15. Independent variables entered into the analyses were river habitat (pool, riffle, run), distance to nearest change in river habitat, presence of available perches, distance to nearest perch, and number of surrounding perches. River habitat is a nominal variable, and was therefore transformed into a set of 0,1 variables that were used in the analysis. To prevent over specification of the model, we considered the variable "run" as the base state and did not include "run" in the model; thus, we determined if being a pool or riffle increased the chance of, for example, being a foraging site. Distance to the nearest perch was placed in a category of 1 to 6, with 1 = 0–10 m, 2 = 11–25 m, 3 = 26–50 m, 4 = 51–75 m, and 5 = >75 m. Number of perches was placed in a category of 0–5, with 0 = no surrounding perches, 1 = less than 5, 2 = 6–10, 3 = 10–20, and 4 = >20 perches available within 75 m. We further explored relationships of variables contributing significantly to logistic regression models using Chi-square goodness-of-fit tests (Zar 1984; FREQ procedure, SAS 1990). We calculated average stream flow and depth for each transect, and compared means using analysis of variance (ANOVA; GLM procedure, SAS 1990) to determine if flow or depth characteristics varied significantly between foraging, perch, or random sites by river habitat type.

RESULTS

Counts of Bald Eagles from 18 aerial surveys conducted every two weeks during both winters of our study ranged from 0–17 ($\bar{x}$ = 7.8) eagles. We observed 31 attempted prey captures of fish (17 successful) at 29 different sites, identified 94 eagle perch sites, and collected habitat data from 131 random sites. Fish species taken by eagles included largescale suckers (*Catostomus macrocheilus*, N = 10), mountain whitefish (*Prosopium williamsoni*, N = 4), and rainbow trout (N = 3).

Due to low sample sizes, and because stream flows were similar during both winters, we lumped habitat data collected during both years of the study for analyses. Significant differences existed between foraging and random sites, foraging and perch sites, and perch and random sites. Foraging sites differed from random sites with regard to river habitat and distance to the nearest change in river habitat (Table 1). Further analysis using Chi-square goodness-of-fit tests revealed that eagles foraged at pools more than expected (number expected = 3.63, actual number = 8, χ^2 = 6.1, P = 0.013, df = 1), and that foraging sites were closer (<15 m) to changes in river habitat more than expected (number expected = 5.6, actual number = 12, χ^2 = 9.5, P = 0.002, df = 1). Where eagles foraged at runs, those runs were shallower than runs available at random (Table 2). Where eagles foraged at riffles, those riffles had slower stream flows than riffles available at random.

Perch sites were similar to foraging sites, but differed with regard to distance to the nearest change in river habitat and the number of surrounding perches (Table 1). As with foraging sites compared to random, foraging sites were closer to changes in river habitat (χ^2 = 9.5, P = 0.002, df = 1) than perch sites. Foraging sites also had fewer surrounding perches than perch sites. Perch sites differed from random with regard to the presence, number of, and distance to surrounding perches. Not all random sites had potential eagle perches available within 75 m. No differences existed between perch and random sites with regard to either river depth or flows (Table 2).

Table 1. Results from three separate stepwise logistic regression procedures comparing habitat between foraging and random, foraging and perching, and perching and random sites for Bald Eagles. Sample sizes are in parentheses.

	PARAMETER ESTIMATE	STANDARD ERROR	WALD CHI-SQUARE	P VALUE	ODDS RATIO
FORAGING (29) vs. RANDOM (131)[a,b]					
Intercept	−1.682	0.583	8.331	0.004	
Pool	1.423	0.549	6.768	0.009[e]	4.175[f]
Distance to Habitat Change	−0.018	0.007	5.736	0.017[e]	0.983
Perch	0.967	0.548	3.115	0.078	2.630
FORAGING (29) vs. PERCHING (94)[c]					
Intercept	4.751	1.584	8.999	0.003	
Distance to Habitat Change	−0.015	0.007	4.514	0.034[e]	0.985
Perch	−1.987	1.234	2.591	0.108	0.137
Number of Surrounding Perches	−2.463	0.631	15.194	0.0001[e]	0.085
PERCHING (94) vs. RANDOM (131)[d]					
Intercept	−8.833	1.469	36.15	0.0001	
Pool	0.955	0.606	2.486	0.115	2.599
Distance to Nearest Perch	0.801	0.334	5.731	0.017[e]	2.227
Perch	2.978	1.031	8.332	0.004[e]	19.65
Number of Surrounding Perches	3.812	0.643	35.15	0.0001[e]	45.24

[a] The first of the two listed site types was the modeled state, thus the sign (+ or −) of the parameter estimate indicates whether an increase in the independent variable was associated with a higher (if +) or lower (if −) probability of being a site of the modeled state.

[b] Model statistics: overall $G = 19.74$ with 3 df ($P = 0.0002$); concordance/discordance: 73.6%/22.8%. Concordance is determined as follows. All possible pairings of foraging and random sites are created. A pair of sites is defined as concordant if the foraging site of that pair is also the site predicted by the logistic regression model (based on predictor variables, e.g., habitat) to be the site more likely to be the foraging site. A pair is discordant if the model predicts (incorrectly) that the random site is more likely to be the foraging site. Percents of the total number of pairs that are concordant or discordant are presented. Ties are not presented.

[c] Model statistics: overall $G = 45.22$ with 3 df ($P = 0.0001$); concordance/discordance: 85.6%/12.2%.

[d] Model statistics: overall $G = 153.86$ with 4 df ($P = 0.0001$); concordance/discordance: 85.1%/3.7%.

[e] Denotes variables contributing significantly to stepwise logistic regression models at $P < 0.05$ level (analyses performed using SAS, procedure logist).

[f] A pool site has an approximately four-fold greater probability of being a feeding site than does a non-pool site.

DISCUSSION

Fish species captured by Bald Eagles can influence foraging behavior and foraging site selection in rivers. Many authors have discussed increased vulnerability of bottom-feeding fish to avian predators (Swenson 1979, Todd et al. 1982, Haywood and Ohmart 1986). In our study, eagles took more benthic-dwelling than pelagic fish. River habitat also may influence Bald Eagle foraging site selection. In our study, eagles foraged more from pools than other river habitats. Hunt et al. (1992) reported that eagles foraged more from pools than other habitats in California's Pit River. In Arizona, nesting Bald Eagles also foraged most at pools (Haywood and Ohmart 1986). On the Boise River, Spahr (1990) reported that eagles were observed at pools more than expected. During winter, most fish species, especially salmonids, seek pools or other areas of low stream velocity to maintain position with minimal energy expenditure (Allen 1969, Cunjak and Power 1986, 1987, Hillman et al. 1987). Because of winter temperatures and low stream flows on the South Fork Boise River, it is likely that during our study, pools were areas of high fish abundance.

Changes in river habitat, especially from pool to riffle or pool to run, usually indicate decreasing water depth and a change in stream flow, both found to be important parameters at foraging sites during our study. We found that eagles foraged at sites which were closer to river habitat changes than were random sites. This suggests that changes in river habitat may be important to foraging Bald Eagles as habitat edges. The edges of habitats containing higher prey densities may represent areas where fish become vulnerable to predation due to

Table 2. Results from ANOVA procedures on average depth and average velocity by river habitat type. *P*-values are from ANOVA, means with different letters are different by Tukey's Studentized Range (HSD) tests at $P < 0.05$ level.

	MEAN DEPTH (m)			MEAN FLOW (m/s)		
HABITAT TYPE	*N*	$\bar{x}$	SE	*N*	$\bar{x}$	SE
POOLS						
Foraging	8	0.80a	0.10	8	0.32a	0.05
Perch	13	0.84a	0.10	13	0.33a	0.05
Random	12	0.92a	0.10	12	0.32a	0.05
F, (df), *P* Value		0.33, (2), 0.72			0.10, (2), 0.90	
RIFFLES						
Foraging	8	0.41a	0.04	8	0.62a	0.05
Perch	16	0.40a	0.04	16	0.80b	0.05
Random	38	0.41a	0.04	38	0.71b	0.05
F, (df), *P* Value		0.02, (2), 0.98			3.54, (2), 0.04	
RUNS						
Foraging	13	0.47a	0.03	13	0.57a	0.03
Perch	64	0.57ab	0.03	64	0.54a	0.03
Random	81	0.61b	0.03	81	0.50a	0.03
F, (df), *P* Value		4.76, (2), 0.01			2.96, (2), 0.06	

decreasing water depth. Haywood and Ohmart (1986) reported that eagles foraged from pools bounded by shallows or riffles where benthic feeding fish were vulnerable to predation. Hunt et al. (1992) also showed that eagles foraged from shallow areas of pools. Wintering Bald Eagles in Grand Canyon, Arizona, foraged more in creeks (i.e. smaller, shallower streams) than rivers (Brown 1993). We found that eagles foraged at runs which were shallower than those available at random. Though water depth influences fish vulnerability, foraging site selection by Bald Eagles also may be influenced by stream flow. We found that eagles foraged at riffles with lower stream flows. Water turbulence is related to stream flow; the faster the flow, the greater the turbulence. Low surface turbulence may be an important component of Bald Eagle foraging sites (Hunt et al. 1992), enabling eagles to better detect fish.

We concur with other authors that physical habitat parameters of rivers or streams are important to Bald Eagle foraging site selection and foraging success. Eagles commonly took prey from habitats where fish were likely most abundant, but concentrated foraging efforts at the edges of those habitats where water was shallower and slower, suggesting that vulnerability of prey also may be important.

ACKNOWLEDGMENTS

Funding and vehicles for this study were provided by the U.S. Forest Service, Boise National Forest. Housing was arranged by Idaho Department of Fish and Game and the U.S. Bureau of Reclamation. Necessary equipment was provided by Boise State University, Idaho Department of Fish and Game, and the U.S. Fish and Wildlife Service. We thank L.L. Donohoo, U.S. Forest Service, for logistic support, advice on design, and help in the field. This manuscript benefitted from reviews by L.L. Donohoo, M.N. Kochert, G. Bortolotti, R. Knight, and J.H. Kaltenecker. Maps were prepared by E. Holzer and M. Spencer. Invaluable help in the field was provided by R. Moore, J. Hilty, J. Weaver, R. Garwood, L. Spain, and B. Zoellick.

LITERATURE CITED

ALLEN, K.R. 1969. Limitations on production in salmonid populations in streams. Pages 3–18 *in* T.G. Northgate [ED.], Symposium on salmon and trout in streams. Univ. British Columbia, Vancouver Canada.

BOVEE, K.D. 1982. A guide to stream habitat analysis using the Instream Flow Incremental Methodology. Instream flow information paper 12. U.S. Fish and Wild. Serv. FWS/OBS-82/86.

BROWN, B.T., R. MESTA, L.E. STEVENS AND J. WEISHEIT. 1989. Changes in winter distribution of Bald Eagles along the Colorado River in Grand Canyon, Arizona. *J. Raptor Res.* 23:110–113.

BROWN, B.T. 1993. Winter foraging ecology of Bald Eagles in Arizona. *Condor* 95:132–138.

CUNJAK, R.A. AND G. POWER. 1986. Winter habitat utilization by stream resident brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*). *Can J. Fish. Aquat. Sci.* 43:1970–1981.

CUNJAK, R.A. AND G. POWER. 1987. Cover use by stream-resident trout in winter: a field experiment. *N. Am. J. of Fisheries Manage.* 7:539–544.

DUNSTAN, T.C. AND J.F. HARPER. 1975. Food habits of Bald Eagles in North-Central Minnesota. *J. Wild. Manage.* 39:140–143.

GEBHARDS, S.V. 1964. Federal aid to fish restoration. Job performance report, Project No. F-51-R-1. Idaho Dept. Fish and Game, Boise, ID U.S.A.

HARRELL, F.E. 1986. The LOGIST procedure. Sugi Supplemental Library Guide, Version 5. SAS Institute, Cary, NC U.S.A.

HAYWOOD, D.D. AND R.D. OHMART. 1986. Utilization of benthic-feeding fish by inland breeding Bald Eagles. *Condor* 88:35–42.

HILLMAN, T.W., J.S. GRIFFITH AND W.S. PLATTS. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. *Trans. Am. Fish. Soc.* 116:185–195.

HUNT, W.G., B.S. JOHNSON AND R.E. JACKMAN. 1992. Carrying capacity for Bald Eagles wintering along a northwestern river. *J. Raptor Res.* 26:49–60.

KALTENECKER, G.S. AND M.J. BECHARD. 1995. Bald Eagle wintering habitat study, upper Boise River Drainage, Idaho. Raptor Res. Ser. No. 9. Boise State Univ., Boise, ID U.S.A.

KALTENECKER, G.S. 1997. Winter ecology of Bald Eagles in the upper Boise River Drainage, Idaho. M.S. thesis, Boise State Univ., Boise, ID U.S.A.

MCEWAN, L.C. AND D.H. HIRTH. 1980. Food habits of the Bald Eagle in north-central Florida. *Condor* 82:229–231.

PLATTS, W.S., W.F. MEGAHAN AND G.W. MINSHALL. 1983. Methods for evaluating stream, riparian, and biotic conditions. U.S. Dept. Ag., Forest Service, Gen. Tech. Report INT-138. Intermountain Forest and Range Experiment Station, Ogden, UT U.S.A.

SAS, 1990. SAS/STAT user's guide, version 6, fourth edition. SAS institute, Cary, NC U.S.A.

SPAHR, R. 1990. Factors affecting the distribution of Bald Eagles and effects of human activity on Bald Eagles wintering along the Boise River. M.S. thesis, Boise State Univ., Boise, ID U.S.A.

STALMASTER, M.V. 1987. The Bald Eagle. Universe Books, New York, NY U.S.A.

STEENHOF, K. 1978. Management of wintering Bald Eagles. U.S. Fish and Wild. Serv. Rep., FWS/OBS-78/79, Harper's Ferry, WV U.S.A.

———, S.S. BERLINGER AND L.H. FREDRICKSON. 1980. Habitat use by wintering Bald Eagles in South Dakota. *J. Wild. Manage.* 44:798–805.

SWENSON, J.E. 1979. The relationship between prey species ecology and dive success in Ospreys. *Auk* 96:408–412.

TODD, C.S., L.S. YOUNG, R.B. OWEN, JR. AND F.J. GRAMLICH. 1982. Food habits of Bald Eagles in Maine. *J Wild. Manage.* 46:636–645.

TREXLER, J.C. AND J. TRAVIS. 1993. Nontraditional regression analyses. *Ecology* 74:1629–1637.

ZAR, J. H. 1984. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, NJ U.S.A.

Received 6 August 1997; accepted 23 June 1998

J Raptor Res. 32(3):221–228
© 1998 The Raptor Research Foundation, Inc.

URBAN, SUBURBAN AND RURAL RED-TAILED HAWK NESTING HABITAT AND POPULATIONS IN SOUTHEAST WISCONSIN

William E. Stout
W2364 Heather Street, Oconomowoc, WI 53066 U.S.A.

Raymond K. Anderson
University of Wisconsin-Stevens Point, Stevens Point, WI 54481 U.S.A.

Joseph M. Papp
Rt. 1 Box 158A, Drummond, WI 54832 U.S.A.

Abstract.—Nesting Red-tailed Hawks (*Buteo jamaicensis*) are becoming increasingly common in urban environments. We described Red-tailed Hawk nesting habitat and reproductive success and compared urban, suburban, and rural nesting locations in southeast Wisconsin. Nest sites were classified as urban, suburban or rural if ≥70%, 30–70%, or ≤30% of the area (706.9 ha, 1.5-km radius) around nests was used for industrial or residential purposes, respectively. Mean success and productivity of breeding Red-tailed Hawks in the metropolitan Milwaukee area from 1989–94 ($N = 426$) was 81.9% (range = 75.3–92.7%) and 1.43 young/breeding pair (range = 1.13–1.91), respectively. Brood size averaged 1.75 young/successful nest (range = 1.61–2.06). Productivity was variable and was significantly higher in 1994 than each of the preceding yr ($P < 0.001$). Based on internest distances, the density of the Red-tailed Hawk nesting population for rural locations was greater than in suburban areas and lowest in urban locations. The amount of natural microhabitat cover around nests (19.6 ha, 0.25-km radius) did not differ for urban, suburban, or rural nest sites ($P = 0.967$) indicating that cover was an important component of the nesting habitat of Red-tailed Hawks. Natural cover comprised about 16% of the landscape area of urban sites and 40% of this area was wooded with the remaining 60% consisting of herbaceous cover. Urban planning should consider the amount of natural cover to allow Red-tailed Hawks and other wildlife to coexist with humans in an urban environment.

Key Words: *Red-tailed Hawk*; Buteo jamaicensis; *urban*; *suburban*; *rural*; *nesting habitat*; nesting density.

Habitat de anidación urbano, suburbano y rural de *Buteo jamaicensis* en el sureste de Wisconsin

Resumen.—La anidación en áreas de *Buteo jamaicensis* es cada vez mas común en ambientes urbanos. Describimos el habitat de anidación de *Buteo jamaicensis* y su éxito reproductivo y comparamos las localidades urbanas, suburbanas y rurales de anidación en el sureste de Wisconsin. Los sitios de los nidos fueron clasificados como urbanos, rurales y suburbanos si ≥70%, ≤30%, y 30–70% del área (706.9 ha, 1.5 km de radio) alrededor del nido eran utilizadas para propósitos industrial o residencial (desarrollo) respectivamenmte. La media del éxito en la productividad de los nidos ocupados por *Buteo jamaicensis* en el área metropolitana de Milwakee entre 1989–94 ($N = 426$) fue de 81.9% (rango = 75.3–92.7%) y 1.43 juveniles/nido ocupado (rango = 1.13–1.91). Tamaño de la nidada promedio de 1.75 juveniles/nido exitoso (rango = 1.61–2.06). La productividad fue variable y significativamente mas alta en 1994 que en cada uno de los añós precedentes ($p < 0.0001$). Con base en la distancia entre nidos se observo que la densidad de la población reproductiva de las localidades rurales, fue mayor que en las áreas suburbanas y fue menor en áreas urbanas. La cantidad de cobertura de microhabitat natural alrededor de los nidos (19.6 ha, 0.25 km de radio) no fue diferente entre los sitios de los nidos urbanos, suburbanos y rurales ($P = 0.967$) lo cual indica que la cobertura es un componente importante del habitat de anidación de *Buteo jamaicensis*. La cobertura natural incluyo el 16% del microhabitat de los sitios urbanos, 40% de esta área eran bosques y el 60% restante eran cobertura de pastizales. La planeación urbana debe considerar la cantidad de cobertura natural requerida para que *Buteo jamaicensis* y la vida silvestre puedan coexistir con los humanos en un ambiente urbano.

[Traducción de César Márquez]

Red-tailed Hawks (*Buteo jamaicensis*) nest in urban environments, yet no comprehensive studies have been published on their urban nesting habitat. Two reports in Michigan document the successful nesting of red-tails in urban settings (Valentine 1978, Hull 1980), and urban nesting also has been reported in Puerto Rico (Santana et al. 1986) and New York (Minor et al. 1993).

Three studies of rural Red-tailed Hawk populations have previously been conducted in Wisconsin (Orians and Kuhlman 1956, Gates 1972, Petersen 1979). Howell et al. (1978) correlated nesting habitat structure and productivity at rural nest sites in Ohio and found that highly productive sites had more than twice as much fallow land, less than half as much cropland, and less than half the number of woodlots than did sites with low productivity. Other studies of red-tails conducted in rural areas throughout North America have described other aspects of red-tail ecology (e.g., Wiley 1975, Fitch and Bare 1978, Adamcik et al. 1979).

Our objectives were to describe Red-tailed Hawk nesting habitat and reproductive success, and to compare urban, suburban, and rural nesting locations in southeast Wisconsin. We determined relative nesting population densities for all three locations based on internest distances and identified important physical components of the nesting habitat.

STUDY AREA

Our study area covered approximately 1100 km^2 located in the metropolitan Milwaukee area in southeast Wisconsin (43°N, 88°W). It included Milwaukee county and parts of Waukesha, Washington, and Ozaukee counties. Milwaukee and Ozaukee counties are bordered by Lake Michigan to the east. Milwaukee county covers an area of 626.5 km^2. The city of Milwaukee covers an area of 248.5 km^2 with a human population of 629 554 (1994 population estimate; 2533 people per km^2). Human population density decreases radially from the city of Milwaukee to suburban communities and to rural areas. Two interstate highways transect the study area. Land use within the study area included agricultural, natural, industrial/commercial, and residential areas.

METHODS

Red-tailed Hawk nests were located from a vehicle from 1 February–30 April, 1987–94 (Craighead and Craighead 1956) and visited at least twice (once within 10 d after the onset of incubation and again when nestlings were 20–35 d of age) during each nesting season to determine productivity (Postupalsky 1974, Steenhof 1987). Woodlots that were not entirely visible from the road early in the season before leaf-out were checked by foot. A breeding pair (i.e., eggs were laid) was considered successful if ≥1 nestling survived to a bandable age (20–35 d). Intrayear internest distances for 1989 and 1990 were measured to determine the nearest breeding pair of Red-tailed Hawks (nearest neighbor; Clark and Evans 1954). These data were used as an index for population nesting density and to compare urban, suburban, and rural densities (Clark and Evans 1954, McGovern and McNurney 1986). We believe that all nests were found in urban and suburban areas and, therefore, the distances between nests in these locations are accurate.

To describe Red-tailed Hawk nesting habitat and to compare urban, suburban, and rural locations, we characterized features of 1989 and 1990 nest sites on four different spatial scales: 1) nest site, 2) habitat, 3) macrohabitat and 4) landscape (Titus and Mosher 1981, Mosher et al. 1986, 1987, Adamus 1995, Stout 1995; Table 1). The nest-site scale described the nest and nest tree and data were collected when nestlings were 20–35-d old Nest exposure (i.e., the open side of the nest) was assigned one of the following values: total access/exposure, N, NE, E, SE, S, SW, W, or NW. The nest tree was classified as being in a woodlot interior (the tree crown did not touch a woodlot edge), on the edge of an interior woodlot clearing (clearing was ≥0.1 ha), savannah (not on an edge), woodlot edge, hedgerow, lone tree, powerline tower, or billboard.

The habitat scale described vegetation within a 0.04-ha circular plot (11.3 m radius) centered on the nest tree and data were collected after fledging through September for 1989 and 1990 nest sites. Canopy, understory, shrub, ground cover, and slope of the plot were described according to Titus and Mosher (1981) and Mosher et al. (1986, 1987). Shrub structure was classified by shrub density, shrub index and density board (Mosher et al. 1986). Slope and slope aspect were determined for sites with a slope ≥2% using a compass and clinometer.

The landscape scale described land use within a 1.5-km radius (706.9 ha) of the nest tree. Data were collected for 1989 and 1990 nest sites, and used for analysis and nest site classification (i.e., as urban, suburban, or rural). The amount of land with natural, agricultural, residential, and industrial cover types within the landscape area was determined from 1990 aerial photos (1 cm = 48 m) with a compensating polar planimeter. The number of individual areas of each cover type was recorded. Natural habitat included woodlots, tree and shrub savannahs, shrublands, herbaceous cover (grasses and forbs, fallow fields, and inactive pastures), and open water. The mean area of open water was <1% (7 ha; maximum = 6.2%, 43.8 ha) and primarily consisted of pothole ponds and, therefore, was included in the natural category. For management recommendations, natural habitat was subdivided into grassland and forest habitat. Agricultural land included row crops (e.g., corn), cover crops (e.g., alfalfa and clover), actively grazed pastures, tree nurseries and orchards. Residential land included human dwellings and other buildings and land associated with them. Industrial land included nonresidential industrial and commercial buildings, pavement, roads, graded land (e.g , gravel pits), mowed land (e.g., cemeteries, airports, mowed land surrounding industrial buildings), and nonmowed land associated with human activity (e.g., freeway intersections, nonmowed land surrounding industrial buildings). Each area was measured separately and combined for analysis. Industrial and residential areas were considered developed. Natural and agricultural areas were considered undeveloped because they are devoid of any buildings or pavement. A nest site was classified as urban if ≥70% of the landscape area (706.9 ha) was developed, rural if ≤30%, and suburban if 30–70% was developed (Stout et al. 1996). Hedgerow length was measured within the landscape area. The Baxter-Wolfe interspersion index was determined from the changes in cover type along the north-south and east-west median lines within the landscape area (Baxter and Wolfe 1972, Mosher et al. 1987). The area and perimeter of woodlots containing nests were measured. Distances to the nearest residence, industrial building and road were recorded and mean distance to buildings was determined by using a point-quarter method of measuring the distance to the nearest building in each of four quadrants; a buffer area (circular area surrounding the nest without buildings) was calculated by using the mean distance to buildings as the radius of a circle (Stout 1995). The macrohabitat scale described land use within a 0.25 km radius (19.6 ha) of the nest for a comparison of land use patterns closer to the nest site. The same variables that were measured at the landscape scale also were determined at the macrohabitat scale.

Nest-site data were collected for all known breeding pairs of Red-tailed Hawks in the metropolitan Milwaukee area for 1989 and 1990. Nest sites that were used in both 1989 and 1990 (in either the same or a different nest tree or structure) were included in the analysis only once. Macrohabitat and landscape-scale data were collected on all urban sites and at least as many suburban and rural sites. According to our definitions, 15 urban nest sites were found. For the urban, suburban, and rural comparison, 22 suburban and 18 rural nest sites were identified. Nest-site and habitat data were collected for these sites where access (landowner permission) was granted.

Categorical data were tested with a Chi-square goodness of fit. Urban, suburban, and rural nest sites were compared using univariate statistics. Frequency distributions were used to determine variables with normal distributions. Log transformations were used when applicable. Quantitative variables with normal distributions were treated with parametric methods (one-way ANOVA). The TUKEY multiple range test was used to identify different groups. Nonparametric methods (Kruskal-Wallis test, Chi-square approximation; Sokal and Rohlf 1981) were used for nonparametric variables. All tests were considered significant when $P \leq 0.05$. The Statistical Package for the Social Sciences (SPSS; Nie et al. 1975) was used for statistical analyses.

RESULTS

Productivity did not differ among urban, suburban, and rural nest sites used by breeding Red-tailed Hawks (Table 1). Mean nesting success for Red-tailed Hawks in the Milwaukee metropolitan area from 1989–94 ($N = 426$) was 81.9% (range = 75.3–92.7%; Table 2). Productivity of breeding pairs for the same 6-yr period averaged 1.43 young/breeding pair (range = 1.13–1.91), and 1.75 young/successful nest (range = 1.61–2.06). Productivity was significantly higher in 1994 than each of the preceding years ($P < 0.001$). Mean internest distance for urban sites was greater than in suburban and rural sites ($P = 0.004$, $P < 0.001$, respectively), and mean internest distance was greater for suburban than rural sites ($P = 0.018$; Table 1).

In 1989 and 1990, we found 89 breeding Red-tailed Hawks nesting in 18 species of trees. Four were on high voltage transmission towers and one was on a billboard. Nests constructed in trees and on unnatural structures occurred in urban, suburban, and rural areas (Stout 1995, Stout et al. 1996). Only one nest-site variable, nest-tree height, was different for urban, suburban, and rural locations indicating behavioral consistency in nest building (Stout 1995, Table 1). Nest structures were in woodlots or on edges of woodlots more often than in hedgerows, totally exposed lone trees, or human-made structures ($\chi^2 = 23.273$, df $= 2$, $P < 0.001$). Nests had a northwest exposure more often than other directions ($N = 88$; Fig. 1; $\chi^2 = 35.955$, df $= 8$, $P < 0.001$). Sloped sites ($N = 41$) were not used more often than nonsloped sites ($N = 38$; $\chi^2 = 0.114$, df $= 1$, $P = 0.736$). When sloped, red-tails used a southeast slope more often than other directions (Fig. 1; $\chi^2 = 19.293$, df $= 7$, $P = 0.007$).

At the habitat scale, the percent slope of plots was greater for suburban sites than for rural sites, the number of shrub species at suburban sites was greater than at both urban and rural sites, and the number of small understory saplings (dbh = 1–4 cm) at suburban sites was greater than at rural sites (Table 1).

At the landscape scale, total hedgerow length within the landscape area, mean building distance, buffer area, nearest residence, industrial structure, building, road, the Baxter-Wolfe interspersion index, and the amount of natural, agricultural, industrial and residential land were different for urban, suburban, and rural sites (Table 1). At the macrohabitat scale, agricultural, industrial, and residential land use were different, but the amount of natural cover (total grassland and forest cover) did not differ among the three sites (Table 1). Natural cover within the macrohabitat area averaged 10.3 ha for all three locations while natural habitat within the larger landscape area averaged 111.3 ha

Table 1. Comparison of productivity, nest site, habitat (0.04-ha circular plot, 11.3-m radius), macrohabitat (19.6-ha, 0.25-km radius) and landscape (706.9 ha, 1.5 km radius) for urban, suburban and rural Red-tailed Hawk nest sites. Nest site and habitat results do not include nests on artificial substrates. Productivity, macrohabitat and landscape results include nests on artificial substrates.

	URBAN NEST SITES	SUBURBAN NEST SITES	RURAL NEST SITES	ONE-WAY ANOVA[b] KRUSKAL-WALLIS TEST[c]	
VARIABLE	MEAN ± SE RANGE (*N*)	MEAN ± SE RANGE (*N*)	MEAN ± SE RANGE (*N*)	F/χ^2	*P*
Productivity	1.27 ± 0.25 0–3 (15)	1.50 ± 0.19 0–3 (22)	1.44 ± 0.22 0–3 (18)	0.593[c]	0.744
Nest Site					
Nest tree height (m)	20.09 ± 1.00^{x} 14.10–26.30 (11)	23.33 ± 0.67^{y} 18.50–28.96 (20)	21.09 ± 0.99^{xy} 14.17–28.65 (16)	3.699[b]	0.033
Habitat (0.04-ha circular plot, 11.3-m radius)					
% Slope	2.7 ± 1.75^{xy} 0–10 (7)	3.6 ± 0.86^{x} 0–16 (21)	1.0 ± 0.46^{y} 0–6 (15)	6.076[c]	0.048
No. shrub species	4.6 ± 0.95^{x} 1–8 (7)	7.4 ± 0.56^{y} 4–12 (21)	4.5 ± 0.84^{x} 0–11 (15)	5.640[b]	0.007
No. small saplings	48.3 ± 9.61^{xy} 0–78 (7)	72.6 ± 8.67^{x} 15–183 (21)	40.9 ± 9.32^{y} 0–113 (15)	6.420[c]	0.040
Macrohabitat Area (19.6-ha, 0.25-km radius)					
Grassland (ha)	4.77 ± 1.38 0.0–17.2 (15)	4.37 ± 0.76 0.0–13.3 (22)	4.53 ± 1.34 0.0–18.5 (18)	0.143[c]	0.813
Forest (ha)	4.83 ± 1.19 0.0–13.3 (15)	6.06 ± 0.72 0.0–15.2 (22)	5.91 ± 1.05 0.3–14.1 (18)	1.528[c]	0.466
Natural (ha)	9.76 ± 1.68 0.0–17.7 (15)	10.62 ± 0.97 2.6–18.8 (22)	10.44 ± 1.49 1.3–19.6 (18)	0.067[c]	0.967
Agricultural (ha)	0.17 ± 0.17^{x} 0.0–2.6 (15)	3.89 ± 1.05^{y} 0.0–15.6 (22)	7.76 ± 1.53^{z} 0.0–18.2 (18)	21.171[c]	<0.001
Industrial (ha)	5.24 ± 1.68^{x} 0.0–18.9 (15)	3.14 ± 0.96^{x} 0.0–16.2 (22)	0.46 ± 0.25^{y} 0.0–3.5 (18)	10.263[c]	0.006
Residential (ha)	4.43 ± 1.16^{x} 0.7–19.0 (15)	1.96 ± 0.56^{y} 0.0–9.1 (22)	0.93 ± 0.45^{y} 0.0–6.4 (18)	15.160[c]	0.001
Landscape					
Woodlot area[a] (ha)	9.93 ± 4.19 0.3–45.4 (11)	8.53 ± 1.27 2.5–20.2 (20)	9.39 ± 2.99 0.3–39.5 (15)	0.164[b]	0.850
Woodlot perimeter[a] (m)	1550 ± 403.0 288–3936 (11)	1425 ± 137.0 768–2688 (20)	1715 ± 440.9 307–6816 (15)	0.348[b]	0.708
Mean building dis. (m)	224 ± 17.7^{x} 68–341 (15)	322 ± 35.4^{y} 79–759 (22)	455 ± 29.9^{z} 150–692 (18)	12.607[b]	<0.001
Buffer area[a] (ha)	17.10 ± 2.35^{x} 1.5–36.5 (15)	40.89 ± 8.85^{x} 2.0–181.0 (22)	62.18 ± 7.27^{y} 7.1–127.5 (18)	9.004[b]	<0.001
Nearest residence[a] (m)	117 ± 10.6^{x} 30–178 (15)	240 ± 26.1^{y} 86–533 (22)	289 ± 34.3^{y} 67–571 (18)	11.327[b]	<0.001
Nearest industry[a] (m)	348 ± 69.8^{x} 48–1080 (15)	397 ± 72.1^{x} 62–1166 (22)	743 ± 97.2^{y} 187–1375 (17)	6.915[b]	0.002
Nearest building[a] (m)	106 ± 11.3^{x} 30–178 (15)	180 ± 15.0^{y} 62–293 (22)	272 ± 33.0^{z} 67–571 (18)	12.620[b]	<0.001
Nearest road[a] (m)	114 ± 14.9^{x} 24–197 (15)	218 ± 28.5^{y} 53–518 (22)	322 ± 48.5^{y} 38–878 (18)	8.292[b]	0.001
Mean internest dis.[a] (m)	2743 ± 319.3^{x} 1327–4968 (15)	1780 ± 120.9^{y} 799–2904 (20)	1316 ± 165.5^{z} 403–2246 (15)	11.322[b]	<0.001

Table 1. Continued.

VARIABLE	URBAN NEST SITES MEAN ± SE RANGE (*N*)	SUBURBAN NEST SITES MEAN ± SE RANGE (*N*)	RURAL NEST SITES MEAN ± SE RANGE (*N*)	ONE-WAY ANOVA[b] KRUSKAL-WALLIS TEST[c] F/χ^2	 *P*
Landscape Area (706.9-ya, 1.5-km radius)					
Baxter-Wolfe Index	18.3 ± 1.36[x] 8–27 (15)	28.8 ± 1.03[y] 21–40 (21)	26.2 ± 1.26[z] 19–37 (18)	19.304[b]	<0.001
Hedgerow length (m)	7619 ± 1087[x] 2208–16080 (15)	10 506 ± 995[xy] 1920–18 432 (22)	12 053 ± 981[y] 3984–18 720 (18)	4.258[b]	0.019
Grassland (ha)	67.20 ± 11.14[x] 0.0–146.3 (15)	137.23 ± 8.57[y] 70.0–231.9 (22)	141.18 ± 22.77[y] 24.7–312.5 (18)	6.707[b]	0.003
Forest (ha)	39.30 ± 6.26[x] 0.0–94.0 (15)	77.82 ± 7.56[y] 31.1–178.9 (22)	103.80 ± 9.70[z] 43.1–187.3 (18)	14.007[b]	<0.001
Natural (ha)	111.27 ± 13.52[x] 16.3–190.2 (15)	221.07 ± 10.68[y] 123.7–329.4 (22)	253.11 ± 29.22[y] 81.3–457.4 (18)	13.166[b]	<0.001
Agricultural (ha)	11.69 ± 4.05[x] 0.0–48.8 (15)	128.05 ± 14.75[y] 20.5–310.3 (22)	309.74 ± 30.76[z] 108.2–534.4 (18)	40.587[c]	<0.001
Industrial (ha)	273.85 ± 35.34[x] 56.6–499.1 (15)	180.45 ± 18.94[y] 39.6–354.2 (22)	53.57 ± 9.77[z] 0.0–123.0 (18)	23.117[b]	<0.001
Residential[a] (ha)	310.00 ± 31.28[x] 153.4–537.2 (15)	177.27 ± 15.65[y] 21.9–331.5 (22)	90.68 ± 9.60[z] 25.5–173.2 (18)	25.905[b]	<0.001

[a] Variables log-transformed for one-way Analysis of Variance (one-way ANOVA).

[b] One-way ANOVA *F* values.

[c] Kruskal-Wallis test χ^2 values (χ^2 approximation).

[xyz] Values followed by the same superscript letter [x], [y] or [z], are not significantly different at the $P \leq 0.05$ level (TUKEY multiple range test[b] or Mann-Whitney U test[c]).

(15.7%) for urban nest sites only, and this natural habitat was interspersed among developed land in an average of 16.4 different tracts.

DISCUSSION

Reproductive success and productivity of breeding Red-tailed Hawks during our 6-yr study was comparable to that of previous studies in Wisconsin (Orians and Kuhlman 1956, Gates 1972, Peterson 1979; Table 2) and an urban/suburban area in New York (Minor et al. 1993). Red-tailed Hawk nest success estimates for North America range from 58–93% (Preston and Beane 1993).

The distance between breeding pairs of Red-tailed Hawks was used as an index of nesting density (McGovern and McNurney 1986). Our mean internest distance of 1.9 km was comparable to other studies (Fitch et al. 1946, Orians and Kuhlman 1956, Gates 1972, Petersen 1979, McGovern and McNurney 1986). Rural nests were significantly closer together than suburban and urban nests, and suburban nests were closer together than urban nests which indicated that nesting density decreased from rural to urban areas. We found rural nests adjacent to suburban nests at the perimeter of our study area. As a result, the nearest breeding pair of red-tails may not have been found in all rural areas making rural nests even closer than our data indicated. Peterson (1979) found a mean internest distance of 1.51 km in rural Wisconsin. Our mean internest distance of 1.32 km between rural nests may indicate that the density of nesting Red-tailed Hawks may have increased in rural southeast Wisconsin over the past 25 yr, possibly because of increased availability of nesting habitat resulting from changes in agricultural practices such as the conservation reserve program (CRP).

The microclimate surrounding nest structures is important in the selection of nest sites by raptors. We found Red-tailed Hawk nests had predominantly northern exposures (primarily NW and NE) and sloped sites had southeast aspects. Speiser and Bosakowski (1988) also found Red-tailed Hawk nests to have southeast facing slope exposures. They suggested that a southeast slope maximizes insulation to the nest on cold mornings and min-

Table 2. Red-tailed Hawk reproductive success from 1989–94 for the metropolitan Milwaukee, Wisconsin area.

				YOUNG/NEST			REPRODUCTIVE SUCCESS	
YR	BREEDING PAIRS	NEST FAILURES	NEST SUCCESS (%)	1	2	3	A[a]	B[b]
1989	59	11	81.4	20	24	4	1.36	1.67
1990	85	21	75.3	19	39	6	1.35	1.80
1991	92	16	82.6	33	40	3	1.33	1.61
1992	83	9	89.2	24	45	5	1.55	1.74
1993	52	16	69.2	16	17	3	1.13	1.64
1994	55	4	92.7	13	22	16	1.91	2.06
Total	426	77	81.9	125	187	37	1.43	1.75

[a] Young/breeding pair.

[b] Young/successful nest.

imizes the possibility of heat stress in the afternoon. Southeast slopes may help to keep nestlings dry by minimizing the effects of predominantly northwest storm winds in Wisconsin while northern nest accesses may provide more shade and reduce heat stress. Several studies also found that nest sites usually have unobstructed access and a commanding view of the surrounding area (Petersen 1979, Bednarz and Dinsmore 1982, Santana et al. 1986, Speiser and Bosakowski 1988, Bechard et al. 1990, Toland 1990, Preston and Beane 1993). Sloped nest sites probably provide this type of nest orientation.

Red-tailed Hawks used similar types of nest sites in urban, suburban, and rural locations, however, suburban nest sites tended to be located on sloped sites and in wetlands, probably because upland sites are developed first. Suburban areas also had the highest land use diversity (Baxter-Wolfe interspersion index) while urban locations had the least amount of land use diversity. Woodlot area and perimeter remained relatively constant for urban, suburban, and rural nesting locations indicating that 9 ha may represent an ideal size woodlot for Red-tailed Hawk nesting sites. Other studies have found that red-tails selected smaller woodlots, open stands, and woodlot edges compared to larger woodlots or closed canopy woodlot interiors (Orians and Kuhlman 1956, Gates 1972, Petersen 1979). Speiser and Bosakowski (1988) found that red-tails nested closer to forest openings than random sites and Howell et al. (1978) reported that the most productive pairs of Red-tailed Hawks used small woodlots.

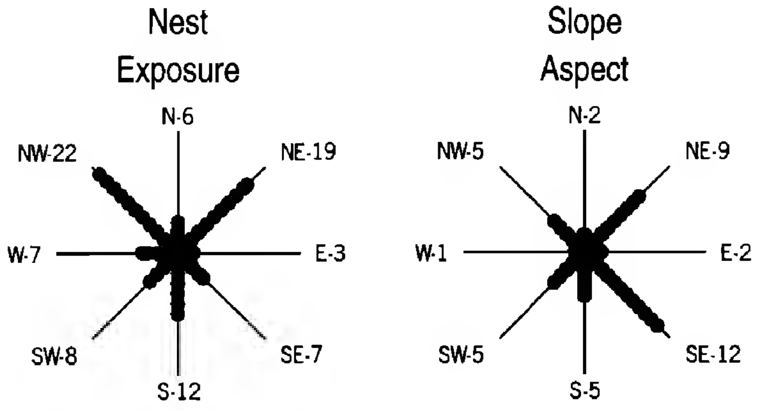

Figure 1. Nest exposure ($N = 84$) and slope aspect ($N = 41$) at Red-tailed Hawk nest sites in southeast Wisconsin. Sample size is indicated for each direction.

Landscape variables (e.g., nearest road, industry, residence) varied significantly and increased from urban to suburban and rural areas. The amount of natural and agricultural land within the landscape scale decreased as the amount of industrial and residential land increased. While the amount of agricultural land increased and residential and industrial land decreased at the macrohabitat scale from rural through suburban and urban areas, the amount of natural cover within the macrohabitat remained consistent for all three areas averaging 10.3 ha indicating that natural cover constitutes an important nesting habitat component for Red-tailed Hawks.

For the purposes of urban planning and development, we believe that managing for important habitat components such as natural cover will enhance the availability of nesting habitat for Red-tailed Hawks in urban areas. Based on our findings, we recommend that at least 16% of urban land be left in natural habitat with approximately 40% wooded and 60% herbaceous cover. This natural habitat should be distributed among residential and industrial land in approximately 16 separate tracts within the landscape area (706.9 ha). Wooded areas should be approximately 9 ha to provide suitable nesting woodlots.

ACKNOWLEDGMENTS

J.W. Hardin and R.W. Miller provided guidance and editorial assistance. D. Hebbert helped collect nest site and habitat data. R.D. Beane, S. Postupalsky, K. Titus and two anonymous reviewers provided comments and suggestions that greatly improved this manuscript. J.A. Reinartz and R. Rogers provided assistance with statistical analyses. The senior author's wife, Vicki, daughter, Jennifer, and son, Tim, provided continual support, patience and assistance in all areas of this research project. This project was partially funded by the U.W.-Stevens Point Graduate School Student Research Fund, and V.W. Debroux. An earlier version of this manuscript was submitted to the University of Wisconsin-Stevens Point, College of Natural Resources, in April 1995 for partial fulfillment of a Master of Science degree in Natural Resources (Wildlife).

LITERATURE CITED

ADAMCIK, R.S., A.W. TODD AND L.B. KEITH. 1979. Demographic and dietary responses of Red-tailed Hawks during a snowshoe hare fluctuation. *Can. Field-Nat.* 93: 16–27.

ADAMUS, P.R. 1995. Validating a habitat evaluation method for predicting avian richness. *Wildl. Soc. Bull.* 23: 743–749.

BAXTER, W.L. AND C.W. WOLFE. 1972. The interspersion index as a technique for evaluation of Bobwhite Quail habitat. Pages 158–165 *in* J.A. Morrison and J.C. Lewis [EDS.], Proc. First Natl. Bobwhite Quail Symp. Okla. State Univ., Res. Found. Stillwater, OK U.S.A.

BECHARD, M.J., R.L. KNIGHT, D.G. SMITH AND R.E. FITZNER. 1990. Nest sites and habitats of sympatric hawks (*Buteo* spp.) in Washington. *J. Field Ornithol.* 61:159–170.

BEDNARZ, J.C. AND J.J. DINSMORE. 1982. Nest sites and habitat of Red-shouldered and Red-tailed Hawks in Iowa. *Wilson Bull.* 94:31–45.

CLARK, P.J. AND F.C. EVANS. 1954. Distance to nearest neighbor as a measure of spatial relationships in populations. *Ecology* 35:445–453.

CRAIGHEAD, J.J. AND F.C. CRAIGHEAD. 1956. Hawks owls and wildlife. The Stackpole Co., Harrisburg, PA and Wildlife Manage. Inst., Washington DC U.S.A.

FITCH, H.S., F. SWENSON AND D.F. TILLOTSON. 1946. Behavior and food habits of the Red-tailed Hawk. *Condor* 48:205–257.

——— AND R.O. BARE. 1978. A field study of the Red-tailed Hawk in eastern Kansas. *Trans. Kansas Acad. Sci.* 81:1–13.

GATES, J.M. 1972. Red-tailed Hawk populations and ecology in east-central Wisconsin. *Wilson Bull.* 84:421–433.

HOWELL, J., B. SMITH, J.B. HOLT AND D.R. OSBORNE. 1978. Habitat structure and productivity in the Red-tailed Hawk. *Bird Banding* 49:162–171.

HULL, C.N. 1980. Additional successful nesting of a Red-tailed Hawk in an urban subdivision. *Jack Pine Warbler* 58:30.

MCGOVERN, M. AND J.M. MCNURNEY. 1986. Densities of Red-tailed Hawk nests in aspen stands in the Piceance Basin, Colorado. *J. Raptor Res.* 20:43–45.

MINOR, W.F., M. MINOR AND M.F. INGRALDI. 1993. Nesting of Red-tailed Hawks and Great Horned Owls in a central New York urban/suburban area. *J. Field Ornithol.* 64:433–439.

MOSHER, J.A., K. TITUS AND M.R. FULLER. 1986. Developing a practical model to predict nesting habitat of woodland hawks. Pages 31–35 *in* J. Verner, M.L. Morrison and C.J. Ralph [EDS.], Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI U.S.A.

———, ——— and ———. 1987. Habitat sampling, measurement and evaluation. Pages 81–97 *in* B.A. Giron Pendleton, B.A. Millsap, K.W. Cline and D.M. Bird [EDS.], Raptor management techniques manual. Nat. Wildl. Fed., Washington, DC U.S.A.

NIE, N.H., C.H. HULL, J.G. JENKINS, K. STEINBRENNER AND D.H. BENT [EDS.]. 1975. Statistical package for the social sciences. McGraw Hill, Inc., New York, NY U.S.A.

ORIANS, G. AND F. KUHLMAN. 1956. Red-tailed Hawk and horned owl populations in Wisconsin. *Condor* 58:371–385.

PETERSON, L. 1979. Ecology of Great Horned Owls and Red-tailed Hawks in southeastern Wisconsin. Wis. Dept. Nat. Res. Tech. Bull. No. 111, Madison, WI U.S.A.

POSTUPALSKY, S. 1974. Raptor reproductive success: some problems with methods, criteria, and terminology. Pages 21–31 *in* F.N. Hamerstrom, B.E. Harrell and R.R. Olendorff [EDS.], Management of raptors. Raptor Res. Rep. 2.

PRESTON, C.R. AND R.D. BEANE. 1993. Red-tailed Hawk (*Buteo jamaicensis*). *In* A. Poole and F. Gill [EDS.], The birds of North America., No. 52. Philadelphia: Acad Nat. Sci.; Am. Ornithol. Union, Washington, DC U.S.A.

SANTANA, E.C., E.N. LABOY, J.A. MOSHER AND S.A. TEMPLE. 1986. Red-tailed Hawk nest sites in Puerto Rico. *Wilson Bull.* 98:561–570.

SOKAL, R.R. AND F.J. ROHLF. 1981. Biometry. W.H. Freeman and Co., New York, NY U.S.A.

SPEISER, R. AND T. BOSAKOWSKI. 1988. Nest site preferences of Red-tailed Hawks in the highlands of southeastern New York and northern New Jersey. *J. Field Ornithol.* 59:361–368.

STEENHOF, K. 1987. Assessing raptor reproductive success and productivity. Pages 157–170 *in* B.A. Giron Pendleton, B.A. Milsap, K.W. Cline, D.M. Bird [EDS.], Raptor management techniques manual. Natl. Wildl. Fed., Washington, DC U.S.A.

STOUT, W.E. 1995. An urban, suburban, rural Red-tailed Hawk nesting habitat comparison in southeast Wisconsin. M.S. thesis, Univ. Wisconsin, Stevens Point, WI U.S.A.

———, R.K. ANDERSON AND J.M. PAPP. 1996. Red-tailed Hawks nesting on man-made and natural structures in southeast Wisconsin. Pages 77–86 *in* D.M. Bird, D.E. Varland and J.J. Negro [EDS.], Raptors in human landscapes. Academic Press, London, U.K.

TITUS, K. AND J.A. MOSHER. 1981. Nest-site habitat selected by woodland hawks in the central Appalachians. *Auk* 98:270–281.

TOLAND, B.R. 1990. Nesting ecology of Red-tailed Hawks in central Missouri. *Trans. Miss. Acad. Sci.* 24:1–16.

VALENTINE, A.E. 1978. The successful nesting of a Red-tailed Hawk in an urban subdivision. *Jack Pine Warbler* 56:209–210.

WILEY, J.W. 1975. Nesting and reproductive success of Red-tailed Hawks and Red-shouldered Hawks in Orange County, California, 1973. *Condor* 77:133–139.

Received 21 November 1997; accepted 19 May 1998

J. Raptor Res. 32(3):229–232
© 1998 The Raptor Research Foundation, Inc.

HIGHWAY MORTALITY OF BARN OWLS IN NORTHEASTERN FRANCE

SYLVIE MASSEMIN[1]
Centre d'Ecologie et de Physiologie Energétiques, CNRS, 23 Rue Becquerel, 67087 Strasbourg, Cedex 2, France

THIERRY ZORN
Office National de la Chasse, 17 Avenue de Wagram, 75017 Paris, Cedex, France

ABSTRACT.—We found a total of 187 road-killed raptors along a 150 km stretch of highway in northeastern France between 1990–94. Of these, 148 were Barn Owls (*Tyto alba*), 15 Long-eared Owls (*Asio otus*), and 10 Tawny Owls (*Strix aluco*). We analyzed different variables including the topography of highways, the types of habitats crossed by highways, and the types of vegetation along highways to determine why so many Barn Owls were killed. Most mortalities (64%) occurred along embanked stretches that crossed open fields and lacked hedges on either side. We concluded that the local population density and flight behavior of Barn Owls were probably related to such high mortality.

KEY WORDS: *Barn Owl;* Tyto Alba; *highway mortality.*

Mortalidad de *Tyto alba* en autopistas del noreste de Francia

RESUMEN.—Encontramos un total de 187 aves rapaces atropelladas a lo largo de un segmento de 150 km de autopista en el noreste de Francia entre 1990–94. De estas, 148 fueron *Tyto alba,* 15 *Asio otus* y 10 *Strix aluco,* para un total de 173 aves rapaces nocturnas. Analizamos las diferentes variables incluyendo la topografía de las autopistas, el tipo de habitats atravezados por la autopista y tipos de vegetación con el fin de determinar las causas de mortalidad de *Tyto alba.* La mayoría de las muertes (64%), ocurrieron a lo largo de terraplenes angostos que cruzaban sitios abiertos sin arbustos a los lados. Concluimos que la densidad poblacional local y el comportamiento de vuelo de *Tyto alba* estaban probablemente relacionados con esta alta tasa de mortalidad.

[Traducción de César Márquez]

Studies indicate that large numbers of raptors, especially Barn Owls (*Tyto alba*) are killed along highways in Europe (Baudvin et al. 1991, de Bruijn 1994, Taylor 1994). In France, there is a predominance of owls killed and, although the numbers vary according to region, Barn Owls (*Tyto alba*) are most commonly killed (Bourquin 1983, Joveniaux 1986, Athanaze 1992). Most Barn owls are killed in autumn and winter (Joveniaux 1986, Athanaze 1992, de Bruijn 1994, Taylor 1994), but there is also a high mortality along highways during the post-fledging period. Whereas the temporal variation in Barn Owl mortality has been shown, little information is available on the spatial variation in mortalities or the causes of death (Joveniaux 1986, Athanaze 1992). Here, we present the results of a study designed to show how such variables as the landscape crossed by the highway, the topography, and the vegetation along the roadway affect Barn Owl mortality.

MATERIAL AND METHODS

Raptors killed by vehicles were collected along a 150-km section of a highway between Strasbourg and Metz (northeastern France) over a 5-yr period (1990–94). In the case of dead Barn Owls, the location of carcasses on the highway was noted as either in the emergency stopping lane, the traffic lanes, or the median strip. We also noted the landscape crossed by the highway (forest, open field including cultivated fields, wasteland, bogs, and concrete), the topography, and the type of vegetation along the sides of the highway. Because Barn Owls fly at an average height of 5 m (Baudvin 1986), we classified this section of embanked and excavated highway (Fig. 1) into the following classes: highly embanked (≥5 m elevation on at least one side), shallow embanked (1–4 m elevation on at least one side), level highway, shallow excavated (1–4 m excavation on at least one side), deeply excavated (≥5 m excavation on at least one side), and embanked/excavated (height not distinguished). Vegetation along

---

[1] Present Address: Section of Ecology, Department of Biology, University of Turku, 20500 Turku, Finland.

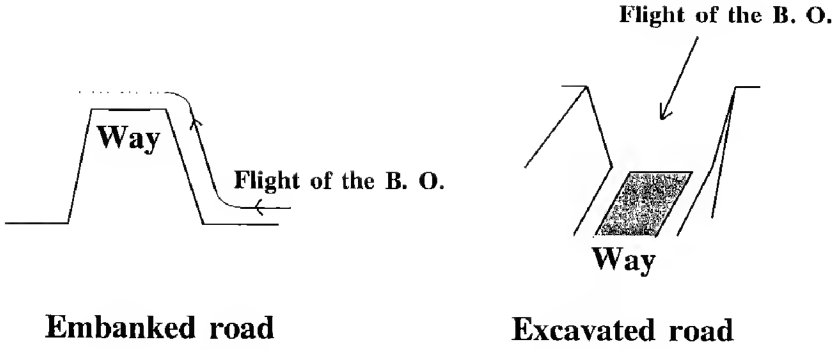

Figure 1. Schematic representation of two highway topographies. The arrow indicates the possible direction of flight of Barn Owls killed.

the side of the highway was classified according to the amount of hedge present (present, absent, and present/absent).

Factorial Correspondence Analysis (FCA) was used to determine which variables explained Barn Owl mortality. A variable highly correlated with one of the principal axes explains a large part of the inertia of this axis (STAT-ITCF, Dervin 1988). Variables identified using this method were verified using χ^2 analysis. For this analysis, the proportion of birds killed in each class of the variable was compared to the kilometric proportion of the class along the highway (proportion of killed birds expected). Therefore, if a variable had no influence on Barn Owl mortality, the number of killed birds collected was the same as the expected number.

RESULTS

A total of 187 road-killed raptors was found representing three owl and one buzzard species. Of the owls, 148 were Barn Owls (86%), 15 Long-eared Owls (*Asio otus*), and 10 Tawny Owls (*Strix aluco*).

Although dead Barn Owls were not distributed evenly along the section of highway studied, on average we found about 1 Barn Owl/5 km of highway. The largest number (59%) was found along a small segment of the highway (23%) (Fig. 2). The direction of traffic did not seem to affect mortality but the majority of owls (55%) were found in the emergency stopping lane. Only a few individuals were found in the median strip (18%).

The results of FCA on the different variables measured indicated that the topography of the highway (mostly the excavated parts) and vegetation along the sides (present or absent) were important variables contributing 99.9% of the first principal component (A1, Fig. 3). Likewise, topography (principally level) and landscape (mainly forest) contributed to 96.4% of the second principal component (A2). Most of the owls were killed along embanked stretches of the highway that lacked roadside hedges and crossed open fields.

Comparison of the number of observed mortalities vs. those expected showed that mortalities did in fact increase along embanked highway stretches ($\chi^2 = 13.78$, $P < 0.05$) that crossed open fields ($\chi^2 = 26.99$, $P < 0.05$). Our findings were most significant when the highway stretch both lacked a hedge and was highly embanked ($\chi^2 = 4.82$, $P < 0.05$). The stretch of highway with the highest raptor mortality (59%) had the highest embankment and lacked hedges ($\chi^2 = 7.39$, $P < 0.05$). Fewer raptors (24%) were found dead in a stretch of the highway with both a high excavation and a hedge ($\chi^2 = 8.4$, $P < 0.05$).

DISCUSSION

Unlike Bourquin (1983) and Joveniaux (1986), who found most Barn Owls killed along highways

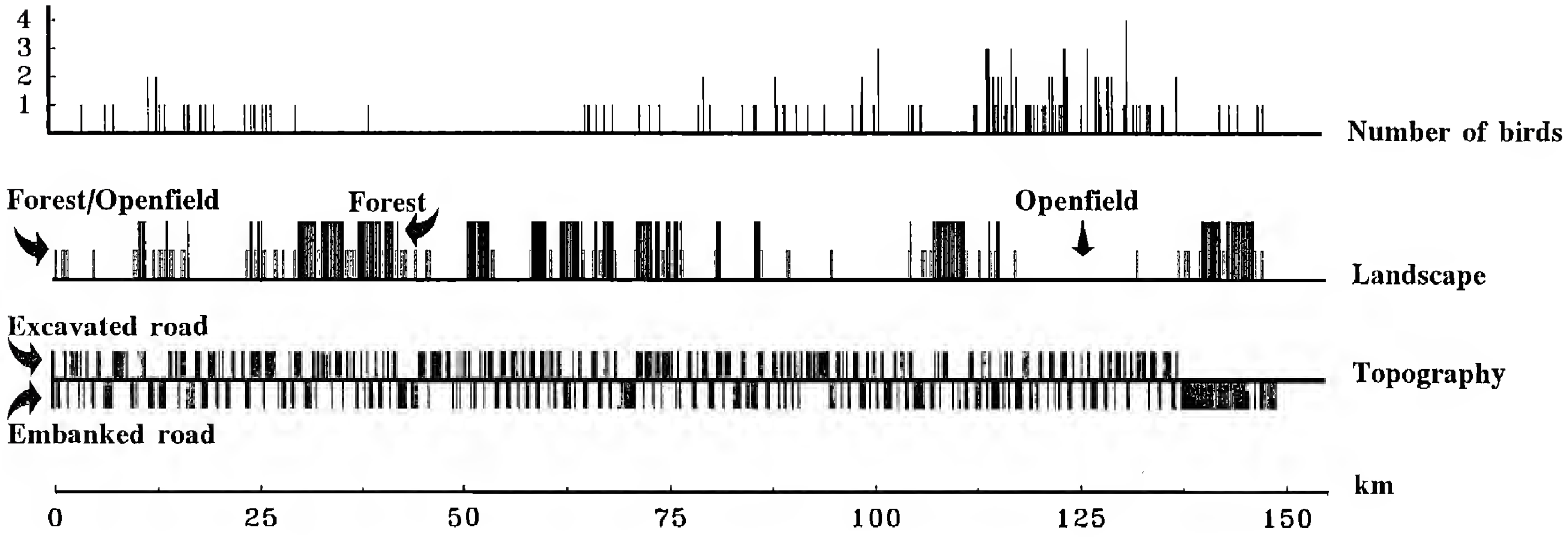

Figure 2. Spatial distribution of Barn Owl mortalities on the highway studied (Strasbourg-Metz). Number of raptors killed, topography, and landscape features are shown. The largest number of dead Barn Owls was found between kilometers 114–149.

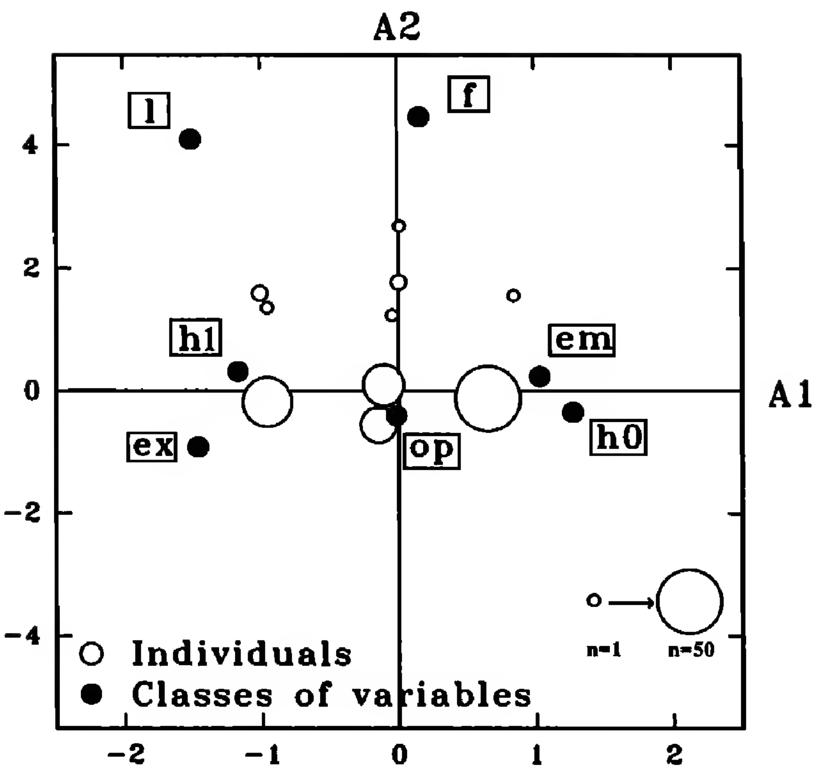

Figure 3. Biplot of the first two principal components of variables recorded at sites where Barn Owls were found killed by vehicles. The first (A1) and the second (A2) principal components explained 33% and 28% of the variance, respectively. Many individuals are on the same coordinates. To achieve a good representation of FCA, numbers are proportional to the diameter of the point (see scale in lower right corner). Forest/open field class (14% of total sector) is included in the open field class, embanked/excavated class (14% of total sector) is included in the embanked class, and the presence/absence of hedge class (15% of total sector) is included in the class presence of hedge. Symbols are as follows: landscape (forest [f] and open field [op]), togography (excavated road [ex], level [l], and embanked road [em]), and hedge (absence [h0] and presence [h1]).

crossing through forests, we found that Barn Owls were principally killed along sections of highway that crossed open fields. This may have been due to the fact that Barn Owls primarily hunt in open landscapes (Michelat and Giraudoux 1992) but it could also be related to the fact that Barn Owls killed along highways crossing through forests may have been attracted to excavated highway stretches that supported hedges on either side. These stretches support numerous small mammals that take refuge at the sides of highways and are undisturbed by agriculture (Spitz 1977, van der Reest 1992). Barn Owls typically prey on small mammals such as common voles (*Microtus arvalis*), woodmice (*Apodemus sylvaticus*), and common shrews (*Sorex araneus*) (Bourquin 1981, Baudvin et al. 1991). Because the sides of highways favor high winter densities of many of these species and make them more available because they are frequently snow-free (Bourquin 1983), these areas could potentially become traps for Barn Owls that are killed by passing vehicles.

Portions of embanked highway that crossed open fields also accounted for many of the Barn Owl mortalities. In this type of a situation, Barn Owls typically fly 2–5 m above the ground while hunting (Baudvin 1986), a height which corresponds to the normal height of passing trucks and cars. Barn Owls may cross embanked highways without climbing increasing the likelihood of impacts. Since we found most dead owls in the emergency lane of the highway, it appears that impacts probably occurred at the edges of the highway, when owls first started to cross the road. The amount of traffic probably had little effect on increasing the mortality (Canteneur 1964, Illner 1992) but the high speed of the traffic (>80 km/hr) probably did increase the danger for owls. It appeared that many of the owls were not killed by direct impact with vehicles but by impact with the ground after they were projected up into the air by turbulence behind vehicles.

We feel that few Tawny and Long-eared Owls were killed along this stretch of highway because they were simply less common in the area. The Tawny Owl is a woodland species and there was little forest habitat along this segment of highway. Likewise, few Long-eared Owls were known to occur in the area.

ACKNOWLEDGMENTS

This study was supported by a grant from the Société des Autoroutes du Nord et de l'Est de la France (SANEF) and by a grant from the French Ministry of the Environment. We are very grateful to the FIR (Fonds d'Intervention pour les Rapaces) for their administrative help. We are indebted to Drs. E. Challet and D. Currie for constructive comments on this manuscript.

LITERATURE CITED

ATHANAZE, P. 1992. Etude de l'impact de l'autoroute A6 sur les populations de rapaces nocturnes. Rapport du Centre Ornithologique Rhône-Alpes (CORA). Univ. Lyon 1, Villeurbanne, France.

BAUDVIN, H. 1986. La reproduction de la Chouette Effraie (*Tyto alba*). *Le Jean le Blanc* XXV.

———, J.C. GENOT AND Y. MULLER. 1991. Les rapaces nocturnes. Sang de la Terre, Paris, France.

BOURQUIN, J.D. 1981. Les petits mammifères vivant le long des autoroutes. *Strasse und Verkehr* 2:43–47.

———. 1983. Mortalité des rapaces le long de l'autoroute Genève-Lausanne. *Nos oiseaux* 37:149–169.

CANTENEUR, R. 1964. Les oiseaux sauvages victimes de la circulation routière dans l'Est de la France. *L'Oiseau et R.F.O.* 34:252–267.

DE BRUIJN, O. 1994. Population ecology and conservation of the Barn Owl (*Tyto alba*) in farmland habitats in Liermers and Achterhoek (the Netherlands). *Ardea* 82:1–109.

DERVIN, C. 1988. STAT-ITCF (institut Technique des Céréales et des Fourrages). Paris, France.

ILLNER, H. 1992. Road deaths of Westphalian owls: methodological problems, influence of road type and possible effects of population levels. Pages 94–100 *in* C.A. Galbraith, I. Taylor, and S. Percival [EDS.], The ecology and conservation of European owls. Joint Nature Conservation Committee, Peterborough, U.K.

JOVENIAUX, A. 1986. Influence de la réalisation d'une autoroute sur les populations de Chouettes Effraies. Rapport d'Environnement Participation et Aménagement (EPA) 13. Lons le Saunier, France.

MICHELAT, D. AND P. GIRAUDOUX. 1992. Activité nocturne et stratégie de recherche de nourriture de la Chouette Effraie (*Tyto alba*) à partir du site de nidification. *Alauda* 60:3–8.

MUSELET, D. 1985. Etude du comportement de l'avifaune lors de la traversée de voies autoroutières, autoroute A10 au Nord d'Orléans, Loiret. Route et faune sauvage. Conseil de l'Europe, Strasbourg, France.

SPITZ, F. 1977. Le campagnol des champs (*Microtus arvalis*) en Europe. *Bull. OEPP* 7:165–175.

TAYLOR, I. 1994. Predator-prey relationships and conservation. University Press, Cambridge, U.K.

VAN DER REEST, P.J. 1992. Kleine zoogdieren in nederlandse wagbermen: oecologie en beheer. *Lutra* 35:1–27.

Received 6 August 1997; accepted 14 February 1998

J Raptor Res. 32(3):233–240
© 1998 The Raptor Research Foundation, Inc.

THE EFFECT OF BURROW SITE USE ON THE REPRODUCTIVE SUCCESS OF A PARTIALLY MIGRATORY POPULATION OF WESTERN BURROWING OWLS (*SPEOTYTO CUNICULARIA HYPUGAEA*)

Eugene S. Botelho[1] and Patricia C. Arrowood[2]
P.O. Box 30001/Dept. 3AF, Department of Biology, New Mexico State University, Las Cruces, NM 88003-0001 U.S.A

Abstract.—We compared the number of nestlings produced by pairs of Burrowing Owls (*Speotyto cunicularia hypugaea*) using burrows in different types of nest sites, use of different types of burrows by resident and migrant males, and burrow type use by returning migrant males and females and the productivity of individuals that switched burrows. The number of nestlings and fledglings produced by pairs nesting in artificial burrows was also compared to the productivity of pairs in natural burrows. We determined that pairs in undisturbed areas used burrows located in or at the base of cliff walls more often than any other burrow type, while pairs in disturbed areas used burrows on flat ground more often. Both resident and migrant males used burrows in or at the base of cliff walls more often in undisturbed areas but, in disturbed areas, they used burrows in flat ground more often. Most males and females that switched burrows from one year to the next produced more nestlings in burrows they left than in new burrows. Pairs which nested in artificial burrows produced significantly more nestlings than those that used natural burrows, but pairs in natural burrows produced significantly more fledglings. Our results suggest the importance of determining burrow sites favored by nesting owls prior to initiation of conservation plans which require protection of areas containing nest holes or installation of artificial burrows.

Key Words: *Burrowing Owl;* Speotyto cunicularia hypugaea; *nest type use; artificial burrows; conservation.*

El efecto de los sitios de madriguera en el éxito reproductivo de una población parcialmente migratoria de *Speotyto cunicularia hypugaca*

Resumen.—Comparamos el número de pichones producidos por pares de *Speotyto cunicularia hypugaea* que utilizaron madrigueras en distintos tipos de sitios de anidación, el uso de distintos tipos de madrigueras por machos residentes y migratorios, el uso de distintos tipos de madrigueras por machos y hembras que retornaron al mismo lugar y la productividad de los individuos que cambiaron madrigueras. El número de pichones producidos por pares que anidaron en las madrigueras artificiales fue comparado con la productividad de los pares que anidaron en las madrigueras artificiales. Determinamos que los pares en áreas no perturbadas utilizaron madrigueras localizadas en la base de paredes en precipicios en mas ocasiones que otro tipo de madrigueras, mientras que los pares en áreas perturbadas utilizaron madrigueras en el suelo con mas frecuencia. Los machos residentes y migratorios utilizaron madrigueras en la base o en los precipicios con mayor frecuencia en las áreas no perturbadas, pero en las áreas perturbadas utilizaron el suelo con mayor frecuencia. Los machos y hembras que cambiaron madrigueras de un año a otro, produjeron mas pichones en la madriguera que dejaron que en la nueva. Los pares que anidaron en madrigueras artificiales produjeron significativamente mas pichones. Nuestros resultados resaltan la importancia de la determinación de sitios de madrigueras para anidación de buhos antes de la iniciación de planes de conservación, los cuales pueden requerir de la protección de áreas que contengan cavidades de nidos o la instalación de madrigueras artificiales.

[Traducción de César Márquez]

---

[1] Present Address: 58 Ohio Street, New Bedford, MA 02745 U.S.A.

[2] Present Address: Department of Wildlife and Fisheries Sciences, New Mexico State University, Las Cruces, NM 88003 U.S.A.

The western Burrowing Owl (*Speotyto cunicularia hypugaea,* from here on referred to as the Burrowing Owl) nests in underground burrows usually dug by other animals (Coulombe 1971, Thomsen 1971, Haug et al. 1993). Its requirement for underground nests may leave it with few choices, depending on the biology of animals that excavate burrows in a particular location (e.g., colonial vs. dispersed fossorial mammals). Conversely, in regions with diverse physiography (e.g., cavities in the cliff faces of dry creeks or rivers), Burrowing Owls may encounter a variety of possible nest site possibilities. Understanding the relationship between the use of different burrow site types and reproductive success in burrowing owls is important in light of recent conservation plans for the species throughout much of its range (Haug et al. 1993).

This study was conducted on a population of Burrowing Owls which nested on the campus of New Mexico State University (NMSU). Partial migration occurs in this population with all females and fledglings migrating from the study area each year. The majority of males, however, reside on the study area throughout the year (resident) but a few migrate (migrants). Resident and migrant males use nesting burrows (either retaining the previously-used one or switching to a new one) and defend the area surrounding their burrows prior to the arrival of females each year. Females begin to arrive in the area in February and immediately choose a mate.

Variation in burrow use by this population led us to consider what factors might affect the productivity of pairs which nested in different types of burrows. Burrow sites with no or low grass cover and high elevation should offer the most protection from predators, thus, males should more commonly use burrows in cliff walls. Compared to flat ground, burrows at the base of cliffs should offer more protection from predators, so they should be used more often than those on flat ground but less than those in cliffs. An abundance of lights in our study area attracted insects, bats and nighthawks (*Chordeiles* spp.), all of which the owls ate. In some lighted areas, the only type of burrow available was on flat ground; in these cases the increased prey availability should have offset increased predation risk. Thus, based on food availability, pairs in disturbed areas should be more successful than those in natural, nonlighted areas. Pairs with a resident male should produce more nestlings due to the increased experience of males in these areas and their opportunity throughout the winter to assess different burrows. We also felt that the use of artificial burrows should enhance reproductive success since they are not susceptible to collapse, are protected from flooding, and are impossible for larger predators to dig out and enter.

In order to test these predictions, we evaluated the reproductive success of Burrowing Owls that used burrows of different types. We also compared the types of burrows used by resident males with that of migrants. To determine the effect of switching nest burrows from one year to the next, we compared the number of nestlings produced by individuals at their current and previous burrow sites. We also compared the number of nestlings produced by pairs which used natural versus artificial burrows. Our results will help to determine if the types of burrows used by Burrowing Owls should be considered as part of future conservation plans, especially plans which involve installation of artificial burrows in areas without natural burrows.

STUDY AREA AND METHODS

Our study area on the NMSU campus encompassed a triangle of approximately 364 ha. The campus included irrigated pastures at its lowest elevation (~3900 m) and Chihuahuan Desert vegetation of approximately 121 ha at its highest elevation (~4100 m). Campus buildings occupied the central part of the triangle. The abundance of rock squirrels (*Spermophilus variegatus*) throughout the campus resulted in a large number of available burrows. Sometimes the squirrels dug shallow burrows which the owls enlarged. Spotted ground squirrels (*Spermophilus spilosoma*) dug smaller burrows which may have been enlarged by the owls. In natural areas, rock squirrels, cottontail rabbits (*Sylvilagus spp.*) and jackrabbits (*Lepus spp.*) also dug burrows. The rabbit population was large in the natural areas. Naturally-occurring crevices, abundant throughout the campus, were also used or enlarged by the owls. Since there were hundreds of shallow and deep burrows present at any one time, the owls had an abundant supply of burrow opportunities.

A total of 59 pairs nested in natural burrows located in two natural and two disturbed areas on the campus of NMSU. No pair was used more than once in this study. We did, however, include different pairs which used the same burrow in different years and we have repeated data for some pairs which switched burrows from year to year. We define a "natural" burrow as any existing cavity either above or below ground that had not been modified by us. We do not mean to suggest that a "natural" burrow was located in a natural (i.e., undisturbed) setting, although some of the burrows used in this study did fall into this category.

We used two natural areas in our study. The first consisted of an abandoned landfill (4.1 ha) inoperative for at least 10 yr prior to this study. Its initial contents had been covered with soil and its base was overgrown with native vegetation. The second area consisted of an earthen dam at one end of a flood control basin (8 ha, Botelho and Arrowood 1996). Both natural areas were located in the remote southeast part of campus and rarely visited by people. They were actually large depressions surrounded by cliffs up to 10 m high. Burrows were located in several different sites including flat ground, above ground and 3–10 m up in the sides or at the base of cliff walls. Vegetation consisted of typical arid Chihuahuan Desert vegetation, dominated by Creosote Bush (*Larra tridentata*) and Mesquite (*Prosopsis spp.*).

Disturbed areas consisted of the university quadrangle (quad) and football stadium (stadium). Owls nested among closely-spaced buildings separated by walkways, lawns, parking lots and other buildings on the quad and at the top of small hills and at the base of cement walls behind each endzone in the stadium (Botelho and Arrowood 1996). Pairs typically used burrows under cement walkways or curbs, especially those in the vicinity of street lights or other types of artificial lighting. Soil was rich loamy topsoil; more durable burrows occurred in this soil type because it was less susceptible to collapse during rainstorms. Vegetation consisted of irrigated cultivated grass on well-manicured lawns with some trees and shrubs (Botelho and Arrowood 1996). In addition, burrows were located at the base of light posts in a large parking lot and in large pipes above ground.

Some burrows in cliff walls in natural areas were located high off the ground, but within human reach from the top of the cliff. Burrows in cliff walls had very little space at their entrances for nestlings to congregate during feedings. As a consequence, nestlings sometimes fell from the front of their burrows and either found shelter in a burrow close to the ground or fell victim to predation. In disturbed areas, burrows in cliff walls were similar to those in natural areas. Burrows in cliff walls in disturbed areas were only available behind each endzone inside the stadium. Owls which used these burrows perched on the tops of bleachers and on fences.

Burrows at the base of cliff walls in natural areas were dug at ground level into the sides of cliff walls. Because of their location at ground level, these burrows had more space at their entrances for nestlings to congregate during feedings and there was no danger of nestlings falling from their burrows. Because of their close proximity to both the ground and a cliff wall, these burrows could be blocked when loose dirt from the cliff poured over their entrances during heavy rains. Burrows were located beneath stone walls behind each endzone in the stadium and at the base of buildings in the quad in disturbed areas. Unlike burrows at the base of cliff walls in natural areas, in disturbed areas some burrows were dug under concrete sidewalks and abutments. Owls which used burrows at the base of cliff walls used buildings or cement walls as perches.

Burrows in flat ground in natural areas were dug directly into the desert floor and were surrounded by sparse vegetation. These burrows had few elevated perch sites and were resistant to erosion but lacked a cliff face which may have increased vulnerability to predation because predators could approach the burrow from all directions. Burrows in flat ground, however, had ample space at their entrances for nestlings to congregate during feedings without the danger of nestlings falling from the burrow. Burrows in flat ground in disturbed areas were located on lawns (often at the base of chain link fences), or under curbs. Owls regularly used man-made perch sites (e.g., fences, walls, and buildings) when nesting in these burrows.

All above ground nesting attempts occurred in disturbed areas. These nest sites consisted of large metal pipes located on flat ground. In one case a pair nested in a drainpipe located in the side of a building.

We constructed 24 artificial burrows. Eight of 24 natural burrows were situated in such a way that we could replace them with artificial burrows. The remaining 16 burrows were left in place and artificial burrows were installed in the vicinity of and adjacent to them. We replaced natural burrows with artificial burrows in winter when breeding was not in progress. Natural burrows were excavated in the evening after we placed a one-way door (this door allowed owls to leave the burrow but not reenter) over the burrow entrance for at least 48 h to ensure that no owls were present inside the burrow during excavation. We oriented the chambers and tunnels of our artificial burrows as close as possible to that of original burrows.

Artificial burrows were completely self contained and consisted of a nesting chamber (a 19 l covered plastic bucket) located at the end of a tunnel made of two 2 5 m × 10 cm PVC pipes (with 2 cm holes drilled every 6 cm for drainage) connected by a right angle PVC connector. A single 10 cm hole was cut into the side of the plastic bucket about two cm from the bottom to allow insertion of the PVC tunnel pipe. A 10 cm hole also was cut into the cover of the bucket; we could insert a hand through this hole to gain access to the nest for weighing and measuring nestlings without removing the entire bucket lid. The cover hole was capped with a PVC lid We drilled three to four holes (each 2 cm in diameter) in the bottom of the plastic bucket for drainage. During installation we placed dirt in the bottom of the bucket and inside the tunnel pipes. To avoid human disturbance, the entire burrow (including covers) was buried. Artificial burrows were not buried under mounds as in Trullio (1995) and Collins and Landry (1977) because some of our early, more obvious artificial burrows were stolen (probably for the PVC pipe) before any owls had begun to use them.

All of the owl pairs used in this study and any young they produced were trapped using either a cage and one-way door trap (Banuelos 1993, PVC tube trap (Botelho and Arrowood 1995), or captured by hand in the artificial nest cavity. Captured owls were banded with USGS aluminum bands and a unique combination of colored plastic bands. We insured that all nestlings were captured by repeated observation and trapping at each burrow until all nestlings were marked on three consecutive observation periods. Because we did not excavate natural burrows, we cannot rule out that some nestlings may have gone undetected. We feel, however, that undetected nestlings, if they did occur, were rare.

Because our data is nonnormal, we used nonparamet-

Table 1. Types of burrows used by nesting Burrowing Owls and the numbers of nestlings they produced in undisturbed and disturbed areas.

	UNDISTURBED AREAS			DISTURBED AREAS		
BURROW TYPE	NO. PAIRS (%)	NO. NESTLINGS (%)	NESTLINGS/PAIR ($\bar{x} \pm 1$ SE)	NO. PAIRS (%)	NO. NESTLINGS (%)	NESTLINGS/PAIR ($\bar{x} \pm 1$ SE)
Vertical cliff	15 (47)	25 (45)	1.7 ± 1.8	4 (15)	0 (0)	0
Base of cliff	11 (34)	14 (25)	1.3 ± 1.7	7 (26)	25 (38)	3.6 ± 2.1
Flat ground	6 (19)	17 (30)	2.8 ± 4.1	12 (44)	41 (62)	3.4 ± 1.8
Above ground	0 (0)	0 (0)	0	4 (15)	0 (0)	0
Total	32	87		27	66	

ric statistics. Our alpha level for significance is 0.05. Means are reported with standard errors ($\bar{x} \pm 1$ SE).

RESULTS

In undisturbed areas, pairs used burrows in cliff walls more often than burrows on flat ground but the difference was not significant ($\chi^2 = 3.80$, df = 2, $0.10 > P > 0.02$; Table 1). In disturbed areas, pairs used burrows in flat ground more than burrows in cliff walls and above ground but, here also, the difference was not significant ($\chi^2 = 6.33$, df = 3, $0.10 > P > 0.05$). Burrows in flat ground in disturbed areas were very common and potential sites in cliff walls were less common than in natural areas because they only occurred in the stadium and banks of the irrigation canal. However, there were numerous burrows in the stadium and along the canal that were dug by squirrels. Burrows in cliff walls in disturbed areas that appeared suitable for nesting were not used. Sites at the base of cliffs were common under the concrete edges of buildings and walls. Even though burrows in culverts and pipes appeared to be common throughout disturbed areas, only four pairs utilized them. These nesting attempts failed.

In undisturbed areas, pairs that used burrows in flat ground produced significantly more nestlings than pairs in the other types of burrows (Kruskal-Wallis test, F = 13.52, df = 2, $P < 0.005$; Table 1). In disturbed areas, pairs that nested in burrows in cliff wall and above ground sites produced no nestlings. Pairs which used burrows at the base of cliff walls and in flat ground produced significantly more nestlings than their counterparts in natural areas (F = 11.40, df = 3, $P < 0.005$; Table 1).

In undisturbed areas, the distribution of breeding resident males was more equal among available burrow types than was the distribution of breeding migrant males (Table 2). Migrant males exclusively used burrows in cliff walls although the highest percentage of resident males also used burrows in cliff walls. The lowest percentage of males in natural areas used burrows in flat ground. In contrast to undisturbed areas, very few migrant and resident males used burrows in cliff walls in disturbed areas. Instead, they mostly used burrows in flat ground with resident males using burrows at the base of cliff walls more often than migrants.

Among migrants that bred in 1993 and returned to breed in 1994 ($N = 15$), 60% changed burrow site types with 67% of males and 50% of females using burrows in different site types in 1994 (Table 3). Among those migrants that bred in 1994 and returned to breed in 1995 ($N = 12$), 58% changed burrow site types. Males that returned in 1995 overwhelmingly used burrows of the same site type (75%), the reverse of what happened in 1994. Seventy-one percent of females, however, used burrows in sites different from those used in 1994. Use of same ($N = 11$) and different ($N = 16$) burrow types over both years by males and females did not differ significantly ($\chi^2 = 0.926$, df = 1, $P > 0.05$).

Table 2. Types of burrow sites used by resident and migrant male Burrowing Owls in undisturbed and disturbed areas.

	UNDISTURBED AREAS		DISTURBED AREAS	
BURROW LOCATION	NO. RESIDENT MALES (%)	NO. MIGRANT MALES (%)	NO. RESIDENT MALES (%)	NO. MIGRANT MALES (%)
Vertical cliff	11 (52)	5 (100)	0 (0)	1 (10)
Base of cliff	8 (38)	0 (0)	6 (43)	1 (10)
Flat ground	2 (10)	0 (0)	7 (50)	7 (70)
Above ground	0 (0)	0 (0)	1 (7)	1 (10)

Table 3. Burrow switching by migrant male and female Burrowing Owls between 1993–94 and 1994–95.

	1993–1994			1994–1995		
BURROW TYPE	MALES	FEMALES	TOTAL	MALES	FEMALES	TOTAL
From base of cliff to base of cliff	1	0	1	0	0	0
to cliff wall	2	1	3	0	1	1
to flat ground	0	0	0	0	0	0
to artificial burrow	0	0	0	0	0	0
From cliff wall to cliff wall	1	1	2	1	0	1
to base of cliff	1	1	2	0	1	1
to flat ground	1	0	1	0	1	1
to artificial burrow	0	1	1	0	0	0
From flat ground to flat ground	1	1	2	2	2	4
to base of cliff	1	0	1	0	0	0
to cliff wall	1	0	1	0	1	1
to artificial burrow	0	0	0	0	0	0
From artificial burrow to artificial burrow	0	1	1	0	0	0
to flat ground	0	0	0	1	2	3
to base of cliff	0	0	0	0	0	0
to cliff wall	0	0	0	0	0	0
Total nestings	9	6	15	4	7	12
Total number switches between years	6	3	9	1	5	7

Among those individuals that used burrows on different sites the following year, two (one male and one female) moved from a burrow in or at the base of a cliff wall to a burrow in flat ground (Table 3). Of the remainder, 18% moved from burrows in cliff walls to burrows at the base of cliffs. Only 11% of the owls that moved from burrows in flat ground to those in or at the base of cliff walls moved to a different site type.

The number of nestlings produced by pairs that bred in our study area in one year and returned to breed again the following year did not differ significantly regardless of burrow type used (Wilcoxon Signed Ranks Test, T = -13, $P = 0.343$, $N = 6$ for males and T = 10, $P = 0.0635$, $N = 4$ for females). On average, females that switched burrow types from one year to the next, produced more nestlings in the burrows they left rather than in their new burrows (Table 4). Males switching burrows from one year to the next produced equal numbers of nestlings in the two sites.

Eight pairs which nested in artificial burrows produced an average of 8.3 ± 3.5 eggs per pair (Table 5). The number of nestlings ranged from 0–8 ($\bar{x} = 3.5 \pm 2.9$). Clutches in all but two artificial burrows partially hatched; the two clutches which failed to hatch were abandoned prior to hatching because the mates died. Of the 28 nestlings produced in artificial burrows, only 12 or 43% fledged. In all but one burrow where all nestlings hatched synchronously, one nestling hatched much later (2–4 d) than the rest and always died. These smaller nestlings usually disappeared from the burrow overnight either through predation or cannibalism. One female was videotaped feeding her youngest nestling to the surviving young. Older nestlings which failed to fledge also disappeared quickly from burrows without a trace. Owls in artificial burrows produced an average of 3.5 ± 2.9 nestlings ($N = 8$ nests) which is significantly higher than production in natural burrows (2.2 ± 1.9 nestlings, $N = 59$ nests; Mann-Whitney U test, Z = -2.07, $N = 67$, $P < 0.02$). When pairs abandoning their burrows prior to hatching were removed from the analysis, owls which used artificial burrows still produced significantly more nestlings ($\bar{x} = 3.3 \pm 1.3$ nestlings for natural and $\bar{x} = 4.7 \pm 2.3$ nestlings for artificial burrows, Z = -1.68, $N = 44$, $P < 0.05$). However, when we compared the number of fledglings produced by the two types of burrows, the number produced by pairs in natural burrows was significantly greater for natural than for artificial burrows ($\bar{x} = 1.9 \pm 1.9$ nestlings for natural and $\bar{x} = 1.5 \pm 1.5$ nestlings for artificial burrows, Z = -2.81, $N = 67$, $P < 0.003$). The average number of fledglings produced by pairs in

Table 4. Number of nestlings produced by migrant Burrowing Owls that returned to the same and different burrow types between 1993–94 and 1994–95.

	MALES			FEMALES		
	CURRENT YEAR	PREVIOUS YEAR	TOTAL	CURRENT YEAR	PREVIOUS YEAR	TOTAL
From base of cliff to base of cliff	0	4	4	0	0	0
to vertical cliff	1	6	7	6	7	13
to flat ground	0	0	0	0	0	0
to artificial burrow	0	0	0	0	0	0
From vertical cliff to vertical cliff	1	0	1	0	0	0
to base of cliff	2	1	3	0	4	4
to flat surface	0	0	0	3	8	11
to artificial burrow	0	0	0	3	3	6
From flat surface to flat surface	3	3	6	6	6	12
to base of cliff	3	3	6	0	0	0
to vertical cliff	3	2	5	3	3	6
to artificial burrow	0	0	0	0	0	0
From artificial to artificial burrow	0	0	0	3	3	6
to flat ground	6	7	13	9	10	19
to base of cliff	0	0	0	0	0	0
to cliff wall	0	0	0	0	0	0
Average for migrants returning to same type of burrow	1.0 ± 1.4	1.8 ± 2.1		2.3 ± 2.9	2.3 ± 2.9	
Average for migrants returning to different type of burrow	1.3 ± 1.9	1.6 ± 2.5		2.0 ± 3.0	2.9 ± 3.6	

Table 5. Hatching and fledging success of eight pairs of Burrowing Owls nesting in artificial burrows form 1993–95. Fledglings are defined as young that were observed flying in their natal territories.

PAIR	CLUTCH SIZE	NO. EGGS HATCHING (%)	NO. FLEDGLINGS (%)
1	6	0 (0)	0 (0)[a]
2	7	0 (0)	0 (0)[a]
3	7	3 (43)	2 (67)
4	7	3 (27)	2 (67)
5	8	7 (64)	4 (57)
6	9	3 (33)	0 (0)
7	11	4 (36)	3 (75)
8	11	8 (73)	1 (13)
Total	66	28	12
Mean	8.3	3.5	1.5
SE	1.9	2.9	1.5
Total[b]	53	28	12
Mean[b]	8.8	4.6	2.0
SE[b]	1.8	2.3	1.4

[a] Nest abandoned.

[b] Abandoned burrows have been omitted.

natural burrows where adults did not abandon was 2.9 ± 1.5, significantly higher than that produced by pairs in artificial burrows (2.0 ± 1.4 fledglings, $Z = -2.97$, $N = 43$, $P < 0.002$).

DISCUSSION

Our prediction that males would use burrows located in sites with high elevation and low grass cover in cliffs more often due to decreased predation was supported for resident and migrant males in natural areas but not disturbed areas. Only one male in a disturbed area used a burrow in a cliff wall despite the apparent availability of cliff sites. One possible reason for not using burrows in cliffs may have been that they were located in the territories of other males not using cliff burrows. Another reason may have been the possible high mortality of fledglings when they fell from their burrows although this seemed unlikely because pairs using burrows in cliff walls were as productive as pairs that used burrows at the base of cliffs in natural areas.

Burrows in cliff walls appeared to be safer from predators because of their height and approach by predators was possible in only one direction. MacCracken et al. (1985) and Green and Anthony (1989) have shown that Burrowing Owls use burrows located in sites on mounds of dirt with low grass cover, but our study shows for the first time that Burrowing Owls can also use sites associated with cliffs. The presence of depressions surrounded by steep cliffs coupled with the tendency of rock squirrels and rabbits to colonize these areas and dig holes in and at the bases of the cliff walls can provide an unusual type of nest site for Burrowing Owls. The only other case we know of where Burrowing Owls have been shown to use burrows in or at the bases of cliffs is in Albuquerque, New Mexico, 155 km north of our study site (Kendall pers. comm.).

Our prediction that pairs should use burrows at the bases of cliffs more often than on flat ground was supported for undisturbed areas but not disturbed areas. Also, pairs that nested at the bases of cliffs in undisturbed areas produced fewer nestlings on average than pairs which nested either in cliff walls or in flat ground. In disturbed areas, however, pairs that nested in the bases of cliffs produced more nestlings than all other burrow types. Larger broods in disturbed areas may have been due to increased prey availability attributed to artificial lighting, especially in the stadium. Also, the larger amount of space at the entrances to burrows in the bases of cliffs better accommodated larger broods and restricted the approach routes of predators. Decreased risk of nestling predation in disturbed areas may have contributed to this trend but we have no data on the effect of predation on the reproductive success of this population.

Most studies of Burrowing Owls have found them occupying burrows in relatively flat ground although some elevation near the burrow is important. Burrowing Owls in Oregon (Green and Anthony 1989), in South Dakota (MacCracken et al. 1985), and in Colorado (Plumpton and Lutz 1993) preferred burrows on high ground with low mean shrub volume or low grass cover, possibly to gain an elevated unobstructed view. Females in this study monitored their surroundings from an elevated site with a clear view and gave alarm calls to which the nestlings responded by running into the burrow. For flightless nestlings to respond quickly, females must produce alarm calls well in advance of a predator's approach making a clear view of the area surrounding the burrow important. In undisturbed areas, pairs using burrows in flat ground were most productive; in disturbed areas such pairs produced only slightly fewer nestlings than pairs at the bases of cliffs.

The lower overall productivity of pairs in undisturbed areas may have been due in part to predation. A pair of Barn Owls (*Tyto alba*) used a burrow located in a cliff wall in the landfill only 2–3 m away from an occupied Burrowing Owl nest and within easy striking distance of up to 13 other nests. Burrowing Owls actively mobbed the Barn Owls as they left their burrow but we are unaware of any predation by the Barn Owls on Burrowing Owls. Also, lack of an available food supply close to their burrows may have lowered productivity, especially among those pairs which nested in areas without the benefit of insects attracted by artificial lighting. Violent storms, which passed through the study area in late summer, may have also resulted in the deaths of small nestlings caught outside their burrows.

Most females and males which returned to a different burrow type from one year to the next produced fewer nestlings in their second breeding attempt than in their first. Decreased reproductive success in new burrows may explain why owls switched burrows infrequently and never accepted artificial burrows installed in the vicinity of their nesting burrow.

An average hatching and fledging success of 42% and 18%, respectively, by pairs which nested in artificial burrows was lower than that found in other studies where artificial burrows have been used (Landry 1979, Olenick 1987). Pairs that nested in artificial burrows produced significantly more nestlings than pairs that used natural burrows even if pairs that failed to hatch any eggs were included in the analysis. In fact, pairs which nested in artificial burrows produced almost one nestling more on average than their counterparts in natural burrows. The opposite was true for fledglings. Pairs that nested in natural burrows produced significantly more fledglings than pairs that used artificial burrows regardless of whether pairs failing to hatch any eggs were included in the analysis. After removing pairs that failed to hatch any eggs, pairs nesting in natural burrows produced almost one more fledgling on average than pairs which used artificial burrows. These results were unexpected because we thought the antipredator advantages of artificial burrows would enhance fledgling production. Nestlings in artificial burrows were captured inside the nest chamber and weighed three to four times per week during the nestling period to determine growth rates for another study. Artificial burrows, however, were not disturbed once clutches were complete and incubation began. Thus, one reason for the observed trend in nestling and fledgling production by pairs in natural and artificial burrows could have been human disturbance during the nestling period and the lack of it during the incubation period.

We suggest that conservation plans for Burrowing Owls involving the use of artificial burrows in areas without natural nesting burrows should consider the characteristics of burrow sites previously used by the owls for nesting. Because some of the owls that switched burrows from year to year suffered decreased nesting success, there may be selection against year to year movement among burrows. Given their nest site fidelity (Haug et al. 1993), disturbance of nest sites could have a devastating impact on Burrowing Owl populations, even if artificial burrows are installed nearby.

This study demonstrates the importance of installing artificial burrows in sites most favored by nesting pairs. Owls in this study nesting in undisturbed areas used burrows located in and at the bases of cliff walls where artificial burrows could not be installed. On average, pairs in artificial burrows produced significantly more nestlings than pairs in natural burrows, indicating that artificial burrows did not contribute to decreased nestling productivity. Furthermore, human disturbance may have played a role in lower fledgling production by pairs in artificial burrows.

ACKNOWLEDGMENTS

We are grateful to the physical plant at NMSU for allowing access to our field sites on campus. Dan Howard provided helpful comments on the manuscript. Dennis Clason provided help with statistical analyses. Betsy Botelho helped with field work. We thank Marc Bechard, Jim Belthoff, Elizabeth Haug, and Paul James for help with revision of this manuscript and Phil Zwank for arranging the loan of night vision equipment from the USFWS Wildlife Cooperative Research Unit on the campus of NMSU. This research was supported by the NMSU Department of Biology, a Frank M. Chapman Award from the American Museum of Natural History, and grants from the New Mexico Council for Higher Education and the Chicago Zoological Society to ESB.

LITERATURE CITED

BANUELOS, G.H.T. 1993. An alternative trapping method for Burrowing Owls. *J. Raptor Res.* 27:85–86.

BOTELHO. E.S. AND P.C. ARROWOOD. 1995. A novel, simple, safe and effective trap for Burrowing Owls and other fossorial animals. *J. Field Ornithol.* 66:380–384.

——— AND ———. 1996. Nesting success of western Burrowing Owls (*Speotyto cunicularia hypugaea*) on natural and human-altered sites. Pages 61–68 *in* D.M. Bird, D.J. Varland and J.J. Negro [EDS.], Raptors in human landscapes. Academic Press, London, U.K.

COLLINS, C.T. AND R.E. LANDRY. 1977. Artificial nest burrows for Burrowing Owls. *N. Am. Bird Bander* 2:151–154.

COULOMBE, H.N. 1971. Behavior and population ecology of the Burrowing Owl (*Speotyto cunicularia*) in the Imperial Valley of California. *Condor* 73:162–176.

GREEN, G.A. AND R.G. ANTHONY. 1989. Nesting success and habitat relationships of Burrowing Owls in the Columbia Basin, Oregon. *Condor* 91:347–354.

HAUG, E.A., B.A. MILLSAP AND M.S. MARTELL. 1993. Burrowing Owl (*Speotyto cunicularia*). Pages 1–20 *in* A.F. Poole and F. Gill [EDS.], The Birds of North America, No. 61. Academy of Natural Sciences, Philadelphia, PA U.S.A. and the American Ornithologists' Union, Washington, DC U.S.A.

LANDRY, R.E. 1979. Growth and development of the Burrowing Owl, *Athene cunicularia.* M.A. thesis, California State University, Long Beach, CA U.S.A.

MACCRACKEN, J.G., D.W. URESK AND R.M. HANSEN. 1985. Vegetation and soils of Burrowing Owl nest sites in Conata Basin, South Dakota. *Condor* 87:152–154.

OLENICK, B. 1987. Reproductive success of Burrowing Owls using artificial burrows in southeast Idaho. *Eyas* 10:38.

PLUMPTON, D.L. AND S.R. LUTZ. 1993. Nesting habitat use by Burrowing Owls in Colorado. *J. Raptor Res.* 27:175–179.

THOMSEN, L. 1971. Behavior and ecology of Burrowing Owls on the Oakland Municipal Airport. *Condor* 73: 177–192.

TRULLIO, L.A. 1995. Passive relocation: a method to preserve Burrowing Owls on disturbed sites. *J. Field Ornithol.* 66:99–106.

Received 3 August 1996; accepted 8 May 1998

J. Raptor Res. 32(3):241–245
© 1998 The Raptor Research Foundation, Inc.

BREEDING-SEASON FOOD HABITS OF BURROWING OWLS (*ATHENE CUNICULARIA*) IN SOUTHWESTERN DOMINICAN REPUBLIC

JAMES W. WILEY[1]
Biological Research Division, U.S. Geological Survey, Reston, VA 22092 U.S.A.

ABSTRACT.—Diet data from 20 Burrowing Owl (*Athene cunicularia*) nests were collected in southwestern Dominican Republic in 1976, 1982, and 1996. Invertebrates (53.3%) comprised the most numerous prey items ($N = 396$) delivered to nests by adult owls, but vertebrates (46.7%) were much better represented than in other studies of Burrowing Owl diet. Among vertebrates, birds (28.3% of all items) and reptiles (14.9%) were most important, whereas mammals (1.0%) and amphibians (2.5%) were less commonly delivered to nests. Vertebrates, however, comprised more than twice (69.2%) of the total biomass as invertebrates (30.8%), with birds (50.4%) and reptiles (12.8%) the most important of the vertebrate prey classes. A positive relationship was observed between bird species abundance and number of individuals taken as prey by Burrowing Owls.

KEY WORDS: Athene cunicularia; *Burrowing Owl; diet; Dominican Republic; ecology.*

Habitos alimenticios durante la epoca reproductiva de *Athene cunicularia* en el suroeste de la República Dominicana

RESUMEN.—Datos de la dieta de 20 nidos de *Athene cunicularia* fueron colectados en el suroeste de la República Dominicana durante 1976, 1982 y 1996. Los invertebrados (53.3%), fueron los items más numerosos ($N = 396$) entregados en los nidos por los buhos adultos, pero los vertebrados (46.7%) fueron mucho mejor representados que en otros estudios sobre hábitos alimenticios de *Athene cunicularia.* Entre los vertebrados, las aves (28.3% de todos los items) y los reptiles (14.9 %) fueron los más importantes mientras que los mamíferos (1.0%) y los anfibios (2.5%) fueron menos comunes. Los vertebrados sin embargo, constituyeron más del doble (69.2%) del total de la biomasa. Los invertebrados (30.8%), aves (50.4%) y los reptiles (12.8%) fueron las clases más importantes de presas. Existió una relación positiva entre la abundancia de especies de aves y el número de individuos capturados como presas por *Athene cunicularia.*

[Traducción de César Márquez]

Burrowing Owls (*Athene cunicularia*) occur from western Canada south through the western U.S., Mexico, Central and South America, and irregularly in Florida and the West Indies. In the Caribbean islands they presently inhabit most of the Bahamas; Cuba, including several cays of the Sabana-Camagüey Archipiélago and the Isla de la Juventud (previously Isla de Pinos); and the western half of Hispaniola, including Gonâve and Beata islands (AOU 1998). The species formerly occurred throughout the Greater Antilles and several of the Lesser Antilles. Populations disappeared from Jamaica (Olson and Steadman 1977, Morgan 1993), the Cayman Islands (Morgan 1977, 1994), and Puerto Rico (Pregill and Olson 1981), possibly as a result of changing climate and habitat conditions, and predation by introduced mammals (Pregill 1981, Pregill and Olson 1981, Wiley 1986a). In the Lesser Antilles, Burrowing Owls recently occurred in St. Kitts, Nevis, Antigua, Redonda, and Marie Galante (AOU 1983). However, the two races endemic to the Lesser Antilles are thought to be extinct: *A. c. guadeloupensis* from Marie Galante, and *A. c. amaura* from Antigua, Nevis, Redonda, and St. Kitts, again partly as a result of predation by exotic animals (Greenway 1967).

In spite of the losses of several populations from former ranges and current concern for the species' survival in the Caribbean, little has been reported on the biology of Burrowing Owls in the region

---

[1] Present Address: Grambling Cooperative Wildlife Project, P.O. Box 841, Grambling State University, Grambling, LA 71245 U.S.A.

Table 1. Prey brought to nests by breeding Burrowing Owls, foothills of the Sierra de Bahoruco, southwestern Dominican Republic, 1976, 1982, and 1996.

	NUMBER OF PREY ITEMS (%)				BIOMASS (%)	
PREY SPECIES	OBSERVED BROUGHT TO NEST (1976)	PREY REMAINS	PELLETS	TOTAL	MEAN PREY BIOMASS (G)	TOTAL
Mammals						
House mouse *Mus musculus*	1(2.8)	1(0.8)	1(0.4)	3(0.8)	21.0	63.0(2.1)
Black rat *Rattus rattus* (young)			1(0.4)	1(0.3)	80.0	80.0(2.6)
Total mammals	1(2.8)	1(0.8)	2(0.9)	4(1.0)		143.0(4.7)
Birds						
Common Ground-Dove *Columbina passerina*		2(1.6)	2(0.9)	4(1.0)	28.9	115.6(3.8)
Antillean Mango *Anthracothorax dominicus*	3(8.3)	5(3.9)	7(3.0)	15(3.8)	5.2	78.0(2.6)
Hispaniolan Emerald *Chlorostilbon swainsonii*		1(0.8)		1(0.3)	4.6	4.6(0.2)
Broad-billed Tody *Todus subulatus*	2(5.6)	6(4.7)	10(4.3)	18(4.6)	8.3	149.4(4.9)
Unidentified tody *Todus* sp.		2(1.6)	3(1.3)	5(1.3)	8.0	40.0(1.3)
Stolid Flycatcher *Myiarchus stolidus*	1(2.8)		1(0.4)	2(0.5)	24.1	48.2(1.6)
Hispaniolan Pewee *Contopus hispaniolensis*		2(1.6)		2(0.5)	11.5	23.0(0.8)
Northern Mockingbird *Mimus polyglottos*						
Adult			1(0.4)	1(0.3)	42.3	42.3(1.4)
Fledgling	1(2.8)	1(0.8)		2(0.5)	35.0	70.0(2.3)
Red-legged Thrush *Turdus plumbeus*		1(0.8)	1(0.4)	2(0.5)	74.0	148.0(4.9)
Black-whiskered Vireo *Vireo altiloquus*		3(2.3)		3(0.8)	19.1	57.3(1.9)
Flat-billed Vireo *Vireo nanus*			1(0.4)	1(0.3)	10.7	10.7(0.4)
American Redstart *Setophaga americana*	1(2.8)		1(0.4)	2(0.5)	8.7	17.4(0.6)
Green-tailed Warbler *Microligea palustris*	1(2.8)	5(3.9)	9(3.9)	15(3.8)	12.5	187.5(6.2)
Ovenbird *Seiurus aurocapillus*			1(0.4)	1(0.3)	18.7	18.7(0.6)
Bananaquit *Coereba flaveola*	1(2.8)	4(3.1)	12(5.2)	17(4.3)	8.7	147.9(4.9)
Black-crowned Palm-Tanager *Phaenicophilus palmarum*		1(0.8)	6(2.6)	7(1.8)	30.5	213.5(7.0)
Greater Antillean Bullfinch *Loxigilla violacea*		2(1.6)		2(0.5)	22.3	44.6(1.5)
Yellow-faced Grassquit *Tiaris olivacea*		1(0.8)	1(0.4)	2(0.5)	8.0	16.0(0.5)
Unidentified bird			10(4.3)	10(2.5)	9.6	96.0(3.2)
Total birds	10(27.8)	36(28.1)	66(28.4)	112(28.3)		1528.7(50.4)
Amphibians						
Eleutherodactylus abbotti	1(2.8)	3(2.3)	6(2.6)	10(2.5)	4.0	40.0(1.3)
Total amphibians	1(2.8)	3(2.3)	6(2.6)	10(2.5)		40.0(1.3)
Reptiles						
Ameiva chrysolaema	1(2.8)		5(2.2)	6(1.5)	8.9	53.4(1.8)
Anolis distichus	2(5.6)	4(3.1)	15(6.5)	21(5.3)	6.7	140.7(4.6)
Anolis semilineatus	3(8.3)	7(5.5)	9(3.9)	19(4.8)	6.4	121.6(4.0)
Sphaerodactylus cryphius	1(2.8)	4(3.1)	5(2.2)	10(2.5)	4.3	43.0(1.4)
Typhlops hectus		1(0.8)		1(0.3)	9.0	9.0(0.3)
Uromacer frenatus		1(0.8)	1(0.4)	2(0.5)	10.5	21.0(0.7)
Total reptiles	7(19.4)	17(13.3)	35(15.1)	59(14.9)		388.7(12.8)
Total vertebrates	19(52.7)	57(44.5)	109(47.0)	185(46.7)		2100.4(69.2)

Table 1. Continued.

PREY SPECIES	NUMBER OF PREY ITEMS (%)				BIOMASS (%)	
	OBSERVED BROUGHT TO NEST (1976)	PREY REMAINS	PELLETS	TOTAL	MEAN PREY BIOMASS (G)	TOTAL
Invertebrates						
unidentified locusts	5(13.9)	21(16.4)	38(16.4)	64(16.2)	2.0	128.0(4.2)
unidentified beetles	8(22.2)	28(21.9)	57(24.6)	93(23.5)	3.1	288.3(9.5)
unidentified tarantulas	3(8.3)	17(13.3)	19(8.2)	39(9.9)	9.7	378.3(12.5)
unidentified centipedes	1(2.8)	5(3.9)	9(3.9)	15(3.8)	9.3	139.5(4.6)
Total invertebrates	17(47.2)	71(55.5)	123(53.0)	211(53.3)		934.1(30.8)
Totals	36	128	232	396		3034.5(100.0)

(Wiley 1986a, 1986b). Here, I report on the high incidence of avian prey I observed in the diet of Burrowing Owls in southwestern Dominican Republic based on direct observations, prey remains, and regurgitated pellets collected at nests.

METHODS

Data were collected in southwestern Dominican Republic from March–June 1976, June–July 1982, and March 1996 from occupied Burrowing Owl nests along the lower slopes of the Sierra de Bahoruco, from Cruce de Limón near Lago Enriquillo (elevation 30 m) south to El Naranjo west of Puerto Escondido (elevation 350 m). The area is in the subtropical dry woodland zone (Union Panamericana 1967) characterized by acacia-cactus woodland that becomes more luxuriant with increasing elevation (Durland 1922). Typical vegetation include cacti (guasábara pilotera *Opuntia antillana*, cagüey *Neoabbottia paniculata*, cayuco *Pilosocereus polygonus*), palmera yarey (*Copernicia berteroana*), bayahonda (*Prosopis juliflora*), *Capparis* spp., baitoa (*Phyllostylon brasiliensis*), aroma (*Acacia farnesiana*), guayacán (*Guajacum officinale*), guayacancillo (*Guajacum sanctum*), almácigo (*Bursera simaruba*), guano (*Coccothrinax argentata*), and doncella (*Byrsonima lucida*). Annual rainfall averages about 455 mm, with peaks in January, April–May, and August–November.

Owls were common in the area and nested wherever suitable substrate was available. I systematically surveyed the owl population in my study area by foot on several occasions, paying special attention to the presence of nest sites. All possible nests were revisited a minimum of four times to confirm their occupancy. In 1976, one 9-ha study plot contained 18 occupied owl nests ($\bar{x}$ = 1 nesting pair per 0.5 ha), concentrated in three colonies of 3, 8, and 7 pairs (mean nearest-neighbor distance = 22.5 ± 6.7 m SD; range = 15–35 m).

All old pellets (i.e., regurgitated castings) and prey remains were cleared from study nests the day before I began each period of data collection. I spent 42 hr observing two occupied nests from blinds placed 4 m from the burrows on 14–15 April (1600–2300 H), 15–16 April (1930–0600 H), 30 April–1 May (2200–0530 H), 1–2 May (1600–2300 H), and 15–16 May (1630–0230 H), 1976. Although observations are preferable to prey remains and pellets for analyzing diets of birds of prey (Snyder and Wiley 1976), prey remains and regurgitated pellets do provide traditional materials for examination of raptor food habits (Errington 1930). I collected remains and pellets at eight nests in 1976, five in 1982, and five in 1996. Observations from blinds were made at different nests than those from which I collected pellets and prey remains. All data were collected during the nestling stage before young emerged from nests. Prey remains and items in pellets were identified by comparison with specimens in the Museo Nacional de Historia Natural (Santo Domingo; MNHN), using a dissecting microscope when needed. Prey biomass was determined from animals captured in or near my study area, from specimen data labels in the MNHN, and from data provided by Anabelle Dod. I surveyed bird populations for relative abundance of species using fixed transects (Emlen 1971, 1977) and mist nets on or adjacent to the Burrowing Owl study area.

RESULTS

A total of 396 prey items was identified at least to order (Table 1). Invertebrates (53.3% of all prey items) made up the most numerous items brought to nests by adult owls. Beetles (23.5% of all items), locusts (16.2%), and tarantulas (9.9%) were the most commonly delivered prey. Birds (28.3%) and reptiles (14.9%) were also important items, whereas mammals (1.0%) and amphibians (2.5%) made up only a minor portion of the prey brought to nests. Among birds, Broad-billed Todies (*Todus subulatus*; 4.6% of items), Bananaquits (*Coereba flaveola*; 4.3%), Antillean Mangos (*Anthracothorax dominicus*; 3.8%), and Green-tailed Warblers (*Microligea palustris*; 3.8%) were the most common items. Among reptiles, anole lizards (*Anolis distichus*, 5.3%

and *A. semilineatus,* 4.8%) were the most frequently delivered species.

At the two nests watched from blinds in 1976, the prey delivery rate averaged 0.86 ± 0.68 (±1 SD) items per hour. A young rat (*Rattus rattus*) observed brought to a nest by an adult was estimated to weigh about 80 g and represented the largest prey item (53% of mean adult Burrowing Owl biomass) in the sample. The largest avian prey items delivered were Red-legged Thrushes (*Turdus plumbeus;* $\bar{x}$ = 74 ± 2.53 g, N = 17; 49% of mean adult owl mass).

Whereas invertebrates were the most numerous items brought to nests by adult owls, vertebrates comprised more than twice (69.2%) as much of the total biomass as invertebrates (30.8%) in the combined samples, with birds (50.4%) and reptiles (12.8%) the most important of the vertebrate prey classes. Mean weights of prey species ranged from 2.0–80.0 g, with vertebrates averaging 18.7 ± 19.0 g and invertebrates 6.0 ± 4.0 g. The greatest range within prey categories occurred among birds, which varied from 4.6–74.0 g.

I found a positive relationship between bird species abundance and numbers of individuals taken as prey by Burrowing Owls (Spearman Rank Correlation, Z = 2.1, P = 0.04, N = 17). I did not evaluate relative abundance of populations of other prey categories but, based on my casual observations, at least reptiles showed some degree of correlation between abundance and numbers taken as Burrowing Owl prey.

DISCUSSION

Burrowing Owls in North America feed primarily on invertebrates (90.9%) and only occasionally eat mammals (6.9%), reptiles and amphibians (2.0%), and birds (0.3%) (summarized in Earhart and Johnson 1970 and Snyder and Wiley 1976). The diet of Burrowing Owls in the West Indies has been reported as consisting of small rodents, small reptiles, frogs, and, especially, large insects, including crickets, grasshoppers, and beetles (e.g., Brudenell-Bruce 1975, Campbell 1978, Dod 1978, Garrido 1992, Kirkconnell et al. 1992). Danforth (1929) reported the contents of two Burrowing Owl stomachs, one of which contained mouse fur, whereas the other contained beetle and centipede parts. Abbott (*in* Wetmore and Swales 1931) found one Burrowing Owl stomach contained one lizard, one scorpion, one mouse, and several insects.

Although avian prey were particularly well-represented in my samples, only occasional mention has been made of Burrowing Owls preying on birds in the West Indies (Brudenell-Bruce 1975, Dod 1978, Kirkconnell et al. 1992), including two cases of cannibalism or scavenging (Abbott *in* Wetmore and Swales 1931, Brudenell-Bruce 1975). Gnatcatchers (*Polioptila* sp.) were among the contents of five Burrowing Owl stomachs collected by Regalado (1975) in Cuba. "R.H.L." (1883) reported the remains of Black-cowled Orioles (*Icterus dominicensis*), Greater Antillean Grackles (*Quiscalus niger*), and Common Ground-Doves (*Columbina passerina*) at Burrowing Owl nests in Haiti. In the southern Bahama Islands, Buden (1974) noted a high proportion of avian prey in Barn Owl (*Tyto alba*) food remains compared with continental samples. He suggested that these results reflected lower abundance of rodents on islands.

Generally, direct observations of prey delivered by raptors to their nests reveal a higher proportion of smaller and more delicate items, such as some arthropods, than do examinations of prey remains and regurgitated pellets (Snyder and Wiley 1976). Thus, small items generally are underrepresented in analyses of remains and pellets. However, the proportion of invertebrates (47.2%) I observed brought to the nests was slightly lower than that represented by remains (55.5%) and pellet contents (53.0%). Collectively, vertebrates (52.8%) were observed brought to the nests as often as invertebrates. The combined data from prey remains and pellet contents revealed higher incidence of invertebrates (53.9%), but vertebrates (46.1%) were not far outnumbered by arthropods. Based on prey biomass, however, vertebrates clearly represented a far more important food source than did invertebrates during the breeding season.

All of the prey species brought to nests by Burrowing Owls were common and, among birds, all but the American Redstart (*Setophaga americana*) and Ovenbird (*Seiurus aurocapillus*) are residents in Hispaniola.

ACKNOWLEDGMENTS

For their assistance in the field, I thank Beth Wiley and Julio E. Cardona. Annabelle Dod, José A. Ottenwalder, and Sixto J. Inchaustegui generously assisted with collections at the Museo Nacional de Historia Natural in Santo Domingo. I thank James R. Belthoff and two anonymous reviewers for their help in improving the manuscript.

LITERATURE CITED

AMERICAN ORNITHOLOGISTS' UNION. 1998. Check-list of North American birds, 7th Ed. A.O.U., Washington, DC U.S.A.

BRUDENELL-BRUCE, P.G.C. 1975. The birds of the Bahamas. New Providence and the Bahama Islands. Taplinger Publ. Co., New York, NY U.S.A.

BUDEN, D.W. 1974. Prey remains of Barn Owls in the southern Bahama Islands. *Wilson Bull.* 86:336–343.

CAMPBELL, D.G. 1978. The ephemeral islands. A natural history of the Bahamas. McMillan Education Limited, London, U.K.

DANFORTH, S.T. 1929. Notes on the birds of Hispaniola. *Auk* 46:358–375.

DOD, A.S. DE. 1978. Aves de la República Dominicana. Mus. Nac. Hist. Nat., Santo Domingo.

DURLAND, W.D. 1922. The forests of the Dominican Republic. *Geogr. Rev.* 1922:206–222.

EARHART, C.M. AND N.K. JOHNSON. 1970. Size dimorphism and food habits of North American owls. *Condor* 72:251–264.

EMLEN, J.T. 1971. Population densities of birds derived from transect counts. *Auk* 88:323–341.

———. 1977. Estimating breeding season bird densities from transect counts. *Auk* 94:455–468.

ERRINGTON, P.L. 1930. The pellet analysis method of raptor food habits study. *Condor* 32:292–296.

GARRIDO, O.H. 1992. Conozca las rapaces. Editorial Gente Nueva, La Habana, Cuba.

GREENWAY, J.C., JR. 1967. Extinct and vanishing birds of the world, 2nd revised ed. Dover Publ., Inc., New York, NY U.S.A.

KIRKCONNELL, A., O.H. GARRIDO, R.M. POSADA RODRIGUEZ AND S.O. CUBILLOS. 1992. Los grupos tróficos en la avifauna cubana. *Poeyana* No. 415:1–21.

MORGAN, G.S. 1977. Late Pleistocene fossil vertebrates from the Cayman Islands, British West Indies. M.S. thesis, Univ. Florida, Gainesville, FL U.S.A.

———. 1993. Quaternary land vertebrates of Jamaica. Pages 417–442 *in* R.M. Wright and E. Robinson [EDS.], Biostratigraphy of Jamaica. Geological Society of Marica Memoir 182, Boulder, CO U.S.A.

———. 1994. Late Quaternary fossil vertebrates from the Cayman Islands. Pages 465–508 *in* M.A. Brunt and J.E. Davies [EDS.], Monographiae biologicae volume 7 Kluwer Academic Publishers, Dordrecht, The Netherlands.

OLSON, S.L. AND D.W. STEADMAN. 1977. A new genus of flightless ibis (Threskiornithidae) and other fossil birds from cave deposits in Jamaica. *Proc. Biol. Soc. Wash.* 90:447–457.

PREGILL, G.K. 1981. Late Pleistocene herpetofaunas from Puerto Rico. *Univ. Kansas Mus. Nat. Hist. Misc. Publ.* 71:1–72.

——— AND S.L. OLSON. 1981. Zoogeography of West Indian vertebrates in relation to Pleistocene climatic cycles. *Annu. Rev. Ecol. Syst.* 12:75–98.

REGALADO RUÍZ, P. 1975. Primer hallazgo de *Speotyto cunicularia* (Molina) anidando en Cuba. *Revista Baracoa* No. 1-2:36–56.

R.H.L. 1833. On the burrowing owl. *Field Naturalist* 1. 459–461.

SNYDER, N.F.R. AND J.W. WILEY. 1976. Sexual size dimorphism in hawks and owls of North America. AOU Monogr. No. 20.

UNION PANAMERICANA. 1967. Reconocimiento y evaluacion de los recursos naturales de la República Dominicana. Estudio para su desarrollo y planificacion. Organización de los Estados Americanos, Washington, DC U.S.A.

WETMORE, A. AND B.H. SWALES. 1931. The birds of Haiti and the Domninican Republic. U.S. Nat. Mus. Bull 155.

WILEY, J.W. 1986a. Habitat change and its effects on Puerto Rican raptors. *Birds of Prey Bull.* No. 3:51–56.

———. 1986b. Status and conservation of raptors in the West Indies. *Birds of Prey Bull.* No. 3:57–70.

Received 21 November 1997; accepted 12 May 1998

SHORT COMMUNICATIONS

J. Raptor Res. 32(3):246–247
© 1998 The Raptor Research Foundation, Inc.

RATES OF OPEN-FIELD FORAGING BY THE MISSISSIPPI KITE (*ICTINIA MISSISSIPPIENSIS*)

E. WILLIAM WISCHUSEN
Department of Biological Sciences, 104 Life Sciences Building, Baton Rouge, LA 70803 U.S.A.

KEY WORDS: *Mississippi Kite;* Ictinia mississippiensis; *foraging success; foraging habitat.*

During the past 50 yr, populations of the Mississippi Kite (*Ictinia mississippiensis*) appear to have recovered from earlier declines (Levy 1971, Parker 1977, Parker and Ogden 1977, Glinski and Ohmart 1983). Although several hypotheses have been proposed to explain the recovery, the reason or reasons remain unclear. The hypotheses propose that habitat changes on the breeding or wintering grounds have allowed for this recovery (Parker and Ogden 1977, Glinski and Ohmart 1983). These hypotheses stress changes in the habitats that either increase the nesting or foraging opportunities for kites. While several studies describe nesting habitats used by Mississippi Kites (Sutton 1939, Jackson 1945, Glinski and Ohmart 1983, Cely 1987), only one documents foraging success (Skinner 1962). Skinner found that kites averaged six successful captures of insects per 40 min interval while foraging over an open field in Alabama. I report on the results of observations of Mississippi Kites foraging over a large open field in southern Louisiana.

STUDY AREA AND METHODS

The study area was located in the Sherburne Wildlife Management Area approximately 5 km southeast of the town of Krotz Springs, in southern Louisiana. The area consisted of an approximately 1000 ha old field surrounded by bottomland hardwood forest on three sides and bordered by Big Alabama Bayou on the fourth. Foraging by Mississippi Kites was observed using 7 × 35 binoculars and a 20× spotting scope. Individual birds were observed for 2-min intervals which was long enough to include one or more forage attempts and short enough to insure that birds would remain in suitable viewing range for the entire period. During each foraging interval, I recorded the number of successful prey captures. Prey capture success or failure was determined by observing whether or not the individual fed following a prey capture attempt. After each foraging interval, a different individual was chosen to observe for the next foraging interval.

During July–August 1995 and May–August 1996, observations of Mississippi Kite foraging were conducted on 10 d between 1000–1530 H. All observations were made under partly cloudy to sunny and calm weather conditions.

RESULTS AND DISCUSSION

The foraging behavior observed was similar to that described by Skinner (1962). Kites soared at heights of 50–100 m and made steep stoops to capture insects. Once prey was captured, the birds would level off and eat the prey while soaring over the field (Skinner 1962). The number of kites observed foraging ranged from 1 to >50, with an average of 4–6 individuals observed each day These numbers are similar to the average of 10 reported by Skinner (1962).

Mississippi Kites were observed for 248 2-min foraging intervals during this period. The kites caught an average of 1.18 ± 0.076 (± 1 SE, range = 0–5) prey items per 2-min foraging interval for a total of 292 total prey items. This is a much higher rate of prey capture than previously reported. Skinner (1962) reported an average open field foraging rate of 6 prey items per 40-min interval or 0.3 prey items per 2-min foraging interval. His observations were limited to a 5-d period in July (Skinner 1962)

On a monthly basis there were significant differences between the rate of prey item capture (ANOVA, F = 3.036, df = 3, 244, $P = 0.03$). The highest success rates were observed during May and the lowest during August (Fig. 1). Some of the differences in foraging rates between months may be explained by juvenile birds foraging in July and August. Although the ages of all foraging birds were not determined, some juveniles were included in the July and August samples. All observed prey captures were insects. Dragonflies and large grasshoppers were very abundant in this field and have been reported as main prey for this species (Wayne 1906, Sutton 1939, Jackson 1945, Skinner 1962).

The results of this study suggest that Mississippi Kites can be much more efficient at foraging over open fields than reported by Skinner (1962). Most importantly, the results point out the need for comparative data of kite foraging in other habitats and in other locations. Together these data would allow for a better understanding of the foraging abilities of this species and might eventually lead to a better understanding of its recovery.

RESUMEN.—Las poblaciones de *Ictinia mississippiensis* en América del Norte han aumentado recientemente Aunque las modificaciones del habitat son consideradas

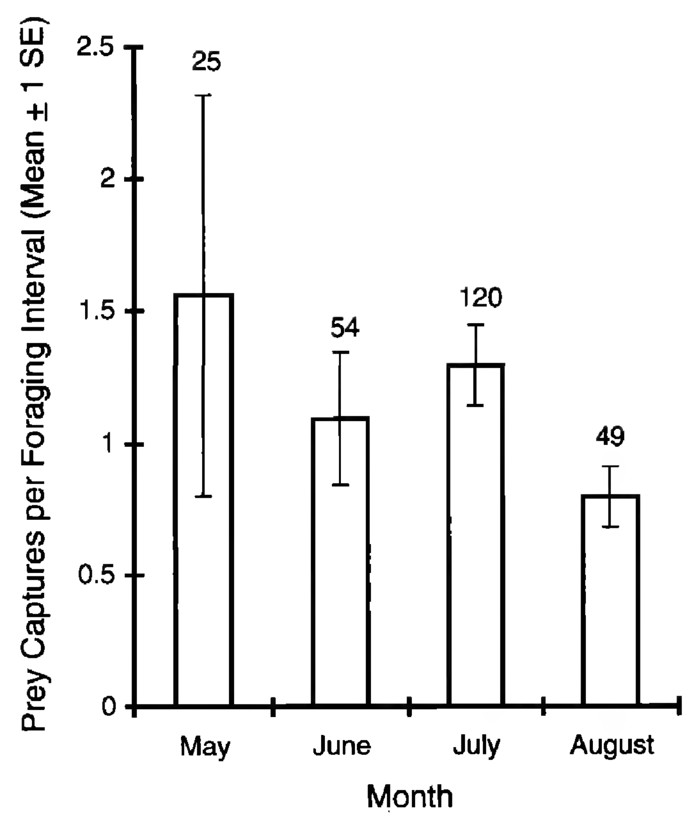

Figure 1. Monthly differences in the mean number of prey items captured by Mississippi Kites (*Ictinia mississippiensis*) during 2-min foraging intervals.

como las causas de este aumento, existen pocos estudios que puedan cuantificar esto. Las observaciones sobre el forrajeo de *Ictinia mississippiensis* en ambientes abiertos en Louisiana sugieren que esta especie es muy eficiente en su forrajeo en este habitat. *Ictinia mississipiensis* captura un promedio de 1.18 presas por cada 2 minutos de intervalo de forrajeo. Esta es una tasa exitosa mucho mayor que las anteriores en ambientes abiertos. Se hace necesaria la comparación de datos de forrajeo colectados en distintos habitats y localidades.

[Traducción de César Márquez]

ACKNOWLEDGMENTS

I am indebted to Dr. John C. Ogden and two anonymous reviewers for their comments and suggestions on an earlier draft of this manuscript and to the Louisiana Department of Wildlife and Fisheries for their assistance

LITERATURE CITED

Cely, J.E. 1987. American Swallow-tailed Kite uses Mississippi Kite nest. *J. Raptor Res.* 21:124.

Glinski, R.L. and R.D. Ohmart. 1983. Breeding ecology of the Mississippi Kite in Arizona. *Condor* 85:200–207.

Jackson, A.S. 1945. Mississippi Kite. *Texas Game and Fish* 3:6–7.

Levy, S.H. 1971. The Mississippi Kite in Arizona. *Condor* 73:476.

Parker, J.W. 1977. Second record of the Mississippi Kite in Guatemala. *Auk* 94:168–169.

——— and J.W. Ogden. 1977. The recent history and status of the Mississippi Kite. *Am. Birds* 33:119–129.

Skinner, R.W. 1962. Feeding habits of the Mississippi Kite. *Auk* 79:273–274.

Sutton, G.M. 1939. The Mississippi Kite in spring. *Condor* 41:41–53.

Wayne, A.T. 1906. A contribution to the ornithology of South Carolina, chiefly the coast region. *Auk* 23:56–68.

Received 12 October 1997; accepted 14 May 1998

J. Raptor Res. 32(3):247–250
© 1998 The Raptor Research Foundation, Inc.

EVALUATION OF NECK-MOUNTED RADIO TRANSMITTERS FOR USE WITH JUVENILE OSPREYS

LAUREN N. GILSON[1]
Raptor Research Center, Department of Biology, Boise State University, 1910 University Drive, Boise, ID 83725 U.S.A.

Key Words: *Osprey;* Pandion haliaetus; *transmitters; necklace; retention.*

Transmitters on necklaces, originally used on game birds, resulted from a modification of neck-mounted markers developed in 1970 in response to selective predation on individuals with back-mounted markers (Pyrah 1970, Amstrup 1980). For larger birds, neck-mounted transmitters are used infrequently; backpack-style harnesses are preferred for their tenacity and durability in long-term research (Day et al. 1980, Marion and Shamis 1977, Young and Kochert 1987). For short-term research using short-lived radio transmitters, mounting methods must be highly reliable for the length of the study but need not be permanent. In a study of fledgling behavior, I mounted radio transmitters around the necks of Os-

---

[1] Present Address: P.O. Box 179 Wakkerstroom, 2480 South Africa.

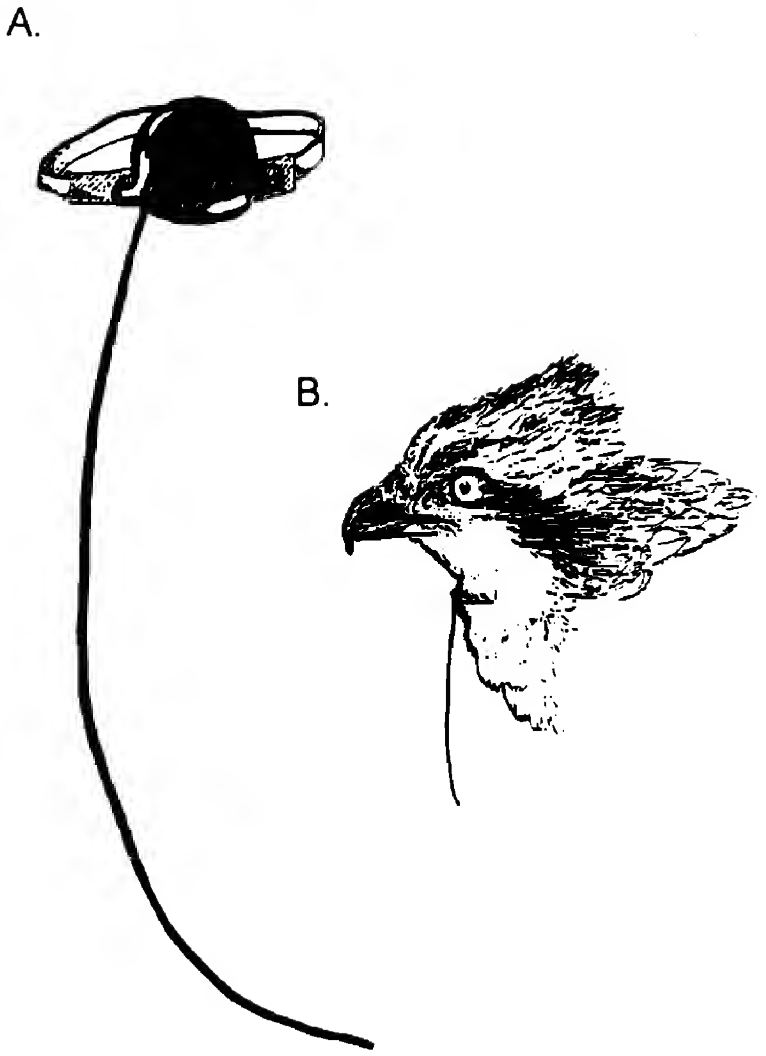

Figure 1. Rubber band style transmitter mount. A) Front view. B) As worn by fledglings.

preys *(Pandion haliaetus)* using standard rubber bands which have been used with Ospreys (C.P. Schaadt pers. comm.) as well as with raptors manned for falconry, and crimped nylon-wound elastic mounts designed and tested in this study. I describe and review the merits and drawbacks of each.

Methods

In June–August 1993, I attached eight modified necklace-style transmitters (ATS Model 2032: 5.6–5.9 g, 90 d battery life) to nestling Ospreys of age 35–45 d at Cascade Reservoir, Valley County, ID. I stitched the radio to a 3 × 4 cm patch of 100% nylon pack cloth rolled into a sleeve and sewn around a size 34 rubber band (0.4 cm width, 12.5–13 cm unstretched circumference; herein referred to as RB, Fig. 1). In July–August 1994, I attached 16 pendant-style transmitters (Merlin Systems: 7.4–7.7 g, 90 d battery life) to Osprey nestlings at Cascade Reservoir. I hung the unit around an Osprey's neck on an adjustable loop of nylon-wrapped elastic (Stretchrite Round Cord Elastic, Rhode Island Textile Company; herein referred to as NWE, Fig. 2). The necklace consisted of two elastic segments with looped ends rejoined by cotton thread, fed through a 1.5 cm segment of metal tubing (Archer Butt Connector, No. 64-3036) and crimped to size.

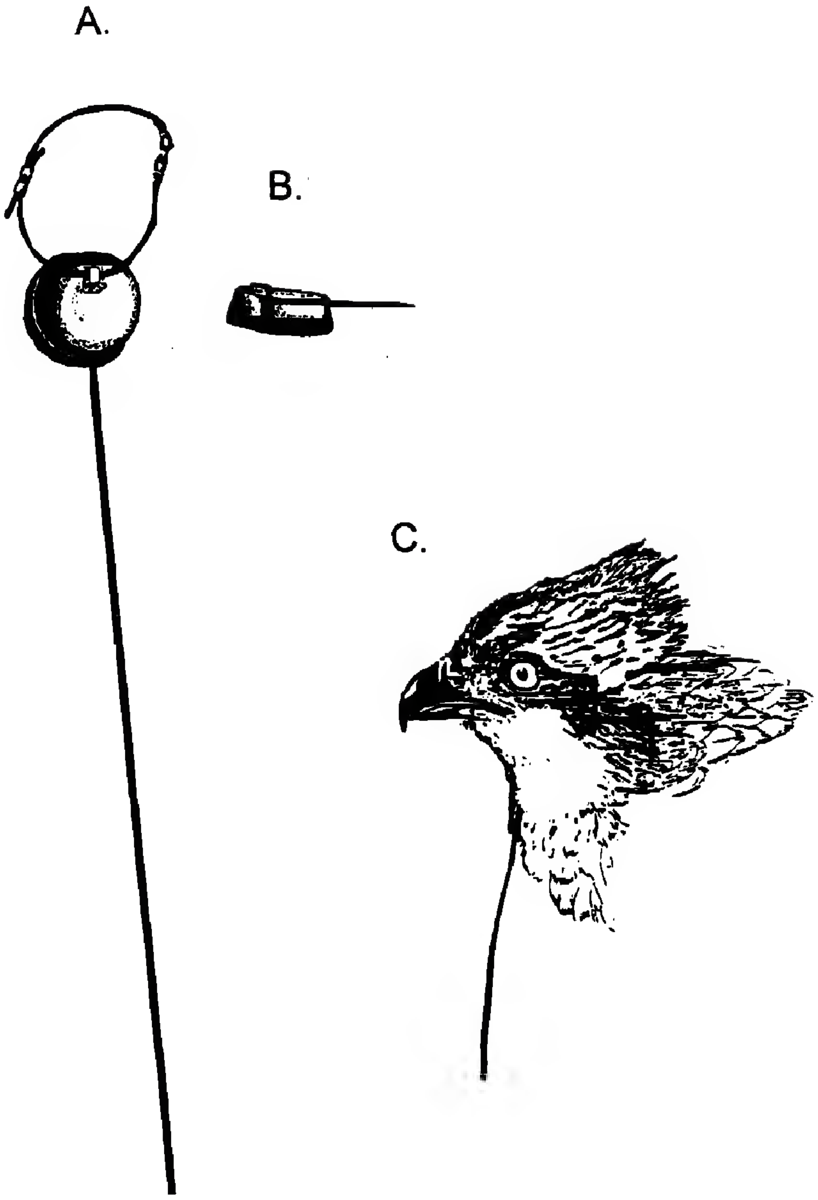

Figure 2. Nylon-wound elastic style transmitter mount. A) Rear view. B) Side view. C) As worn by fledglings (mounting bracket and antenna toward the Osprey's breast).

Results

Ospreys shed or removed seven of nine RB mounts in 1993. I recovered five of these. On two units, the rubber bands were broken and the three others were intact. All recovered transmitters remained firmly attached to the pack cloth.

RB mounts were lost at an average of 35 d (range 21–44 d) after application. In 115 hr of observations, I saw only one juvenile Osprey pull at its transmitter. Its sibling also preened around the necklace, both on the second day after application. Transmitter positions occasionally shifted, indicating that the unit moved freely about the Osprey's neck.

Ospreys shed or removed eight of 16 NWE mounts in 1994. The elastic pulled out of the crimping on one unit, two separated at the break-away loops, the elastic was not recovered with four units, and the eighth unit was not recovered. NWE mounts failed at an average of 26 d (range 18–37 d). In 504 hr of observation, I observed no Osprey young pulling at their own or their nest mates' transmitters. All shed or removed transmitters were lost 18–44 d after application. The two mounting styles differed significantly in mean length of retention (Wilcoxon Rank Sum Test, S = 73.5, Z = 1.97, $P = 0.04$; NWE $\bar{x}$ = 25.7 d, RB $\bar{x}$ = 34.8 d) but not in variance in retention ($F_{6,7} = 1.63$, $P = 0.53$; NWE s = 2.18, RB s = 2.98). The locations of transmitter losses were not random: 13 of 15 were recovered at or below nest or perch sites ($\chi^2 = 6.2$, $P = 0.016$).

Two Ospreys, one in each year, pulled the necklace material into their mouths. Whereas the 1993 Osprey fed normally despite transmitter position and shed the unit without incident at the reservoir shore two days later, the 1994 Osprey fed itself with difficulty. I trapped this fledgling at its natal nest and removed the transmitter. The elastic caused only minor abrasion of tissue at the corners of the mandibles. Necklaces did not obstruct feeding of any other juvenile Ospreys.

Transmitters that were retained through dispersal from the breeding area (one in 1993, four in 1994) were worn by Ospreys for an average of 34 d (range 28–43 d). Four shed units that were replaced and subsequently retained through dispersal provided an additional 6–18 d of information before the Ospreys dispersed.

Discussion

The main benefits of necklace style transmitter mounts are their low cost, easy construction, rapid application in the field, and minimal physical impact to their recipients. I prepared both styles of necklace mounts ahead of time, then simply slipped them over the heads and worked them under the feathers in the field. This greatly reduced the length of disturbance and amount of stress incurred by juveniles during marking and measurement. Per unit, both styles required under 30 min to prepare, cost less than $1.00 to construct, and took only minutes to attach. However, I found drawbacks to both styles I tested.

RB mounts were faster to attach than NWE mounts, but could not be adjusted to fit snugly. Rubber bands should have outlasted the battery life of the transmitter and dropped off after the unit was defunct. However, the elasticity of rubber bands apparently enabled fledglings to pull the transmitters off without breaking the bands. Although this behavior was never observed in the field, seven of nine (one remounted) units were either shed prematurely or successfully removed, three with the rubber band unbroken.

NWE cord had less stretch than rubber bands, permitting a tighter fit to each individual. However, nestlings scratched at their necks more often in 1994 than in 1993. I observed two Ospreys scratching at their necks 1 wk before both shed their transmitters: one separated at the loops and the other was missing its elastic. Preening resulted in two Ospreys being bridled by their necklaces, indicating that beak preening was not strong enough to break the mounts, but also that neither style fit sufficiently snugly. Were a talon to become hooked under the elastic, the downward force of an Osprey's leg was probably sufficient to pull the elastic out of the crimping or snap the threads at the break-away loops. I recovered no units on which the nylon cord was broken, in contrast to three of nine broken RB mounts. Transmitter removals in 1994 may thus have been attributable to scratching or preening in response to irritation, possibly caused by the crimped metal tubing. The recovery of most shed units below platforms or trees supports the notion that transmitter loss was associated with a behavior that is performed while perched.

All methods of transmitter mountings vary in physical impact to their bearers and in retention. Many studies that have employed neck-mounted transmitters or visual markers have encountered loss of markers, injury to the bearer, or death by starvation or predation (Hawkins and Simpson 1985, Small and Rusch 1985, Marks and Marks 1987, MacInnes and Dunn 1987, Pekins 1987, Sorenson 1989, Ely 1990, Samuels et al. 1990), thus leading most researchers to avoid neck mounts. However, Marcström et al. (1989) found significantly higher survival of pheasants with neck-mounts than with backpacks. Necklace mounts require a minimum of skill and time to attach in the field and cannot damage developing wing and tail feathers of young birds.

Tail mounts have been used effectively with many adult raptors including Ospreys (Kenward 1985a, Hagan and Walters 1990, Phelps 1993) but have resulted in damaged or loss of the retrices to which they were attached (Samuels and Fuller 1994). Backpacks are widely recommended and widely used for raptor studies, yet improperly fitted backpacks have entangled feet (Nicholls and Warner 1968) and can damage growing body feathers of young birds (Kenward 1985b). Backpacks also sometimes affect behavior which can result in selective predation on their bearers (Small and Rusch 1985, Marcström et al. 1989). Although backpacks are more costly and require more time and skill to attach properly, the benefits of greater retention and reduced impact on behavior may outweigh the costs. Kenward (1985b) recommended tail mounts as best for raptors "... unless the retrices of young birds are not yet fully grown," which precludes their use with pre-fledging juveniles. He also recommended anklet mounts over backpacks for juveniles, but found the transmission range of anklet-mounted radios was more readily reduced by low or ground perching, as is often observed in young Ospreys in this population.

In addition to cost-effectiveness and rapid deployment in the field, I used neck-mounted transmitters to reduce the risks to juveniles of damage to developing wing and tail feathers, interference with normal development of flight and hunting skills, selective predation on already vulnerable juveniles, or possible electrocution of young Ospreys with tail or backpack mounts. In general, while the neck-mounted transmitters in this study did not appear to cause damage to their bearers, antennae may have been annoying, as juvenile Ospreys were observed biting at antennae while feeding and shaking their heads to move the antennae out of the way. Whereas necklace-style transmitter mounts were easy to construct and apply in the field, their retention was generally low, and appeared to be obtrusive in several instances. Neither neck-mounting method proved sufficiently durable for use with juvenile Ospreys. Both styles were shed early in the post-fledging period, which lasts 30–35 d at this study area.

Four possible means of improving NWE mounts are: (1) use tubing with "teeth" that could effectively bite into the elastic to prevent it from pulling out, (2) securing the NWE inside of the tubing with a drop of cyanoacrylate glue at each end, (3) knotting the end of the NWE against the ends of the crimped tubing, or (4) stitching through the elastic rather than crimping it. I was unable to locate "toothed" tubing small enough to effectively crimp the 1/8" NWE cord. Although gluing and/or stitching through might damage the elastic itself, it may secure the mount better than crimping alone, extending the effectiveness while still remaining a temporary mounting method.

RESUMEN.—El uso de accesorios para sujetar radio-transmisores temporalmente son aconsejables para las necesidades de investigación en el corto plazo, sin embargo los transmisores deben ser retenidos durante la duración del estudio. Con el fin de valorar la eficiencia de transmisores de collar en las aves rapaces, evalúe la retención de dos diseños en individuos juveniles de *Pandion haliaetus*: una banda de caucho y un cordón elástico de nylon. Los transmisores montados con bandas de caucho fueron significativamente retenidos por mas tiempo que los de cordon elástico de nylon. Sinembargo ninguno de los dos fue retenido hasta el períódo requerido para ser recomendados para realizar investigaciones de campo.

[Traducción de César Márquez]

LITERATURE CITED

AMSTRUP, S.C. 1980. A radio collar for game birds. *J. Wildl. Manage.* 44:214–217.

DAY, G.I., S.D. SCHEMNITZ AND R.D. TABER. 1980. Capturing and marking wild animals. Pages 61–88 *in* S.D. Schemnitz [ED.], Wildlife management techniques manual, 4th ed. Wildl. Soc., Inc., Washington, DC U.S.A.

ELY, C.R. 1990. Effects of neck bands on the behavior of wintering Greater White-fronted geese. *J. Field Ornithol.* 61:249–253.

HAGAN, J.M., III AND J.R. WALTERS. 1992. Foraging behavior, reproductive success, and colonial nesting in Ospreys. *Auk* 107:506–521.

HAWKINS, L.L. AND S.G. SIMPSON. 1985. Neckband a handicap in an aggressive encounter between Tundra Swans. *J. Field Ornithol.* 56:182–184.

KENWARD, R.E. 1985a. Radio transmitters tail-mounted on hawks. *Ornis Scand.* 9:220–223.

———. 1985b. Raptor radio-tracking and telemetry. *ICBP Tech. Publ.* 5:409–420.

MACINNES, C.D. AND E.H. DUNN. 1987. Effects of neck bands of Canada Geese nesting at the McConnell River. *J. Field Ornithol.* 59:239–246.

MARCSTRÖM, V., R.E. KENWARD AND M. KARLBOM. 1989. Survival of Ring-necked Pheasants with backpacks, necklaces, and leg bands. *J. Wildl. Manage.* 53:808–810.

MARION, W.R. AND J.D. SHAMIS. 1977. An annotated bibliography of bird marking techniques. *Bird-Banding* 48:42–61.

MARKS, S.J. AND V.S. MARKS. 1987. Influence of radio collars on survival of Sharp-tailed Grouse. *J. Wildl. Manage.* 51:468–471.

NICHOLLS, T.H. AND D.W. WARNER. 1968. A harness for attaching radio transmitters to large owls. *Bird Banding* 39:209–214.

PEKINS, P.J. 1987. Effects of poncho-mounted radios on Blue Grouse. *J. Field Ornithol.* 59:46–50.

PHELPS, J.M., III. 1993. Factors affecting foraging and productivity of Ospreys (*Pandion haliaetus*) at Cascade Reservoir, Idaho. M.S. thesis, Boise State Univ., Boise, ID U.S.A.

PYRAH, D. 1970. Poncho markers for game birds. *J. Wildl. Manage.* 34:466–467.

SAMUELS, M.D. AND M.R. FULLER. 1994. Wildlife radiotelemetry. Pages 370–416 *in* T.H. Bookhout [ED.], Research and management techniques for wildlife and habitats. Wildl. Soc., Bethesda, MD U.S.A.

———, D.H. RUSCH AND S.R. CRAVEN. 1990. Influence of neck bands on recovery and survival rates of Canada Geese. *J. Wildl. Manage.* 54:45–53.

SMALL, R.J. AND D.H. RUSCH. 1985. Backpacks versus ponchos: survival and movements of Ruffed Grouse. *Wildl. Soc. Bull.* 13:163–165.

SORENSON, M.D. 1989. Effects of neck collar radios on female Redheads. *J. Field Ornithol.* 60:523–528.

YOUNG, L.S. AND M.N. KOCHERT. 1987. Marking Techniques. Pages 125–156 *in* B.A. Giron Pendleton, B.A Millsap, K.W. Cline and D.M. Bird [EDS.], Raptor management techniques manual. Natl. Wildl. Fed., Washington, DC U.S.A.

Received 10 December 1996; accepted 9 May 1998

J Raptor Res. 32(3):251–254
© 1998 The Raptor Research Foundation, Inc.

ORGANOCHLORINES AND MERCURY IN PEREGRINE FALCON EGGS FROM WESTERN NORTH CAROLINA

TOM AUGSPURGER
U.S. Fish and Wildlife Service, P.O. Box 33726, Raleigh, NC 27636-3726 U.S.A.

ALLEN BOYNTON[1]
*North Carolina Wildlife Resources Commission, Nongame and Endangered Wildlife Program,
Morganton, NC 28655 U.S.A.*

KEY WORDS: *Peregrine Falcon;* Falco peregrinus; *mercury; organochlorine pesticides; North Carolina.*

In North Carolina, Peregrine Falcons (*Falco peregrinus anatum*) were historically regarded as an uncommon breeding bird in the western mountains (Pearson et al. 1919, 1942) prior to their contaminant-induced extirpation in the eastern U.S. (Hickey 1969). Surveys in the early 1970s indicated peregrines no longer bred in North Carolina or at any historical nest sites east of the Mississippi River (Fyfe et al. 1976). Efforts to restore peregrines to the breeding bird fauna of North Carolina included protection of nesting habitat and the introduction of 81 young peregrines, produced in captivity by the Peregrine Fund, Inc. and private breeders (Barclay 1988), at seven sites between 1984–91. Introduced birds first bred in 1987 and successfully so in 1988. One to five pairs have bred each year producing a mean of 0.0–2.5 young per occupied territory and an average of 0.81 young per occupied territory from 1987–95 (Boynton and Currie 1993, C. McGrath pers. comm.).

Peregrine Falcons in western North Carolina occupy a portion of the Southern Appalachian recovery region; an objective of the recovery plan is the establishment of 20–25 nesting pairs in this region (U.S. Fish and Wildlife Service [hereafter USFWS] 1979, 1991), a level that has not been attained. Although contaminants were not suspected to be limiting productivity, no analyses had been performed on this new population prior to this assessment. Our objectives were to quantify organochlorine and mercury concentrations and shell thickness from western North Carolina peregrine eggs and to evaluate their significance relative to reproduction.

METHODS

Between 1990–93, five Peregrine Falcon eggs were collected from four clutches in three breeding territories after they were either incubated past term or abandoned. Eggs were stored frozen until harvested into acid-rinsed and solvent-rinsed glass jars for analyses.

The USFWS Patuxent Analytical Control Facility analyzed for 20 organochlorine compounds and mercury. Organochlorine analyses were by gas-liquid chromatography with peak confirmation of p,p′-DDE by gas chromatography/mass spectrometry under methods adapted from Cromartie et al. (1975). Mercury determination was performed by cold vapor atomic absorption spectrophotometry as described by the Joint Mercury Residues Panel (1961).

The lower limit of detection was 0.01 ppm wet weight for organochlorines and 0.04 ppm wet weight for mercury. A procedural blank indicated no background contamination of analytical equipment or reagents. Results of duplicate analyses and spike recoveries for mercury (104%), DDT metabolites (54.7–107%), lindane (69.6%), and chlordane metabolites (65.1–89.8%) were within acceptable ranges for method precision and accuracy. Residues reported here were not adjusted for recoveries.

We used a regression equation for American Kestrel (*Falco sparverius*) eggs to estimate egg volume (Wiemeyer et al. 1986). Because samples had dehydrated from exposure and refrigeration, we adjusted all residue concentrations for moisture loss using egg weight to volume ratios and assuming a specific gravity of 1.0 (Stickel et al. 1973). All contaminant concentrations are reported as parts per million (ppm) fresh wet weight. Contaminant concentrations for the two eggs collected in 1991 from Whiterock Cliff were averaged prior to calculating geometric means. Consequently, geometric means are based on clutches ($N = 4$) rather than individual eggs ($N = 5$).

Eggshell thicknesses of the five specimens and shell fragments from four additional nests were determined with a Federal Model 35 bench comparator thickness gauge at the Western Foundation for Vertebrate Zoology. At least 10 measurements of each sample were made for each mean reported.

RESULTS AND DISCUSSION

DDE, a metabolite of DDT, was considered causative in the extirpation of the Peregrine Falcon after it was found that DDE-induced eggshell thinning of around 20% was invariably associated with declining populations (Risebrough and Peakall 1988). Corresponding DDE residues

---

[1] Present address: Virginia Department of Game and Inland Fisheries, Route 1, Box 107, Marion, Virginia 24354 U.S.A.

Table 1. Concentrations of mercury (Hg) and organochlorines measured in eggs of North Carolina Peregrine Falcons. Concentrations are ppm fresh wet weight and corresponding shell thicknesses are in mm.

LOCATION AND YEAR	SHELL THICKNESS	Hg	p,p′ DDE	TOTAL DDT[a]	TOTAL CHLORDANE[b]	DIELDRIN	LINDANE
Chimney Rock Rutherford County 1993	0.332	0.14	1.47	1.70	0.50	0.12	0.02
Looking Glass Rock Transylvania County 1990	0.329	0.07	5.72	6.13	1.95	0.42	0.03
Looking Glass Rock Transylvania County 1992	0.386	0.11	3.96	4.26	1.04	0.18	0.01
Whiterock Cliff Madison County 1991	0.328	0.05	5.14	5.70	1.32	0.54	0.02
Whiterock Cliff Madison County 1991	0.321	0.10	2.64	3.03	1.71	0.66	0.01

[a] Total DDT defined here as summed o,p′-DDD, o,p′-DDE, o,p′-DDT, p,p′-DDD, p,p′-DDE, and p,p′-DDT.

[b] Total chlordane defined here as sum of alpha chlordane, cis-nonachlor, gamma chlordane, heptachlor epoxide, oxychlordane, and trans-nonachlor.

of 15–20 ppm have been associated with population level average eggshell thinning of this magnitude (Peakall and Kiff 1988). The geometric mean concentration of p,p′-DDE in North Carolina Peregrine Falcon clutches was 3.37 ppm (Table 1). Total metabolites of DDT ranged from 1.70–6.13 ppm with a geometric mean concentration of 3.73 ppm. These DDE concentrations are well below those associated with population declines (Peakall and Kiff 1988).

Arithmetic mean eggshell thicknesses were 0.339 mm for the five whole eggs and 0.334 mm when the shell fragments from four additional nests were included. Assuming a pre-1947 eggshell thickness for eastern Peregrine Falcons of 0.360 mm (Burns et al. 1994), the values reported here are approximately 7% thinner than normal, pre-DDT era eggshells. Peakall and Kiff's (1988) summary of thinning data indicate that extirpated or declining Peregrine Falcon populations occurred whenever mean population-level eggshell thinning exceeded 17% (except for intensively managed populations). Fyfe et al. (1988) suggested 14.5% thinning as a minimum estimate of the threshold below which there is no appreciable effect on peregrine productivity. Since the stock for the restored eastern Peregrine Falcon population was derived from several sources, eggshell thickness comparisons must necessarily be approximate.

The geometric mean concentration of total chlordane components and metabolites (1.11 ppm) was composed primarily by oxychlordane (0.30 ppm) and heptachlor epoxide (0.22 ppm). Peakall et al. (1990) considered ≥1.5 ppm heptachlor epoxide to be critical for producing adverse effects in peregrine eggs.

Dieldrin levels in raptors were reviewed by Peakall et al. (1990); they derived an egg screening value for determining adverse effects in Peregrine Falcons of 1–4 ppm. Peakall (1996) reported a "no-observed-effect" level of dieldrin at 0.7 ppm. North Carolina Peregrine Falcon eggs were well below this range with a geometric mean of 0.27 ppm.

Lindane has been used to control balsam wooly adelgid (*Adelges piceae*) infestation of fraser fir (*Abies fraseri*) in western North Carolina. The geometric mean concentration of lindane in our samples was 0.02 ppm. Lindane has not been associated with avian impairment in the wild, and levels well above those reported here have been detected in bird eggs without any apparent effects (Wiemeyer 1996).

Mercury has been shown to cause mortality and reproductive impairment in wild birds (Eisler 1987). The geometric mean mercury concentration of 0.10 ppm in North Carolina Peregrine Falcon eggs is below estimates of 0.5–1.0 ppm used by others as screening values for reproductive effects on peregrines (Peakall et al. 1990) and other raptors (Bowerman et al. 1995).

The geometric mean concentration of DDE in peregrine eggs from the Southern Appalachian recovery region was approximately half that reported for eggs from the mid-Atlantic and northeastern U.S. (Gilroy and Barclay 1988, Burns et al. 1994). The lower DDE concentrations in peregrine eggs from the Southern Appalachian recovery region may be due to different prey preferences. Preliminary data indicate that western North Carolina peregrines depend largely on resident birds, particularly Rock Doves (*Columba livia*), Mourning Doves (*Zenaida macroura*), Northern Flickers (*Colaptes auratus*), and Blue Jays (*Cyanocitta cristata*) (Boynton and Currie 1993). Additional documentation of prey preferences is ongoing.

Factors possibly limiting productivity include nest predation, inclement weather, inexperience, poor food supply, and human disturbance (Boynton and Currie 1993). While a larger data set is advisable before ruling out pollutant effects, our current data suggest that environmental contaminants are not limiting productivity of peregrines in the Southern Appalachian recovery region. The USFWS (1995) has indicated its intention to remove the Peregrine Falcon from the list of Endangered and Threatened Species. These data may serve as a baseline in future monitoring of the Southern Appalachian population which may be a requirement following delisting.

RESUMEN.—Concentraciones de pesticidas organoclorados y mercurio fueron encontrados en huevos colectados de cinco halcones peregrinos de una población reestablecida en el noroeste de Carolina. Las concentraciones de la media geométrica de p,p′ DDE (3.37 ppm en peso fresco), total de metabolitos de chlordane (1.11 ppm), dieldrin (0.27 ppm), lindano (0.02 ppm), y mercurio (0.10 ppm) estaban generalmente por debajo de los niveles asociados al fracaso reproductivo. Los 0.334 milímetros de la media aritmética del grueso de la cáscara de los huevos, fué del 7% mas delgada de lo normal encontrado en la "pr–era" del DDT de la población original del este de Estados Unidos.

[Traducción de César Márquez]

ACKNOWLEDGMENTS

We thank John Moore of Patuxent Analytical Control Facility for coordinating chemical analyses. The data were part of USFWS Regional Study ID#93-4PER. Stanley Wiemeyer, Chuck Henny, Tom Cade, Robert Currie, Chris McGrath, and Randy Wilson provided helpful reviews of earlier versions of the manuscript.

LITERATURE CITED

BARCLAY, J.H. 1988. Peregrine restoration in the eastern United States. Pages 549–558 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise, ID U.S.A.

BOWERMAN, W.W., J.P. GIESY, D.A. BEST AND V.J. KRAMER. 1995. A review of factors affecting productivity of Bald Eagles in the Great Lakes region: implications for recovery. *Environ. Health Perspect.* 103:51–59.

BOYNTON, A.C. AND R. CURRIE. 1993. Productivity of reintroduced Peregrine Falcons in western North Carolina. *Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies* 47:386–393.

BURNS, S.A., W.M. JARMAN, T.J. CADE, L.F. KIFF AND B J WALTON. 1994. Organochlorines and eggshell thinning in Peregrine Falcon *Falco peregrinus* eggs from the eastern United States, 1986–1988. Pages 709–716 *in* B.-U. Meyburg and R.D. Chancellor [EDS.], Raptor conservation today. Proceedings of the IV world conference on birds of prey and owls. Pica Press, London, U.K.

CROMARTIE, E., W.L. REICHEL, L.N. LOCKE, A.A. BELISLE, T.E. KAISER, T.G. LAMONT, B.M. MULHERN, R.M. PROUTY AND D.M. SWINEFORD. 1975. Residues of organochlorine pesticides and polychlorinated biphenyls and autopsy data for Bald Eagles, 1971–72. *Pest. Monit J.* 9:11–14.

EISLER, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. USFWS Biol. Rep. 85(1.10). U.S. Fish Wildl. Serv., Laurel, MD U.S.A

FYFE, R.W., S.A. TEMPLE AND T.J. CADE. 1976. The 1975 North American Peregrine Falcon survey. *Can. Field Nat.* 90:228–273.

———, R.W. RISEBROUGH, J.G. MONK, W.M. JARMAN, D.W. ANDERSON, L.F. KIFF, J.L. LINCER, I.C.T. NISBET, W. WALKER, II AND B.J. WALTON. 1988. DDE, productivity, and eggshell thickness relationships in the genus *Falco.* Pages 319–335 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise, ID U.S.A.

GILROY, M.J. AND J.H. BARCLAY. 1988. DDE residues and eggshell characteristics of reestablished peregrines in the eastern United States. Pages 403–411 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise ID U.S.A.

HICKEY, J.J. 1969. Peregrine Falcon populations: their biology and decline. Univ. Wisconsin Press, Madison, WI U.S.A.

JOINT MERCURY RESIDUE PANEL. 1961. Recommended methods of analysis of pesticide residues in food stuffs. *Analyst* 86:608–614.

PEAKALL, D.B. 1996. Dieldrin and other cyclodiene pesticides in wildlife. Pages 73–97 *in* W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood [EDS.], Environmental contaminants in wildlife: interpreting tissue concentrations. Lewis Publishers, Boca Raton, FL U.S.A.

——— AND L.F. KIFF. 1988. DDE contamination in peregrines and American Kestrels and its effect on reproduction. Pages 337–350 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise, ID U.S.A.

———, D.G. NOBLE, J.E. ELLIOT, J.D. SOMERS AND G. ERICKSON. 1990. Environmental contaminants in Canadian Peregrine Falcons, *Falco peregrinus*: a toxicological assessment. *Can. Field Nat.* 104:244–254.

PEARSON, T.G., C.S. BRIMLEY AND H.H. BRIMLEY. 1919. Birds of North Carolina. North Carolina geologic and economic survey. Vol. IV. Edwards and Broughton Printing Co., Raleigh, NC U.S.A.

———. 1942. Birds of North Carolina. North Carolina Department of Agriculture. Bynum Printing Co., Raleigh, NC U.S.A.

RISEBROUGH, R.W. AND D.B. PEAKALL. 1988. The relative importance of several organochlorines in the decline of Peregrine Falcon populations. Pages 449–462 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise, ID U.S.A.

STICKEL, L.F., S.N. WIEMEYER AND L.J. BLUS. 1973. Pesticide residues in eggs of wild birds: adjustment for loss of moisture and lipid. *Bull. Environ. Contam. Toxicol.* 9: 193–196.

U.S. FISH AND WILDLIFE SERVICE. 1979. Eastern Peregrine Falcon recovery plan. Washington, DC U.S.A.

———. 1991. First update of Peregrine Falcon (*Falco peregrinus*), eastern population, revised recovery plan. Newton Corner, MA U.S.A.

———. 1995. Endangered and threatened wildlife and plants; advance notice of a proposal to remove the American Peregrine Falcon from the list of endangered and threatened wildlife. *Federal Register* 60: 34406–34409.

WIEMEYER, S.N. 1996. Other organochlorine pesticides in birds. Pages 99–115 *in* W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood [EDS.], Environmental contaminants in wildlife: interpreting tissue concentrations. Lewis Publishers, Boca Raton, FL U.S.A.

———, R.D. PORTER, G.L. HENSLER AND J.R. MAESTRELLI. 1986. DDE, DDT + dieldrin: residues in American Kestrels and relations to reproduction. USFWS, Fish Wildl. Tech. Rep. 6. Washington, DC U.S.A.

Received 23 January 1997; accepted 11 May 1998

J. Raptor Res. 32(3):254–256
© 1998 The Raptor Research Foundation, Inc.

IMPORTANCE OF BIRDS AND POTENTIAL BIAS IN FOOD HABIT STUDIES OF MONTAGU'S HARRIERS (*CIRCUS PYGARGUS*) IN SOUTHEASTERN SPAIN

JOSÉ A. SÁNCHEZ-ZAPATA AND JOSÉ F. CALVO
Departamento de Ecología e Hidrología, Facultad Biología, Universidad de Murcia,
Campus de Espinardo, 30100 Espinardo, Murcia, Spain

KEY WORDS: *Montagu's Harrier,* Circus pygargus; *diet; mediterranean.*

Methods used to study the diets of raptors include the analysis of pellets, stomach contents, prey remains in nests or under perches, and direct observation of prey delivered to nests. In many species, including harriers (*Circus* spp.), analysis of prey remains only appears to underestimate the proportion of smaller prey and overestimate the occcurence of large prey (Schipper 1973, Simmons et al. 1991, Mañosa 1994, Real 1996). Several researchers have studied the diet of Montagu's Harrier (*Circus pygargus*) during the breeding season (Pérez-Chiscano and Fernández-Cruz 1971, Pérez-Chiscano 1974, Corbacho et al. 1995, Thiollay 1968, Helmich 1986), on migration (Castroviejo 1969), and while wintering (Cramp and Simmons 1980, Cormier and Baillon 1991). Extensive studies of their breeding diet in Spain (Hiraldo et al. 1975, Arroyo 1997), Holland and France (Schipper 1973), and Britain (Underhill-Day 1993) indicate that small birds and mammals are important prey in northern and central Europe, whereas in southern Europe invertebrates appear to be numerically important, together with small birds (Underhill-Day 1993, Hiraldo et al. 1975). The goal of our study was to assess the importance of birds in the diet of Montagu's Harriers and to test how different study methods affect the results of such a food habits study.

STUDY AREA AND METHODS

The diets of three pairs of Montagu's Harriers breeding in a wadi or "rambla" in a mediterranean semiarid region in southeastern Spain (Suárez et al. 1996) were studied during 1995. Ajauque is a small wetland located in the most arid sector (average annual rainfall = 30 cm) of Murcia in southeastern Spain. Ajauque rambla drains an impermeable watershed of sedimentary marls. In arid and semiarid lands of the region, ramblas are more productive than surrounding lands owing to their vegetation that consists mainly of reeds (*Phragmites australis*) and halophytic plants. The Ajauque rambla is part of a protected

Table 1. Montagu's Harrier prey during courtship (April), incubation (May), nestling (June), and fledging (July) periods, and all data pooled (All). Proportion of each prey and prey group by number (%) and by weight (%W).

PREY	APRIL	%	MAY	%	JUNE	%	JULY	%	ALL	%	%W
BIRDS	22	59	31	68	15	44	13	59	81	58	86
Carduelis chloris	1	3	0	0	1	3	0	0	2	1	3
Carduelis carduelis	0	0	1	2	1	3	0	0	2	1	2
Carduelis spp.	2	5	0	0	2	7	1	5	5	4	5
Galerida theklae	2	5	0	0	0	0	0	0	2	1	3
Galerida sp.	1	3	1	2	1	3	2	9	5	4	8
Calandrella rufescens	1	3	0	0	0	0	0	0	1	1	1
Alaudidae ind.	1	3	9	20	1	3	0	0	11	8	14
Cisticola juncidis	0	0	0	0	1	3	0	0	1	1	1
Saxicola torquata	0	0	0	0	3	9	3	13	6	4	4
Unidentified passerine	14	38	20	44	5	15	7	32	46	33	45
REPTILES	4	11	7	15	4	12	1	5	16	11	7
Psammodromus algirus	2	5	2	4	0	0	0	0	4	3	2
Unidentified lizzard	2	5	5	11	4	12	1	5	12	8	6
MAMMALS	0	0	2	4	0	0	0	0	2	2	5
Mus spp.	0	0	1	2	0	0	0	0	1	1	1
Oryctolagus cuniculus	0	0	1	2	0	0	0	0	1	1	4
INVERTEBRATES	11	30	6	13	15	44	8	36	40	29	2
Anacridium aegyptium	7	19	0	0	5	15	0	0	12	9	2
Unidentified Orthoptera	0	0	2	4	1	3	2	9	5	3	0
Orictes nasicornus	1	3	0	0	0	0	0	0	1	1	0
Unidentified Coleoptera	0	0	4	9	7	21	5	22	16	12	0
Unidentified invert.	3	8	0	0	2	6	1	5	5	4	0
TOTAL	37		46		34		22		139		

area that includes 69.4 ha of wetland where 3–5 pairs of Montagu's Harriers regularly breed on dense salty-shrub (*Sarcocornia fruticosa* and *Scirpus spp.*) and hunt in surrounding shrubsteppes and cropland.

We divided the breeding season into four periods of approximately 30 d each: courtship period (April), incubation period (May), nestling period (June), and fledging period (July). We used three different methods to assess food habits: (1) we collected pellets found under perches used by male harriers, (2) we collected prey remains found under perches used by both males and females, and (3) we identified prey during aerial transfers by males to females and young. Perches were visited weekly during the breeding period and all prey remains were identified and removed. Prey were identified using reference collections. Weights of prey were assigned following Hiraldo et al. (1975) and Donázar (1988). We used percentage uniformity tests (Sokal and Rohlf 1969) to compare the different diets determined using the different study methods.

RESULTS AND DISCUSSION

When we pooled all the data, small birds were the most numerous prey (58%) followed by invertebrates (29%), lizards (12%), and mammals (1%) (Table 1). Overall, small birds including small land, shrub and edge-nesting passerines (Thela lark, *Galerida theklae*; Lesser Short-toed Lark, *Calandrella rufescens*; Common Stonechat, *Saxicola torquata*; Citting Cisticola, *Cisticola juncidis*; and finches, *Carduelis* spp.) made up to 84% of the prey by weight. During the courtship and incubation periods, nestling passerines and eggs accounted for 60–86% of the birds eaten. Orthopterans and coleopterans were the second prey group by number, followed by a small lizard *Psammodromus algirus* which is the most common lizard species in salty shrub-steppes in southeastern Spain (Hernández et al. 1993). Small mammals occurred infrequently in the diet probably because of their low densities and nocturnal nature, especially in arid and semiarid Mediterranean regions (Herrera and Hiraldo 1976, Sánchez 1994).

The proportion of birds found in pellet samples (48%, $N = 83$) was smaller than in samples of prey remains (78%, $N = 37$) and aerial transfers (72%, $N = 19$), though there were only significant differences between pellets and prey remains ($t = 3.23$, df $= 2$, $P = 0.0012$). Several authors have suggested that pellets provide a less biased source of information on raptor diets because many prey species found in pellets are seldom found in prey remains at nests or under perches (Schipper 1973, Simmons et al. 1991, Real 1996). Because pellet samples underestimated the number of small birds in our study, it seems that the opposite appears to be true in the diet of Montagu's Harriers. Most of bird prey recorded were nestlings or juveniles which are easy to digest and difficult to find in pellets (Underhill-Day 1993), whereas invertebrate and lizard remains such as scales in pellets were easily detected. Birds have also been shown to be the main prey of Montagu's Harriers both in terms of numbers and weight in most of Europe during the breeding season (Schipper 1973, Hiraldo et al. 1975, Underhill-Day 1993, Corbacho et al. 1995, Arroyo 1997). Small mammals such as voles (*Mictrotus* spp.) and young hares (*Lepus* spp.) have been shown to be only locally or temporally important (Thiollay 1968, Arroyo 1997) even though most studies have only considered pellets.

RESUMEN.—Se analiza la dieta del Aguilucho cenizo en un humedal del Sureste de España durante el periodo reproductor utilizando tres métodos diferentes; egagrópilas, restos de presas en posaderos y transferencia de presas. Los pájaros se revelan como el componente principal de la dieta tanto en número como en biomasa, seguidos por los invertebrados y reptiles. Se observaron sesgos en la proporción de aves que aparecen en la dieta según el método de estudio.

[Traducción Autores]

ACKNOWLEDGMENTS

This study had financial support from the Dirección General del Medio Natural of Murcia Region and LIFE projects (UE). Sergio Eguía, Andrés Giménez, Javier Royo, José María Caballero, José Joaquín Hernández, Asunción Andreu and Trino Ferrández helped during the field work. Andrés Millán and Eulalia Clemente helped to identify invertebrate remains. José Antonio Donázar revised an early draft and Beatriz Arroyo, Keith L Bildstein and an anonymous referee made valuable comments on the manuscript.

LITERATURE CITED

ARROYO, B.E. 1997. Diet of Montagu's Harrier *Circus pygargus* in central Spain: analysis of temporal and geographic variation. *Ibis* 139:664–672.

CASTROVIEJO, J. 1969. Sobre paso y alimentación de *Circus pygargus* en el NW de España. *Ardeola* 14:216–217.

CORMIER, J.P. AND F. BAILLON. 1991. Concentracion de busards cendrés *Circus pygargus* (L.) dans la région de 'bour (Sénégal) durante l'hiver 1988–1989: utilisation du milieu et régime alimentaire. *Alauda* 59:163–168.

CORBACHO, C., A. MUÑOZ AND P. BARTOLOMÉ. 1995. Espectro trófico del aguilucho cenizo (*Circus pygargus* L.) en Extremadura. *Alytes*:441–448.

CRAMP, S. AND K.E. SIMMONS. 1980. The birds of western Paleartic, Vol. II. Oxford Univ. Press, Oxford, U.K.

DONÁZAR, J.A. 1988. Variaciones en la alimentación entre adultos reproductores y pollos en el Búho Real (*Bubo bubo*). *Ardeola* 35:278–284.

HELMICH, J. 1986. Notas sobre el ritmo de actividad y la alimentación del aguilucho cenizo en Agosto y Septiembre en Extremadura. *Alytes* 5:69–79.

HERNÁNDEZ, V.F. DICENTA, F. ROBLEDANO, M. GARCÍA, M.A. ESTEVE AND L. RAMÍREZ. 1993. Anfibios y reptiles de la Región de Murcia. Cuadernos de Ecología y Medioambiente, No. 1. Univ. Murcia, Murcia, Spain.

HERRERA, C.M. AND F. HIRALDO. 1976. Food niche and trophic relationships among European owls. *Ornis Scand.* 7:29–41.

HIRALDO, F., F. FERNANDEZ AND F. AMORES. 1975. Diet of the Montagu's Harrier (*Circus pygargus*) in southwestern Spain. *Doñana Acta Vertebrata* 2:25–55.

MAÑOSA, S. 1994. Goshawk diet in a Mediterranean area of northeastern Spain. *J. Raptor. Res.* 28:84–92.

PÉREZ-CHISCANO, J.L. 1974. Sumario informe sobre alimentación de rapaces en el NE de Badajoz. *Ardeola* 19:331–336.

——— AND M. FERNÁNDEZ-CRUZ. 1971. Sobre *Grus grus* y *Circus pygargus* en Extremadura. *Ardeola,* Vol. Especial· 509–574.

REAL, J. 1996. Biases in diet study methods in the Bonelli's Eagle. *J. Wildl. Manage.* 60:632–638.

SÁNCHEZ, J.A. 1994. Ecología de las aves de presa de la región de Murcia. M.S. thesis, Univ. Murcia, Murica, Spain.

SCHIPPER, W.J.A. 1973. A comparison of prey selection in sympatric harriers (*Circus*) in western Europe. *Le Gerfaut* 63:17–120.

SIMMONS, R.E., D.M. AVERY AND G. AVERY. 1991. Biases in diets determined from pellets and remains: correction factors for a mammal and bird-eating raptor. *J Raptor Res.* 25:63–67.

SOKAL, R.R. AND F.J. ROHLF. 1969. Biometry. Freeman and Co., San Francisco, CA U.S.A.

SUÁREZ, M.L., M.R. VIDAL, J.F. CALVO, M.A. ESTEVE, R. GÓMEZ, A. GIMÉNEZ, J.A. PUJOL, J.A. SÁNCHEZ, M. PARDO, J. CONTRERAS AND L. RAMÍREZ. 1996. Zone humide d'Ajauque-Rambla Salada. Pages 40–52 *in* C. Morillo and J.L. Gonzalez [EDS.], Management of Mediterranean wetlands, III. Ministerio de Medio Ambiente, Gran Vía San Francisco, Madrid.

THIOLLAY, J.-M. 1968. La pression de predation estivale du busard cendré *Circus pygargus L.* sur les populations de *Microtus arvalis* en Vendeé. *Terre Vie* 22:321–342.

UNDERHILL-DAY, J.C. 1993. The foods and feeding rates of Montagu's Harriers *Circus pygargus* breeding in arable farmland. *Bird Study* 40:74–80.

Received 6 August 1997; accepted 19 May 1998

J. Raptor Res. 32(3):257–260
© 1998 The Raptor Research Foundation, Inc.

PREY BROUGHT TO RED-SHOULDERED HAWK NESTS IN THE GEORGIA PIEDMONT

DOUG L. HOWELL[1] AND BRIAN R. CHAPMAN
Daniel B. Warnell School of Forest Resources, University of Georgia, Athens, Georgia 30602-2152 U.S.A.

KEY WORDS: *Red-shouldered Hawk;* Buteo lineatus; *prey; nests; food habits; Georgia; Piedmont.*

The Red-shouldered Hawk (*Buteo lineatus*) is a common breeding species throughout the southeastern U.S. Despite its wide distribution, information on its food habits in the Southeast is largely anecdotal (Burleigh 1958, Janik and Mosher 1982). Although food habit studies in several geographic regions have documented the breadth the Red-shouldered Hawk's diet (Craighead and Craighead 1956, Snyder and Wiley 1976, Bednarz and Dinsmore 1985, Parker 1986), they differ with respect to the importance of certain prey classes in the diet. While this could represent variation within and between regions, it may be an artifact of the methods used to quantify prey (Marti 1987), or due to the failure to report results in terms of biomass (Steenhof 1983). Our objectives were to quantify the prey brought to nests of the Red-shouldered Hawk in Georgia and to compare food habits in Georgia with those reported elsewhere.

STUDY AREA AND METHODS

The study was conducted on the 5718-ha Bishop F. Grant Memorial Forest (BGF), located in Putnam and Morgan counties, approximately 14 km north of Eatonton (83°28′N, 33°25′W), in east-central Georgia. The area lies within the Piedmont physiographic province, a peneplain dissected by numerous streams to form a rolling topography (Brender 1973). Elevation ranges from 120–220 m above sea level. Average annual rainfall is approximately 120 cm, with peak precipitation occurring in winter (USDA-SCS 1965, 1976).

Over 60% of BGF consists of natural or planted stands of loblolly pine (*Pinus taeda*). Bottomland hardwood forests (7%) exist along the area's major drainages. These include Big Indian Creek, Glady Creek, and Little River. Dominant vegetation includes green ash (*Fraxinus pennsylvanicus*), sweetgum (*Liquidambar styraciflua*), box elder (*Acer negundo*), sycamore (*Platanus occidentalis*), overcup oak (*Quercus lyrata*), water oak (*Q. nigra*), and willow oak (*Q. phellos*). Upland hardwood stands (23%) consisting of mixed oaks (*Quercus* spp.) and hickories (*Carya* spp.), blackgum (*Nyssa sylvatica*), sweetgum, and winged elm (*Ulmus alata*) lie adjacent to bottomland corridors, or are associated with major drainage basins. The remainder of BGF is maintained as pasture for cattle grazing and hay production, or is planted as wildlife food plots. Several small reservoirs provide irrigation, public fishing, and waterfowl habitat.

We monitored prey deliveries to eight occupied Red-shouldered Hawk nests within ($N = 6$) and around ($N = 2$) BGF from 3 April–14 July 1994. Old nests were located prior to leaf-out, and then rechecked for signs of occupancy. Observations totaling 103 hr were made with a 20-45× spotting scope and 8× binoculars from a ground blind placed within 20 m the base of the nest tree. Observation periods were normally 4–6 hr and were allocated randomly to cover all daylight hours (0600–1800 H) Most nests were observed over one time interval at least once each week from early in incubation until the young had fledged. Nest sites were checked periodically for remains of prey beneath the nest. We compared the observational data with those of prey remains to insure that we counted only those prey remains that could not have been seen during observations from blinds. Regurgitated pellets normally contained only hair or feathers, and were excluded from the analysis. Prey items were identified to species or the lowest possible taxonomic category.

We calculated the percent frequency of each prey item from the total number of items delivered to nests and collected from prey remains. The percent biomass contribution of each prey item was calculated by multiplying the frequency of occurrence of each prey item by its mean body mass. When possible, we derived biomass directly from prey collected on the study area. Otherwise, we estimated prey biomass (Marti 1987) from the literature (Golley 1962, Steenhof 1983, Dunning 1984). Large insects were assumed to weigh 1 g, the average mass obtained from representative samples collected on the study area. The masses of unidentified prey were assumed to be similar to the mean mass of the most closely related, identified taxa.

RESULTS AND DISCUSSION

All Red-shouldered Hawk nests were located in bottomland forests, or in upland hardwood stands adjacent to the bottomland corridor (Moorman and Chapman 1996). Mean nest height was 17.6 m (range = 12.2–21.3 m, $N = 8$). Six of eight nests fledged at least one young (range = 1–2) for an average of 1.8 young per successful nest.

A total of 181 prey items (Table 1) was identified by observations made from blinds ($N = 144$) and remains collected beneath nests ($N = 37$). Prey delivered to nests averaged 36.1 g (range = 1–487 g). Vertebrates represented 76.2% of the prey by numbers and 97.2% of prey biomass. Vertebrate prey included nine species of mammals, nine species of birds, eight species of reptiles, and four species of amphibians. Invertebrates represented 23.8% of the prey by numbers, but were insignificant in

---

[1] Present address: Directorate of Public Works and the Environment, AFZA-PW-DW, Wildlife Branch, Fort Bragg, NC 28307-5000 U.S.A.

Table 1. Food habits of Red-shouldered Hawks (*Buteo lineatus*) in the Bishop F. Grant Memorial Forest, Putnam and Morgan counties, Georgia in 1994. Prey items from eight nests identified from visual observations in blinds (N = 144) and prey remains beneath nests (N = 37). Prey listed taxonomically by class. N = number of individuals, %N = percent occurrence of prey, Mass = mean prey biomass in grams, and %B = percent of total biomass.

Prey Species	*N*	%*N*	Mass	%B
Oligochaeta				
Unidentified earthworms	8	4.4	6	0.7
Crustacea				
Unidentified crayfish (*Cambarus* spp.)	17	9.4	7	1.8
Insecta				
Unidentified beetles	5	2.8	1	0.1
Unidentified grasshoppers and crickets	8	4.4	1	0.1
Unidentified caterpillars	5	2.8	1	0.1
Amphibia				
Spotted salamander (*Ambystoma maculatum*)	1	0.6	12	0.2
Two-lined salamander (*Eurycea bislineata*)	5	2.9	7	0.5
Unidentified salamanders	10	5.5	9	1.4
Southern toad (*Bufo terrestris*)	3	1.7	20	0.9
Unidentified toads (*Bufo* spp.)	4	2.2	20	1.2
Southern leopard frog (*Rana utricularia*)	10	5.5	38	5.8
Unidentified frogs (*Rana* spp.)	13	7.2	38	7.7
Reptilia				
Snapping turtle (*Chelydra serpentina*)	11	6.1	24[a]	4.0
Green anole (*Anolis carolinensis*)	3	1.7	15	0.7
Eastern fence lizard (*Sceloporus undulatus*)	3	1.7	17	0.8
Unidentified skinks (*Eumeces* spp.)	4	2.2	18	1.1
Black racer (*Coluber constrictor*)	3	1.7	77	3.5
Black rat snake (*Elaphe obsoleta*)	2	1.1	190	5.8
Eastern kingsnake (*Lampropeltis getulus*)	1	0.5	190	2.9
Rough green snake (*Opheodrys aestivus*)	1	0.5	15	0.2
Unidentified water snakes (*Nerodia* spp.)	5	2.9	125[a]	9.7
Eastern garter snake (*Thamnophis sirtalis*)	10	5.5	64	9.8
Aves				
Mourning Dove (*Zenaida macroura*)	1	0.5	119	1.8
Carolina Wren (*Thryothorus ludovicianus*)	1	0.5	21	0.3
American Robin (*Turdus migratorius*)	1	0.5	78	1.2
Pine Warbler (*Dendroica pinus*)	1	0.5	12	0.2
Unidentified warbler (*Dendroica* sp.)	1	0.5	12	0.2
Common Yellowthroat (*Geothlypis trichas*)	1	0.5	10	0.1
Kentucky Warbler (*Oporonis formosus*)	1	0.5	14	0.2
Hooded Warbler (*Wilsonia citrina*)	2	1.1	11	0.3
Northern Cardinal (*Cardinalis cardinalis*)	3	1.7	45	2.1
Indigo Bunting (*Passerina cyanea*)	1	0.5	15	0.2
Unidentified passerines	2	1.1	17	0.5
Mammalia				
Short-tailed shrew (*Blarina brevicauda*)	2	1.1	13	0.4
Eastern mole (*Scalopus aquaticus*)	1	0.5	40	0.6
Eastern cottontail (*Sylvilagus floridanus*)	1	0.5	230[a]	3.5
Eastern chipmunk (*Tamias striatus*)	4	2.2	110	6.7
Eastern gray squirrel (*Sciurus carolinensis*)	1	0.5	487	7.5
White-footed mouse (*Peromyscus leucopus*)	3	1.7	18	0.8
Unidentified mice (*Peromyscus* spp.)	2	1.1	18	0.6
Golden mouse (*Ochrotomys nuttalli*)	2	1.1	18	0.6
Hispid cotton rat (*Sigmodon hispidus*)	6	3.3	85	7.8
Pine vole (*Microtus pinetorum*)	4	2.2	26	1.6
Unidentified voles (*Microtus* spp.)	3	1.7	26	1.2
Unidentified rodents	5	2.9	33	2.6
TOTAL	181	100.0		100.0

[a] Specimens represent subadult animals.

terms of prey biomass (2.8%; Table 1). All of the crustaceans identified were crayfish (Cambaridae), and the oligochaetes were earthworms (Lumbricidae).

Amphibians (25.6%) were the most frequently delivered prey items to Red-shouldered Hawk nests. Frogs (*Rana* spp.) were numerically important as prey and, collectively, they represented 41.4% of amphibian prey delivered to nests. Reptiles (38.5%) and mammals (33.9%) contributed most to total prey biomass (Table 1), followed by amphibians (17.7%), birds (7.1%), crustaceans (1.8%), oligochaetes (0.7%), and insects (0.3%). The species contributing most to total prey biomass were eastern garter snakes (*Thamnophis sirtalis*) and water snakes (*Nerodia* spp.).

Red-shouldered Hawks in the Georgia Piedmont preyed upon a variety of food items. Although their food habits were similar to those reported in previous studies in which the majority of prey taken were amphibians, reptiles, mammals, and crayfish (Craighead and Craighead 1956, Snyder and Wiley 1976, Bednarz and Dinsmore 1985, Parker 1986), the importance of certain prey classes such as amphibians and reptiles differed. Craighead and Craighead (1956) and Bednarz (1979), for instance, reported food habits based largely on percentages of prey occurring in pellets and found that small mammals were the preferred food of nesting Red-shouldered Hawks in Michigan and Iowa, respectively. Snyder and Wiley (1976) reported that invertebrates were the dominant foods found in Red-shouldered Hawk stomachs. Parker (1986) used visual observations to identify food items delivered to nests in Missouri, and found that amphibians were the most frequently delivered prey. Overall, frogs (*Rana* spp.) were the most frequently delivered prey to Red-shouldered Hawk nests in our study, and contributed most to total prey biomass. Craighead and Craighead (1956), Bednarz (1979), and Parker (1986) all found frogs to be important foods of Red-shouldered Hawks. However, studies relying on pellet analysis alone may underestimate the proportions of amphibians in the diet because amphibians are often completely digested and leave little osseous remains in pellets (Errington 1932). Snyder and Wiley (1976) and Portnoy (1974) both reported a higher incidence of frogs in the diet than suggested by pellet analysis.

Eastern garter snakes, unidentified water snakes and hispid cotton rats (*Sigmodon hispidus*) also were important prey, in terms of both numbers and biomass. Garter snakes were reported in Red-shouldered Hawk diets in Michigan (Craighead and Craighead 1956) and Iowa (Bednarz 1979). Cotton rats have never been reported as important prey, but distribution of cotton rats does not extend into the northern portion of the Red-shouldered Hawk's range, where voles (*Microtus* spp.) and mice (*Peromyscus* spp.) are taken more frequently (Craighead and Craighead 1956, Bednarz 1979). Snapping turtles (*Chelydra serpentina*) were numerically important prey items, but contributed little to overall prey biomass because individuals taken by the hawks were small.

Red-shouldered Hawks in our study rarely brought birds to nests and birds contributed little to total prey biomass (7.1%). Only Craighead and Craighead (1956) found that birds were important prey items based on the frequency of occurrence of avian species within pellets

Although invertebrates, particularly crayfish, were frequently delivered to Red-shouldered Hawk nests in our study, they contributed little to total prey biomass (2.8%). Snyder and Wiley (1976) found that 55.6% of a Red-shouldered Hawk's diet included invertebrates. The invertebrate component of their study probably was overestimated because they did not examine their data in terms of biomass. In addition, their study was based largely on analysis of stomach contents, which could contain items of secondary origin, particularly insects, ingested incidentally as stomach contents of prey.

Of the prey delivered to Red-shouldered Hawk nests in this study, 60% (108 of 181) were those frequently associated with bottomland forests, marshes, or wet meadows. Red-shouldered Hawks we equipped with radio transmitters (Howell and Chapman 1997) were located most often foraging within bottomland forests close to water, small beaver (*Castor canadensis*) ponds, wet meadows, or areas containing many seasonally or permanently flooded pools. Other researchers also have demonstrated the importance of these habitats as foraging sites for Red-shouldered Hawks (Henny et al. 1973, Portnoy 1974, Bednarz 1979, Parker 1986, Bloom et al. 1993). Red-shouldered Hawks in our study foraged in the bottomland forest habitat and used the variety of foods within it, rather than specializing on particular prey species, a result consistent with Bednarz and Dinsmore (1985). The most important foods of Red-shouldered Hawks during the nesting season were reptiles and amphibians, particularly snakes and frogs, associated with the bottomland forest. Small mammals may become more important during the winter months, given the seasonality of the preferred prey (Craighead and Craighead 1956) and also may increase in importance as buffer foods during extremely dry conditions (Bednarz and Dinsmore 1985)

Resumen.—Las presas de *Buteo lineatus* fueron estudiadas durante la estación reproductiva en un área de pinos de manejo intensivo en la región fisiogeográfica del piedemonte de Georgia. Un total de 1881 items fueron entregados a los pichones ($N = 144$) y, colectados como restos de presas debajo de los nidos ($N = 37$). Los vertebrados representaron el 76.2% de las presas en números y el 97.2% de la biomasa, incluyendo nueve especies de aves, ocho especies de reptiles y cuatro especies de anfibios. Los invertebrados representaron el 23.8 % de las presas en números pero fueron insignificantes en términos de biomasa (2.8%). Serpientes, ranas y roedores fueron las presas mas frecuentemente entregadas y de mayor contribución a la biomasa total de presas. Sesenta por ciento de las presas entregadas en los nidos fueron aquellas asociadas a habitats del sotobosque, lo cual sugiere que *Buteo lineatus* forrajea extensivamente en este habitat.

[Traducción de César Márquez]

ACKNOWLEDGMENTS

Funding was provided by the Daniel B. Warnell School of Forest Resources at the University of Georgia and McIntire Stennis Project No. GEO-0074-MS. We thank Kim A James for assistance with data collection and species identification. Special recognition goes to the staff of Bishop F. Grant Memorial Forest, particularly A. Harris and J. Gallagher, for housing and on-site logistical support. We thank C.E. Moorman, who provided field assistance throughout the project, and D.E. Andersen, M.J. Bechard, J.C. Bednarz, J.J. Dinsmore, A.S. Johnson, D.G. Krementz, C.D. Marti, and K. Titus who reviewed earlier drafts of the manuscript.

LITERATURE CITED

BEDNARZ, J.C. 1979. I. Productivity, nest sites, and habitat of Red-shouldered and Red-tailed Hawks in Iowa. II. Status, habitat utilization, and management of Red-shouldered Hawks in Iowa. M.S. thesis, Iowa State Univ., Ames, IA U.S.A.

——— AND J.J. DINSMORE. 1985. Flexible dietary response and feeding ecology of the Red-shouldered Hawk, *Buteo lineatus,* in Iowa. *Can. Field-Nat.* 99:262–264.

BLOOM, P.H., M.D. MCCRARY AND M.J. GIBSON. 1993. Red-shouldered Hawk home-range and habitat use in southern California. *J. Wildl. Manage.* 57:258–265.

BRENDER, E.V. 1973. Silviculture of loblolly pine in the Georgia Piedmont. Georgia Forest Research Council Rep. No. 33. Atlanta, GA U.S.A.

BURLEIGH, T.D. 1958. Georgia birds. Univ. Oklahoma Press, Norman, OK U.S.A.

CRAIGHEAD, J.J. AND F.C. CRAIGHEAD, JR. 1956. Hawks, owls, and wildlife. Stackpole Co., Harrisburg, PA U.S.A.

DUNNING, J.B., JR. 1984. Body weights of 686 species of North American birds. Monog. No. 1. West. Bird Banding Assoc., Eldon Publ. Co., Cave Creek, AZ U.S.A.

ERRINGTON, P.L. 1932. Technique of raptor food habits study. *Condor* 35:19–29.

GOLLEY, F.B. 1962. Mammals of Georgia. Univ. Georgia Press, Athens, GA U.S.A.

HENNY, C.J., F.S. SCHMID, E.L. MARTIN AND L.L. HOOD. 1973. Territorial behavior, pesticides, and the population ecology of Red-shouldered Hawks in central Maryland, 1943–1971. *Ecology* 54:545–554.

HOWELL, D.L. AND B.R. CHAPMAN. 1997. Home range and habitat use of Red-shouldered Hawks in Georgia. *Wilson Bull.* 109:131–144.

JANIK, C.A. AND J.A. MOSHER. 1982. Breeding biology of raptors in the Appalachians. *Raptor Res.* 16:18–24.

MARTI, C.D. 1987. Raptor food habits studies. Pages 67–80 *in* B.A.G. Pendleton, B.A. Millsap, K.W. Cline and D.M. Bird [EDS.], Raptor management techniques manual. Nat. Wildl. Fed., Sci. Tech. Ser. 10. Washington, DC U.S.A.

MOORMAN, C.E. AND B.R. CHAPMAN. 1996. Nest-site selection of Red-shouldered Hawks and Red-tailed Hawks in a managed forest. *Wilson Bull.* 108:357–368.

PARKER, M.A. 1986. The foraging behavior and habitat use of breeding Red-shouldered Hawks (*Buteo lineatus*) in southeastern Missouri. M.A. thesis, Univ. Missouri, Columbia, MO U.S.A.

PORTNOY, J.W. 1974. Some ecological and behavioral aspects of a nesting population of Red-shouldered Hawks (*Buteo lineatus lineatus*). M.S. thesis, Univ. Massachusetts, Amherst, MA U.S.A.

SNYDER, N.F.R. AND J.W. WILEY. 1976. Sexual size dimorphism in hawks and owls of North America. *Ornith. Monogr.* 20:1–96.

STEENHOF, K. 1983. Prey weights for computing percent biomass in raptor diets. *Raptor Res.* 17:15–27.

USDA-SCS. 1965. Soil survey of Morgan County, Georgia. USDA Soil Conservation Service.

USDA-SCS. 1976. Soil survey of Baldwin, Jones, and Putnam counties, Georgia. USDA Soil Conservation Service.

Received 10 April 1997; accepted 11 May 1998

J. Raptor Res. 32(3):261–264
© 1998 The Raptor Research Foundation, Inc.

PEREGRINE FALCONS (*FALCO PEREGRINUS*) NEST IN A QUARRY AND ON HIGHWAY CUTBANKS IN ALASKA

ROBERT J. RITCHIE
ABR, Inc. Environmental Research and Services, P.O. Box 80410, Fairbanks, AK 99708 U.S.A.

TERRY J. DOYLE
Tetlin National Wildlife Refuge, P.O. Box 779, Tok, AK 99780 U.S.A.

JOHN M. WRIGHT
Alaska Department of Fish and Game, 1300 Mile College Road, Fairbanks, AK 99701 U.S.A.

KEY WORDS: *Peregrine Falcon;* Falco peregrinus; *nesting; artificial nests; disturbance; Alaska.*

In Alaska, the Peregrine Falcon (*Falco peregrinus*) nests primarily on cliffs, although nesting on low relief dirt banks does occur (Cade 1960). As peregrine populations have recovered and increased in Alaska, sites that once may have been described as suboptimal for nesting have been occupied. Further, man-made structures and altered habitats might be expected to attract at least occasional use as densities of these falcons, as well as altered habitats, continue to increase. Examples of human developments in Alaska that might attract nesting peregrines include towers, quarries, and road cutbanks. Some of these artificial nesting habitats have been used elsewhere in the breeding range of the species (White et al. 1988, Cade et al. 1996), and have been used in Alaska by Gyrfalcons (*Falco rusticolus;* White and Roseneau 1970, Ritchie 1992). In this paper, we describe recent Peregrine Falcon nesting at two well-trafficked highway cutbanks along the Alaska Highway in east central Alaska and at a quarry site on the Seward Peninsula in northwestern Alaska.

HIGHWAY SITES

In 1995, we located a pair of Peregrine Falcons at a quarried road cut along the Alaska Highway in east-central Alaska. The nest (EN) was located on a broad, rocky ledge (approximately 1 m × 1 m), approximately 10 m from the base of the cut (Fig. 1). There was no rock overhang above the nest, but steep rock sides adjacent to the scrape sheltered the site partially from severe weather. Subsequent visits that year revealed the female incubating two eggs, both of which hatched and both young fledged. In 1996, an adult, presumably the female, was observed incubating two eggs but the nest eventually failed. In 1997, a pair of falcons again occupied this site, four eggs were incubated, and three young were present on 31 July.

The second nest (WN) was located along the highway on a cutbank approximately 300 km west of EN. Although the nest occurred on a well-shaded, natural ledge, approximately 50 m above the highway (Fig. 2), the lower third of the cliff had been quarried during widening of the road. This location was first identified as a possible nesting location by biologists during a general bird survey in the area; a pair was observed in May 1997 (M. Ambrose and C. McIntyre pers. comm.). Our observations that summer revealed that a pair nested on the bluff and that one young fledged in mid-August.

Peregrine Falcons are a common breeding bird along the Tanana River and some of its major tributaries adjacent to the Alaska Highway. More than 25 pairs occur on riparian cliffs along the main channel (Bente and Wright 1995). Off-river sites on bluffs bordering older portions of the floodplain or in upland areas (like EN) are present but more limited than are sites fronting the main river channel. Some of these off-river sites have been occupied, but after most cliff habitats along the river had been used (B. Ritchie, unpubl. data).

Although initial sightings of falcons at these two sites were not associated with rigorous Peregrine Falcon surveys in the area, we think that it is unlikely that peregrines occupied these sites during the previous decade. The EN site had been checked by one of the authors since 1988. The WN site is along a route frequented by biologists interested in peregrine use of cliffs in the area.

QUARRY SITE

The third site (QN) was located east of Nome, northwestern Alaska, in a rock quarry fronted by a well-maintained, two-lane gravel road. The quarry was cut in graduated 6–8 m steps to an elevation of approximately 110 m, on a 185 m tall headland on the coast of the Bering Sea. Peregrines were first reported there in 1988, when a pair of adults with one young was observed in late July (T. Booth and B. Nelson pers. comm.). From 1989–91, a pair of defensive peregrines was observed each year, but

Figure 1. A view of a quarried road cut used by Peregrine Falcons for nesting (EN) along the Alaska Highway.

no young were found during intensive searches. Two empty nest scrapes were found near the top of the quarry in niches in vertical step faces of the blasted rock in 1989. In 1996 and 1997 members of birding tours occasionally observed single peregrines at the site (P. Bente pers. comm.) and Common Ravens (*Corvus corax*) nested in the quarry.

As of 1991, more than 35 peregrine nest sites were known for the northwest coast of Alaska, on sea cliffs and dirt bluffs from the mouth of the Yukon River to Cape Prince of Wales (Bente and Wright 1992). The closest neighboring sites were approximately 60 km of our quarry site. Peregrines may have been at the quarry before 1988, but it is unlikely. Birding tours regularly drove by the site to reach premier birding spots and the quarry itself was checked occasionally by agency representatives.

ATYPICAL SITES

We think that at least two factors make these sites of special interest. First, all three sites were located in what can be described as moderate to high disturbance zones. Both sites in interior Alaska were along busy highways. Traffic records for EN identified over 600 vehicles/day during May–August (U.S. Customs, ALCAN Station, unpubl. data), the period during which peregrines would have courted, laid and incubated eggs, and raised young (Cade 1960). Traffic at WN undoubtedly was much greater due to regular commuter activity associated with nearby communities. In addition, heavy equipment (e.g., rock crusher) operated behind EN in the years it was successful.

The road beneath QN was used daily by numerous vehicles after mid-May (1988–91, 1996–97; J. Wright, unpubl. data). The quarry was in operation periodically during our visits. In July 1988, a rock crusher was operating at the base of the quarry within 200 m of the nest with young. From 1989–91, the quarry was not in operation when we stopped 2–4 times each summer to check for peregrines. In 1996 and 1997, blasting and crushing operations regularly occurred.

While it is true that a few other peregrine nests in Alaska have been found close to roads, the volume of and proximity to traffic are substantially less at these than levels and distances recorded at EN and WN. Numerous cases of peregrines using high-traffic areas (e.g., occupied buildings and traffic-laden bridges) have been described in urban areas (Cade et al. 1996, Bell et al. 1996). However, it is likely that many of these birds (especially in eastern North America) were captive-reared and released in urban areas (Cade and Bird 1990). These "urban" birds may have been more tolerant of human activity. To our knowledge, this phenomenon of using human altered habitats, within high disturbance areas, has not been reported for remote and wild populations in North America.

Confirmed records for Peregrine Falcons nesting on man-modified structures in Alaska are lacking. Peregrines were reported defending a large microwave tower at a Dewline Site on Barter Island in northern Alaska (D. Nigro pers. comm.), but proof of nesting was not established. In earlier years, Peregrine Falcons had successfully nested on a coastal dirt bluff within 2 km of the tower site (F. Mauer pers. comm.). Quarried sites undoubtedly provide suitable habitat for nesting and may be more attractive where natural cliffs are limited or occupied. As peregrine populations have recovered elsewhere from

Figure 2. A view from the nest scrape on a quarried road cut used by Peregrine Falcons for nesting (WN) along the Alaska Highway.

the low numbers in the 1970s and prime habitats are reoccupied, quarries have been used (Australia: White et al. 1988, Olsen 1995; Britain: Ratcliffe 1988; U.S.: Cade et al. 1996). It is important to note, however, that while peregrines were successful in nesting at all of our sites during some years, they only definitely nested in one of six years at the QN site and failed during incubation in one of three years at EN. We cannot rule out disturbance as a factor in unsuccessful nesting.

The three sites described herein are close analogs of natural cliff habitats in their respective regions. With the exception of the levels of human disturbance, each provided basic requirements for successful nesting: good ledges and protection from ground predators. Additionally, each appeared to be proximal to enough habitats used by common prey species for successful hunting. Prey remains gathered at WN and EN were typical of prey for peregrines in interior Alaska (Cade 1960). For example, shorebirds (53%), ducks and gulls (18%) and passerines (29%) comprised the prey items found at the EN site ($N = 17$). Similar species have been gathered at Tanana River nests (B. Ritchie, unpubl. data).

As populations of the Peregrine Falcon continue to recover in Alaska, they appear to be increasing in areas which once were thought to contain suboptimal nesting sites (e.g., off-river areas, subalpine cliffs). As reported here, some pairs have been able to use "suboptimal" (i.e., disturbed areas) cliffs modified by humans. It will be interesting to monitor the continued recovery of peregrines in Alaska as man-modified habitats increase in distribution, abundance, and diversity. We suspect that a growing number of birds will tolerate human activity and attempt to use these areas for nesting.

Resumen.—*Falco peregrinus* anidó exitosamente en tres habitats alterados de Alaska. Dos de los sitios fueron en canteras a lo largo de congestionadas autopistas en el interior de Alaska. El tercer sitio ocupado fue una de las caras de una cantera activa en el noreste de Alaska. Estas son las primeras observaciones registradas de halcones peregrinos anidando en este tipo de habitats en lugares remotos como Alaska. Aun más, todos los sitios estaban cercanos a carreteras con bastante tráfico. De continuar en aumento el número de peregrinos y de sitios alterados en Alaska, sospechamos que mas halcones peregrinos utilizarán estos habitats y estructuras.

[Traducción de César Márquez]

Acknowledgments

The authors would like to thank Clayton White, Ted Swem and Dan Varland for reviewing a draft of this paper and making insightful comments for its improvements. We would also like to thank a number of people who provided their observations of Peregrine Falcons at sites described in our paper: Michelle Ambrose; Peter Bente and Bob Nelson, Alaska Department of Fish and Game; Carol McIntyre, National Park Service; Tony Booth, Fran Mauer, and Henry Timm, U.S. Fish and Wildlife Service; and Debbie Nigro, ABR, Inc.

Literature Cited

Bell, D.A., D.P. Gregoire and B.J. Walton. 1996. Bridge use by Peregrine Falcons in the San Francisco Bay Area. Pages 15–24 *in* D.M. Bird, D.E. Varland and J.J. Negro [Eds.], Raptors in human landscapes. Academic Press, London, U.K.

Bente, P.J. and J.M. Wright. 1992. Documentation of active Peregrine Falcon nest sites. Alaska Dept. Fish and Game, Fed. Aid in Wildl. Restor., Final Rep., Proj. SE 2-6, Juneau, AK U.S.A.

——— and ———. 1995. Documentation of Peregrine Falcon nest sites in relation to state land use proposals. Alaska Dept. Fish and Game, Fed. Aid in Wildl. Restor., Final Rep. Proj. SE 2-8, Part II, Juneau, AK U.S.A.

Cade, T.J. 1960. The ecology of peregrines and Gyrfalcons in Alaska. *Univ. Calif. Publ. Zool.* 63:151–290.

——— and D.M. Bird. 1990. Peregrine Falcons, *Falco peregrinus*, nesting in an urban environment: a review. *Can. Field-Nat.* 104:209–218.

———, M. MARTELL, P. REDIG, G. SEPTON AND H. TORDOFF. 1996. Peregrine Falcons in urban North America. Pages 3–13 *in* D.M. Bird, D.E. Varland and J.J. Negro [EDS.], Raptors in human landscapes. Academic Press, London, U.K.

OLSEN, P. 1995. Australian birds of prey: the biology and ecology of raptors. John Hopkins Univ. Press, Baltimore, MD U.S.A.

RATCLIFFE, D.A. 1988. The peregrine population of Great Britain and Ireland, 1965–1985. Pages 147–157 *in* T.J. Cade, J.H. Enderson, C.G. Thelander and C.M. White [EDS.], Peregrine Falcon populations: their management and recovery. The Peregrine Fund, Inc., Boise, ID U.S.A.

RITCHIE, R.J. 1992. Effects of oil development on providing nesting opportunities for Gyrfalcons and Rough-legged Hawks in northern Alaska. *Condor* 93:180–184

WHITE, C.M. AND D.G. ROSENEAU. 1970. Observations on food, nesting, and winter populations of large North American falcons. *Condor* 72:13–115.

———, W.B. EMISON AND W.M. BREN. 1988. Atypical nesting habitat of the Peregrine Falcon in Victoria, Australia. *J. Raptor Res.* 22:37–43.

Received 29 December 1997; accepted 17 May 1998

J. Raptor Res. 32(3):264–266
© 1998 The Raptor Research Foundation, Inc.

LICE (PHTHIRAPTERA: AMBLYCERA, ISCHNOCERA) OF RAPTORS IN HUNGARIAN ZOOS AND REHABILITATION CENTERS

SZABOLCS SOLT
H-9083 Écs, Petőfi u. 60., Hungary

KEY WORDS: *Phthiraptera; louse infestations; rehabilitation centers; injured raptors; Hungary.*

Here, I describe louse (Phthiraptera: Amblycera, Ischnocera) infestations of raptors kept in zoos and rehabilitation centers in Hungary and conclude that injuries increase the frequency and extent of such infestations. Fifty-five individuals of 18 species of raptors from the families Accipitridae, Falconidae, Tytonidae, and Strigidae were examined in 1995–96 at the Zoological Parks of Győr and Veszprém (14 and 12 birds, respectively), the raptor rehabilitation center of Fertő-Hanság National Park at Kőszeg (11 birds), and the rehabilitation center of Hortobágy National Park at Góréstanya (18 birds). There was no regular use of insecticides to control louse infestations at any of these sites. Many of the raptors were badly injured by electrocution from high voltage transmission lines or by illegal shooting. Injuries often resulted in extensive damage to wings and legs.

Lice were collected using forceps during 10-min visual examinations while the birds were immobilized by assistants. Twenty birds (36%) were found to carry lice resulting in a total of 373 lice (86 males, 196 females, and 91 nymphs, Table 1) collected. Eight species of lice were found, two of which were typical parasites of galliform hosts and presumably originated from dead chickens supplied as food.

Avian grooming partially serves to control the spread of ectoparasites (Marshall 1981). Grooming, such as foot scratching, eliminates lice on the head and bill preening removes lice from other body parts (Clayton 1991, Rózsa 1993). Since I assumed that birds with major limb injuries were presumed to preen less frequently, I compared the numbers of lice on 11 injured and 28 healthy raptors. Because of the aggregated distribution of lice on different individuals (Rékási et al. 1997), I used a one-tailed Mann-Whitney U-test as a nonparametric statistic. Statistical analyses were carried out by InStat 2.01.

Avian lice can be viewed as representatives of a single ecological guild of ectoparasites. Thus, their abundance can be expressed as total louse numbers (belonging to any species) living on the same individual bird (Rózsa 1997). When comparisons were made between raptors with damaged limbs versus intact limbs, there was a significant difference in total louse abundance ($U = 48.5$, $P < 0.001$, Fig. 1). Lice also show considerable site-specificity on their hosts (e.g., Perez et al. 1996); therefore different louse taxa should show different responses to decreased grooming abilities of injured raptors. Species of the genus *Craspedorrhynchus,* for instance, are typically distributed on the head and nape of raptors and birds scratch using their feet to remove them (Gallego et al 1987). Limb-damaged birds naturally show much less foot scratching, either because they lack the use of one leg which prohibits them from reaching their heads with the other one, or because they have broken wings which distorts normal foot scratching movements at least on one side of the body. In fact, the abundance of *Craspe-*

Table 1. Lice collected from healthy and injured raptors in Hungarian zoos and rehabilitation centers.

RAPTOR SPECIES	NO. HEALTHY BIRDS	NO. INJURED BIRDS	LOUSE SPECIES	NO. FROM HEALTHY BIRDS	NO. FROM INJURED BIRDS
Accipiter gentilis	1	1	*Menopon gallinae*[a]	0	2
Buteo buteo	11	6	*Craspedorrhynchus platystomus*	8	102
			Colpocephalum buteonis	0	31
			Degeeriella fulva	7	35
			Menopon gallinae[a]	0	4
			Lipeurus caponis[a]	0	1
Buteo rufinus	1	0	*Craspedorrhynchus platystomus*	8	0
			Degeeriella fulva	20	0
Circus aeruginosus	3	0	—	0	0
Circaetus gallicus	1	0	—	0	0
Falco peregrinus	1	0	—	0	0
Falco subbuteo	0	1	—	0	0
Falco tinnunculus	8	1	*Degeeriella rufa*	40	27
Aquila heliaca	0	1	*Degeeriella fulva*	0	2
Aquila pomarina	0	1	*Craspedorrhynchus naevius*	0	73
Hieraetus pennatus	1	0	*Degeeriella fulva*	12	0
Theratopius ecaudatus	1	0	—	0	0
Asio otus	2	0	—	0	0
Athene noctua	4	0	—	0	0
Bubo bubo	1	0	—	0	0
Nyctea scandiaca	2	0	—	0	0
Stryx aluco	2	0	—	0	0
Tyto alba	5	0	*Kurodaia subpachygaster*	1	0

[a] Lice specific to galliforms.

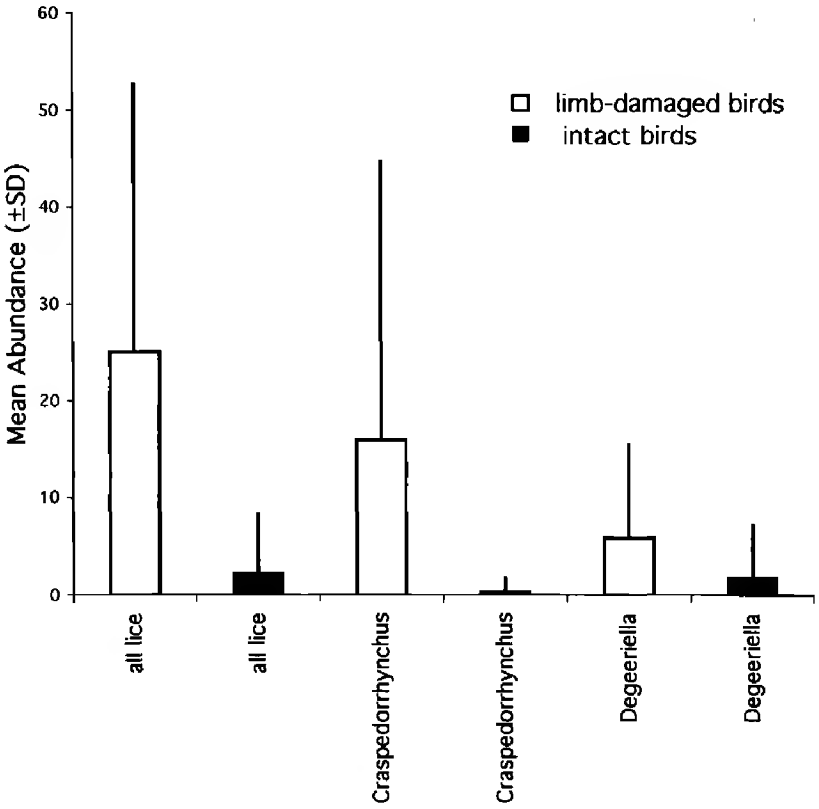

Figure 1. Mean number (±SD) of all lice and only lice belonging to the genera *Craspedorrhynchus* and *Degeeriella* on limb-damaged (11 birds) and healthy raptors (28 birds).

dorrhynchus lice was significantly greater in injured versus healthy raptors ($U = 101.5$, $P < 0.05$).

Contrastingly, lice of the genus *Degeeriella* are typically distributed on the wings, especially the primary and secondary feathers. Raptors typically preen with their beaks to eliminate them (Clay 1958). Limb-damaged birds usually exhibit normal preening activities except for preening of broken wings. *Degeeriella* lice appeared to be more abundant on limb-damaged raptors in my examination, though this difference was not significant ($U = 119.5$, $P > 0.10$).

The correlation between major injuries of raptor limbs and increased louse infestations seemed to be related to the impairment of their grooming behavior. Site-specific lice such as those that infest the heads of raptors were found to be common. Since wing- or leg-damaged raptors are at times important for raptor breeding and repatriation programs, it is important that they be routinely examined for ectoparasite infestations and may need special attention to control their lice.

RESUMEN.—Cincuenta y cinco individuos de 18 especies de aves rapaces fueron examinados con el fin de encontrar infestaciones de piojos (Phthiraptera: Amblycera, Ischnocera) en Hungría en 1995 y 1996. Un total de 373 piojos de 8 especies fueron encontrados en 20 aves. Los daños mayores causados en alas y patas como electrocutaciones, aparentemente lograron disminuir la habilidad de acicalarse de las aves aumentando la frecuencia de los piojos. Debido a que estas aves fueron mas propensas a ser infestadas por ectoparásitos, especialmente aquellas con problemas en alas y patas, hubo que prestarles mayor atención para el control de piojos.

[Traducción de César Márquez]

ACKNOWLEDGMENTS

I thank the staff of Zoos and National Parks for providing access to captive raptors, József Rékási for his help in louse identification and Lajos Rózsa for comments on the manuscripts.

LITERATURE CITED

CLAY, T. 1958. Revisions of Mallophaga genera. *Degeeriella* from the Falconiformes. *Bull. Brit. Mus. (Nat. Hist.) Entomol.* 7:5–207.

CLAYTON, D.H. 1991. Coevolution of avian grooming and ectoparasite avoidance. Pages 258-289 *in* J. E. Loye and M. Zuk [EDS.], Bird-parasite interactions. Oxford Univ. Press, Oxford, U.K.

GALLEGO, J., M.P.M. MATEO AND J.M. AGUIRRE. 1987. Malófagos de rapaces Espanolas. II. Las especies del género *Craspedorrhynchus* Kéler, 1938 parásitas de falconiformes, con descripción de tres especies nuevas. *Eos* 63:31–66.

MARSHALL, A.G. 1981. The ecology of ectoparasitic insects. Academic Press, London, U.K.

PEREZ, J.M, I. RUÍZ-MARTÍNEZ AND J.E. COOPER. 1996. Occurrence of chewing lice on spanish raptors. *Ardeola* 43:129–138.

RÉKÁSI, J., L. RÓZSA AND J.B. KISS. 1997. Patterns in the distribution of avian lice (Phthiraptera: Amblycera, Ischnocera). *J. Avian Biol.* 28:150–156.

RÓZSA, L. 1993. An experimental test of the site-specificity of preening to control lice in feral pigeons. *J. Parasitol.* 79:968–970.

———. 1997. Patterns in the abundance of avian lice (Phthiraptera: Amblycera, Ischnocera). *J. Avian Biol* 28:249–254.

Received 21 November 1997; accepted 12 May 1998

LETTERS

J. Raptor Res. 32(3):267
© 1998 The Raptor Research Foundation, Inc.

EVIDENCE OF SPOTTED KESTREL (*FALCO MOLUCCENSIS*) NESTING IN THE ROOFS OF SUMBA'S TRADITIONAL HOUSES

On the Island of Sumba, Indonesia, the persistent, animist religion of ancestor worship has widely preserved the traditional thatched house structure. These houses have a striking roof that is low-sided but high-peaked. The houses associated with clan ancestors have higher roofs and are preferably placed on hilltops. Traditional villages are found scattered throughout the rolling country, much of which is used for extensive cattle and horse raising, but ancestral houses are also in small towns.

During a tour of Sumba, from 10–14 August 1997, I made short visits to nine traditional villages. In five of these, and also near traditional houses in the center of Waikabubak (the second largest town of Sumba with a population of about 15 000), I found Spotted Kestrels (*Falco moluccensis*). I found no more than one pair at each location and the kestrels called from trees close to the houses or hunted in adjacent fields. In the village of Praigoli, southeast of Waikabubak, I also observed an adult kestrel enter the top of a roof and emerge after some seconds to stand near the presumed nesting cavity. About 10 km to the west, I saw similar behavior in a village near Morossi Beach; on this occasion the kestrel entered the roof top with a large orthopteran in its bill and it stayed inside longer. Both houses were occupied, but the upper part of the Sumbanese houses are undisturbed because the owners believe them to be places reserved for the spirits of their ancestors. I was intrigued by some possible link between this belief and the presence of kestrels, but an apparently well-informed local guide was unable to give me any relevant information on this subject.

Although the Spotted Kestrel is already listed among the raptors that are attracted to towns by opportunities to nest in buildings (Brown 1976, Birds of prey, Ross International Books, Ltd., London, U.K.), finding it nesting in traditional villages suggests much older association with human dwellings. The distribution of this Indonesian endemic is centered on the biogeographic region called Wallacea. The extensive grassland that is found in this region, which is especially vast on Sumba, may be the result of human activity but has existed long enough to support a distinctive bird fauna (White and Bruce 1986, The Birds of Wallacea, B.O.U. Checklist No. 7, London, U.K.). Although it is rather an opportunistic species, the Spotted Kestrel has shown its preference for open habitats in Wallacea in the past (e.g., Rensch 1931, *Mitt. Zool. Mus. Berlin* 17:451–637) when the region was more forested.

During my tour, Sumba's grassland seemed especially rich in orthopteran prey. However, the Spotted Kestrel probably has to cope with a scarcity of prominent rocks to nest on Sumba, especially where I saw evidence of nesting birds. Also, trees with hollows are scarce in these grasslands. High-peaked thatched roofs situated on hilltops probably make up for this deficiency. Unfortunately for kestrels, the thatching practice is being challenged by longer-lasting, though thermally less insulating, sheet-iron roofs.

I found no previous report of this apparently common nesting habit. This may be because the Spotted Kestrel looks similar to the Common Kestrel (*Falco tinnunculus*) and did not draw the attention of early field naturalists, Sumba is still visited infrequently by tourists who might tend to report these birds; Sumba's traditional villages offer many other attractions that might distract people from the species, and birders avoid human settlements with Spotted Kestrels to search for more "memorable" species. In fact, the Spotted Kestrel remains a little known species over all of its range.

I thank T. Cade, D.E. Varland, and an anonymous reviewer for text improvement.—**Tiziano Londei, Dipartimento di Biologia, Università di Milano, Via Celoria 26, 20133 Milano, Italy.**

J. Raptor Res. 32(3):268
© 1998 The Raptor Research Foundation, Inc.

CARIBOU ANTLERS AS NEST MATERIALS FOR GOLDEN EAGLES IN NORTHWESTERN ALASKA

There are few published records of antlers in Golden Eagle (*Aquila chrysaetos*) nests (three for Scotland [L. MacNally, 1977, The ways of an eagle, Collins and Harvill Press, London, U.K. and S. Gordon, 1955, The golden eagle: king of birds, Citadel Press, New York, NY U.S.A.] and an unspecified number for Colorado [R. Olendorff, 1975, Golden eagle country, Alfred A. Knopf, New York NY U.S.A.]). We found many caribou (*Rangifer tarandus*) antlers in three Golden Eagle nests in the Cape Kruzenstern region (67–68°N, 163–164°W) of northwestern Alaska. At one nest that was situated at 270 m elevation on a cliff face, three young and two adults were present in 1986. This nest contained numerous male and female antlers composing 10–15% of the matrix of the nest. The second nest in which antlers were found was situated 240 m elevation atop a 2-m cliff in 1988. We counted 21 antlers, some of which were very large male antlers, all in the perimeter of the nest (ca 10–15% of the bulk). The third nest was situated at 300 m elevation on a cliff face in 1988 and it contained numerous antlers, mostly of female caribou (ca 10% of nest bulk). We did not find antlers in one other Golden Eagle nest on the Cape and in five other nests along the Noatak River immediately to the east.

Because all three nests were within 13 km of each other and no two were simultaneously occupied, there was a possibility that all were alternate nests of the same pair of eagles with a preference for antlers. Another explanation for the local importance of antlers is that the lack of sizable sticks, the similarity of antlers to sticks, and the abundance of antlers, especially the much smaller female caribou antlers, lead the eagles to substitute antlers for sticks. C.M. White and J.R. Haugh (pers. comm.) saw parts of antlers in a nest in northwestern Alaska and Y. Potopov (pers. comm.) found a small caribou antler in a tree nest along the Kolyma River in northeastern Siberia. We also found a single deer (probably *Odocoileus hemionus*) antler in a nest in Montana.—**David H. Ellis, USGS Patuxent Wildlife Research Center, HCR 1 Box 4420, Oracle, AZ 85623 U.S.A. and Richard L. Bunn, AFZC-ECM-NR (Wildlife), Building 302, Fort Carson, CO 80913-5000 U.S.A.**

THE RAPTOR RESEARCH FOUNDATION, INC.
(Founded 1966)

OFFICERS

PRESIDENT: Michael N. Kochert
VICE-PRESIDENT: David E. Andersen
SECRETARY: Patricia A. Hall
TREASURER: Jim Fitzpatrick

BOARD OF DIRECTORS

NORTH AMERICAN DIRECTOR #1:
Brian A. Millsap
NORTH AMERICAN DIRECTOR #2:
Petra Bohall Wood
NORTH AMERICAN DIRECTOR #3:
Karen Steenhof
INTERNATIONAL DIRECTOR #1:
Massimo Pandolfi
INTERNATIONAL DIRECTOR #2:
Reuven Yosef
INTERNATIONAL DIRECTOR #3:
Michael McGrady
DIRECTOR AT LARGE #1: Patricia L. Kennedy
DIRECTOR AT LARGE #2: John A. Smallwood
DIRECTOR AT LARGE #3: James C. Bednarz
DIRECTOR AT LARGE #4: César Márquez Reyes
DIRECTOR AT LARGE #5: Lloyd Kiff
DIRECTOR AT LARGE #6: Robert Kenward

****

EDITORIAL STAFF

EDITOR: Marc J. Bechard, Department of Biology, Boise State University, Boise, ID 83725 U.S.A.

ASSOCIATE EDITORS

Allen M. Fish
Gary R. Bortolotti
Charles J. Henny
Fabian Jaksić
Daniel E. Varland

BOOK REVIEW EDITOR: Jeffrey S. Marks, Montana Cooperative Research Unit, University of Montana, Missoula, MT 59812 U.S.A.
SPECIAL PUBLICATIONS EDITOR: Daniel E. Varland, Rayonier, 3033 Ingram Street, Hoquiam, WA 98550
SPANISH EDITOR: César Márquez Reyes, Instituto Humbolist Colombia, AA. 094766, Bogotá 8, Colombia

The Journal of Raptor Research is distributed quarterly to all current members. Original manuscripts dealing with the biology and conservation of diurnal and nocturnal birds of prey are welcomed from throughout the world, but must be written in English. Submissions can be in the form of research articles, letters to the editor, thesis abstracts and book reviews. Contributors should submit a typewritten original and three copies to the Editor. All submissions must be typewritten and double-spaced on one side of 216 × 278 mm (8½ × 11 in.) or standard international, white, bond paper, with 25 mm (1 in.) margins. The cover page should contain a title, the author's full name(s) and address(es). Name and address should be centered on the cover page. If the current address is different, indicate this via a footnote. A short version of the title, not exceeding 35 characters, should be provided for a running head. An abstract of about 250 words should accompany all research articles on a separate page.

Tables, one to a page, should be double-spaced throughout and be assigned consecutive Arabic numerals. Collect all figure legends on a separate page. Each illustration should be centered on a single page and be no smaller than final size and no larger than twice final size. The name of the author(s) and figure number, assigned consecutively using Arabic numerals, should be pencilled on the back of each figure.

Names for birds should follow the A.O.U. Checklist of North American Birds (6th ed., 1983) or another authoritative source for other regions. Subspecific identification should be cited only when pertinent to the material presented. Metric units should be used for all measurements. Use the 24-hour clock (e.g., 0830 H and 2030 H) and "continental" dating (e.g., 1 January 1990).

Refer to a recent issue of the journal for details in format. Explicit instructions and publication policy are outlined in "Information for contributors," *J. Raptor Res.*, Vol. 27(4), and are available from the editor.

The Raptor Research Foundation, Inc. gratefully acknowledges a grant and logistical support provided by Boise State University to assist in the publication of the journal.

****

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.

The Journal of Raptor Research (ISSN 0892-1016) is published quarterly and available to individuals for $33.00 per year and to libraries and institutions for $50.00 per year from The Raptor Research Foundation, Inc., 14377 117th Street South, Hastings, Minnesota 55033, U.S.A. (Add $3 for destinations outside of the continental United States.) Periodicals postage paid at Hastings, Minnesota, and additional mailing offices. POSTMASTER: Send address changes to *The Journal of Raptor Research,* OSNA, P.O. Box 1897, Lawrence, KS 66044-8897, U.S.A.

Printed by Allen Press, Inc., Lawrence, Kansas, U.S.A.

Copyright 1998 by The Raptor Research Foundation, Inc. Printed in U.S.A.

♾ **This paper meets the requirements of ANSI/NISO Z39.48-1992 (Permanence of Paper).**

RAPTOR RESEARCH FOUNDATION, INC., AWARDS

Recognition for Significant Contributions[1]

The **Dean Amadon Award** recognizes an individual who has made significant contributions in the field of systematics or distribution of raptors. Contact: **Dr. Clayton White, 161 WIDB, Department of Zoology, Brigham Young University, Provo, UT 84602 U.S.A.** Deadline August 15.

The **Tom Cade Award** recognizes an individual who has made significant advances in the area of captive propagation and reintroduction of raptors. Contact: **Dr. Brian Walton, Predatory Bird Research Group, Lower Quarry, University of California, Santa Cruz, CA 95064 U.S.A.** Deadline: August 15.

The **Fran and Frederick Hamerstrom Award** recognizes an individual who has contributed significantly to the understanding of raptor ecology and natural history. Contact: **Dr. David E. Andersen, Department of Fisheries and Wildlife, 200 Hodson Hall, 1980 Folwell Avenue, University of Minnesota, St. Paul, MN 55108 U.S.A.** Deadline: August 15.

Recognition and Travel Assistance

The **James R. Koplin Travel Award** is given to a student who is the senior author of the paper to be presented at the meeting for which travel funds are requested. Contact: **Dr. Petra Wood, West Virginia Cooperative Fish and Wildlife Research Unit, P.O. Box 6125, Percival Hall, Room 333, Morgantown, WV 26506-6125 U.S.A.** Deadline: established for conference paper abstracts.

The **William C. Andersen Memorial Award** is given to the student who presents the best paper at the annual Raptor Research Foundation Meeting. Contact: **Ms. Laurie Goodrich, Hawk Mountain Sanctuary, Rural Route 2, Box 191, Kempton, PA 19529-9449 U.S.A.** Deadline: Deadline established for meeting paper abstracts.

Grants[2]

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. Kimberly Titus, Alaska Division of Wildlife Conservation, P.O. Box 20, Douglas, AK 99824 U.S.A.** Deadline: September 10.

The **Leslie Brown Memorial Grant** for $500–$1,000 is given to support research and/or the dissemination of information on raptors, especially to individuals carrying out work in Africa. Contact: **Dr. Jeffrey L. Lincer, 15644 Kingman Rd., Poway, CA 92064 U.S.A.** Deadline: September 15.

---

[1] Nominations should include: (1) the name, title and address of both nominee and nominator, (2) the names of three persons qualified to evaluate the nominee's scientific contribution, (3) a brief (one page) summary of the scientific contribution of the nominee.

[2] Send 5 copies of a proposal ($\leq$5 pages) describing the applicant's background, study goals and methods, anticipated budget, and other funding.