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**COVER:** Northern Barred Owl (*Strix varia varia*) and northern Spotted Owl (*S. occidentalis caurina*).  
Painting by Patty Walcott, P.O. Box 545, Tenino, WA 98589 U.S.A. (e-mail: sunnmoon@scattercreek.com)

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## DISTRIBUTION, NUMBERS, AND SITE CHARACTERISTICS OF SPOTTED OWLS AND BARRED OWLS IN THE CASCADE MOUNTAINS OF WASHINGTON<sup>1</sup>

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**ABSTRACT.**—We analyzed distribution, numbers, and site characteristics of northern Spotted Owls (*Strix occidentalis caurina*) and northern Barred Owls (*Strix varia varia*) in Gifford Pinchot National Forest, Washington from July 1978–November 2001. Spotted Owl site-centers averaged significantly higher in elevation and in areas with steeper slopes than Barred Owl site-centers. Relative percentage of Barred Owl detections increased 8.6% annually during the study period. Plots of 0.8-km radius (201 ha) centered on Spotted and Barred owl sites differed significantly from random plots ( $N = 500$ ); plots of both species contained more forest  $\geq 180$  yr old and fewer ha of forest 50–79 yr old, and Spotted Owl plots contained more forest 130–179 yr old. Spotted Owl site-centers were in significantly older forest than Barred Owl and random site-centers. Occupied Spotted Owl sites with timber harvest since 1978 contained significantly more forest  $\geq 80$  yr old than unoccupied sites with harvest. Occupancy of Spotted Owl sites in 2001 relative to number of Barred Owl sites, and slope and elevation of Spotted Owl sites, could be traced directly or indirectly to the presence of Barred Owls.

**KEY WORDS:** *northern Barred Owl*; *Strix varia varia*; *northern Spotted Owl*; *Strix occidentalis caurina*; *occupancy*; *site characteristics*; *Washington*; *Cascade Mountains*.

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### DISTRIBUCIÓN, NÚMEROS Y CARACTERÍSTICAS DE SITIOS DE *STRIX OCCIDENTALIS* Y *STRIX VARIA* EN LAS MONTAÑAS CASCADES DE WASHINGTON

**RESUMEN.**—Analizamos la distribución, los números, y las características de los sitios de *Strix occidentalis caurina*, considerados en peligro de extinción en los Estados Unidos, y de *Strix varia varia*, que están invadiendo el noroeste de los Estados Unidos. El área de estudio abarcó 217 812 ha en el Bosque Nacional Gifford Pinchot, Washington, y estudiamos estos búhos desde julio de 1978 a noviembre del 2001. En promedio, los sitios de *Strix occidentalis* estaban en lugares significativamente más altos en elevación y más inclinados que los de *Strix varia*. El porcentaje relativo de detecciones de *Strix varia* aumentó 8.6% al año. Los sitios de un radio de 0.8 km (201 ha) de *Strix occidentalis* y las de *Strix varia* difirieron de los sitios al azar ( $N = 500$ ) porque los sitios de ambas especies tenían más áreas de bosques de por lo menos 180 años de edad y menos áreas de bosques de 50–79 años de edad, y los sitios de *Strix occidentalis* tenían más áreas de 130–179 años de edad. Los centros de los sitios de *Strix occidentalis* estaban situados en bosques más viejos que los de *Strix varia* y los de sitios al azar. Los sitios ocupados por *Strix occidentalis* con tala de árboles desde 1978 tenían más áreas de bosques de por lo menos 80 años de edad que los sitios inocupados con tala de árboles. La ocupación de los sitios de *Strix occidentalis*

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<sup>1</sup> The views herein reflect those of the authors and are not necessarily those of the U.S. Fish and Wildlife Service

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en 2001 en relación al número de los sitios de *Strix varia*, y la inclinación y elevación de los sitios de *Strix occidentalis*, puede estar directamente o indirectamente relacionada a la presencia de *Strix varia*.

[Traducción de Kent Livezey]

The northern Spotted Owl (*Strix occidentalis caurina*) is a resident of forest areas in southwestern British Columbia, western Washington and Oregon, and northwestern California. It was listed as a "threatened" subspecies in 1990, primarily due to evidence that the population was declining concurrent with loss of habitat (Federal Register 55:26 114–26 194, 26 June 1990, Gutiérrez 1994). The Barred Owl (*Strix varia*) was originally widespread throughout southeastern Canada, eastern United States, and eastern Mexico (Rignall 1973). In the early 1900s, the northern Barred Owl (*S. v. varia*) began to expand its range westward, moving across southern Canada and south into Washington, Oregon, and California. Barred Owls were first detected in Washington in 1965 (Rogers 1966), Oregon in 1974 (Taylor and Forsman 1976), and California in 1981 (Evens and LeValley 1982). The range of the Barred Owl now nearly completely overlaps that of the northern Spotted Owl. Barred Owls are larger than Spotted Owls, are aggressive toward them (e.g., Hamer et al. 1989, Leskiw and Gutiérrez 1998), and may compete with them (Hamer et al. 1989, Herter and Hicks 2000, Kelly et al. 2003).

Although a number of authors have suggested that Barred Owls compete with Spotted Owls, the range expansion of the Barred Owl is so recent that there have been only a few studies in which relationships between Barred Owls and Spotted Owls have been investigated (Hamer 1988, Hamer et al. 1989, 1994, 2001, Herter and Hicks 2000, Kelly 2001, Kelly et al. 2003). In this paper, we present data from a 24-yr study (1978–2001) during which we monitored the distribution, number, and habitat associations of Barred and Spotted owls in southwestern Washington. Our objectives were to: (1) describe temporal changes in the relative number and distribution of Barred and Spotted owls; (2) compare habitats occupied by both species, and (3) evaluate occupancy of territories by Spotted Owls as a function of the presence or absence of Barred Owls.

#### METHODS

**Study Area.** The study area was the Cowlitz Valley Ranger District of the Gifford Pinchot National Forest (GPNF). This 217 812-ha area was located on the west slope of the Cascade Mountains in southwestern Wash-



Figure 1. Location of study area (shaded area) in southern Washington.

ington (Fig. 1). Forested vegetation on the study area was dominated by Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), mountain hemlock (*Tsuga mertensiana*), Pacific silver fir (*Abies amabilis*), noble fir (*Abies procera*), subalpine fir (*Abies lasiocarpa*), and western redcedar (*Thuja plicata*). Small, local areas of deciduous hardwoods (1146 ha), such as bigleaf maple (*Acer macrophyllum*), red alder (*Alnus rubra*), and black cottonwood (*Populus trichocarpa*), were mostly in riparian areas.

Much of the study area exhibited the effects of large-scale fires, resulting in large blocks of contiguous forest of the same general age. Forest harvest began in the study area in about 1947. By 2001, forested areas 50–129 yr old were still relatively untouched by timber harvest, whereas forested areas  $\geq 130$  yr old either were heavily fragmented by logging outside of wilderness or remained in contiguous blocks within wilderness. In some areas, older forest was found in narrow bands along valley bottoms, intergrading into younger forest on upper slopes and ridges. The four main forest zones in the study area, with approximate elevation limits, were western hemlock (<914 m), Pacific silver fir (914–1373 m), mountain hemlock (1372–1708 m), and subalpine fir (1707 m-tree-line) (GPNF unpubl. data). The Douglas-fir, a pioneer species that repopulates areas denuded by forest fire, was present throughout the study area within all four forest zones. In addition, old-growth Douglas-fir were present throughout the study area as remnants left after forest fires. In 2001, the study area was composed of ca. 52% forest  $\geq 80$  yr old, 33% forest <80 yr old, and 15% non-

forested areas (rock, wet-mesic, dry meadow/brush, water). The study area contained ca. 2630 km of roads.

The Northwest Forest Plan (USDA and USDI 1994a, 1994b) placed every ha of federally-administered land within the range of the northern Spotted Owl into one of many land-use allocations (LUAs). Within the study area, the LUAs in which timber harvest was permitted were matrix (49 716 ha) and an adaptive management area (42 741 ha), and the LUAs in which no timber harvest was permitted were late-successional reserves (55 275 ha), congressionally-reserved areas (42 721 ha), and administratively-withdrawn areas (27 359 ha). The Northwest Forest Plan permits certain forest activities in late-successional reserves, such as thinning in stands  $\leq 80$  yr old, when they are beneficial to the creation of late-successional forest characteristics. Critical habitat units (Federal Register 57:1796–1838 15 Jan 1992) (81 597 ha) overlaid all other LUAs except congressionally-reserved areas; timber harvest was permitted in critical habitat if it overlaid an adaptive management area or matrix.

**Survey Methods.** Spotted and Barred owls respond to imitations of their calls, and Barred Owls readily respond to Spotted Owl calls (Forsman et al. 1984, McGarigal and Fraser 1985, Hamer 1988, Reid et al. 1999). Amplified tape recordings or voice imitations of Spotted Owl calls were used to detect Spotted Owls and Barred Owls. Hybrid Spotted  $\times$  Barred owls were identified by their distinctive five- or six-note calls that are “strikingly different” from Spotted Owl four-note calls and Barred Owl eight-note calls (Hamer et al. 1994). Surveys followed accepted protocols (Forsman 1983). Spotted Owl habitat was identified within each survey area and survey stations were located to achieve complete coverage of the habitat. Survey stations were 0.2–0.8 km apart within Spotted Owl habitat, typically along roads. Each survey was conducted for at least 10 min per station unless a Spotted Owl, Barred Owl, or Great Horned Owl (*Bubo virginianus*) was detected, in which case calling was stopped to minimize disturbance to Spotted Owls and potential confrontations between owl species.

Survey effort varied from year to year. Forest Service surveys were conducted according to several region-wide survey programs, including random-area surveys and surveys surrounding proposed timber sales, targeting portions of the entire study area each year with no single year achieving total coverage. However, by 1992, the entire study area had been surveyed cumulatively. From 1994–2001, R. Pearson achieved nearly complete coverage in even years and in 2001, while the other years had partial coverage. Because of this, our total sites for Spotted and Barred owls represent cumulative totals rather than the total in 2001.

**Data Analysis.** In our analysis, a Spotted or Barred owl “site” was analogous to an individual owl territory. Sites were designated based on the presence of nests, fledged young, or repeated detections of paired or single owls in the same general area. Resident single sites required  $\geq$  three detections of a male or female owl on three different survey outings. Pair sites required  $\geq$  one detection of both a male and a female that was  $>1.6$  km from a known site for Spotted Owls and  $>0.8$  km from a known site for Barred Owls. The 1.6-km requirement for Spotted Owl sites was waived in one instance when a new pair site

with fledglings was identified  $<1.6$  km from two known sites, and adults of the adjacent pairs were also detected.

For Barred Owls, we also used a category called “potential sites” that included suspected territories that were based on one or two detections of a single owl in an area that was  $\geq 0.8$  km from any other Barred Owl sites. The 0.8-km cutoff for designation of potential sites was based on Hamer et al. (1989), who reported a 218-ha mean summer home-range for Barred Owls (a circular plot of 0.8-km radius includes 201 ha). For each site, we plotted a “site-center” based on the location of the nest tree, fledged young, multiple detections of a pair, or multiple detections of single owls. For Spotted Owl pairs that changed nest trees or centers of activity within sites, we used the most recent locations to plot site-centers. For comparison of forest-stand age at the site-center, we identified the age of the forest stand that included the site-center. We considered Spotted Owl sites unoccupied if they were occupied by a pair for  $\geq 1$  yr, and were subsequently surveyed  $\geq 10$  times during the last 5 yr of the study (1997–2001) with no detections of Spotted Owls within a 1.6-km radius. We excluded sites that were not surveyed  $\geq 10$  times during the last 5 yr of the study ( $N = 16$ ) from analyses of Spotted Owl site occupancy. We analyzed landscape attributes around site-centers based on comparisons of vegetation composition within a 0.8-km radius centered on the owl site-centers and random locations, which we called “core-plots” (Hunter et al. 1995, Meyer et al. 1998, Swindle et al. 1999, Herter and Hicks 2000, Kelly 2001). We centered random sites on the first 500 randomly generated, terrestrial locations below 1524 m in elevation. We excluded random locations above 1524 m elevation because we detected no Spotted or Barred owls above 1524 m, and excluded non-forested areas (rock, wet-mesic, dry meadow/brush, and water) from vegetation analyses. To approximate Spotted Owl home ranges in Fig. 2, we used a 2.9-km radius circle (2670 ha) following Herter and Hicks (2000). For each owl site, we determined a “nearest-neighbor distance” which was the distance to the nearest known site-center of the same species (Hamer et al. 1989, Anthony 2001). We estimated the “minimum nearest-neighbor distance” for Spotted Owls by measuring the distance to the nearest nest trees being used by adjacent pairs in the same year. We found no Barred Owl nest trees, so for this species we estimated minimum nearest-neighbor distance by taking the mean of the 10 shortest distances between Barred Owl site-centers.

We used ArcView version 3.1.1 (Environmental Systems Research Institute, Inc., Redlands, CA) to analyze spatially geographic, physiographic, vegetation, and owl data. GPNF geographic information system database supplied basic forest-age data. We divided the forested area into five age groups, as follows: 0–49-yr-old forest (mainly young trees growing on harvested areas); 50–79-yr-old forest (natural stands growing in burned areas without any late-successional characteristics); 80–129-yr-old forest (forest developing late-successional characteristics); 130–179-yr-old forest (an equal sub-division between forest  $\geq 80$  yr old and forest  $<180$  yr old to test for potential differences of younger and older mature forest); and forest  $\geq 180$  yr old (forest with late-successional characteristics). In western Washington, Douglas-fir stands usually begin to develop late-successional characteristics by the

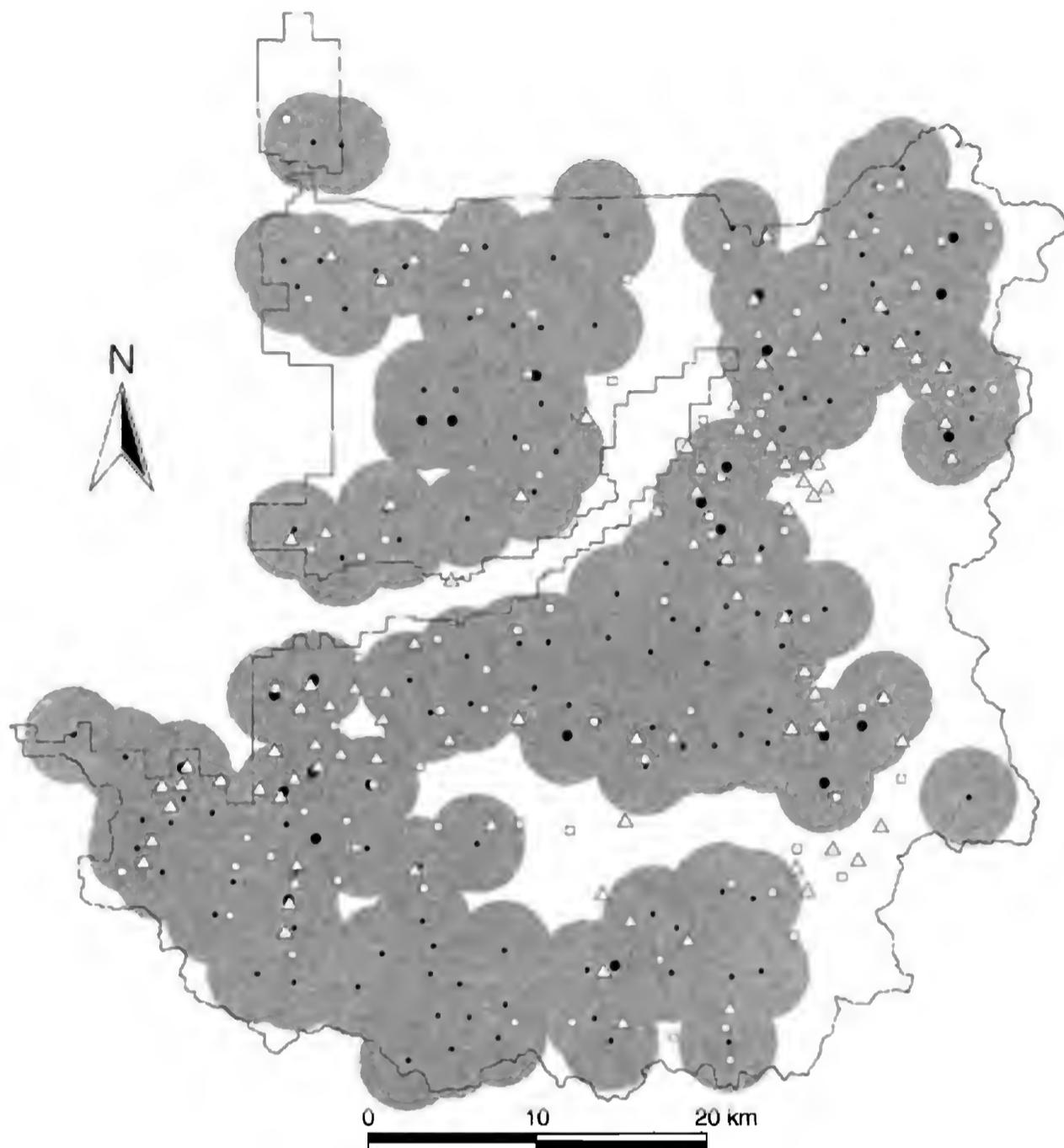


Figure 2. Locations of Spotted Owl occupied sites ( $N = 120$ , small dots), Spotted Owl unoccupied sites ( $N = 25$ , large dots), Barred Owl sites ( $N = 98$ , triangles), and potential Barred Owl sites ( $N = 76$ , boxes). Gray circles represent Spotted Owl home-range circles of 2.9-km radius.

time they are  $\geq 80$  yr old, including relatively large live and dead trees with some multi-layered canopies, and some large woody debris on the forest floor (USDA and USDI 1994a). We estimated the 1946 forest-age throughout the study area by subtracting 55 yr from the age in 2001 of each forested area. For areas where the 2001 forest-age was  $< 55$  yr, such as clearcut and burned areas, we estimated an original yr of origin based upon surrounding forest (e.g., a clearcut surrounded by forest with a 1729 yr-of-origin was given the same yr-of-origin) and then subtracted 55 yrs.

We used nonparametric statistical tests because most of our data did not fit a normal distribution. Statistical tests (two-tailed Mann-Whitney  $U$ -test, Spearman coefficient of rank correlation) were carried out using SYSTAT Version 10 (SPSS Inc., Chicago, IL). We followed Sokal and Rohlf (2000) to calculate  $\chi^2$  tests. Significance level for all tests was  $\alpha = 0.05$ . Sample sizes in statistical tests were presented only when they departed from  $N = 500$  for ran-

dom sites,  $N = 145$  for Spotted Owl sites and  $N = 98$  for Barred Owl sites. All means were expressed as  $\bar{x} \pm 1$  SD.

For comparison of aspect between random, Spotted and Barred owl site-centers, we conducted chi-square tests of the frequency of site-centers within each of four directions (northeast, southeast, southwest, and northwest), north vs. south, and east vs. west, and we included only those random points that were in forest stands  $\geq 80$  yr old ( $N = 250$ ) to ensure that results would indicate possible selection rather than availability. Aspects for random site-centers were not equally distributed in each of the four directions due to naturally occurring differences in the landscape.

We examined change in percentage of Barred Owl detections relative to all *Strix* detections. For this analysis, we determined the first and last years to be included in our sample of owl detections over time as follows: (1) we started with 1982 because it was the first yr with  $\geq 25$  total *Strix* detections, and (2) we ended with 2000 because sur-

Table 1. Mean ( $\pm$ SD) elevation (m) and slope (%) of Spotted Owl, Barred Owl, and random site-centers in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.<sup>a</sup>

	SPOTTED OWL ( <i>N</i> = 145)	BARRED OWL ( <i>N</i> = 98)	RANDOM ( <i>N</i> = 500)
Elevation	966.2 $\pm$ 197.2 <sup>b,c</sup>	812.5 $\pm$ 275.9 <sup>b,d</sup>	1070.3 $\pm$ 282.6 <sup>c,d</sup>
Slope	54.1 $\pm$ 14.5 <sup>b,c</sup>	39.2 $\pm$ 18.6 <sup>b,d</sup>	46.2 $\pm$ 17.16 <sup>c,d</sup>

<sup>a</sup> All significant Mann-Whitney *P*-values < 0.001.

<sup>b</sup> Mean differed from random site-centers.

<sup>c</sup> Mean differed from Barred Owl site-centers.

<sup>d</sup> Mean differed from Spotted Owl site-centers.

veys in 2001 emphasized detecting Barred Owls more than in previous years. Because there were no Barred Owl detections in 1982, but there were Barred Owl detections in previous and in later years, we estimated the 1982 Barred Owl percentage as the mean of the detections for this species for the 9 yr centered on 1982 (1978–86). We were unable to test the effect of Barred Owl presence on Spotted Owl site occupancy in sites that had not been harvested due to small sample sizes.

## RESULTS

**Total Detections and Sites.** We recorded 2170 Spotted Owl and 521 Barred Owl detections during July 1978–November 2001. Based on these detections, we identified 145 Spotted Owl sites, 98 Barred Owl sites, and 76 potential Barred Owl sites in the study area (Fig. 2). Spotted Owl sites were centered on nests (*N* = 43), young (*N* = 41), locations of adults of both sexes detected at the same time (*N* = 44), locations of adults of both sexes detected at different times (*N* = 11), and resident singles (*N* = 6). Barred Owl sites were centered on young (*N* = 8), locations of adults of both sexes detected at the same time (*N* = 51), locations of adults of both sexes detected at different times (*N* = 21), and resident singles (*N* = 18). We believe we found virtually all of the Spotted Owl sites in the study area sometime during 1978–2001, and

that the actual total number of Barred Owl pair sites in our study area was probably closer to our total (*N* = 174) for both the Barred Owl sites and the potential Barred Owl sites.

**Distribution and Numbers.** Both Spotted and Barred owl site-centers were significantly lower in elevation than random, and Spotted Owl site-centers were significantly higher in elevation than Barred Owl site-centers (Table 1). Spotted Owl site-centers were situated in areas of significantly steeper slope than random and Barred Owl site-centers. Barred Owl site-centers were situated in areas of significantly less slope than random. None of the tests of aspect showed any significant difference between Spotted Owl, Barred Owl, and random site-centers for the four directions ( $\chi^2_3 = 1.63$ – $5.99$ , *P* > 0.10) or for north vs. south or east vs. west ( $\chi^2_1 = 0.30$ – $3.30$ , *P* > 0.05).

Table 2. Mean ( $\pm$ SD) area (ha) of five forest-age classes within 0.8-km radius plots (201 ha) around Spotted Owl, Barred Owl, and random site-centers in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.<sup>a</sup>

FOREST-AGE CLASS (yr)	SPOTTED OWL ( <i>N</i> = 145)	BARRED OWL ( <i>N</i> = 98)	RANDOM ( <i>N</i> = 500)
0–49	47.9 $\pm$ 32.1 <sup>b</sup>	42.8 $\pm$ 33.0	43.5 $\pm$ 42.8 <sup>c</sup>
50–79	10.8 $\pm$ 26.3 <sup>b</sup>	12.2 $\pm$ 33.6 <sup>b</sup>	28.8 $\pm$ 52.4 <sup>c,d</sup>
80–129	28.8 $\pm$ 42.2	32.3 $\pm$ 45.8	35.0 $\pm$ 51.2
130–179	17.2 $\pm$ 31.2 <sup>b</sup>	14.2 $\pm$ 30.2	10.7 $\pm$ 24.7 <sup>c</sup>
$\geq 180$	82.4 $\pm$ 44.6 <sup>b</sup>	84.6 $\pm$ 54.1 <sup>b</sup>	57.1 $\pm$ 49.6 <sup>c,d</sup>

<sup>a</sup> Significant Mann-Whitney *P*-values ranged from *P* < 0.02 to *P* < 0.001. None of the means differed between Spotted Owl and Barred Owl sites (all Mann-Whitney *P*-values > 0.20).

<sup>b</sup> Mean differed from random core-plots.

<sup>c</sup> Mean differed from Spotted Owl core-plots.

<sup>d</sup> Mean differed from Barred Owl core-plots.

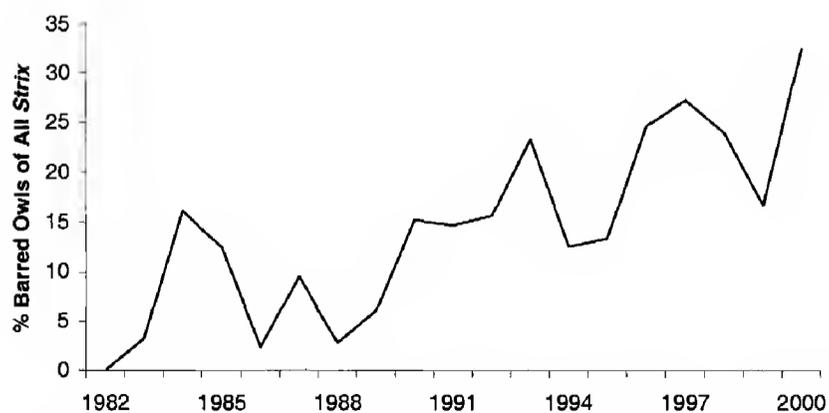
Figure 3. Percent of Barred Owl detections relative to all *Strix* detections by year, 1982–2000.

Table 3. Age of forest stand at site-centers of Spotted Owl, Barred Owl, and random sites in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.

FOREST-AGE CLASS (yr)	STUDY AREA		SPOTTED OWL (N = 145)		BARRED OWL (N = 98)		RANDOM (N = 442) <sup>a</sup>	
	Ha	%	No.	%	No.	%	No.	%
0–49	41 951	24.1	0	0	1 <sup>b</sup>	1.0	111	25.1
50–79	28 193	16.2	1	0.7	4	4.1	78	17.6
80–129	34 934	20.1	15	10.3	19	19.4	89	20.1
130–179	11 814	6.8	10	6.9	7	7.1	26	5.9
≥180	57 252	32.9	119	82.1	67	68.4	138	31.2
Total	174 144		145		98		442 <sup>b</sup>	

<sup>a</sup> The 58 random site-centers that fell in non-forested areas (rock, wet-mesic, dry meadow/brush) were excluded from this analysis.

<sup>b</sup> A 42-yr-old clearcut that retained some remnant old-growth trees.

The percent of Barred Owl detections ( $N = 403$ ) relative to all *Strix* detections ( $N = 2431$ ) increased significantly annually from 1982–2000 ( $r_s = 0.790$ ,  $P < 0.001$ ,  $N = 19$ ). Overall, Barred Owl detections increased 8.6% annually from 1982–2000, and there did not appear to be any leveling-off of this increase (Fig. 3). Thirty-three percent (78 of 240) of the *Strix* detections in 2000 were Barred Owl detections, and 53% (113 of 215) of the *Strix* detections in 2001 (when detecting Barred Owls was emphasized) were Barred Owl detections. The minimum nearest-neighbor distances for Barred Owls and Spotted Owls were 1.1 km and 1.6 km, respectively.

**Hybrids.** We detected two hybrids between Spotted and Barred owls. The first, which had a distinctive five-note call, was detected once in 1994 and not afterward. The second hybrid, which had a distinctive six-note call, was audibly detected annually from 1995–2000.

**Site Characteristics.** Spotted and Barred owl sites did not differ relative to the amount of different forest-age classes within core-plots (Table 2). On

average, core-circles of both species contained more forest ≥180 yr old and less forest 50–79 yr old than random plots. Spotted Owl plots also contained more 130–179-yr-old forest and 0–49-yr-old forest than random plots.

Number of Spotted Owl site-centers in each forest-age group differed from Barred Owls ( $\chi^2_4 = 9.61$ ,  $P < 0.05$ ) and random ( $\chi^2_4 = 130.31$ ,  $P < 0.001$ ), and that of Barred Owl site-centers differed from random ( $\chi^2_4 = 61.56$ ,  $P < 0.001$ ; Table 3). Mean age of forest stand at site-centers was significantly greater ( $U = 5844.5$ ,  $P < 0.02$ ) for Spotted Owls ( $254.7 \pm 76.5$  yr) than for Barred Owls ( $228.3 \pm 101.5$  yr). Thirty-three percent of forest in the study area was comprised of stands ≥180 yr old, and 28% of random site-centers were in these old-forest stands. However, 82% of Spotted Owl and 68% of Barred Owl site-centers were in these old-forest stands (Table 3).

**Site Occupancy.** Of the 129 Spotted Owl sites with ≥10 surveys during the last 5 yr of the study, 25 (19.4%) apparently were unoccupied by Spotted Owls by 2001. There were significantly more

Table 4. Mean ( $\pm$ SD) elevation (m) and slope (%) of occupied and unoccupied Spotted Owl site-centers and Barred Owl site-centers in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.<sup>a</sup>

	OCCUPIED SPOTTED OWL (N = 104)	UNOCCUPIED SPOTTED OWL (N = 25)	BARRED OWL (N = 98)
Elevation	982.9 $\pm$ 143.9 <sup>b</sup>	886.4 $\pm$ 291.8	812.5 $\pm$ 275.9 <sup>d</sup>
Slope	57.1 $\pm$ 12.1 <sup>b,c</sup>	43.0 $\pm$ 19.2 <sup>d</sup>	39.2 $\pm$ 18.6 <sup>d</sup>

<sup>a</sup> All significant Mann-Whitney  $P$ -values  $< 0.001$ .

<sup>b</sup> Mean differed from Barred Owl site-centers.

<sup>c</sup> Mean differed from unoccupied Spotted Owl site-centers.

<sup>d</sup> Mean differed from occupied Spotted Owl site-centers.

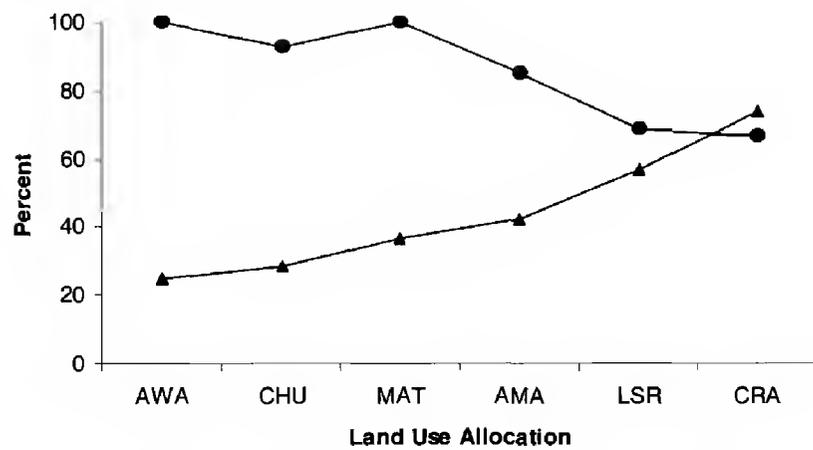


Figure 4. The percent of Spotted Owl sites that were occupied (circles) compared to the percent of Barred Owl sites relative to all occupied *Strix* sites (triangles) by Land Use Allocation. Timber harvest was permitted in critical habitat units (CHU), matrix (MAT), and the adaptive management area (AMA), and was not permitted in administratively-withdrawn areas (AWA), late-successional reserves (LSR), and congressionally-reserved areas (CRA).

Barred Owl site-centers in: (1) unoccupied ( $0.44 \pm 0.51$ ) than occupied ( $0.14 \pm 0.35$ ) Spotted Owl circles of 0.8-km radius ( $U = 915.5$ ,  $P = 0.001$ ), (2) unoccupied ( $0.84 \pm 0.99$ ) than occupied ( $0.44 \pm 0.69$ ) Spotted Owl circles of 1.6-km radius ( $U = 1016.0$ ,  $P = 0.049$ ), and (3) unoccupied ( $2.40 \pm 1.83$ ) than occupied ( $1.32 \pm 1.38$ ) Spotted Owl home-range circles of 2.9-km radius ( $U = 845.5$ ,  $P = 0.005$ ).

Occupied Spotted Owl sites were significantly steeper in slope and were significantly higher in elevation than Barred Owl sites (Table 4). Compared to unoccupied Spotted Owl sites, occupied Spotted Owl sites were also significantly steeper in slope, but were not significantly different in elevation. Unoccupied Spotted Owl sites were not significantly different than Barred Owl sites in slope or elevation.

The number of ha of forest  $\geq 80$  yr old was not significantly different in occupied ( $130.1 \pm 36.3$  ha) vs. unoccupied ( $123.4 \pm 37.8$  ha) ( $U = 1445.0$ ,  $P = 0.388$ ) Spotted Owl core-plots. However, when considering core-plots in which some forest harvest had taken place from 1978–2001 ( $20.6 \pm 16.2$  ha, range = 0.1–72.6 ha), occupied sites ( $125.5 \pm 31.4$  ha,  $N = 83$ ) had significantly more forest  $\geq 80$  yr old than unoccupied sites ( $109.9 \pm 28.6$  ha,  $N = 19$ ) ( $U = 1022.0$ ,  $P = 0.045$ ).

The percent of Spotted Owl sites that were occupied in each LUA was inversely proportional to the percent of Barred Owl sites relative to all occupied *Strix* sites ( $r_s = -0.899$ ,  $P < 0.01$ ,  $N = 6$ ;

Table 5. Reserve areas and timber harvest relative to number of occupied Spotted Owl sites ( $N = 104$ ), Barred Owl sites, and all Barred Owl (including potential Barred Owl sites) sites ( $N = 174$ ) in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.

	TIMBER MANAGEMENT			
	CRIT-ICAL HABITAT	LATE SUCCESSIONAL RESERVE	NO HARVEST <sup>a</sup>	HARVEST PERMITTED <sup>b</sup>
Spotted Owl sites	57	35	44	60
Barred Owl sites	55	47	64	34
All Barred Owl sites	89	73	102	72

<sup>a</sup> Includes late-successional reserve, congressionally-reserved areas, and administratively-withdrawn areas.

<sup>b</sup> Includes matrix and adaptive management area.

Fig. 4). In areas specifically allocated to benefit Spotted Owls (critical habitat units and late-successional reserves), the number of Barred Owl sites approached or surpassed the number of occupied Spotted Owl sites (Table 5). The numbers of occupied Spotted Owl sites and Barred Owl sites differed between areas in which timber harvest was permitted and was not permitted ( $\chi^2_1 = 10.77$ ,  $P < 0.005$ ). There were more Spotted Owl sites than Barred Owl sites in areas with timber harvest and fewer Spotted Owl sites than Barred Owl sites in areas without timber harvest. When including potential Barred Owl sites, there were more than twice as many Barred Owl sites than Spotted Owl sites in areas without timber harvest, whereas in areas with timber harvest, there were slightly more Barred Owl sites than Spotted Owl sites (Table 5).

#### DISCUSSION

**Distribution and Numbers.** Within the study area, more gradual slopes were along valley-bottoms and, in some cases, higher-elevation plateaus. Barred Owls generally appeared to favor the less-steep valley bottoms, while Spotted Owls were found more often in the steeper upland areas, a distinction also noted by Herter and Hicks (2000). Our analysis did not show that aspect had any influence on site selection by Spotted or Barred owls. Although studies in comparatively warmer regions of California indicated that Spotted Owls may select roosts or nests on north aspects during summer (Barrows 1981, North et al. 2000), studies in the more mesic conditions typical of western

Table 6. Mean ( $\pm$ SD) area (ha) of five forest-age classes in 0.8-km radius plots (201 ha) around 1946 Spotted Owl, 2001 Spotted Owl, 1946 random, and 2001 random site-centers in the Cowlitz Valley Ranger District, Gifford Pinchot National Forest, Washington.<sup>a</sup>

FOREST-AGE CLASS (yrs)	1946 SPOTTED OWL (N = 145)	2001 SPOTTED OWL (N = 145)	1946 RANDOM (N = 500)	2001 RANDOM (N = 500)
0–49	22.4 $\pm$ 39.3 <sup>c,d,e</sup>	47.9 $\pm$ 32.1 <sup>b,e</sup>	46.8 $\pm$ 63.9 <sup>b,e</sup>	43.5 $\pm$ 42.8 <sup>b,c,d</sup>
50–79	18.9 $\pm$ 38.2	10.8 $\pm$ 26.3 <sup>e</sup>	21.7 $\pm$ 45.9 <sup>e</sup>	8.8 $\pm$ 52.4 <sup>c,d</sup>
80–129	17.5 $\pm$ 33.0 <sup>c,d</sup>	28.8 $\pm$ 42.4 <sup>b</sup>	10.3 $\pm$ 24.5 <sup>b,e</sup>	35.0 $\pm$ 51.2 <sup>d</sup>
130–179	22.0 $\pm$ 43.5	17.2 $\pm$ 31.2 <sup>e</sup>	16.2 $\pm$ 34.6	10.7 $\pm$ 24.7 <sup>c</sup>
$\geq$ 180	106.1 $\pm$ 59.6 <sup>c,d</sup>	82.4 $\pm$ 44.6 <sup>b,e</sup>	80.2 $\pm$ 70.5 <sup>b</sup>	57.1 $\pm$ 49.6 <sup>c,d</sup>

<sup>a</sup> Significant Mann-Whitney *P*-values ranged from  $P < 0.020$  to  $P < 0.001$ .

<sup>b</sup> Mean differed from 1946 Spotted Owl core-plots.

<sup>c</sup> Mean differed from 2001 Spotted Owl core-plots.

<sup>d</sup> Mean differed from 1946 random core-plots.

<sup>e</sup> Mean differed from 2001 random core-plots.

Oregon and Washington indicated little selection for aspect by Spotted Owls (Forsman et al. 1984, this study).

At the northern edge of the range of the Spotted Owl, surveys suggested that there were four times as many Barred Owl sites than Spotted Owl sites both in British Columbia during the late 1980s (Dunbar et al. 1991) and in North Cascades National Park, Washington, during 1993–96 (R. Kuntz and R. Christopherson unpubl. data). Hamer et al. (1989) found that Barred Owls were twice as abundant as Spotted Owls in the northern Cascade Mountains east of Mt. Baker in the late 1980s. Barred Owls were almost as numerous as Spotted Owls in the mid 1990s just north of Mt. Rainier (Herter and Hicks 2000) and, in our study area, it appeared that they were at least as numerous as Spotted Owls in 2001.

Our shorter minimum nearest-neighbor distance for Barred Owls vs. Spotted Owls, which also was found by Hamer et al. (1989), could be a result of the wider variety of prey that Barred Owls use (Hamer et al. 2001), thereby allowing Barred Owls to “pack” closer together. The shortest nearest-neighbor distance for Spotted Owls (1.6 km) in Forsman et al. (1984) was equal to our shortest nearest-neighbor distance.

**Hybrids.** Apparently, Spotted  $\times$  Barred owl hybrids are rare. Kelly (2001) gathered reports of visual sightings of only 24 adult and 26 juvenile Spotted  $\times$  Barred owl hybrids in Washington and Oregon from 1974–99.

**Site Characteristics.** Our data suggest that Spotted Owl site-centers were more often in older for-

est than Barred Owl and random site-centers. Also, territories of Spotted and Barred owls tended to be centered in areas characterized by higher concentrations of old forest and lower concentrations of 50–79-yr-old forest than were available at random. These findings were in agreement with most previous studies of Spotted Owls (Bart and Forsman 1992, Hunter et al. 1995, Swindle et al. 1999), but the similarity of forest age in Spotted and Barred owl core-plots was a somewhat surprising result. Barred Owls have been thought to be more habitat generalists than Spotted Owls (Hamer et al. 1989, Herter and Hicks 2000, Hamer et al. 2001), and seem to use much more varied habitats throughout the Pacific Northwest (Smith et al. 1997, Kelly and Forsman 2003). However, in the eastern United States and eastern Canada, Barred Owls are considered an old-growth forest species (Dunstan and Sample 1972, Devereux and Mosher 1984, Elody and Sloan 1985, Laidig and Dobkin 1995) and, in some areas, Barred Owls do not breed in or defend territories in younger forests (Haney 1997). In Manitoba, Barred Owl plots contained fewer clearcut, burned, and young forest areas than random plots, and were associated more with areas of high crown-closure than random plots (Hinam and Duncan 2002). Because Barred Owls use forests that are similar to those used by Spotted Owls in our study area, this may make it difficult to manage forests to benefit Spotted Owls over Barred Owls.

Spotted Owl core-plots in our study area contained more old forest ( $\geq$ 130 yr) than random sites, but contained less forest 50–79 yr old than

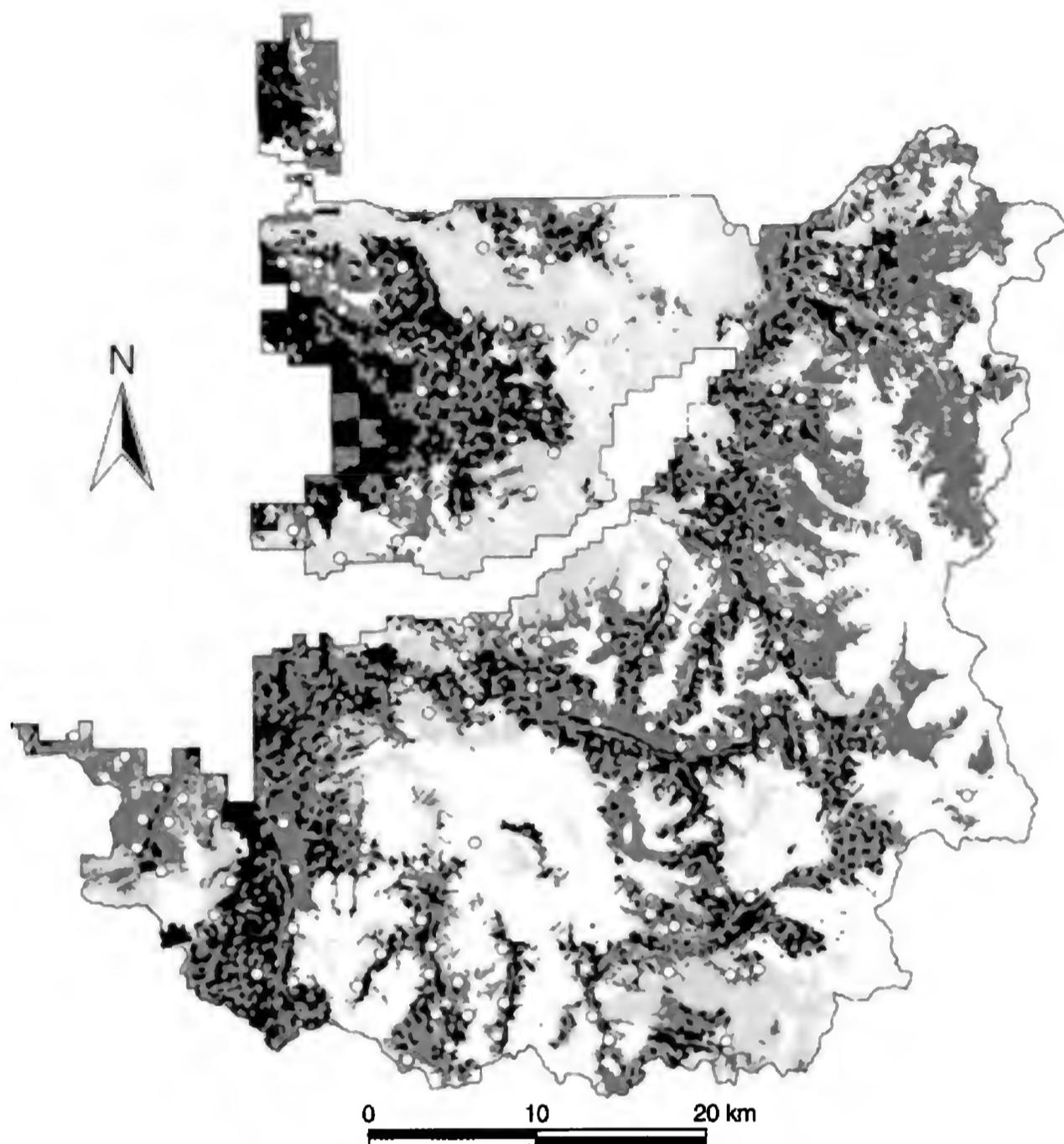


Figure 5. Forest  $\geq 80$  yr old in 1946 that was lost to forest harvest by 2001 (black areas), forest  $\geq 80$  yr old that remained in 2001 (dark gray areas), forest that became 80 yr old from 1946–2001 (light gray areas), and the 145 Spotted Owl site-centers (circles).

random. This finding, coupled with results from other studies that Spotted Owls are an old-forest associated species (USDA and USDI 1994a, 1994b), could lead to an expectation that Spotted Owl core-plots should contain less forest 0–49 yr old than random areas. However, Spotted Owl core-plots in our study area contained significantly more ha of 0–49-yr-old forest than random plots. To further examine this counter-intuitive result, we adjusted stand age to approximate the forest in the study area before any forest harvest took place (in 1946) to compare the forest-age distribution in Spotted Owl and random sites in 1946 vs. 2001 (Table 6). Spotted Owl core-plots in 1946 contained significantly less 0–49-yr-old forest than 1946 random sites, with a mean less than one-half that of

random (22.8 vs. 46.8 ha). In 1946, the 0–49-yr-old forest was consolidated into large, contiguous areas caused by forest fires in the early 1900s, while in 2001, 0–49-yr-old forest, the result of thousands of clearcuts, was widely distributed and was intermixed with older forest. The distribution of Spotted Owl sites in 2001 mirrored very closely the distribution of forest  $\geq 80$  yr old in 1946, excluding the areas lost to forest harvest. Spotted Owls appeared to have persisted in areas of older forest. Other than a few instances, Spotted Owls had yet to repopulate the areas that had grown into forest  $\geq 80$  yr old by 2001 (Fig. 5).

Spotted Owl core-plots in our study area included more forest  $\geq 130$  yr old than random, but did not contain more forest 80–129 yr old than ran-

dom. However, forest practices reduced the amount of forest  $\geq 130$  yr old by 25 921 ha and increased the amount of forest 80–129 yr old by 22 672 ha. Replacing stands  $\geq 130$  yr old with stands 80–129 yr old could negatively affect Spotted Owls.

Our results were similar to previous studies in that Barred Owls were most abundant in lowland forests and less common in upland areas (Laidig and Dobkin 1995, Haney 1997, Mazur et al. 1997, Hamer 1988, Mazur et al. 1998). However, these owls also occurred in upland areas, as has been reported in previous studies (Boxall and Stepney 1982, Dunbar et al. 1991, Dark et al. 1998, Wright and Hayward 1998). Collectively, these results suggest that riparian zones and lowland forests were more suitable habitat for Barred Owls in our study area, and that upland forests were less likely to be occupied by Barred Owls.

The large standard deviations for elevation and slope (Table 1) indicated considerable variation among sites occupied by both Barred Owls and Spotted Owls. For example, there were four cases where Spotted Owl core-plots were dominated by 50–79-yr-old forest (53–96% cover), with only small areas of old forest or remnant older trees remaining.

**Site Occupancy.** When we analyzed only those core-plots in which some timber harvest had taken place since 1978, unoccupied Spotted Owl sites had significantly less forest  $\geq 80$  yr old remaining than occupied sites, indicating that forest harvest contributed to these sites becoming unoccupied. Occupancy of Spotted Owl sites, relative to (1) number of Barred Owl sites within Spotted Owl plots, (2) slope of Spotted Owl sites, and (3) elevation of Spotted Owl sites could be directly or indirectly traced to the presence of Barred Owls. There were significantly more Barred Owl sites within 0.8-km, 1.6-km, and 2.9-km radius circles centered on Spotted Owl site-centers in unoccupied Spotted Owl sites than in occupied Spotted Owl sites. Occupied Spotted Owl sites were on significantly steeper slopes and were significantly higher in elevation than Barred Owl sites, whereas unoccupied Spotted Owl sites were not significantly different than Barred Owl sites in slope or elevation.

Our results suggest that Spotted Owls are more likely to abandon a site if Barred Owls take up residence close to that site. Similarly, Kelly et al. (2003) found that occupancy of Spotted Owl sites

declined after Barred Owls were detected within 0.8 km, but occupancy was not affected when Barred Owls were located  $>0.8$  km from Spotted Owl site-centers. We suggest that a combination of habitat lost due to timber harvest and the presence of Barred Owls may work synergistically to put Spotted Owl pairs at risk of losing their territories.

Sites in the eastern Washington Cascade Mountains, unoccupied for 4 ( $N = 2$ ), 6 ( $N = 3$ ), and 8 ( $N = 1$ ) yr were reoccupied by Spotted Owls; the marked, replacement Spotted Owls in these cases tended to use the same core areas and, in some sites, even nested in the same nest tree used by previous owls (T. Fleming pers. comm.). Therefore, some of our unoccupied sites may become reoccupied in the future.

**Spotted Owl Conservation.** Existing metapopulation conservation strategies of the northern Spotted Owl are dependent upon Spotted Owls surviving and reproducing in reserves, and being able to move between reserves via hospitable habitats to facilitate genetic interchange and, if necessary, recolonization (Levins 1968, USDA and USDI 1994a, Caughley and Gunn 1996, Gutiérrez and Harrison 1996, Noon and McKelvey 1996). Our data suggested that Barred Owls caused a reduction in the Spotted Owl population by physically excluding them from historic territories and making those territories unavailable for recolonization, as was suggested by Dunbar et al. (1991), Dark et al. (1998), and Kelly (2001). There is an increasing body of evidence that Barred Owls physically attack Spotted Owls (E. Forsman, J. Mowdy, T. Snetsinger, and G. Stagner pers. comm.), and sometimes may kill them (Leskiw and Gutiérrez 1998). Aside from direct competition for space, it is also likely that Barred and Spotted owls compete for prey (Hamer et al. 2001). In addition, Barred Owls may negatively affect dispersing, juvenile Spotted Owls by creating a hostile environment that inhibits the occupation of vacated Spotted Owl territories and other suitable areas. In our study area, presence of Barred Owls had a greater effect on Spotted Owl site occupancy than did the status of the area as a Spotted Owl reserve. In fact, our data suggest that Barred Owls were more numerous in the reserve areas than Spotted Owls. Continued loss of old-growth and mature forest may reduce the ability of Spotted Owls to persist in the presence of Barred Owls.

We recommend continuing the long-term Spotted Owl demography studies (Franklin et al. 1999)

to track occupancy and reproduction of northern Spotted Owls range-wide. Also, this work should include surveys for Barred Owls to determine if the presence of this species affects Spotted Owl occupancy and reproduction.

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# TEMPORAL AND SPATIAL STABILITY OF RED-TAILED HAWK TERRITORIES IN THE LUQUILLO EXPERIMENTAL FOREST, PUERTO RICO

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**ABSTRACT.**—We mapped Red-tailed Hawk (*Buteo jamaicensis*) territories in the Luquillo Experimental Forest (LEF) of Puerto Rico in 1998. We combined our 1998 data with that collected during previous studies of Red-tailed Hawks in the LEF to examine population numbers and spatial stability of territorial boundaries over a 26-yr period. We also investigated potential relationships between Red-tailed Hawk territory sizes and topographic and climatic factors. Mean size of 16 defended territories during 1998 was  $124.3 \pm 12.0$  ha, which was not significantly different from our calculations of mean territory sizes derived from data collected in 1974 and 1984. Aspect and slope influenced territory size with the smallest territories having high slope and easterly aspects. Territory size was small compared to that reported for other parts of the species' range. In addition, there was remarkably little temporal change in the spatial distribution, area, and boundaries of Red-tailed Hawk territories among the study periods. Further, there was substantial boundary overlap (21–27%) between defended territories among the different study periods. The temporal stability of the spatial distribution of Red-tailed Hawk territories in the study area leads us to believe the area might be at or near saturation.

**KEY WORDS:** *Buteo jamaicensis; Red-tailed Hawk; Puerto Rico; spatial stability; temporal stability; territorial characteristics.*

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## ESTABILIDAD TEMPORAL Y ESPACIAL DE LOS TERRITORIOS DEL GAVILAN DE COLA ROJA EN EL BOSQUE EXPERIMENTAL DE LUQUILLO, PUERTO RICO.

**RESUMEN.**—Hicimos mapas de los territorios del gavilán de cola roja (*Buteo jamaicensis*) en el Bosque Experimental de Luquillo (BEL) en Puerto Rico en 1998. Combinamos nuestros datos de 1998 con aquellos colectados durante estudios previos con gavilanes de cola roja en el BEL para examinar los números de la población y la estabilidad espacial de los límites de los territorios en un periodo de 26 años. Además investigamos las posibles relaciones entre los tamaños de los territorios del gavilán y factores topográficos y climáticos. El tamaño medio de los 16 territorios defendidos durante 1998 fue  $124.3 \pm 12.0$  ha, lo cual no fue significativamente diferente de nuestros cálculos del tamaño medio de los territorios a partir de datos colectados en 1974 y 1984. La cara y la pendiente influyeron en el tamaño del territorio, teniendo los territorios más pequeños altas pendientes y caras orientales. El tamaño del territorio fue pequeño comparado con los reportados para otras partes dentro del rango de la especie. Adicionalmente, hubo notablemente poco cambio temporal en la distribución espacial, área y límites de los territorios del gavilán de cola roja entre los periodos estudiados. Además hubo un traslape substancial en los límites (21–27%) en los territorios defendidos entre los diferentes perio-

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dos estudiados. La estabilidad temporal en la distribución espacial de los territorios del gavián de cola roja en el área de estudio no lleva a creer que el área puede estar en o cerca de su saturación.

[Traducción de César Márquez]

The Red-tailed Hawk (*Buteo jamaicensis*) is possibly the most ubiquitous raptor in North America, with a distribution from Alaska and northern Canada south through most of Mexico and into Central America, and east across the West Indies (Snyder and Snyder 1991). It occupies a diversity of ecotypes, but is typically considered a raptor of open country characterized by interspersed woodlots (Preston and Beane 1993). Red-tailed Hawks are primarily sit-and-wait predators, scanning open areas from elevated perch sites (Preston and Beane 1993). However, these hawks also occupy tropical forest areas (Snyder and Snyder 1991, Preston and Beane 1993). The Red-tailed Hawk population in the Luquillo Experimental Forest (LEF) of Puerto Rico is one of the densest (1.6 pairs/km<sup>2</sup>) ever reported for the species (Snyder and Snyder 1991, Preston and Beane 1993). Furthermore, Santana and Temple (1988) found the density of Red-tailed Hawks was greatest in the mountainous rain forest of the LEF and lowest in the open-country lowlands. This is intriguing because the LEF is a closed-canopy forest and quite atypical from the usual open and mixed vegetation ecotypes commonly occupied by Red-tailed Hawks (Howell et al. 1978, Preston and Beane 1993).

We mapped Red-tailed Hawk territories in the LEF study area in 1998. Here we combine our data with that collected during previous studies of these species in the LEF from 1972–75 (Snyder et al. 1987; hereafter 1970s study) and from 1981–83 (Santana and Temple 1988; hereafter 1980s study) to examine population stability and the spatial stability of territorial boundaries across three decades. We also explore potential relationships of topographic and climatic factors to territory sizes in the study area.

#### STUDY AREA

Our study area was an irregularly shaped 32-km<sup>2</sup> area within the Luquillo Experimental Forest (Snyder et al. 1987). The LEF is a subtropical rain forest in the Luquillo Mountains, located at the east end of the island of Puerto Rico (18°N, 65°W), and is part of the Caribbean National Forest in the U.S. Department of Agriculture National Forest system. The study area ranges from 300–1050 m in elevation and includes “tabonuco forest” (subtropical wet forest life zone), “colorado forest” (lower montane wet forest life zone), and “cloud forest” (lower montane rain forest life zone) (Ewel and Whitmore

1973). In general, as elevation increases, mean tree height, diameter, basal area, and species richness decreases, while stem density increases (Brown et al. 1983). Annually, the study area receives more than 1000 rain showers, with rainfall ranging from 354–485 cm (García-Martino et al. 1996). Temperatures at the highest elevations range from 17–20°C (García-Martino et al. 1996).

#### METHODS

A territory is usually considered as that area in which a resident individual or pair excludes conspecifics (Powell 2000). Red-tailed Hawks in the LEF defend their territories year-round, attacking any conspecifics that stray into their territory (Snyder et al. 1987). Observations of intraspecific encounters between neighboring Red-tailed Hawks allow mapping of territory boundaries (Janes 1984, Santana and Temple 1984, Snyder et al. 1987).

Intraspecific territorial encounters typically involve the resident Red-tailed Hawk diving from a soar or initiating a direct flapping chase. Intruders are pursued by one or both members of the resident pair to a boundary of the territory, at which point the intruder stops fleeing and both resident and intruder will circle up near each other while vocalizing. The interaction ends when both resident and intruder retreat into their respective territories, often with one individual of a pair flying above and to the rear of its mate, after which the pairs drop down into their territories, presumably near the nest area. The sequence of soaring residents chasing intruding neighbors to a boundary, then circling and screaming until both sets of hawks return to the relative centers of their territories may be repeated every few hours throughout the day with several neighboring conspecifics. The chases consistently stop at the same locations, which we interpreted as the territorial boundaries.

The hunting behaviors of Red-tailed Hawks in the LEF also facilitate territory mapping. The almost constant updrafts, created by a northeasterly laminar flow of air against the Luquillo Mountains, allow Red-tailed Hawks to expend little energy while hunting in flight (Snyder and Snyder 1991). Thus, the majority of hunting occurs while in flight rather than from a perch (Snyder et al. 1987, Santana and Temple 1988).

We replicated earlier studies of Red-tailed Hawks in the LEF (Snyder et al. 1987, Santana and Temple 1988) by making visual observations of hunting Red-tailed Hawks and their interactions with conspecific neighbors and intruders from 5–20 January 1998, which is the onset of the nesting period in the study area (Snyder et al. 1987). We made our observations from many of the same overlooks used by Snyder et al. (1987) and Santana and Temple (1988). These consisted primarily of clear-viewing locations along roads, the top of rocks, slides, hillsides and escarpments, and the Mt. Britton and Yokahú lookout towers. Some of the previously-used road overlooks were no longer suitable due to vegetation changes. Fog precluded use of some of the high-elevation sites on some days, so we made the sampling from these lookouts a

priority when good viewing conditions prevailed. We began observations at about 0800 H and continued until 1700 H unless fog or rain developed. We recorded data directly onto photocopies of a 1981 1:20 000 U.S. Forest Service topographic map with 10-m contour intervals. We used the map copies to record observed flight lines for individual hawks, locations of perch points, copulation sites, nest sites, and locations of intraspecific interactions relative to obvious landmarks. We numbered observations sequentially and recorded observation times, vocalizations, details of individual birds' molts and markings, and the details of intraspecific interactions observed. We used fresh map copies each day and used additional map photocopies during the day as needed to maintain clarity of the records.

Due to the amount of time they spent in flight, hawks were easiest to detect initially when soaring aloft or when sunlight reflected off their wings as they circled. Hawks flying against the background of forest canopy were harder to locate and track visually. When we identified regularly-used perches, we subsequently could often locate perched hawks and observe them as they moved within their defended area. Once an individual hawk was sighted, we visually tracked the bird until it was lost from sight in clouds, the forest canopy, or behind topographical features. Hawks were in view for periods ranging from several minutes to an hour or more. When interactions between two or more Red-tailed Hawks occurred, each observer chose an individual or a pair to follow exclusively. Thus, it was advantageous to have multiple observers.

We easily plotted locations of perched hawks and points where hawks dove under the canopy. Apart from brief and intermittent cloud buildup and subsequent rain showers, sunlight was always available and we estimated locations of hawks flying within ca. 10 m of the forest canopy using the shadow they cast on the canopy. Errors of a few meters due to sun angle likely had only minor influence on our estimation of hawk locations. Observations were facilitated by the topography of the study area; we could accurately gauge a hawk's position and movements relative to cliff faces, drainage bottoms, and other topographic or landmark features of known size, elevation, and distance from the observation point. Finally, two observers frequently recorded data independently from different observation points. The resulting estimations of hawk locations were then compared between the two observers to verify the location plotted and improve the accuracy. In these ways, we were able to develop a sense of hawk size at different distances and make estimations of their location relative to the ground when they were soaring. Although some error undoubtedly occurred, we used methods identical to those used during the 1970s (Snyder et al. 1987) and 1980s (Santana and Temple 1988), so estimates should have equivalent bias and comparisons among study periods are valid. Based on the fact that we estimated visual locations against a topographically complex background at distances of only a few hundred meters, we suggest that our error is not substantially different from, and may be less than, error associated with radiotelemetry triangulations.

We used location points, flight lines, and behavioral observations to map the estimated area of each pair's

territory. We used ArcView 3.2 (Environmental Systems Research Institute, Inc.) to plot outermost location points for members of each Red-tailed Hawk pair onto a digital map of the study area. We then employed the ArcView Animal Movements extension (Hooge and Eichenlaub 1997) to calculate a 100% minimum convex polygon (MCP) territory size estimate for each pair. In the same manner, we created MCP home range size estimates of 11 Red-tailed Hawk territories mapped during the Snyder et al. study (H. Snyder unpubl. data) and of the 16 Red-tailed Hawk territories mapped by Santana and Temple (1984) to compare with our data.

We report means and standard errors for our calculations of territory sizes and overlap during each study period. We calculated the mean size for each territory that was used and also the proportion of the study area that comprised territories in all three periods. We used an analysis of variance to examine changes in territory sizes among the study intervals and an analysis of variance on ranks to examine changes in territorial overlap for those territories that were occupied consistently during all three studies. We used a paired *t*-test to examine differences in territory size among territories detected in the 1980s study and 1998, but not those detected during the 1970s because the defined study area was smaller.

We used a U.S. Geological Survey 30-m Digital Elevation Model (DEM) to characterize topographic features of each home range. Using ArcView Spatial Analyst (Environmental Systems Research Institute, Inc.), we derived slope and aspect grids for the study area. These grids were used to calculate mean elevation, slope, and aspect for each home range. Mean aspect was the circular mean (Fisher 1993). We also created a 10-m contour map of the study area. Using the ArcView XTools extension (Oregon Department of Forestry), we calculated the lengths of each 10-m elevation contour within each home range. We obtained a measure of topographic complexity by dividing length of contours by home range size. Our measure of topographic complexity was analogous to the Land Surface Ruggedness Index (LSRI) proposed by Beasom et al. (1983).

We used Pearson correlations and multiple regression to examine the relationship between elevation, slope, aspect, topographic complexity, and territory size. We used a Kruskal-Wallis test to determine if years could be pooled for further analysis. We tested for differences in mean aspect between years using Watson's  $Y_r$  test (Fisher 1993). All linear variables were examined for normality. Variables that were not normally distributed were transformed using natural log transformations. Principal components analysis (PCA) was used as a variable reduction method if independent variables were highly correlated. Several linear regression models were created. Mean aspect, a circular variable, was treated as a linear model with cosine (aspect) and sine (aspect) as the independent variables (Fisher 1993). For all regressions, territory size was considered the response variable. We used Akaike's Information Criterion (AIC) to select the best regression model (Burnham and Anderson 1998). Because we suspected that different features of the topography might have influenced territory size at different elevations, territories were divided into high and low elevation groups using 700 m as an arbitrary breakpoint to divide

the sample approximately in half. Regression models were reanalyzed in the manner described above using these two groups. All statistical analyses were conducted using SPSS for Windows 10.0.7 (SPSS, Inc.).

## RESULTS

Our results were consistent with those from the previous studies in terms of both number of territorial pairs and the individual territorial boundaries. We obtained adequate territory data for 16 pairs of Red-tailed Hawks in the study area. Mean size of defended Red-tailed Hawk territories during our study was  $124.3 \pm 12.0$  ha (i.e., 1 pair/1.24 km<sup>2</sup>). This was similar to our calculations of mean territory sizes of  $139.0 \pm 15.0$  ha for 11 pairs from the 1970s study (Snyder unpubl. data) and  $153.7 \pm 15.8$  ha for 16 pairs from the 1980s study (Santana and Temple 1984). Assuming territory size related to breeding density, the density of breeding pairs of Red-tailed Hawks in the LEF was at the high end of the range of nesting densities reported for the species in 10 different studies (1 breeding pair/1.3–24.9 km<sup>2</sup>; McGovern and McNurney 1986). In addition, there was substantial boundary overlap between defended territories. Territorial incursions by neighboring Red-tailed Hawks resulted in territory overlap that was greatest in the 1970s study ( $\bar{x} = 27.3 \pm 8.8\%$ ), lowest in the 1980s study ( $\bar{x} = 21.3 \pm 3.3\%$ ), and in between the two during our 1998 study ( $\bar{x} = 24.3 \pm 4.2\%$ ), but these were not significant differences ( $H_2 = 0.022$ ,  $P = 0.989$ ).

In general, there was remarkably little temporal change in the spatial distribution, area, and boundaries of Red-tailed Hawks among the study periods (Figs. 1–3). Some territories, such as A and B, appeared to show slight increases and decreases among study periods (see Figs.). Other territories appeared to have disappeared altogether, but were replaced by new territories nearby. For example, territory I of the 1970s study (Fig. 1) may have shifted to become territory Q in the 1980s study (Fig. 2), but neither territory was occupied in 1998 (Fig. 3). Likewise, territory K in the 1970s study (Fig. 1) and K in 1998 (Fig. 3) may have also been R or P in the 1980s study (Fig. 2).

Nine territories (A, B, C, D, E, F, G, H, and J; Figs. 1–3) that were detected in all three study periods appeared to have been relatively consistent spatially. Although there were temporal fluctuations in the boundaries of these territories, over half of the area ( $\bar{x} = 55.3 \pm 2.9\%$ ) within each

territory was common to that territory in all three studies. Furthermore, the territories did not temporally differ in sizes ( $F_{2,24} = 0.726$ ,  $P = 0.494$ ).

There were some territories not detected or mapped during the 1970s study, due to a more restricted study area than in the 1980s study or our 1998 study. Three of these (L, M, and O) appeared spatially similar between the 1980s and 1998 (Figs. 2 and 3). Territory N in the 1980s study (Fig. 2) appeared to have dissolved into two territories (N and T) in 1998 (Fig. 3). When pooling T into N for 1998, there was no significant difference in size between the 1980s and 1998 territories at locations L, M, N, and O ( $t_3 = 0.065$ ,  $P = 0.952$ ).

None of the linear topographic variables differed among years (Elevation,  $\chi^2_2 = 2.384$ ,  $P = 0.304$ ; Slope,  $\chi^2_2 = 1.274$ ,  $P = 0.529$ ; Topographic Complexity,  $\chi^2_2 = 1.636$ ,  $P = 0.441$ ; Aspect,  $\chi^2_2 = 0.028$ ,  $P = 0.986$ ); therefore, we pooled all years (Table 1). Territory size was transformed using the natural logarithm. All linear variables except elevation were negatively correlated with territory size (Elevation,  $r = 0.041$ ,  $P = 0.793$ ; Slope,  $r = -0.401$ ,  $P = 0.008$ ; Topographic Complexity,  $r = -0.406$ ,  $P = 0.007$ ). Slope and topographic complexity were strongly correlated with each other ( $r = 0.994$ ,  $P < 0.001$ ); therefore, they could not be used as independent variables in the same regression model. A new variable was created based on component I of the PCA used to combine slope and topographic complexity. Because of the extremely low correlation of elevation and territory size, elevation was not used as an independent variable in any regression model.

Seven regression models were tested. Models 1–3 were single-variable linear regressions of slope, topographic complexity, and the PCA variable, respectively. Model 4 was a linear regression with cosine (aspect) and sine (aspect) as the independent variables. Models 5–7 were all possible pair-wise combinations of the four topographic variables.

Size of territory was not independent of slope and aspect. Territory size decreased as slope increased. Generally, the smallest territories had aspects facing southeast. The model with the best data fit predicts territory size using the formula:

$$\text{Territory size} = e^{5.6762 - 0.038 \text{ Slope} + 0.1395 \text{ Cosine (Aspect)} - 0.1404 \text{ Sine (Aspect)}}$$

The slope-aspect model received the smallest AIC value, and the best single-variable model was based on topographic complexity. The topographic vari-

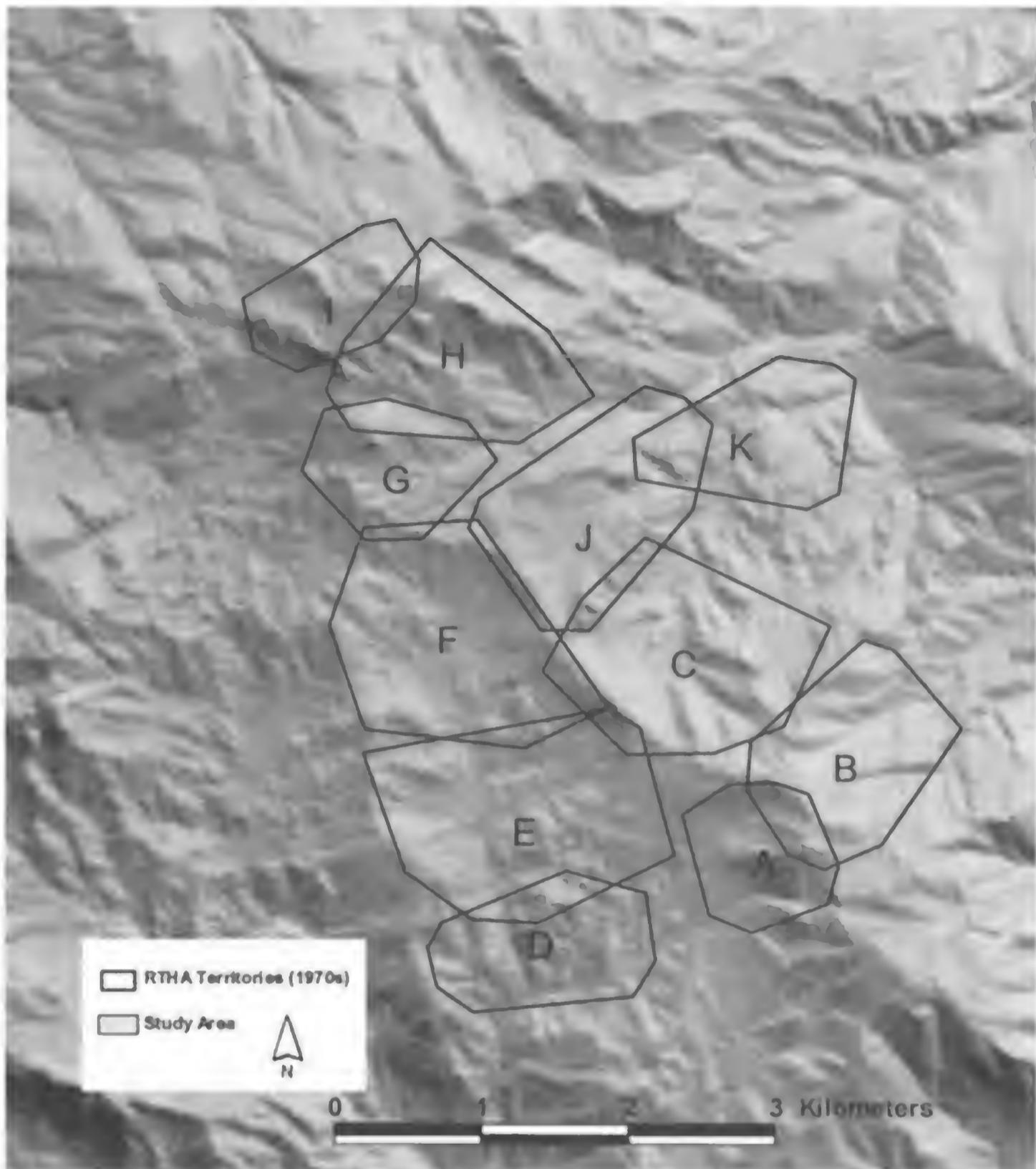


Figure 1. Estimated spatial distribution of Red-tailed Hawk (RTHA) territories in the Luquillo Experimental Forest, Puerto Rico, during the early 1970s (H. Snyder unpubl. data).

ables that influenced territory size varied by elevation. Low-elevation territories were most influenced by aspect and topographic complexity. High-elevation territories were most influenced by aspect.

#### DISCUSSION

We suspect the temporal and spatial stability of Red-tailed Hawk territories in the LEF might be related to the hawks' apparent use of permanent geographic features such as hilltops and ridgelines

to delineate territory boundaries and corners. These geographic features might serve an important function by funneling the updrafts from the tradewinds. This allows Red-tailed Hawks on the east and northeast sides of the highest peaks and ridgelines to soar or hover in the strong updrafts, from which they can hunt and defend any part of the territory with a minimum expenditure of energy. In addition, the relatively constant updrafts allow the Red-tailed Hawks to be on the wing from pre-dawn to twilight. This is especially true at high-

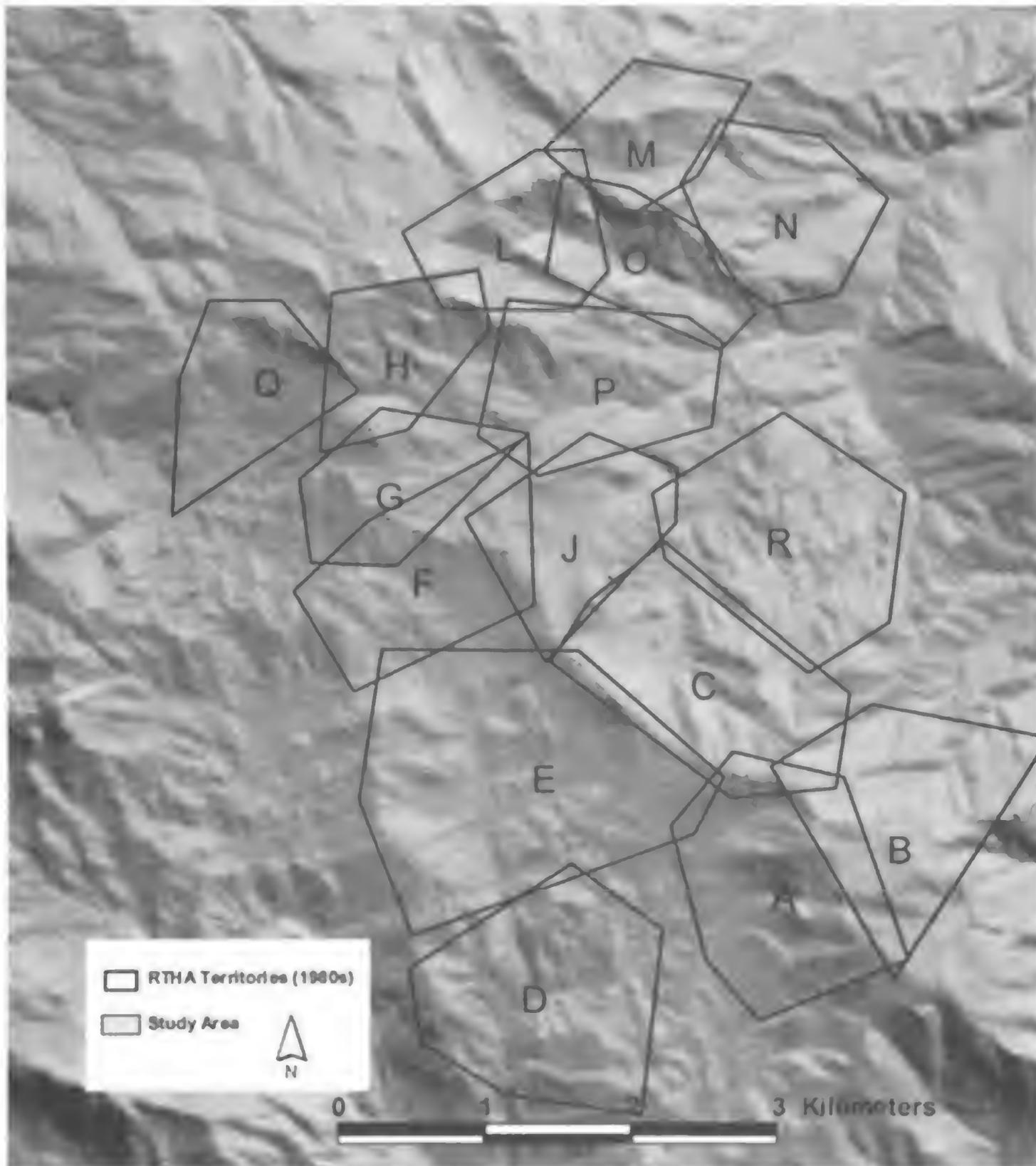


Figure 2. Estimated spatial distribution of Red-tailed Hawk (RTHA) territories in the Luquillo Experimental Forest, Puerto Rico, during the early 1980s, based on Santana and Temple (1984).

er elevations where the updrafts should be stronger. Thus, the hawks are not constrained by time of day, as is the case in areas where soaring is dependent upon thermal updrafts. For example, Red-tailed Hawks occupying areas on the lee side of ridges usually gained and maintained altitude by extended flapping and circling rather than soaring.

The role of topographic complexity in territory size is difficult to interpret. Increasing complexity

may create more regions within a given territory in which updrafts occur, resulting in more areas in which a hawk can efficiently soar. Topographic complexity may also indicate a greater surface area within a territory than would be estimated by the MCP estimate, which is based on a flat surface area. However, we found a very high correlation between slope and topographic complexity. This might indicate that the LSRI method (Beasom et al. 1983) of measuring topographic complexity is



Figure 3. Estimated spatial distribution of Red-tailed Hawk (RTHA) territories in the Luquillo Experimental Forest, Puerto Rico, during January 1998 (this study).

overly sensitive to slope, conveying little additional information about the area.

There appeared to be little variability in territorial boundaries and areas among the three studies, despite a passage of 26 yr and forest damage associated with Hurricane Hugo in 1989. The hurricane struck the LEF study area with 4 hr of winds reaching as high as 227 km/hr (Scatena and Larsen 1991). Resulting damage to the forest was deemed so severe that forest composition and dy-

namics were affected for several years (Brokaw and Grear 1991). Removal of canopy trees by the hurricane resulted in numerous openings in the forest canopy (Brokaw and Grear 1991, Scatena and Larsen 1991, Waide 1991). Openings in the canopy might have facilitated hunting by Red-tailed Hawks and, although there are no supportive data, might have resulted in an increase in Red-tailed Hawks in the LEF. If any increase occurred, however, it was likely short-lived, as the Red-tailed Hawk den-

Table 1. Mean elevation (m), slope and aspect (degrees), and topographic complexity (m/ha) of Red-tailed Hawk territories, Luquillo Experimental Forest, Puerto Rico. Data derived from 1972–75 (Snyder et al. 1987), 1981–83 (Santana and Temple 1984), and 1998 (this study) were pooled.

VARIABLE	MEAN	STAN-	MINI-	MAXI-
		DARD		
		ERROR	MUM	MUM
Elevation	671.1	23.2	300.9	956.0
Slope	19.5	0.6	12.8	28.2
Aspect	107.1	14.4	12.0	289.0
Topographic complexity	395.5	12.3	269.0	583.8

sity in 1998 was not different than the pre-hurricane density (Santana and Temple 1984, Snyder et al. 1987).

The 1970s study (Snyder et al. 1987) and 1980s study (Santana and Temple 1984) did not report the number of days of Red-tailed Hawk observation. However, it is obvious our study of 15 days was quite limited in duration compared to these earlier investigations. The differences we found in some territory boundaries might be attributable to the time constraints of our study compared to the earlier studies (Santana and Temple 1984, Snyder et al. 1987), and that our survey was conducted in January, which is at the start of the nesting cycle for resident Red-tailed Hawks and when they are presumably most aggressive in territorial defense (Snyder et al. 1987). However, there was no statistical difference in the territory sizes or boundary overlap among the study periods. Thus, we believe our data accurately depict territory sizes, boundaries, and Red-tailed Hawk density at the onset of the 1998 breeding season.

There appeared to be a number of individual Red-tailed Hawks that were unattached to a mate or territory passing through the study area and they traversed the defended territories by maintaining high altitudes. Territorial residents would occasionally soar upward, but remained below the intruder as it passed over the area. As the intruder approached a boundary, the adjacent resident Red-tailed Hawks would rise up toward the intruder. In this way, a high-altitude intruder would be “escorted” across the study area by each successive pair of resident hawks. These encounters appeared to be non-aggressive as long as the intruder maintained a high altitude and continued moving

across the area. Only when an intruder began descending or circling were chases initiated. None of the high-altitude intruders were observed diving into known territories. Rather, they invariably “vanished” into the distance after having passed over several occupied territories. Therefore, we assumed that these intruders were not, in fact, adjacent or distant neighbors who were cruising above the landscape. In addition to an unknown number of adult Red-tailed Hawks that flew over the territories, we observed at least two immature Red-tailed Hawks that appeared to be living inconspicuously among the territories in the study area. Immature Red-tailed Hawks were rarely observed soaring. Rather, they were observed moving low above the canopy of the forest, and they quickly dove out of sight into the canopy when detected and pursued by a resident.

Territorial behavior among birds can lead to generally even distribution of individuals within a population (Newton 1998). High site-fidelity among individuals in a population might lead to relatively long-term spatial stability of territory areas. Furthermore, stability of a population over several years suggests that densities might be limited by competition for territorial space (Newton 1991). Our data, combined with that of Santana and Temple (1984) and Snyder et al. (1987), suggest there was a high degree of temporal stability in the density and spatial distribution of Red-tailed Hawks in the LEF that may be linked to geographic features. Furthermore, the temporal stability of the spatial distribution of Red-tailed Hawk territories in the study area leads us to believe the area is at or near saturation.

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## WINTERING RAPTOR USE OF HYBRID POPLAR PLANTATIONS IN NORTHEASTERN OREGON

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**ABSTRACT.**—We studied wintering raptor use of hybrid poplar (*Populus* spp.) plantations in comparison to surrounding cover types in the Columbia Basin of northeastern Oregon. Diurnal raptors were surveyed in shrub-steppe, poplar plantations, and irrigated croplands. Logistic regression analyses suggested that the three most common raptors, Red-tailed Hawks (*Buteo jamaicensis*), American Kestrels (*Falco sparverius*), and Northern Harriers (*Circus cyaneus*) were associated with croplands, interiors of 1-yr-old plantations, and plantation edges. Shrub-steppe was also selected as a significant predictor of sites with American Kestrels. The best model for the Northern Harrier also included the interior of 2-yr-old plantations, but excluded croplands and edges of older plantations. Plantations and plantation edges appeared to be used by wintering raptors disproportionately to their availability. Our data suggest that maximizing plantation edges and managing for a variety of plantation ages within this landscape will likely provide suitable habitat for wintering raptors in this region.

**KEY WORDS:** *American Kestrel; Falco sparverius; Northern Harrier; Circus cyaneus; Red-tailed Hawk; Buteo jamaicensis; cropland; hybrid poplar; Populus spp.; shrub-steppe.*

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### USO DE PLANTACIONES DE ALAMO HÍBRIDO POR RAPACES DURANTE EL INVIERNO EN EL NORORIENTE DE OREGON

**RESUMEN.**—Estudiamos el uso por rapaces durante el invierno de plantaciones de álamo híbrido (*Populus* spp.) en comparación a los tipos de cobertura de los alrededores en la cuenca del Columbia en el nororiente de Oregon. Las rapaces diurnas fueron estudiadas en plantaciones, campos irrigados y hábitats de estepas arbustivas. El análisis de regresión logística sugiere que la mayoría de rapaces están asociadas con los campos, los interiores de las plantaciones de un año, y los bordes de las plantaciones. La estepa arbustiva fue seleccionada además como un pronosticador efectivo del cernícalo (*Falco sparverius*). El mejor modelo para el aguilucho norteño (*Circus cyaneus*) incluyó además los interiores de plantaciones de 2 años de edad, pero excluyó los campos y bordes de plantaciones más viejas. Las plantaciones parecieron ser un hábitat importante para rapaces invernantes, especialmente como sitios de percha. Nuestros datos sugieren que para la mayoría de especies, maximizar los bordes de las plantaciones y proveer una variedad de edades en la plantación dentro de este paisaje probablemente proporcionará el mejor hábitat para las rapaces durante el invierno.

[Traducción de César Márquez]

The development of short-rotation woody crop (SRWC) plantations in North America is becoming increasingly common with the forest-products industry due to the fast-growing nature of these trees. Nearly 80 000 ha of SRWC plantations have been established in the United States, and over 69 million ha have been identified as suitable for

SRWC plantations (Alig et al. 2000). Although a variety of tree species are grown in these plantations, hybrid poplars (*Populus* spp.) are the most commonly planted species in the Pacific Northwest. In the past 15 yr, over 28 000 ha of poplar plantations have been established in the Pacific Northwest to produce fiber for paper and dimensional lumber products (Heilman et al. 1995). Poplars are also used in conservation plantings, as buffers along riparian areas, and in phytoremediation projects across North America (Isebrands 2000).

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Little is known about the habitat value of these plantings for raptors. However, Bogliani (1988), Prater (1993), Bogliani et al. (1994), Sergio and Bogliani (2000) have examined bird use of poplar plantations in Europe. Bogliani et al. (1994) and Sergio and Bogliani (2000) reported that poplar plantations supported nesting Eurasian Hobby (*Falco subbuteo*) densities of up to one pair per 50 km<sup>2</sup>. Prater (1993) also commented that poplar plantations provided hunting habitat for the Barn Owl (*Tyto alba*) in Great Britain. Moreover, Schmitz (1986) demonstrated that Common Buzzards (*Buteo buteo*) used poplar plantations in Belgium, although at lower relative densities than other bird species.

Fewer studies have examined raptor use of poplar plantations in North America. Moser (2002) found that nine Northern Saw-whet Owl (*Aegolius acadicus*) pairs nested successfully in poplar plantations when 25 nest boxes were provided. Christian et al. (1997) studied bird use of poplar plantations in the Midwest in comparison to bird use of surrounding croplands and natural forests. They reported Red-tailed Hawks (*Buteo jamaicensis*) used forests adjacent to plantations, but were not found in the plantations.

Because poplar plantations commonly replace cropland, and sometimes native habitat, it is important that we understand the use of these plantations by wildlife. The objective of our study was to determine which cover types were the best predictors of raptor abundance in a landscape dominated by shrub-steppe, poplar plantations, and croplands in northeastern Oregon.

#### STUDY AREA

The 20 105 ha study area was a complex of poplar plantations and irrigated croplands, surrounded by shrub-steppe vegetation (Franklin and Dyrness 1988) in the Columbia Basin of northeastern Oregon. Shrub-steppe, plantation, and cropland cover types comprised 47%, 35%, and 18% of the study area, respectively. The area contained relatively high densities of primitive roads in plantations and croplands (>1.9 km/km<sup>2</sup>), and lower densities in shrub-steppe (<1 km/km<sup>2</sup>). Topography varied from flat to slightly undulating and elevations ranged from 150–250 m. Mean annual precipitation was ca. 22 cm (Ruffner 1978).

Shrub-steppe vegetation varied depending on soil type (Franklin and Dyrness 1988). The Quincy sand soil types were dominated by indian ricegrass (*Oryzopsis hymenoides*), needle-and-thread (*Stipa comata*), and antelope bitterbrush (*Purshia tridentata*). The finer soils were dominated by basin big sagebrush (*Artemisia tridentata*), bluebunch wheatgrass (*Pseudoregnaria spicatum*), Idaho

fescue (*Festuca idahoensis*), and Sandberg's bluegrass (*Poa sandbergii*). This vegetation was also found in undisturbed, unfarmed corners (<2 ha each) surrounding agricultural circles. Corners that had been mowed or cultivated were usually dominated by invader species such as cheatgrass (*Bromus tectorum*) and Russian thistle (*Salsola kali*), or planted with a cover crop such as wheat. Perch sites were relatively scarce in this cover type, and when available consisted of power poles, fence posts, and shrubs.

Drip-irrigated plantations were first established in 1994, primarily on former croplands irrigated by center-pivot systems. Plantations representing six age classes of trees (1–6-yr-old) were available for study. All trees were planted on ca. 2.5 × 3 m spacing and scheduled for harvest within 8–12 yr. Plantations differed in both tree height and understory composition depending on age. On average, poplars grew 3–5 m/yr. During the first 2–3 yr of growth, understory vegetation was often dense (>90% cover). Common understory plants included the native horseweed (*Conyza canadensis*) and redroot pigweed (*Amaranthus palmeri*), and several introduced invaders, including Russian thistle, tumble mustard (*Sisymbrium altissimum*), and cheatgrass. Each plantation was sprayed with an herbicide-mixture of glyphosate and 2,4-D, and cultivated once during yr one and two. However, most competing plants quickly recolonized given the favorable growing conditions, and impacts to rodent populations were short-term (Moser et al. 2002). Complete canopy closure occurred following yr three. At this time, the understory became nearly devoid of live plants (<5% cover) and was dominated by organic material primarily composed of leaf litter. Perch sites were abundant in all plantation age classes.

Agricultural fields in the study area contained a cover crop such as winter wheat, were fallow with wheat or corn stubble, or were planted with a perennial crop such as alfalfa or grass. These fields were not irrigated during the study period. Perch sites were more abundant than in shrub-steppe, but less abundant than plantation cover types and included hay stacks, power poles, small trees, and irrigation equipment.

#### METHODS

Cover types were surveyed with equal intensity, and included shrub-steppe, tree plantations by age-class (1–6-yr-old), and croplands. Tree plantations were further stratified into interior and edge categories, with the exception of 6-yr-old plantations, which due to limited area had only edge transects. Plantation-interior categories were bordered on all sides by plantations. Plantation-edge categories were defined as the convergence of plantations and either shrub-steppe or cropland cover types. These habitat cover types used in the analysis are described in Table 1.

We conducted road surveys through the center of each of the cover type categories once per week during January and February, for a total of eight surveys in 1999 and nine in 2000. Primitive road sections were classified as either containing powerlines or not, and road sections were randomly selected for surveys until each cover type had equal amounts of powerline-influenced road sections. Approximately 6.4 km of transect was selected as

Table 1. Indicator variables used in the development of logistic regression models for wintering raptors in north-eastern Oregon, 1999 and 2000.

VARIABLE	DESCRIPTION
YR	1999 or 2000
NAT	Shrub-steppe habitat
AG	Irrigated cropland
I1	Interior of 1-yr-old plantations
I2	Interior of 2-yr-old plantations
I3	Interior of 3-yr-old plantations
I4	Interior of 4-yr-old plantations
I5	Interior of 5-yr-old plantations
E1	Edge of 1-yr-old plantations
E2	Edge of 2-yr-old plantations
E3	Edge of 3-yr-old plantations
E4	Edge of 4-yr-old plantations
E5	Edge of 5-yr-old plantations
E6	Edge of 6-yr-old plantations

four 1.6 km sections of road in each of the cover types. Transects were located  $\geq 0.8$  km apart to reduce the probability of sampling the same bird twice (Fuller and Mosher 1987). Birds detected within 100 m of the transect were assumed to be using that cover type. Surveys were conducted between sunrise and 1100 H. Successive surveys were conducted in opposite directions to reduce any time-of-day bias. Surveys were conducted from a vehicle by two trained observers (all observers had a minimum of a B.S. in Wildlife Biology and were trained formally during a 5-d period to identify local raptors and to conduct surveys) to identify local raptors while using binoculars and spotting scopes to aid in raptor identification. The vehicle was traveling continuously at 10–30 km/hr, depending on visibility within each cover type. Raptor detection rates were influenced by both survey effort (i.e., time) and visibility. In general, visibility was greatest in shrub-steppe and croplands, and decreased with increasing plantation age. Therefore, we adjusted vehicle speed so that it was greater in open cover types, and slower in dense cover types. Although this methodology resulted in greater survey time in older plantations, we felt any survey effort bias was appropriately offset by the lower detection rates of birds in the heavier cover types (Millsap and LeFranc 1988). We stopped the vehicle to identify birds when necessary (Fuller and Mosher 1987). Surveys were not conducted during inclement weather (Ralph et al. 1993).

All birds, whether perched or flying, were recorded and included in the analyses as using a particular cover type for two reasons. First, although most of the species we studied hunt primarily from perches, many of them will also hunt while flying. Two of the three most common raptors we observed (Red-tailed Hawk and American Kestrel) hunt from both a perch and while flying (Preston and Beane 1993, Smallwood and Bird 2002). Furthermore, the Northern Harrier (*Circus cyaneus*) hunts almost exclusively while flying (MacWhirter and Bildstein 1996). Second, we were not always able to de-

termine whether we flushed a bird as we approached its perch, or whether it was flying before we detected it. Therefore, we assumed that if a bird was detected perched in or flying through a cover type, that it was using that cover type.

Chi-square goodness-of-fit analysis of the count distribution indicated the data were not normally distributed, but rather conformed to the geometric distribution ( $P > 0.05$ ). Attempts to transform the data to a normal distribution failed due to the low-frequency of the transect counts, the discrete nature of count data, and the prevalence of transects with no raptor detections. Thus, the count data for each species were converted to a presence/absence variable, and multiple logistic regression was used to determine the elements of the experimental design that contributed significantly to raptor presence. Logistic regression with indicator variables provides insight into the magnitude of effects of individual habitat attributes on raptor presence through the relative value of the coefficient estimate. Coefficient estimates with higher values indicate habitats with a higher probability of raptor presence. Sufficient observations existed to develop individual regression models for Red-tailed Hawk, Northern Harrier, and American Kestrel. Count data were coded according to habitat attributes using indicator variables (Table 1). Additionally, a temporal variable (YR) was included to determine the significance of year to the survey as this was an initial element of the experimental design (Table 1). Because raptor counts were generally higher in 1999 than in 2000, the YR variable was coded as 1 for 1999, and 0 for 2000. Thus, each analysis of habitat associations contained 14 indicator variables. Parameter estimates were developed using Maximum Likelihood estimation (StatSoft, Inc. 1995). Indicator variables for  $P > 0.05$  were removed individually until only significant variables remained in the model. All analyses were performed with Statistica for Windows (StatSoft, Inc. 1995).

The classification ability of each model was tested by randomly sub-sampling 25% of the original data. The predicted values for each model were then compared to the observed values for this sub-sample. It should be noted that these data were used in the original model development, and cannot be considered independent. Nevertheless, they still provided some information on the classification ability of the models.

## RESULTS

We detected 374 diurnal raptors of 12 species during 1999 and 2000 (Table 2). The most common raptors were Red-tailed Hawk, Northern Harrier, and American Kestrel. Species detected at relatively low rates included Rough-legged Hawk (*Buteo lagopus*), Ferruginous Hawk (*B. regalis*), Northern Goshawk (*Accipiter gentilis*), Cooper's Hawk (*A. cooperii*), Sharp-shinned Hawk (*A. striatus*), Golden Eagle (*Aquila chrysaetos*), Prairie Falcon (*Falco mexicanus*), Merlin (*F. columbarius*), and Snowy Owl (*Nyctea scandiaca*). All but the Prairie Falcon were observed at least once within a plan-

Table 2. Total raptor detections by cover type in northeastern Oregon, 1999 and 2000.

	COVER TYPE <sup>1</sup>													TOTAL	PERCENT PERCHED
	NAT	AG	E1	E2	E3	E4	E5	E6	I1	I2	I3	I4	I5		
Total	11	49	17	21	35	57	68	42	49	11	6	6	2	374	89.2
Red-tailed Hawk	1	21	5	4	15	28	44	20	6	2	0	2	1	149	86.8
Northern Harrier	1	6	7	6	1	10	6	2	27	7	4	3	0	80	26.6
American Kestrel	3	17	2	11	11	9	9	6	7	0	0	0	0	75	89.5
Rough-legged Hawk	4	2	1	0	5	3	5	2	1	0	0	0	0	23	87.3
Cooper's Hawk	0	0	0	0	0	3	0	0	2	1	1	0	0	7	100.0
Ferruginous Hawk	0	0	0	0	0	0	0	6	0	0	0	0	0	6	83.3
Golden Eagle	1	0	1	0	0	0	1	0	2	0	0	0	0	5	100.0
Sharp-shinned Hawk	0	0	0	0	0	0	1	1	1	0	0	0	0	3	100.0
Snowy Owl	0	1	0	0	0	0	0	0	1	0	0	0	0	2	100.0
Merlin	1	0	0	0	1	0	0	0	0	0	0	0	0	2	100.0
Prairie Falcon	0	1	0	0	0	0	0	0	0	0	0	0	0	1	100.0
Northern Goshawk	0	0	1	0	0	0	0	0	0	0	0	0	0	1	100.0
Unidentified raptor	0	1	0	0	2	4	2	5	2	1	1	1	1	20	85.5

<sup>1</sup> See Table 1 for description of cover types.

Table 3. Logistic regression models used to identify cover types significant to raptor presence in northeastern Oregon, 1999–2000. All regression models are significant ( $P \leq 0.001$ ).

DEPENDENT VARIABLE	INTERCEPT	INDICATOR VARIABLE	COEFFICIENT	<i>P</i>
Red-tailed Hawk	-3.92	AG	2.89	<0.001
		I1	1.58	0.001
		E1	1.54	0.003
		E2	1.45	0.008
		E3	2.37	<0.001
		E4	3.24	<0.001
		E5	3.74	<0.001
		E6	3.46	<0.001
Northern Harrier	-3.25	I1	2.22	<0.001
		I2	0.93	0.008
		E1	1.07	0.002
		E2	1.23	<0.001
		E4	1.23	<0.001
American Kestrel	-5.40	NAT	1.89	0.004
		AG	3.55	<0.001
		I1	2.81	<0.001
		E2	3.32	<0.001
		E3	3.23	<0.001
		E4	2.96	<0.001
		E5	3.10	<0.001
		E6	3.79	<0.001
		YR	0.78	<0.001

<sup>1</sup> See Table 1 for description of indicator variables.

tation. Most of the raptors detected (except for Northern Harriers) were observed on a perch (Table 2).

Logistic regression models developed for Red-tailed Hawk, Northern Harrier, and American Kestrel, were all significant ( $P < 0.001$ ; Table 3). The cover type categories selected as contributors to predictability of Red-tailed Hawk presence were AG, I1, and E1–E6 ( $P < 0.008$ ; Table 3). Cover type categories selected for the Northern Harrier model included AG, I1–I2, and E1–E2, and E4 ( $P = 0.008$ ; Table 3). The cover type categories selected as contributors to predictability of American Kestrel presence were NAT, AG, I1, and E1–E6 ( $P = 0.004$ ; Table 3). The I3–I6 cover type categories were not selected for any model ( $P > 0.05$ ). Survey year was a significant component for the American Kestrel model only. More Kestrels were detected in 1999 than in 2000 ( $P < 0.001$ ; Table 3). The indicator variables selected for inclusion or exclusion in the logistic models correctly predicted raptor presence 92% of the time within 10% of the observed rate. Variables that did not classify as well included AG and E4 for the Red-tailed Hawk model, and I1 for the Northern Harrier model (Table 4).

#### DISCUSSION

Although we detected 12 species of raptors in this study, only three were detected frequently enough to be considered common. All of the hab-

Table 4. Predicted versus observed classification rates of logistic regression models for Red-tailed Hawk (RTHA), Northern Harrier (NOHA), and American Kestrel (AMKE) presence in various cover types.

INDICA- TOR VARI- ABLE	PREDICTED RATE			OBSERVED RATE		
	RTHA	NOHA	AMKE	RTHA	NOHA	AMKE
NAT	0.00	0.00	0.03	0.00	0.00	0.00
AG	0.26	0.00	0.14	0.07	0.00	0.07
I1	0.09	0.26	0.07	0.13	0.38	0.13
I2	0.00	0.09	0.00	0.00	0.05	0.00
I3	0.00	0.00	0.00	0.00	0.07	0.00
I4	0.00	0.00	0.00	0.06	0.00	0.00
I5	0.00	0.00	0.00	0.00	0.00	0.00
E1	0.09	0.10	0.00	0.00	0.11	0.00
E2	0.08	0.12	0.11	0.07	0.14	0.07
E3	0.18	0.00	0.10	0.18	0.00	0.09
E4	0.34	0.12	0.08	0.50	0.13	0.06
E5	0.46	0.00	0.09	0.45	0.00	0.09
E6	0.39	0.00	0.17	0.42	0.00	0.25

<sup>1</sup> See Table 1 for description of indicator variables.

itats studied were used by at least one raptor species. However, raptor distribution was most affected by croplands and poplar plantations in this study. In fact, only American Kestrel distribution was significantly affected by native shrub-steppe habitat. Our results are not surprising, as a number of other studies have reported relatively high numbers of certain raptor species in agricultural landscapes (Parker and Campbell 1984, Craig et al. 1986, Meunier et al. 2000), probably due to both a large number of available perch sites (e.g., utility poles) as well as relatively high densities of small mammal prey in these settings. The distribution of utility lines and poles within treatment categories in our study was intentionally made equal so as not to confound the results. Nevertheless, plantations still provided significantly more perch sites than shrub-steppe or even cropland cover types. Thus, it is not surprising that most raptors were more commonly detected in or near plantations. Furthermore, prey densities are relatively high in both croplands and young poplar plantations (Christian et al. 1997, Moser et al. 2002). Because most Red-tailed Hawks and American Kestrels were detected on perches, it is likely they were hunting in those habitats (Preston and Beane 1993, Smallwood and Bird 2002). Likewise, most Northern Harriers were detected flying, again suggestive of hunting behav-

ior for this species (MacWhirter and Bildstein 1996).

Hanowski et al. (1997) found that surrounding cover types affected songbird use of hybrid poplar plantations in the midwestern U.S. We believe this phenomenon to be true in our study as well. Raptor detections were generally greatest along the edges of plantations when compared to the interiors, with the exception of the Northern Harrier which commonly hunts in open, brushy habitats, similar to those provided by the younger plantations. Most of the birds studied were not well adapted to hunting in the dense plantation habitats, with the exception of the *Accipiter* spp. The edge habitat created by the convergence of plantations with cropland and shrub-steppe habitats provided ideal hunting areas by providing perch sites near open habitat suited for prey species such as rodents.

Raptors contribute to important ecosystem processes, such as predator-prey dynamics (Hanski et al. 1991), and thus may be important in dampening the effects of nuisance rodent outbreaks (Korpimäki and Norrdahl 1991). In addition, these birds may be indicators of ecosystem quality due to their high-trophic-level status. Due to the ecological and economic importance of this bird assemblage, it is important that plantation managers consider these birds when designing and managing plantations. Due to the affinity of raptors toward both the interiors and edges of plantations, maintaining a diversity of plantation ages and creating within-plantation heterogeneity (Hanowski et al. 1997) will provide more structural diversity and edge, and thus likely create more favorable habitat for wintering raptors.

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## MORTALITY OF WINTERING OSPREYS AND OTHER BIRDS AT AQUACULTURE FACILITIES IN COLOMBIA

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**ABSTRACT.**—Ospreys (*Pandion haliaetus*) winter throughout the country of Colombia. Recoveries of banded Ospreys indicate that many are shot in the country with the number increasing since the 1970s. The increased incidence of shooting has coincided with the development of aquaculture facilities in Colombia that raise tilapia (*Oreochromis* spp.). Because these facilities typically lose production to birds such as Ospreys that depredate fish, we conducted a survey of 83 facilities in three states or departments in Colombia in 2001 to determine the species of birds that take fish at aquaculture facilities and the numbers that are killed each year. Our results showed that bird depredation occurs at aquaculture facilities throughout the country, but mostly in the southern portion of the country in the department of Huila. Facility managers reported shooting Ospreys in all three departments with as few as five individuals shot annually in Antioquia in northern Colombia to as many as 270 shot annually in Huila. In addition, facility managers reported shooting nine other species of birds including Green Kingfishers (*Chloroceryle americana*), Great Kiskadees (*Pitangus sulphuratus*), Snowy Egrets (*Egretta thula*), Great Egrets (*Casmerodius albus*), Olivaceous Cormorants (*Phalacrocorax olivaceus*), Black-crowned Night-Herons (*Nycticorax nycticorax*), White-necked Herons (*Ardea cocoi*), Cattle Egrets (*Bubulcus ibis*), and Striated Herons (*Butorides striatus*), for a total estimate of >9000 birds shot in the three departments annually. A number of alternative methods to shooting had been used to reduce losses to birds including the installation of netting, overhead wires, scarecrows, and noise making devices, but, neither these methods nor shooting, were effective in deterring avian predators.

**KEY WORDS:** *Osprey; Pandion haliaetus; aquaculture facilities; mortality; Oreochromis* spp.; *shooting; tilapia.*

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### MORTALIDAD DE AGUILAS PESCADORAS (*PANDION HALIAETUS*) INVERNANTES Y OTRAS AVES EN INFRAESTRUCTURA PISCICOLA EN COLOMBIA

**RESUMEN.**—El águila pescadora (*Pandion haliaetus*) pasa el invierno en todo el territorio colombiano. La recaptura de águilas pescadoras anilladas indican que muchas son cazadas en este país con cifras en aumento desde 1970. La incidencia del número de águilas muertas coincide con el desarrollo de la acuicultura en Colombia y la cría de tilapia roja (*Oreochromis* spp.). Esta industria tradicionalmente ha tenido problemas con aves depredadoras de peces por lo cual realizamos una encuesta en 83 granjas piscícolas en tres departamentos de Colombia en el 2001 para determinar las especies que consumen peces y el número de aves eliminadas anualmente. Nuestros resultados mostraron que la depredación por aves en la infraestructura piscícola ocurre en todo el territorio pero el problema es más severo en la porción sur del país en el departamento del Huila. Los propietarios de las granjas reportaron que eliminan águilas pescadoras en los tres departamentos, con pocos individuos en Antioquia (al norte de Colombia) equivalente a 5 individuos, y un número máximo en el Huila de 270 individuos. Adicionalmente los propietarios reportaron que cazan otras nueve especies de aves las cuales incluyen a *Chloroceryle americana*, *Pitangus sulphuratus*, *Egretta thula*, *Casmerodius albus*, *Phalacrocorax olivaceus*, *Nycticorax nycticorax*, *Ardea cocoi*, *Bubulcus ibis*, *Butorides striatus*, para un total estimado de >9000 aves eliminadas anualmente en los tres departamentos. Los propietarios de las granjas reportaron que han probado otros métodos de control aparte de las armas de fuego, para reducir las pérdidas tales como la instalación de redes

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protectoras, cuerdas de alambre, espantapájaros, aparatos con sonidos, pero ninguno ha resultado eficiente in incluyendo el uso de las armas de fuego para la reducción de las pérdidas económicas causadas por aves.

[Traducción de los autores]

Ospreys (*Pandion haliaetus*) breed throughout North America (Palmer 1988, Poole 1989, Johnson 1990, Poole et al. 2002). During the 1950s and 1960s, pesticide contamination threatened many populations and declines were widespread across the breeding range (Poole 1989). Since then, populations have rebounded and, today, the Osprey is again a common species in coastal, lake, and riverine habitats. The Osprey is migratory with populations from locations in the western U.S. wintering in Mexico and Central America (Henny and Van Velzen 1972, Melquist et al. 1978, Johnson and Melquist 1991, Martell et al. 2001) and populations from coastal areas of the eastern U.S. and Canada and the Great Lakes region of the Midwest wintering in South America (Martell et al. 2001, Poole et al. 2002). Ospreys banded as nestlings in Maryland, Virginia, New York, New Jersey, Michigan, and Wisconsin have mainly been recovered in Colombia, Venezuela, Ecuador, and Brazil indicating that northern South America is a primary wintering area for eastern and Midwestern populations (Henny and Van Velzen 1972, Poole and Agler 1987, Niemuth 1991).

The Osprey is unusual because it is exclusively piscivorous eating a wide variety of fresh and salt-water fish species (Poole 1989, Poole et al. 2002). It has traditionally been called the "fish hawk" in North America and "aguila pescadora" in South America and it has been shot, had its nests robbed or torn down, and otherwise been persecuted in areas where people believe it competes with them for sport or commercial fisheries. The Osprey is protected from shooting and other forms of human-caused mortality in North America but, on its Caribbean and Central and South American wintering grounds, there are few laws restricting the killing of birds. Band recoveries of dead Ospreys indicate that shooting and trapping continue to threaten the species on its wintering grounds (Poole and Agler 1987, Santana and Temple 1987, Ewins and Houston 1992). The Osprey is also unusual because it does not breed until three years of age and immatures remain on the wintering grounds continuously until they become sexually mature (Henny and Van Velzen 1972, Poole 1989).

Therefore, shooting may selectively eliminate younger individuals and possibly decrease recruitment of new breeders into populations. Currently, there is insufficient information to judge the extent and severity of the South American shooting threat, despite the fact that it may have increased since the 1970s (Poole and Agler 1987, Ewins and Houston 1992).

Since the 1970s, fish farming or aquaculture has become a new and thriving business in Latin America. Production systems consisting of extensive ranching operations where fry are released into reservoirs and later harvested as adults have been built in Mexico, Central and South America (Fitzsimmons 2000). Many aquaculture facilities specialize in the production of tilapia (*Oreochromis* spp.), which are now produced in virtually every country of the Americas. By 1998, the annual production of tilapia had grown to 201 067 mt (metric tons) in Latin America and the U.S. imported 72 428 mt of live weight fish in 2000 (Fitzsimmons 2000). Aquaculture facilities typically raise tilapia in shallow ponds that are <1 ha in size, but, with facilities consisting of >30 ponds, they can have large areas of impounded water. Tilapia are surface-feeders and, when thousands of these bright red fish come to the surface to feed, they become easy prey for piscivorous birds.

With its tremendous warm water resources, Colombia has become one of the leading aquaculture fish producers in South America (Fitzsimmons 2000). Commercial fish production began in the 1980s and by 1996, 22 states or departments were producing a mean of 25 063 mt of fish a year (Instituto Nacional de Pesca y Acuicultura de Colombia [INPA] unpubl. data). Some commercial facilities raise rainbow trout (*Oncorhynchus mykiss*), but most are dedicated to the production of tilapia and Colombia produces a mean of 15 000 mt of tilapia each year (Fitzsimmons 2000). There is an obvious potential for large numbers of piscivorous birds to be killed at these aquaculture facilities and, as such, these facilities may have some effect in redistributing populations of resident and wintering piscivorous birds in the country. Because of the escalating numbers of Ospreys that are shot at

aquaculture facilities and brought to rehabilitation facilities in Colombia (pers. observ.) and increasing concern from the birding community about the numbers of Ospreys being killed each year (Nielson 1998), we undertook a survey to assess the extent of the threat to North American Ospreys wintering in Colombia from shooting, trapping, and other human-caused mortality at aquaculture facilities throughout the country.

#### STUDY AREAS AND METHODS

We chose three departments, Antioquia in northern Colombia, Valle del Cauca in west-central Colombia, and Huila in southern Colombia, to conduct our survey. We chose these departments because they provided a good representation of the variation in the sizes and production of aquaculture facilities in the country, and each department had ca. 30 commercial aquaculture facilities that were either licensed by the Corporacion de Valle del Cauca or by INPA. Antioquia (05°26'–08°52'N, 73°53'–77°07'W) has a total of 28 licensed aquaculture facilities and was the largest of the three departments surveyed covering an area of 63 612 km<sup>2</sup>. The department of Valle del Cauca (03°04'–05°02'N, 72°42'–74°27'W) encompasses a 22 140 km<sup>2</sup> area of west-central Colombia and has 32 licensed aquaculture facilities. The department of Huila in the southern portion of Colombia (01°33'–03°47'N, 74°28'–76°36'W) is the smallest of the three departments covering an area of 19 890 km<sup>2</sup> and it has 27 licensed aquaculture facilities.

Of the 87 licensed aquaculture facilities in the three departments, we visited all except four to interview facility owners and managers who were familiar with the daily operations of the facilities. The four facilities not visited were in the department of Valle del Cauca, and they were not surveyed either because the owners declined to participate or because they were in locations where conditions made them too dangerous to visit. During each visit, we administered a standard questionnaire to owners or managers who were familiar with the daily operations of facilities. Before administering the questionnaire, a statement signed by officials of INPA and the Ministry of the Environment was read stating that all answers would be kept confidential and that no legal proceedings would result from any answers given to the questioner. The questionnaire was administered verbally, and we completed the answers on the questionnaire forms as the interviewees responded to them. The questions asked for information on the size of the facility (number of ponds and ha of impounded water), its annual fish production (mt), if birds were a problem because they impacted annual fish production, the species of birds depredating fish, the seriousness of the impact by each species (on a scale from 1–5 with 1 being a species with one or a few individuals infrequently depredating ponds and 5 being a species with several individuals depredating ponds on a daily basis), if birds were shot, which species of birds were shot, estimated number of each species shot each year (1–10, 11–20, 21–50, 51–75, 76–100, 101–200, or >200 shot annually), and alternative methods to shooting that had been used to decrease the depredation

Table 1. Questionnaire administered to aquaculture facility owners and managers to estimate bird depredation and mortality at aquaculture facilities in the departments of Antioquia, Valle del Cauca, and Huila in Colombia.

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Facility name
Permit number
Location
Name of water source
Water temperature (°C)
Number of ponds
Area of impounded water (ha)
Fish species cultivated
Annual fish production (mt)
Annual income (pesos)
Do you have problems with bird depredation?
How much do you estimate you lose annually to bird depredation? (pesos)
Which species of birds are a problem and rank each species in terms of the seriousness of the problem (1 = none or little problem, 2 = slight problem, 3 = moderate problem, 4 = serious problem, 5 = severe problem with fish taken daily).
Do you shoot problem birds? If yes, estimate the numbers of each species that are shot annually (1–10, 11–20, 21–50, 51–75, 76–100, 101–200, >200).
Have you used other methods of deterring birds from taking fish at your facility? If yes, please describe each method and rate its effectiveness.

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problem (Table 1). Most of the interviewees knew the local common names for the species of birds that caused depredation losses at their farms. When there was any question about the identity of a species of bird, we used color photographs and color plates in Hilty and Brown (1986) to help interviewees identify the species. To estimate the numbers of each species shot annually at each facility, we used the midpoints of the ranges given by interviewees for the numbers of birds they shot each year. To rank species in terms of the seriousness of the threat they posed to the production of fish, we averaged the rankings given by the facility owners in each department.

#### RESULTS

A total of 82 of the 83 aquaculture facilities surveyed reported experiencing depredation losses to the following 10 species of birds: Green Kingfisher (*Chloroceryle americana*), Great Kiskadee (*Pitangus sulphuratus*), Snowy Egret (*Egretta thula*), Great Egret (*Casmerodius albus*), Olivaceous Cormorant (*Phalacrocorax olivaceus*), Black-crowned Night-Heron (*Nycticorax nycticorax*), White-necked Heron (*Ardea cocoi*), Cattle Egret (*Bubulcus ibis*), Striated Heron (*Butorides striatus*), and Osprey (Table 2). Of the 82 facilities with depredation losses, 35 reported

Table 2. Mean ranking of birds that pose depredation problems at aquaculture facilities in the departments of Antioquia, Valle del Cauca, and Huila in Colombia and estimates of the number of each species shot annually.

SPECIES	ANTIOQUIA		VALLE DEL CAUCA		HUILA		TOTAL SHOT
	RANKING <sup>1</sup>	NO. SHOT	RANKING <sup>1</sup>	NO. SHOT	RANKING <sup>1</sup>	NO. SHOT	
Green Kingfisher	2.6	44	2.5	65	3.5	1836	1945
Great Kiskadee	1.7	0	1.7	0	2.5	1692	1692
Snowy Egret	1.1	0	1.1	0	2.9	1621	1621
Great Egret	1.8	45	2.25	65	2.75	931	1041
Olivaceous Cormorant	1.8	230	1.4	19	1.0	288	537
Black-crowned Night-Heron	1.0	0	2.6	105	2.0	318	423
White-necked Heron	1.6	5	2.0	13	1.4	403	421
Cattle Egret	1.0	15	1.0	0	1.7	400	415
Striated Heron	1.3	0	2.5	35	1.1	364	399
Osprey	1.5	5	2.6	40	2.8	270	315
Total shot		356		342		8323	9021

<sup>1</sup> 1 = none or little problem, 2 = slight problem, 3 = moderate problem, 4 = serious problem, 5 = severe problem with fish taken daily.

shooting birds killing a combined estimate of >9000 birds per year. Facilities in all three departments reported Green Kingfishers as causing the most serious losses and managers at these facilities reported killing more kingfishers (ca. 2000/yr) than any other species. Facilities also experienced depredation losses to Great Kiskadees, which took pelletized fish food and fingerlings, and managers at these facilities shot an estimated 1700 kiskadees per year. As a group, ciconiiforms were viewed as causing the most serious depredation losses at fish farms with Snowy Egrets, Great Egrets, and Black-crowned Night-Herons generally considered to be the biggest threats to facility production. Managers of facilities shot an estimated total of >3000 egrets and night-herons. An estimated 315 Ospreys were shot each year at facilities. Ospreys were considered to cause the most serious depredation losses in southern Colombia in the department of Huila. The one facility that did not experience bird depredation problems was located in Antioquia and it raised primarily rainbow trout that were grown in completely-covered raceways and protected from birds.

Aquaculture facilities in Antioquia ranged in size from 1–400 ponds ( $\bar{x} = 36.8 \pm 74.8$ ,  $N = 28$ ,  $\pm$ SD) with 0.01–26.0 ha of impounded water ( $\bar{x} = 1.8 \pm 5.1$ ,  $N = 28$ ) raising a mean of  $79.02 \pm 155.9$  mt of fish per year (range = 1.5–600,  $N = 28$ ). Twenty-two of the facilities raised mostly rainbow trout and the remaining six raised mainly tilapia. A total of 27 facilities in Antioquia experienced losses in pro-

duction to bird depredation. At 23 facilities, birds ranked as causing the most serious depredation losses were Green Kingfishers, Great Egrets, Olivaceous Cormorants, Great Kiskadees, and White-necked Herons that took fish either while perching on overhead wires or while wading along the edges of ponds (Table 2). Only four facilities in Antioquia ranked Ospreys as causing the most serious production losses, and all of these facilities specialized in tilapia production. Eight facility managers said they shot an estimated 230 Olivaceous Cormorants, 45 Great Egrets, 44 Green Kingfishers, 15 Cattle Egrets, and 5 White-necked Herons annually, but only 5 Ospreys per year.

Valle del Cauca, the second-largest, fish-producing department in Colombia, had a mean annual fish production of 4560 mt, most of which was tilapia. The aquaculture facilities consisted of fewer ponds ( $\bar{x} = 13.1 \pm 13.3$  ponds, range = 1–56,  $N = 28$ ) but they were much larger in size ( $\bar{x} = 5.5 \pm 5.8$  ha of impounded water, range = 0.4–18.2,  $N = 28$ ), than in Antioquia. Twenty of the facilities raised a mean of  $67.6 \pm 90.6$  mt (range = 5–240) of tilapia each year. All 28 of the facilities surveyed in Valle del Cauca reported losses in fish production to bird depredation. Black-crowned Night-Herons, Ospreys, Green Kingfishers, Striated Herons, and Great Egrets were considered to be problem species with mean rankings >2 (Table 2). Fourteen facility managers reported Ospreys to be a serious problem species. Only four facility managers in Valle del Cauca said they shot birds and

estimated killing 105 Black-crowned Night-Herons, 65 Great Egrets, 50 Striated Herons, 40 Ospreys, and 20 White-necked Herons each year.

The department of Huila was the smallest fish producer of the three departments surveyed, producing a mean of 2132 mt of fish per year, nearly 94% of which was tilapia. Aquaculture facilities in Huila had more ponds than in either Antioquia or Valle del Cauca ( $\bar{x} = 62.1 \pm 139.7$  ponds, range = 3–650,  $N = 27$ ), but they were smaller in size and covered a mean of  $4.1 \pm 6.1$  ha (range = 0.2–25,  $N = 27$ ). Nevertheless, five of the facilities in Huila were very large consisting of >20 ponds and >15 ha of impounded water. Facilities in Huila were very productive, producing a mean of  $99.08 \pm 132.42$  mt of tilapia (range = 2.5–840,  $N = 27$ ) annually. In Huila, all 27 aquaculture facilities surveyed reported losses in fish production to bird depredation. Green Kingfishers, Great Egrets, and Great Kiskadees were again ranked as causing serious depredation problems (Table 2). Twenty-two facility managers reported Ospreys to be a serious depredation problem and 11 of these reported Ospreys to be their most serious problem species. Unlike the other departments, aquaculture facilities in Huila experienced serious depredation losses to Snowy and Cattle Egrets. Birds were shot at 23 facilities with an estimated 8323 birds killed annually including 1836 Green Kingfishers, 1692 Great Kiskadees, 1621 Snowy Egrets, 931 Great Egrets, 403 White-necked Herons, 400 Cattle Egrets, 364 Striated Herons, 318 Black-crowned Night-Herons, and 270 Ospreys. One facility manager reported shooting >100 Ospreys at a communal roost tree located on the Magdalena River, where as many as 50 Ospreys would roost each night. The number he shot did not reduce the depredation problem at his facility because as many as 20 Ospreys fed there daily. The day we visited the facility, we observed 10 Ospreys taking fish from his ponds.

#### DISCUSSION

Of the 22 departments in Colombia that have INPA-licensed commercial aquaculture facilities, annual fish production averages <100 mt in the departments of Arauca, Atlantico, Casanare, Cauca, Cesar, Choco, Guaviare, and Vichada, and <1500 mt in the departments of Boyaca, Cundinamarca, Risaralda, Putumayo, and Santander. Most of this fish production is rainbow trout so we did not consider facilities in any of these departments to pose serious shooting threats to birds be-

cause these facilities are relatively small and trout are mainly raised in covered raceways that are protected from birds. The departments of Antioquia, Cordoba, Huila, Meta, Tolima, and Valle del Cauca each average >2000 mt of fish production annually, ranging from a low of 2100 mt in Cordoba to a high of 6589 mt in Meta. These departments have the largest aquaculture facilities and raise mainly tilapia in large, shallow ponds that are prone to bird depredation. Therefore, these departments have the greatest potential for developing bird depredation problems and the shooting of birds at aquaculture facilities.

The results of our survey showed that several species of birds depredate aquaculture facilities in Colombia with the depredation problem apparently increasing from north to south in the country. The most common solution to the problem that has been used by facility managers is shooting, and shooting appears to increase from as few as 100 birds shot in Antioquia in northern Colombia to as many as 8000 shot annually in Huila in southern Colombia. This increase appears to be due to a shift in the emphasis of fish production at aquaculture facilities in the country from mostly trout production at facilities in northern Colombia to mostly tilapia production in southern Colombia. In northern Colombia, most facility managers do not consider the bird depredation problem to be severe enough to warrant killing birds and only eight of them said they shot kingfishers, herons, egrets, and Ospreys. In Huila, facility managers appear to see bird depredation as a more serious problem. This attitude was reflected in the numbers and variety of birds they shot. All of the fish farm managers who shot birds, also said that shooting was not an effective method of decreasing their losses because new birds simply replaced birds that were shot.

The numbers of Ospreys reported shot each year also increased from five in Antioquia to over 270 in the southern department of Huila. Ospreys were considered to be only a minor problem in Antioquia causing production losses at only those facilities that emphasized tilapia production. The shooting in Huila increased correspondingly with an increase in the productivity of tilapia in this department. Warmer water temperatures in this department are more conducive to the production of tilapia. Tilapia are raised in large, shallow impoundments averaging nearly 5 ha in size. At any time, aquaculture facilities in Huila have as many

as 1 million tilapia in various stages of growth. The combination of the large amount of impounded water and the multitude of easily-captured fish prey are natural lures for Ospreys.

Of the species that were shot, only Great Egrets, Snowy Egrets, Cattle Egrets, Black-crowned Night-Herons, and Ospreys are considered to be boreal migrants in northern South America (Davis 1993, Parsons and Master 2000, McCrimmon et al. 2001, Ridgley and Greenfield 2001). All, excluding the Osprey, are also resident species that breed as well as winter in Colombia (Hilty and Brown 1986); therefore, it was impossible to assess the overall threat of Colombian aquaculture facilities to boreal migrants. We recovered a total of six USGS bands that owners had removed from dead birds, but had not reported to the Bird Banding Laboratory. All were from dead Ospreys. Two were from Ospreys that had been banded as nestlings in Maine and one each came from Wisconsin, Virginia, New York, and Connecticut, further supporting the view that Colombia is an important wintering area for Ospreys from the eastern and Midwestern U.S.

Aquaculture of tilapia first began in the Americas in the 1960s and 1970s. Currently, Mexico is the biggest producer of tilapia but the industry is rapidly growing in Honduras, Costa Rica, Cuba, Jamaica, Colombia, Ecuador, Peru, Venezuela, and Brazil (Fitzsimmons 2000). Future production trends include further intensification of production in every country with Brazil expected to become the biggest producer of tilapia in the next 20 yr (Fitzsimmons 2000). Ospreys winter in all of these countries, therefore, if this trend continues, we could expect an increase in the shooting of Ospreys on their wintering grounds. In North America, the construction of reservoirs for irrigation and flood control have had a substantial effect on the redistribution of breeding Ospreys, particularly in the western U.S. (Henny et al. 1978a, 1978b, Swenson 1981, Poole 1989). A similar phenomenon may be currently underway in Latin America. Aquaculture facilities may redistribute wintering populations of Ospreys away from natural habitats such as coastal, lake, and river areas to man-made reservoirs, where they are lured by the abundance of easily-captured fish prey. Unlike North America, where legislative controls and public education has reduced the risk of Ospreys being shot (Poole and Agler 1987), in Latin America shooting continues to be a very real threat. Many of the countries lack

any laws that prevent shooting and those that do frequently lack enforcement. Shooting is an easy method of eliminating problem birds and band recovery data from Central and South America support that this is the usual method of reducing the avian depredation problem. Of the Ospreys banded in Canada, 39% have been recovered shot and the number shot appears to have increased since the 1970s, which coincides with the timing of the first construction of aquaculture facilities in Latin America (Poole and Agler 1987, Poole 1989, Ewins and Houston 1992). Populations of Ospreys continue to increase throughout North America (Poole et al. 2002) indicating that the mortality at aquaculture facilities has probably not yet reached numbers great enough to impact populations of North American Ospreys. Nevertheless, in view of the current growing global trend in fish farming, mortality from shooting will only increase in years to come, perhaps developing into a serious problem for Osprey populations in the eastern and Midwestern U.S.

Our survey showed that fish farm owners had tried a variety of nonlethal bird deterrents to reduce their annual losses to birds. The most widely-used alternatives to shooting were noise-making devices, such as guns, cannons, and fireworks, and patrols by people who flushed birds from the edges of ponds. None of these methods had proven effective because problem birds apparently either became habituated to the noise makers or simply learned to ignore people patrolling ponds. A few facility managers had tried using scarecrows and dogs to frighten birds away. Neither of these methods was effective after problem birds became accustomed to them. Another widely-used deterrent was the installation of netting around ponds. Nets were frequently used to reduce losses to birds such as herons and egrets that hunt along the edges of ponds. Owners reported that the devices were relatively effective provided the netting was maintained along the edges of ponds. Such maintenance was labor intensive and it was difficult to make certain that the netting was in place at all times. Only five fish farms reported having tried using overhead lines to deter depredation from aerial foragers such as Ospreys (Salomon and Conte 1981). The five that tried this method found it to be relatively inexpensive and potentially effective in reducing losses to large aerial-hunting birds like Ospreys.

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## BLOOD CHEMISTRY, CYTOLOGY, AND BODY CONDITION IN ADULT NORTHERN GOSHAWKS (*ACCIPITER GENTILIS*)

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**ABSTRACT.**—A bird's physiological state contributes to its reproductive success and survival, yet few baseline physiological data have been published for wild raptors. Mean levels of protein, cholesterol, calcium, uric acid, aspartate aminotransferase, lactate dehydrogenase, and creatine kinase were measured in 29 Northern Goshawks (*Accipiter gentilis*) during 1998–99. None of these substances were significantly different between males ( $N = 8$ ) and females ( $N = 2$ ). Levels of all substances were highly variable among individual birds and unrelated to the body condition index (mass/wing chord  $\times$  tail length  $\times$  culmen length). Total white blood cell count estimates and differential white blood cell counts were not significantly different between the sexes. Of the blood cell measures, only the percent of heterophils and lymphocytes, and the ratio of heterophils to lymphocytes (H/L) differed between birds in good body condition and birds in relatively poor body condition. The H/L ratio has recently been proposed as a reliable measure of stress. Thus, variable H/L ratios between groups of wild birds may indicate differences in stress levels and overall health.

**KEY WORDS:** *Northern Goshawk; Accipiter gentilis; blood chemistry; H/L ratio; body condition.*

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### QUÍMICA SANGUÍNEA, CITOLOGÍA, Y CONDICIÓN CORPORAL EN ADULTOS DEL AZOR NORTEÑO (*ACCIPITER GENTILIS*)

**RESUMEN.**—El estado fisiológico de una ave contribuye a su éxito reproductivo y supervivencia, hasta el momento muy pocos datos fisiológicos de línea base han sido publicados para rapaces silvestres. Los niveles medios de proteínas, colesterol, ácido úrico, aminotransferasa aspartato, lactato deshidrogenasa, y creatina quinasa se midieron en 29 azores norteños (*Accipiter gentilis*) durante 1998–99. Ninguna de estas sustancias fue significativamente diferentes entre machos ( $N = 8$ ) y hembras ( $N = 2$ ). Los niveles de todas las sustancias fueron altamente variables entre aves individuales y no estuvieron relacionadas con el índice de condición del cuerpo (masa/cuerda alar  $\times$  longitud de la cola  $\times$  longitud del culmen). El conteo de leucocitos totales estimados y el conteo diferencial de leucocitos, no fueron significativamente diferentes entre sexos. Entre las medidas de las células sanguíneas, únicamente el porcentaje de heterofilos y linfocitos, y la razón de heterofilos a linfocitos (H/L) difirió entre aves con buena condición corporal y aves en condición corporal relativamente pobre. La razón H/L ha sido propuesta recientemente como una medida confiable de estrés. Así, las razones H/L variables entre grupos de aves silvestres pueden indicar diferencias en los niveles de estrés y en la salud global.

[Traducción de César Márquez]

Northern Goshawks (*Accipiter gentilis*) are found throughout forested portions of the intermountain west (Squires and Reynolds 1997) where they are classified as a “sensitive species” by the United States Forest Service (Beals and Harris 1996, Squires and Reynolds 1997) and a “species of spe-

cial concern” by the Idaho Department of Fish and Game (Beals and Harris 1996). Basic knowledge of adult animal health, including blood-chemistry information, is needed for the management of vulnerable species (Ferrer and Dobado-Berrios 1998), such as the goshawk. However, past research on free-ranging goshawk life history or management does not include an assessment of adult health from blood chemistry or cytology (blood cell) measures. These measures

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provide information regarding nutritional and immunological status, and help us understand ecological and behavioral problems (Ferrer 1990, Stein et al. 1998). Two objectives were addressed in this study: (1) determine baseline means and ranges of blood chemistry and cytology levels in adult breeding goshawks, and (2) determine the relationships among blood chemistry, blood cytology, and body condition.

Blood chemistry is an indirect method of assessing health as blood acts as a means to mobilize and transport nutrients, metabolic products, immune cells, and hormones (Brown 1996). Metabolic substances present in blood plasma reflect avian nutritional health and overall condition (Snyder and Terry 1986, Mauro 1987, Ferrer 1993).

Much like plasma metabolites, plasma enzymes also reflect avian health. Enzymes such as aspartate aminotransferase (AST), lactate dehydrogenase (LDH), and creatine kinase (CK) help regulate metabolic reactions and respond to changes in the body (Mauro 1987). Interpretation of plasma enzyme levels is complicated by natural variation unless levels are significantly higher or lower than reference samples (Ivins et al. 1986, Campbell 1995).

Differential white blood cell counts provide information on the activity level of a bird's immune system and stress response (Dhabhar et al. 1996). Heterophils and lymphocytes are the predominant white blood cells found in raptors (Lavin et al. 1992, Stein et al. 1998). The ratio of heterophils to lymphocytes (H/L) has proven to be a reliable measure of stress in poultry and wild birds and a more accurate measure of stress than individual cell numbers or corticosterone levels (Gross and Siegel 1983, Maxwell 1993, Vleck et al. 2000).

Finally, morphological characteristics measured to provide a body condition estimate can be indicative of individual health (Brown 1996). Body mass is relative to structural size (i.e., wing, tail, culmen length), which is defined as the nutrient-reserve independent size of an individual (Piersma and Davidson 1991). Birds in good condition should have a relatively high ratio of mass to structural size (Brown 1996). We hypothesized that goshawks assessed to be in good body condition by this index would exhibit relatively high levels of blood chemistry measures (e.g., protein, calcium) indicative of adequate nutrition and low levels of cytology measures (e.g., H/L ratio) indicative of an activated immune system, high stress, or disease.

#### MATERIALS AND METHODS

This study was conducted from 1998–99 in the west-central mountains of Idaho. The study area encompassed parts of the Boise, Payette, Snake, and Salmon River drainages where elevations range from 460–3050 m (Steele et al. 1981). Public and private lands were surveyed for goshawks in Washington, Adams, Idaho, Valley, and Boise counties. The study area is a mix of lowland meadows and rangeland and higher elevation mixed coniferous forests. A more complete description of the vegetation and climatic conditions can be found in Hanauska-Brown (2000).

We trapped adult goshawks with a live Great Horned Owl (*Bubo virginianus*) lure to elicit aggressive responses from nesting goshawks and capture them in a dho-gaza net (Detrich and Woodbridge 1994). Upon capturing one adult from a nest, we reset the dho-gaza net in an attempt to catch the other adult.

We collected up to 500  $\mu$ l of whole blood after puncturing the brachial vein with a sterile lancet. Plasma was sent to IDEXX Veterinary Services Laboratory, Inc. (Portland, OR) which measured protein, cholesterol, calcium, uric acid, and three blood enzymes: AST, CK, and LDH. Additional substances were measured in fewer than 10 individuals: albumin, globulin, phosphorous, potassium, sodium, amylase, and alkaline phosphatase. Descriptive statistics only are presented for these variables as statistical analyses were deemed inappropriate.

We evaluated adjusted Sidak *P*-values (Zar 1996) from multiple pairwise comparisons (MULTTEST, SAS Release 6.12, 1996) to determine significant differences between samples collected from females and males, with  $N \geq 8$  samples for each sex (Zar 1996). Adjusted Sidak *P*-values are slightly less conservative than the Bonferroni test and control for comparison-wise error (Zar 1996). We performed Spearman correlation analyses of metabolic substances and enzymes when  $N \geq 8$  individuals for each sex. Assumptions of normality and heterogeneity were tested prior to statistical comparisons, and correlations and nonparametric tests were performed when assumptions were violated.

A blood smear was made for each bird and sent to IDEXX Veterinary Services Laboratory, Inc. The laboratory estimated total white blood cell numbers and percentages of cell types including: heterophils, eosinophils, monocytes, and lymphocytes. We used multiple pairwise comparisons and Sidak *P*-values again to evaluate mean differences between the sexes. We also performed Spearman correlation analyses for all cytological measures.

For the calculation of body condition, we measured mass, unflattened wing chord length, tail length, and culmen length. Wing chord length was measured to the tip of the longest primary, tail length to the tip of the longest rectrix, and culmen length the distance between bill tip and anterior end of the cere. The mass:structural size ratio was determined by dividing mass (g) by the product of wing chord length (mm), tail length (mm), and culmen length (mm).

Preliminary results from our data demonstrated blood metabolite and enzyme measures ( $N = 7$ ) were significantly correlated, as were white blood cell measures ( $N = 6$ ). Thus, we predicted that a suite of blood chemistry and cytology measures, rather than any individual mea-

Table 1. Metabolites and metabolic enzymes measured in Northern Goshawks in west-central Idaho during the breeding seasons of 1998 and 1999 along with ranges reported for 24 captive adult female and male goshawks (Benyon et al. 1996).

SUBSTANCE	FEMALES			MALES			BENYON ET AL. (1996)
	N	MEAN (SE)	RANGE	N	MEAN (SE)	RANGE	RANGE
Total protein (g/dL)	21	1.8 (0.2)	0.9–3.4	8	2.3 (0.2)	1.3–3.2	2.6–4.2
Cholesterol (mg/dL)	21	155.1 (14.8)	33.0–286.0	8	201.0	125.0–282.0	154.7–444.7
Uric acid (mg/dL)	21	12.6 (2.3)	1.8–40.2	8	20.9 (6.6)	9.4–63.0	8.6–14.4
Calcium (mg/dL)	21	4.7 (0.5)	1.8–9.5	8	5.9 (0.7)	3.3–8.0	8.6–10.9
Phosphorous (mg/dL)	4	3.7 (0.7)	2.4–5.0	4	2.2 (0.5)	1.2–3.7	NA
Potassium (mEq/L)	5	3.0 (0.3)	2.5–4.0	4	2.5 (0.3)	1.9–3.1	NA
Sodium (mEq/L)	5	158.6 (7.4)	136.0–181.0	4	159.8 (2.3)	155.0–165.0	NA
A/G ratio <sup>a</sup>	4	0.7 (0.1)	0.5–1.0	4	0.7 (0.03)	0.6–0.7	NA
Globulin (g/dL)	4	1.6 (0.2)	1.2–2.2	4	1.6 (0.1)	1.4–1.8	NA
Albumin (g/dL)	4	1.1 (0.1)	0.9–1.2	4	1.0 (0.1)	0.9–1.2	NA
AST (IU/L) <sup>b</sup>	21	286.2 (26.2)	108.0–561.0	8	375.5	199.0–556.0	176.0–409.0
LDH (IU/L) <sup>c</sup>	21	347.6 (47.9)	136.0–840.0	8	616.4	322.0–1040.0	120.0–906.0
CK (IU/L) <sup>d</sup>	21	127.0 (12.3)	56.0–285.0	8	160.6	56.0–412.0	218.0–775.0
Amylase (IU/L)	4	900.5 (44.1)	790.0–1006.0	4	864.0	567.0–1195.0	NA
Alk. phos. <sup>e</sup> (IU/L)	4	20.5 (4.7)	9.0–31.0	4	20.5 (4.8)	15.0–35.0	NA

<sup>a</sup> A/G = albumin/globulin ratio.

<sup>b</sup> AST = aminotransferase.

<sup>c</sup> LDH = L-lactate dehydrogenase.

<sup>d</sup> CK = creatine kinase.

<sup>e</sup> Alk. phos. = alkaline phosphatase.

sure, would best represent goshawk physiological health. We hypothesized that goshawks with higher body condition indices would exhibit a more healthy physiological profile compared to birds in relatively poor body condition. To test this hypothesis we reduced the physiological data from each bird to an individual factor score using principle components analysis (StatSoft 1998). The factor score was a multi-dimensional physiological representation of goshawk health. In this analysis, we used the factor scores associated with the first eigenvector. Subsequently, factor scores for the 10 goshawks in the best and the worst body condition (as indicated by body condition index) were evaluated using a nonparametric two-sample comparison test (Mann-Whitney *U*-test; Bailey 1995). We used the body condition index as the measure of overall condition as it was independent of the other physiological measures (Hanuska-Brown 2000) and has been used to represent health in other studies (Rising and Somers 1989, Brown 1996). We combined sexes in the analysis because there were no differences in blood chemistry, cytology, or body condition means. Two-sample comparisons were performed on factor scores derived from the entire suite of physiology variables ( $N = 13$ ), the combined metabolic and enzyme variables ( $N = 7$ ), and the suite of blood cell variables ( $N = 6$ ). For significant two-sample comparisons, factor loadings from the principle components analysis were used to help determine which variables had greatest influence on the factor scores (StatSoft 1998). Factor loadings can be interpreted as the correlations between factors (i.e., the set of factor scores

for all birds) and the individual physiological variables (StatSoft 1998). Thus, the magnitude and sign of factor loadings can be used to determine the relative importance of individual variables in explaining the two-sample comparison results.

## RESULTS

Metabolite, enzyme, and white blood cell concentrations were measured in 21 adult female and 8 adult male goshawks (Tables 1–2). There were no between-sex differences in metabolite or enzyme levels ( $P > 0.50$ ). Data on metabolic substances and enzymes from 24 captive adult goshawks (Benyon et al. 1996) are provided for comparison with our data (Tables 1–2). There were no between-sex differences in total white blood cell estimate, individual cell type, or H/L ratio for goshawks measured in this study ( $P > 0.45$ ); however, we note that the differences between female and male monocyte numbers approached significance ( $t = -3.10$ ,  $P = 0.07$ ). The correlation analysis of all blood chemistry variables showed that all metabolites and enzymes were correlated ( $P \leq 0.05$ ) with the exception of the enzyme CK. Correlation analysis of the blood cell

Table 2. White blood cell total estimates ( $\times 10^9/1$ ) and white blood cell types (%) measured in Northern Goshawks in west-central Idaho along with ranges reported for 43 captive adult female and male goshawks (Benyon et al. 1996).

WBC	FEMALES			MALES			BENYON ET AL. (1996)
	<i>N</i>	MEAN (SE)	RANGE	<i>N</i>	MEAN (SE)	RANGE	RANGE
Total estimate	21	9.8 (1.0)	4.0–20.0	8	8.7 (1.2)	4.0–13.0	4.0–11.0
Monocytes	21	2.3 (0.4)	0.0–6.0	8	5.3 (1.3)	0.0–12.0	1.0
Lymphocytes	21	38.2 (2.5)	16.0–58.0	8	46.8 (5.0)	26.0–66.0	14.0–19.0
Eosinophils	21	19.6 (1.4)	6.0–33.0	8	13.6 (2.3)	4.0–23.0	7.0
Heterophils	21	40.0 (2.4)	19.0–58.0	8	34.3 (5.0)	12.0–54.0	36.0–65.0
H/L ratio <sup>a</sup>	21	1.27 (0.17)	0.37–3.31	8	0.89 (0.23)	0.18–2.00	NA

<sup>a</sup> Heterophils to lymphocytes.

counts and white blood cell estimates demonstrated significant patterns ( $P \leq 0.05$ ) among several variables. The mean body condition ratio was  $4.39 \pm 0.08$  (range = 3.81–4.96) for females and  $4.90 \pm 0.03$  (range = 3.86–6.16) for males (Table 3). There was no significant difference between female and male body condition ratios (Wilcoxon signed rank,  $z = 1.51$ ,  $P = 0.13$ ).

The two-sample comparison test using all physiological variables ( $N = 13$ ) for the 10 goshawks with the highest and lowest body condition scores detected no difference ( $N = 20$ ,  $U = 27$ ,  $P = 0.25$ ). Similarly, the comparison using the combined metabolite and enzyme variables ( $N = 7$ ) showed no difference between goshawks of different body condition ( $N = 20$ ,  $U = 36$ ,  $P = 0.29$ ). Two-sample comparison tests on factor scores derived from blood cell variables ( $N = 6$ ), however, showed a difference between good and poor body condition goshawks ( $N = 20$ ,  $U = 3$ ,  $P < 0.01$ ). Factor loadings for each variable indicated that the H/L ratio (negative loading) was the variable primarily responsible for the observed difference. These results are consistent with paired comparisons performed on individual variables that demonstrated high lymphocyte numbers, low heterophil num-

bers, and low H/L ratios for goshawks in good condition (Hanauska-Brown 2000). There was a significant negative correlation between H/L and body condition ( $N = 30$ ,  $r^2 = -0.48$ ,  $P < 0.01$ ).

#### DISCUSSION

Concerns over the population status of goshawks throughout the intermountain west region (Kennedy 1997) prompted us to seek a means of indexing the health of adult breeding goshawks. We based this index on morphological measurements, blood chemistry, and cytological characteristics. We focused on these variables because other researchers have questioned the assessment of population viability using occurrence, density, or annual productivity estimates (e.g., Van Horne 1983, Franklin et al. 2000).

Our data represent baseline metabolite, enzyme, and blood cytology levels for a sample of free-ranging goshawks in central Idaho. The only other baseline data available for goshawks are from “normal, healthy” male and female goshawks of various ages, housed in rehabilitation centers (Benyon et al. 1996). Captive birds are provided with a continuous food source, but also experience a multitude of stressors (e.g., confinement and handling) not

Table 3. Morphology measures from Northern Goshawks in west-central Idaho.

MORPHOLOGY	FEMALES			MALES		
	<i>N</i>	MEAN (SE)	RANGE	<i>N</i>	MEAN (SE)	RANGE
Mass (g)	21	1006.8 (14.8)	870–1134	8	766.9 (15.7)	710–824
Wing (mm)	21	353.8 (1.7)	341–372	8	325.4 (3.8)	310–337
Tail (mm)	21	261.7 (1.2)	249–272	8	225.1 (2.4)	212–232
Culmen (mm)	21	24.8 (0.3)	23.2–29.9	8	21.1 (0.3)	19.7–25.1

present in the wild (Stein et al. 1998). Captive birds receive a steady, high protein diet (Gee et al. 1981, Garcia-Rodriguez et al. 1987) that provides an adequate supply of amino acids for the synthesis of blood proteins (Ferrer et al. 1987). Indeed, total protein was higher for the captive goshawks than for the birds sampled in this study. Wild goshawks may experience periods of fasting (Newton 1979) and may be forced to specialize on certain prey species (Newton 1979, Younk 1996) leading to a mineral or dietary deficiency (Snyder and Terry 1986). Birds in the wild engage in activities (e.g., courtship, territoriality, nestling care) that reduce the time available for foraging, which also may lead to dietary deficiencies, particularly in poor-quality habitat.

Calcium levels were lower in wild than in captive goshawks (Benyon et al. 1996), which also can be attributed to dietary differences. Captive birds receiving chicken carcasses, rabbits, and other 'whole' food items (Gee et al. 1981) obtain calcium from ingesting bones and other body parts. Conversely, captive birds fed only red meat (including muscle, heart, and liver) were found to be deficient in calcium (Graham and Halliwell 1986). Dietary calcium in the wild comes from a varied diet including whole prey items (Graham and Halliwell 1986, Squires and Reynolds 1997). However, wild birds may not be able to capture a steady diet of suitable prey, and low levels of calcium may reflect fasting periods (Halliwell 1981).

The mean uric acid level for females was within the range reported for captive goshawks, but the mean for males was above the reported range (Benyon et al. 1996) by more than 6 mg/dL. Uric acid, a nitrogenous waste product, is typically elevated in birds with high protein diets (Bell and Sturkie 1965, Gee et al. 1981). High uric acid levels in free-ranging birds can also be attributed to decreased body condition or food stress (Handrich et al. 1993, Balbontín and Ferrer 2002, Casado et al. 2002). Males are under high energy demands during the breeding season as they hunt for the adult female and young, as well as for themselves. Mean cholesterol levels from sampled birds were within the range of cholesterol values reported for captive goshawks (Benyon et al. 1996).

The three enzymes we measured are quite variable in captive goshawks (Benyon et al. 1996). Such individual differences are common in avian species due to natural variation in habitat, genetics, body condition, environmental influences, and

other factors (Gee et al. 1981, Hoffman et al. 1985, Stein et al. 1998). For example, Stein et al. (1998) documented more variability in wild populations of American Kestrels (*Falco sparverius*) and Red-tailed Hawks (*Buteo jamaicensis*) than in captive birds. Mean levels of AST and LDH measured in this study were within the wide range of enzyme levels in captive goshawks (Benyon et al. 1996). However, lower levels of CK were measured in wild versus captive birds (Table 1). Average or low CK levels can indicate enhanced muscle strength and endurance (Apple and McGue 1983, Knuth and Chaplin 1994). For example, flight training in captive Red-tailed Hawks improved the structural integrity of muscles and lowered plasma CK levels (Knuth and Chaplin 1994). The low levels of CK observed in this study may reflect increased muscle strength and flight endurance (Apple and McGue 1983, Knuth and Chaplin 1994) in the wild birds.

Individual variation due to proximate factors may play a major role in variable blood chemistry measures, but many other influences can affect the measurement of health in wild birds. Garcia-Rodriguez et al. (1987) demonstrated diurnal and circadian rhythms of calcium, uric acid, and cholesterol levels in raptors. We could not control the time of day of sample collection because of the logistical constraints of trapping breeding goshawks. Time of sample collection in this study ranged from 0700–1900 H. Variation in the levels of protein, uric acid, calcium, and cholesterol in individual goshawks in this study, therefore, may be attributed to recent feeding, fasting periods, or circadian and diurnal rhythms.

Geographic location also may affect blood chemistry measures. Soil types and environmental contamination affect physiology, as raptors incorporate locally-obtained nutrients and toxins from their prey items (Hoffman et al. 1985, Ferrer and Dobado-Berrios 1998). Differences in total protein, uric acid, cholesterol, and calcium were found between two populations of Adalbert's Eagle (*Aquila adalberti*) foraging in different regions of Spain (Ferrer and Dobado-Berrios 1998). Qualitative differences in each population's diet were suggested as the cause of the physiological differences (Ferrer and Dobado-Berrios 1998). Similarly, differences in prey abundance, prey availability, and habitat in central Idaho may be reflected in goshawk blood chemistry measures. For example, goshawks in the western region of the study area likely consumed more ground squirrels (*Spermophilus* spp.)

than goshawks in the northern regions, which probably consumed more birds (pers. observ.).

Goshawk samples from this study showed increased numbers of lymphocytes, monocytes, and eosinophils compared to the ranges reported for captive goshawks (Benyon et al. 1996). Increased levels of particular leukocytes or white blood cell total estimates can indicate physiological stress or decreased immunocompetence (Smith and Bush 1978, Kontecka et al. 1999). A shift from normal values in any direction can indicate disease (Campbell and Dein 1984). Blood loss or tissue damage at the collection site can also affect white blood cell counts. Our blood collection method may have overestimated circulating white blood cells because hemorrhage induced by puncturing the wing vein produces tissue damage that triggers clotting and attracts white blood cells. Capillary blood in humans (e.g., blood collected into capillary tubes after finger pricks) has higher white blood cell counts than blood collected intravenously (Daece et al. 1988, Kayiran et al. 2003). We minimized this potential change in white blood cells by collecting blood immediately upon puncturing the vein.

The H/L ratio was significantly different between birds in relatively good, compared to birds in relatively poor, condition. Such differences suggest decreased immunocompetence or higher rates of stress in birds of poor condition. The H/L ratio has been shown to be a less variable indicator of stress than differential counts and more reliable than corticosterone levels in reflecting chronic stress (Vleck 2002).

One of the main objectives of this work was to explore the relationships between a noninvasive body condition index and the blood chemistry and cytology of wild breeding goshawks. We expected less variability in the blood parameters we observed and stronger relationships among the various physiological measurements and body condition than we detected. Our small sample size undoubtedly influenced variability in our measures, and some of the assumptions underlying the mass/length technique we used to assess body condition have recently been questioned (Green 2001). Furthermore, considering that all birds sampled were breeding, our "poor" condition birds may not have been in poor condition relative to the overall wild population. Despite these concerns, our finding that the relative differences in body condition among breeding birds were reflected in the H/L

ratio highlights the potential sensitivity and utility of this technique.

Our results also illustrate the difficulty in evaluating the physiological state of a bird at any given time. Metabolite and enzyme levels appear to be influenced by too many external and individual factors to serve as accurate barometers of individual health. Despite the limitations inherent in physiological studies of wild raptors such as ours, the H/L ratio and body condition index hold promise as indicators of adult goshawk health. Further study of free-living raptor physiology is warranted to assess the value of the H/L ratio as a measure of individual health.

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## PRODUCTIVITY AND POPULATION TRENDS OF OSPREYS IN THE KAWARTHA LAKES REGION, ONTARIO, 1978–2001

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**ABSTRACT.**—Osprey (*Pandion haliaetus*) populations were monitored extensively in the Kawartha Lakes region of Ontario, Canada, from 1978–2000, and intensively monitored in a subsample of these lakes, particularly Sturgeon Lake, (1991–2001). The number of occupied nests within the Kawartha Lakes region increased from 18 nests in 1978 to 89 nests in 1992 following the ban of organochlorine pesticides and the provision of artificial nesting structures. Surveys in 1996 and 2000, however, revealed 78 and 66 occupied nests, respectively, indicating a recent population decline or stabilization. The Osprey breeding colony at Sturgeon Lake followed very similar trends in nest numbers, peaking in the mid-1990s, followed by a decline and apparent population stabilization by 2001. The mean annual productivity in this colony between 1986 and 2001 was 1.17 chicks per occupied nest, and exceeded the reported replacement level of 0.8 chicks per nest. Population fluctuations and trends in overall annual production of fledglings were similar between the Sturgeon Lake breeding colony and birds breeding at lower densities in neighboring lakes. Estimated potential recruitment to the Sturgeon Lake colony, based on the number of fledglings produced 3–5 years previously, was positively related to population size in a given year ( $r^2 = 0.65$ ); but very few banded nestlings returned to the colony as breeders. Factors affecting productivity and population trends seem to operate similarly at local and regional scales. As very few fledged juveniles returned to the Sturgeon Lake colony as breeders, yet the population was maintained, we suggest there was probably considerable dispersal of juveniles among local natal sites throughout the region.

**KEY WORDS:** *Osprey; Pandion haliaetus; Ontario; population; productivity.*

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### PRODUCTIVIDAD Y TENDENCIAS POBLACIONALES DE ÁGUILAS PESCADORAS EN LA REGIÓN DE LOS LAGOS KAWARTHA, ONTARIO, 1978–2001

**RESUMEN.**—Las poblaciones de Águila pescadora (*Pandion haliaetus*) fueron monitoreadas extensivamente en la región de los lagos Kawartha de Ontario, Canadá, desde 1978–2000, y monitoreadas intensivamente en una submuestra de estos lagos, particularmente en el Lago Esturión, (1991–2001). El número de nidos ocupados dentro de la región de los lagos Kawartha incremento desde 18 nidos en 1978 a 89 nidos en 1992 luego de la prohibición de los pesticidas organoclorados y de proveer estructuras artificiales para anidación. Los estudios en 1996 y 2000 sin embargo, revelaron 78 y 66 nidos ocupados, respectivamente, indicando un reciente declive en la población o su estabilización. La colonia de anidación del águila pescadora en el Lago Esturión siguió tendencias muy similares en el número de nidos, alcanzando su pico a mediados de los 90s, seguido por un declive y una aparente estabilización de la población para el 2001. la productividad media anual en esta colonia entre 1986 y 2001 fue 1.17 polluelos por nido ocupado, y excedió el nivel de reposición reportado de 0.8 polluelos por nido. Las fluctuaciones de la población y las tendencias en la producción anual global de juveniles fueron similares entre la colonia de anidación del Lago Esturión y las aves que anidaron en más bajas densidades en lagos vecinos. El reclutamiento potencial estimado para la colonia del Lago Esturión, basada en los números de juveniles producidos 3–5 años atrás, se relacionó positivamente con el tamaño de la pobla-

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ción en un año dado ( $r^2 = 0.65$ ); aunque, muy pocos juveniles anillados retornaron a la colonia como reproductores. Los factores que afectan la productividad y las tendencias poblacionales parecen operar similarmente a escalas locales y regionales. A pesar que muy pocos de los juveniles levantados retornaron a la colonia del Lago Esturión como reproductores, aun la población se mantuvo, sugerimos que allí probablemente hubo una considerable dispersión de juveniles entre los sitios natales locales de toda la región.

[Traducción de César Márquez]

Osprey (*Pandion haliaetus*) populations, like those of many fish-eating birds throughout North America, declined due to pesticide-related effects in the 1950s and 1960s, subsequently recovering following restrictions in the use of organochlorine pesticides and other chlorinated hydrocarbons in the early 1970s (Spitzer et al. 1983, Ewins et al. 1995, Houghton and Rymon 1997). Many programs have been implemented to monitor Osprey populations in North America and to improve recruitment (Spitzer et al. 1983, Steidl et al. 1991, Castellanos and Ortega-Rubio 1995, Ewins 1996, Witt 1996).

In 1978, the Ontario Ministry of Natural Resources began monitoring the Osprey population of the Kawartha Lakes region of south-central Ontario within the lower Great Lakes basin. Once common throughout the area, this population suffered a decline during the previous decades and the main objectives at that time were to assess and monitor population trends, and to enhance population growth and reproductive success through the creation of artificial nesting sites (Barker 1986, 1988). In 1991, the Canadian Wildlife Service initiated a study to assess the suitability of Ospreys as indicators of chlorinated hydrocarbon contaminants within the Great Lakes basin, as well as to determine if Ospreys breeding on the Great Lakes were continuing to suffer any adverse effects of these contaminants (Ewins et al. 1995, Martin et al. 2003). Ospreys on several lakes within the Kawartha Lakes system were also monitored to provide comparative data on what was assumed to be a clean inland breeding population. However, PCB concentrations in the eggs and plasma of nestling Ospreys in Sturgeon Lake were found to be considerably higher than those of birds nesting on the Canadian Great Lakes and in the Kawartha Lakes region (Martin et al. 2003). By 1996, when it had been demonstrated that contaminant levels were low elsewhere in the study area, the Canadian Wildlife Service ceased Osprey monitoring activities in the Great Lakes basin except in the Stur-

geon Lake colony, where monitoring was continued until 2001.

We report here the population growth and fluctuations of Ospreys in the Kawartha Lakes region from 1978–2001 including data from survey reports from Ontario Ministry of Natural Resources from 1978–91 with those collected by the Canadian Wildlife Service and Kawartha Lakes Friends of the Osprey Society from 1991–2001. As well, we examine post-recovery patterns of productivity and adult returns of Ospreys in a selected small breeding colony.

#### STUDY AREA AND METHODS

The study area was primarily within the Kawartha Lakes region in south-central Ontario, a transitional zone bordering the Canadian Shield and the Great Lakes-St. Lawrence Lowlands ecozones (Fig. 1). This area consists of a chain of 17 shallow, mainly eutrophic lakes interconnected by a series of rivers and man-made canals. The water levels are controlled in some areas through the use of dams, and the littoral regions of some lakes were flooded as a result of the construction of the Trent-Severn Waterway over the last two centuries. Although the total area is approximately 6600 km<sup>2</sup>, much of this land is under agricultural use and unsuitable Osprey habitat.

The entire Kawartha Lakes region, as well as adjoining portions of the Trent-Severn Waterway including Scugog Lake, the Otonabee River, and Rice Lake, were searched for Osprey nests by Ontario Ministry of Natural Resources in 1978–80, 1985, 1986, 1988, and 1991, and by the Canadian Wildlife Service in 1992. Suitable nesting habitat was searched in early June along routes with known nest sites, both by boat and by vehicle on roads near the lakes. Nest checks were done with a spotting scope or an extendible pole with an attached mirror. Second visits in July were made to nests throughout the region in 1986, 1988, and 1991; we calculated productivity on a per nest basis for these years (M. Barker unpubl. data). In 1988, 1991, 1996, and 2000, the entire region was also surveyed by fixed-wing aircraft from which nests may be viewed easily (Fuller and Mosher 1987), during the nestling period (mid-June). Furthermore, we had access to nest site information collected by local Osprey enthusiasts. Thus, we believe that the majority of occupied Osprey nests in the Kawartha Lakes region were located during these surveys.

A subset of Kawartha Lakes at the northwest end of the chain, including Sturgeon, Balsam, Canal, and Mitchell lakes, and Emily Creek (44°36'N, 7°20'W to 44°36'N, 79°05'W; Fig. 1) were surveyed annually by boat from

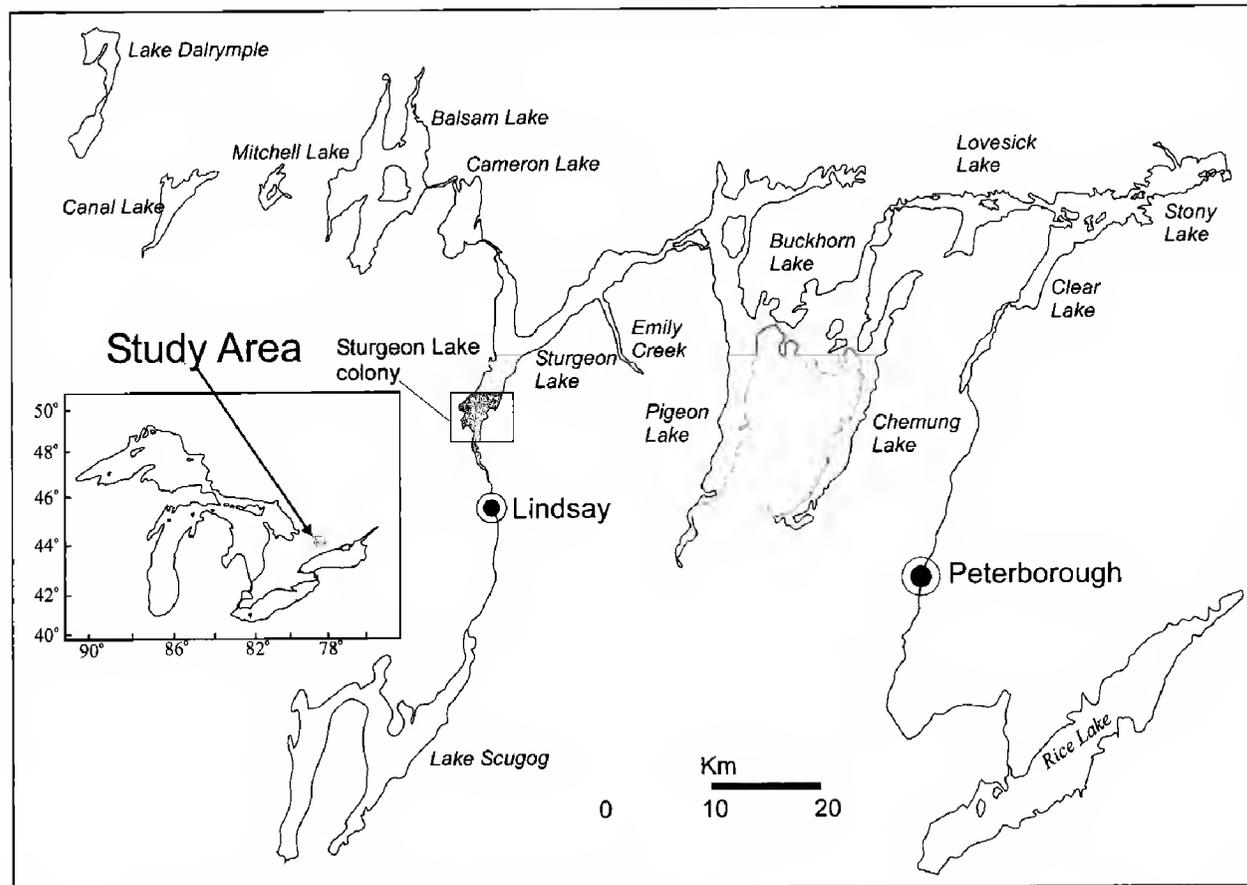


Figure 1. Kawartha Lakes study area indicating its position in the Great Lakes basin and the Sturgeon Lake Osprey breeding colony.

1991–96. The south end of Sturgeon Lake near the town of Lindsay was the site of a fairly dense nesting “colony” of Ospreys, in which up to 17 occupied nests occurred within a 3.5 km radius. In 1991, PCB contaminant levels in this colony were determined to be unusually high (egg  $x = 7.1$  ppm; range: 2.02–26.5 ppm wet weight), and thus intensive annual monitoring in this colony continued until 2001.

Ospreys in Sturgeon Lake and the other northwestern lakes nested mostly on flooded stumps, whose tops were within a meter of the water, and on man-made elevated platforms, although some nests were located on live or dead trees and utility poles (P. Martin unpubl. data). All known nests on each lake were visited twice a year by boat. The first visit was made in late May or early June, during mid- to late incubation, and was a check for nest occupancy. We did not visit during the laying period or early incubation to avoid causing nest abandonment. Thus, it was possible some nests were destroyed before our initial visit, but this probability remained constant among years and probably did not represent a bias in monitoring. The second visit was in late June or early July when the chicks were 5–6 wk old (banding age). At this time we banded nestlings with both a US Department of Interior issued aluminum numeric band and a single colored plastic band. Each color of plastic band indicated a yearly cohort, and was used by all Osprey banding projects throughout the Great Lakes. The number of chicks remaining in the nest at this time were considered to be fledged although it is possible that some mortality may have occurred between banding and nest departure.

The occupied nest (ON) was used as the census unit, and was defined as a nest attended by a pair of Ospreys, regardless of any breeding attempt. Therefore, an occu-

ried nest was considered equivalent to a pair of potential breeders. A successful nest was one in which at least one young was reared to banding age. Productivity rate was defined as the number of chicks reared to banding age per occupied nest. Production was the total number of chicks fledged in the overall population or group (Steenhof and Kochert 1982). We calculated the adjusted recruitment productivity (ARP), which is an estimate of the potential for new breeders in a given year (Spitzer 1980, Spitzer et al. 1983). Spitzer et al. (1983) calculated that 50% of Ospreys first bred at 3, 30% at 4, and 20% at 5 yr of age. Thus, the number of potential breeders recruited into the population in a given year was weighted by these proportions for the number of fledglings 3, 4, and 5 yr prior, using the equation:

$$[1] \quad (0.5 \times Y_{t-3} + 0.3 \times Y_{t-4} + 0.2 \times Y_{t-5}),$$

where  $Y$  is the number of fledglings, and  $t$  is the year. To convert recruitment to productivity, the weighted sum of fledglings was divided by the number of occupied nests the year before their recruitment,  $O_{t-1}$ .

From 1992–96, most adults (all female) Ospreys at Sturgeon Lake were trapped using a dome-shaped noose carpet (Bloom 1987) tied over the nests during late incubation when nest-site tenacity was strong and chances of nest abandonment were low. The banding operation took less than 15 min and birds were always observed to return to their nest. Banded adults were of unknown age. Each year during the initial nest check, an effort was made to read the large 2-digit bands of breeding birds with  $8 \times 42$  binoculars or a spotting scope. Birds were not recaptured on the nest for the purpose of determining their identity.

**Data Analysis.** We estimated changes in the size of the

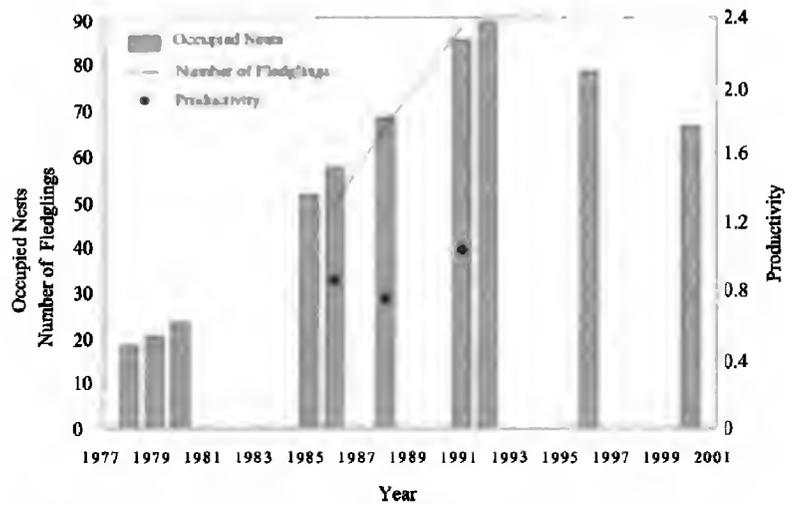


Figure 2. Productivity, number of occupied nests, and total number of Osprey fledged 1978–2001 in the entire Kawartha Lakes region, Ontario.

Osprey population of the entire Kawartha Lakes region study area from 1978 onward, and in the Sturgeon Lake colony from 1986–2001, using least squares regression.

We evaluated the proportion of fledglings from Sturgeon Lake and the four neighboring lakes that later returned to breed. We included only juveniles that were banded from 1991–98, as juveniles fledged after 1998 would not be expected to breed until at least 2002. We compared recruitment and ARP with the annual percent change in population size using least squares regression. Only 6 yr of data (1996–2001) were available for analysis, as a minimum of 5 yr prior to the given year were required for the estimation of ARP. Data were insufficient for other areas within the Kawartha Lakes region to conduct this analysis. We calculated the encounter rates of adults in Sturgeon Lake from 1994–2001. The identities of breeding birds at each nest were tabulated from 1994 onward. Adults were identified by band numbers where possible, or were listed as either unbanded or unknown (band unreadable or unable to determine presence of band). The presence of plastic color bands was recorded.

Weather data were collected by Environment Canada at the Peterborough Airport from 1994–2001 and were accessed on the Ontarioweather webpage (Patrick 2002). For the months of May and June, we calculated the means for the daily mean, maximum, and minimum temperatures, windspeed, and dewpoint. We examined the relationships between spring weather variables and Osprey population dynamics at Sturgeon Lake.

## RESULTS

**Kawartha Lakes Region: Population Size.** From 1978–92 there was a four-fold increase in the number of occupied nests throughout the Kawartha Lakes region study area (Fig. 2;  $r^2 = 0.997$ ,  $P < 0.001$ ), representing a mean annual rate of increase of 10.3%; this rate of increase was greatest between 1980 and 1985 (15.9%). However, the population had declined 9% by 1996 and 22% by 2000, relative to maximum nest numbers of 1992.

**Sturgeon Lake: Population Size.** The number of

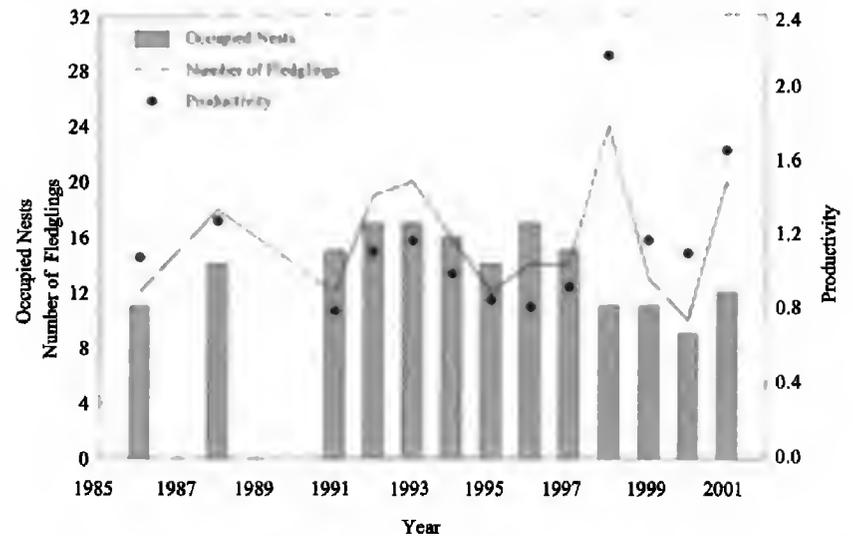


Figure 3. Productivity, number of occupied nests, and total number of Osprey fledged 1985–2001 at the Sturgeon Lake colony, Ontario.

occupied nests within the Sturgeon Lake colony increased from 11 in 1986 to 17 in both 1992 and 1993 ( $r^2 = 0.91$ ,  $F_{1,2} = 21.37$ ,  $P = 0.04$ ), fluctuated through 1996, then declined steadily to a low of nine in 2000, although there was a slight resurgence of nests in 2001 ( $r^2 = 0.69$ ,  $F_{1,7} = 15.31$ ,  $P = 0.006$ ; Fig. 3).

**Sturgeon Lake: Productivity.** Mean annual productivity at the Sturgeon Lake colony ranged from 0.8–2.18, with a mean of 1.17 fledglings per occupied nest (Fig. 3). There was no relationship between the total number of fledglings produced in the colony and the number of occupied nests in the same year, from 1986–2001 ( $r^2 = 0.05$ ,  $P = 0.45$ ); however, nest success was significantly related to mean annual per nest productivity ( $r^2 = 0.72$ ,  $P < 0.001$ ). The percent change in population size was positively correlated to ARP ( $r^2 = 0.71$ ,  $F_{1,4} = 9.93$ ,  $P = 0.035$ ). Similarly, there was a positive relationship between potential recruitment and the population size ( $r^2 = 0.65$ ,  $F_{1,4} = 7.38$ ,  $P = 0.053$ ).

Detailed population and productivity data were available for Ospreys nesting at much lower densities in four adjacent lakes within the northwestern portion of the Kawartha Lakes (Balsam, Canal, and Mitchell lakes, and Emily Creek), providing a local comparison to the colony at Sturgeon Lake. The total number of occupied nests from these lakes ranged from 8–20 nests (Fig. 4). The population fluctuations observed in the Sturgeon Lake colony between 1986 and 2000 were similar to those in nearby lakes ( $r^2 = 0.44$ ,  $P = 0.052$ ) and both groups reached their lowest numbers in 2000. The annual changes in the total number of fledglings produced per year was similar between the

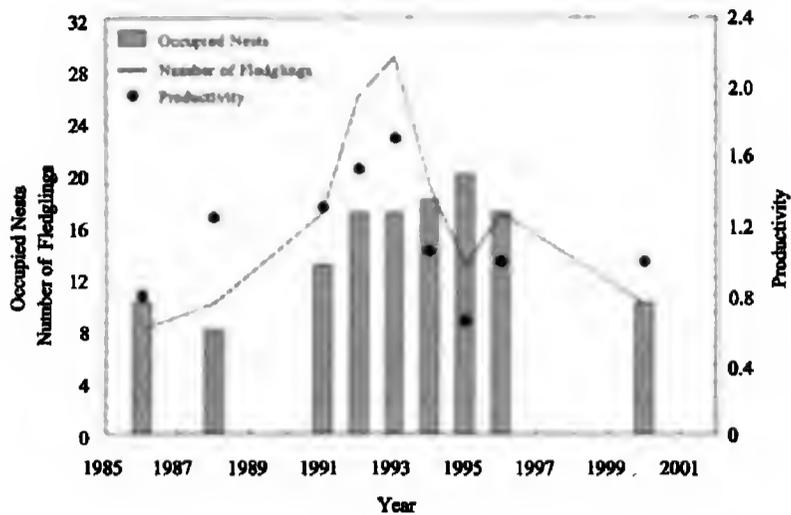


Figure 4. Productivity, number of occupied nests, and total number of Osprey fledged 1985–2001 at Balsam, Canal, and Mitchell lakes, and Emily Creek, Ontario.

Sturgeon Lake colony and the neighboring lakes ( $r^2 = 0.51$ ,  $F_{1,7} = 7.28$ ,  $P = 0.03$ ; however, mean annual productivity per nest was not correlated between the two areas ( $r^2 = 0.34$ ;  $P = 0.17$ ). Nevertheless, mean annual productivity did not differ between the two groups pooled across years (dependent  $t$ -test,  $t_8 = 1.73$ ,  $P = 0.13$ ).

**Resightings of Banded Juveniles.** In the northwestern portion of the Kawartha Lakes a total of 179 chicks were banded between 1991 and 1998, and were thus available for recruitment into the breeding population by 2001, our last year of study. Only one bird banded as a juvenile was confirmed breeding at Sturgeon Lake; a total of three are known to have returned to the Kawartha Lakes region. As well, in 1997, a male Osprey that had been banded with a plastic color band as a juvenile in 1993 bred successfully at Sturgeon Lake. However, the natal location of this bird is unclear, as in 1993, color banding of annual cohorts was being conducted throughout the Great Lakes basin. Band recoveries were obtained for three other birds banded as juveniles. Two were found dead in their first autumn in Ohio and Michigan. The third was found dead on its wintering grounds in Lesser Antilles during its second winter.

**Returns of Banded Adults.** Of 18 adults banded at the nest in the Sturgeon Lake colony between 1992 and 1996, 15 were females. Two males returned to breed at the colony three years subsequent to banding, while one was only observed in the year of banding. Of the females, two bred at the colony only the year of banding and were not observed in subsequent years. There were pronounced declines in the proportion of adults returning on years 2 and 6 (Fig. 5). Females bred at

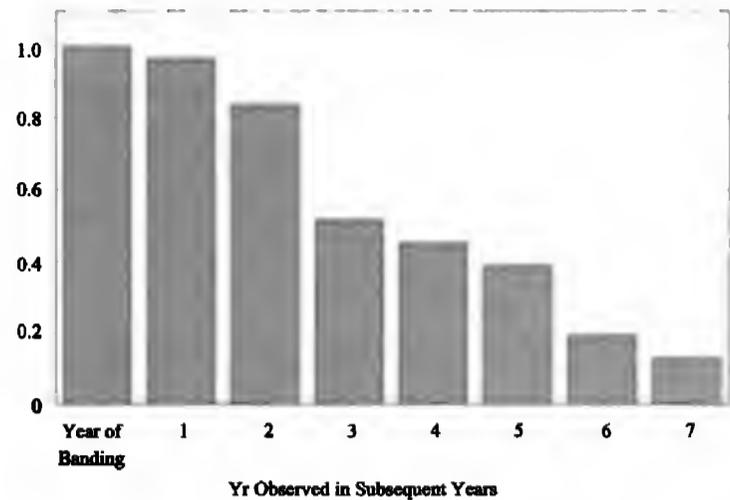


Figure 5. Cumulative proportion of banded adult Ospreys returning per year from year of banding.

the colony a modal frequency of 2 yr (six individuals) and a mean of 3.44 (SD = 2.09) years; seven was the maximum number of years females bred at the colony (two individuals). Thus, the modal minimum age at last breeding was 5 yr and the maximum age was 10 yr. Females with a high proportion of successful nests tended to have many breeding attempts ( $r^2 = 0.64$ ,  $F_{1,12} = 8.35$ ,  $P = 0.014$ ). However, females were equally likely to be successful in the final year observed breeding (65.4%) than any other year of breeding (64.3%;  $\chi^2 < 0.005$ ,  $P = 0.97$ ). There was an equal probability that successful and unsuccessful females would switch nest sites the subsequent year ( $\chi^2 = 2.23$ ,  $P = 0.14$ ).

**Effects of Weather.** During the years 1994–2001, we found significant relationships between mean productivity of Ospreys in the Sturgeon Lake colony and weather conditions for May, but not for June. There was a significant positive relationship between productivity and mean daily temperature ( $r^2 = 0.63$ ,  $F_{1,6} = 10.41$ ,  $P = 0.018$ ), mean maximum daily temperature ( $r^2 = 0.48$ ,  $F_{1,6} = 5.50$ ,  $P = 0.058$ ), mean minimum daily temperature ( $r^2 = 0.65$ ,  $F_{1,6} = 11.30$ ,  $P = 0.015$ ) and mean dewpoint ( $r^2 = 0.67$ ,  $F_{1,6} = 12.02$ ,  $P = 0.013$ ). There were no significant relationships between productivity and mean, maximum, and gust wind speed.

## DISCUSSION

Osprey populations throughout North America have rebounded from the declines of the mid-century induced by organochlorine pesticide exposure, following the ban on the use of these chemicals (Postupalsky 1989, Ewins et al. 1995, Henny and Kaiser 1996, Houghton and Rymon 1997). Although the Osprey population of the Kawartha

Lakes region in the lower Great Lakes basin has also expanded by four-fold from 1978–92, its status pre-DDT era is unknown. By the year 2000, however, this regional population had contracted to 74.2% of its maximum expansion achieved in 1992. Similar leveling off or slight decline of Osprey populations, following periods of rapid growth, has occurred elsewhere in the Great Lakes basin. Osprey populations increased dramatically during the post-DDT era in Michigan (Postupalsky 1989); however, during the latter half of the 1990s, populations in the Upper and Lower Peninsula declined by 22% and 9%, respectively (S. Postupalsky pers. comm.). An Oregon population experienced similar increases in size throughout the 1970s and 1980s as contaminant levels declined (Henny and Kaiser 1996). In contrast, instead of reaching a plateau, this population more than doubled from 1995–2001 (USGS 2002). The authors suspect that recent acceptance by the Ospreys of utility poles and channel markers as nesting sites accounts for this sudden population increase (USGS 2002), suggesting that appropriate nest sites were limiting previously. Sturgeon Lake appeared to have a surplus of established nesting sites. Given that there were always some unoccupied nest platforms, and that we frequently observed the construction of new nests on exposed stumps, it seems that availability of nest sites probably was not limiting this population of Ospreys. A common strategy in encouraging Osprey population expansion has been to erect artificial nest sites (Ewins 1996, Witt 1996). Our results suggest that even at high population densities, nest site availability was not a limiting factor. The Sturgeon Lake breeding colony seems to have reached a plateau and a management strategy based on further construction of artificial nests is unlikely to result in further growth. Nevertheless, there is anecdotal evidence (R. Skitch, Ontario Power Generation pers. comm.) that in the eastern section of the Kawartha Lakes region (Rice and Stoney lakes), there has been a dramatic increase in the use of utility poles in 2001 and 2002. This tendency to use utility poles may have facilitated an increase in the regional population overall.

Although the eggs of Sturgeon Lake Ospreys were found to contain relatively elevated egg PCB concentrations when examined in 1991–92, there was no apparent correlation between productivity with either DDE or PCB concentrations throughout the Canadian Great Lakes basin, including the Kawartha Lakes region (Martin et al. 2003). Un-

fortunately, there is neither contaminant nor population data available for Canadian Great Lakes Osprey from the 1960s and 1970s with which to document a causal relationship. Nevertheless, it is unlikely that current levels of contamination adversely impact reproduction in the Sturgeon Lake colony (Poole 1989).

The proportion of nests that were successful as opposed to the total number of nests occupied in a given year was strongly correlated to mean annual productivity, suggesting that the factors contributing to nest failure also affected partial mortality within a nest. Depredation was clearly evident for three complete clutches of eggs, whereas four complete broods of chicks were obviously depredated. Although only three Osprey nestlings were actually found dead in the nest at time of banding that were suspected to have died from starvation, there were many instances of partial brood reduction (in either the egg or nestling stage). In some cases, single unhatched eggs remained in the nest at banding, possibly as a result of infertility or inclement weather during incubation. The latter possibility is supported by the positive relationship between May temperatures and productivity. Flooding caused by storms, as well as controlled water level changes as a result of damming, are suspected to be the cause of destruction of nest contents in low, overwater stump nests (P. Martin, M. Barker, and B. Puxley unpubl. data). Other studies have also indicated that poor weather conditions impacted fledging success of Ospreys (Poole 1982, Machmer and Ydenberg 1990) but see Stinson (1978) and Stinson et al. (1987). Trends in annual production of successful fledglings were similar between Sturgeon Lake and the neighboring lakes, suggesting that regulatory factors (e.g., weather) operate at a scale that encompasses the entire Kawartha Lakes region in a given year.

Estimates of the productivity required to maintain stable populations for Ospreys ranged from 0.8 fledglings per nest (Spitzer et al. 1983) to between 1.22 and 1.30 (Henny and Ogden 1970). An important distinction between these studies (Spitzer et al. 1983, Henny and Ogden 1970) was that the populations studied had very different trajectories of growth. The population studied by Spitzer (1980) was recovering, while those examined by Henny and Ogden (1970) were generally declining. Nevertheless, 0.8 fledglings per nest is typically accepted as the productivity “break-even” point for Ospreys in the northeastern United States and

Canada (Ewins et al. 1995). This rate was met or exceeded in each of the 11 years of study at the Sturgeon Lake colony and in all but one year in the neighboring lakes. Adjusted recruitment productivity (ARP), which was based on fledgling production from 3–5 yr prior to any given year, was highly correlated with population size. Thus, reduced productivity from 1995 through 1997 directly correspond to declines in the breeding population from 1998 through 2000. Also, the sudden increase in productivity in 1998 corresponded to the increased breeding population of 2001.

The productivity “break-even” point does not account for differences in adult survivorship or dispersal. Although we banded adult birds we were unable to estimate longevity as the ages of the birds were unknown at time of banding. Of 15 adult females banded, two marked in 1992 were observed to breed on Sturgeon Lake for seven seasons. Assuming they were first time breeders when initially banded in 1992 and were 3 yr old at time of banding, these birds were a minimum of 10 yr old during their last known breeding year of 1998. Of 17 known-aged females who reached breeding age banded as nestlings in a recovering population in Michigan during the 1970s and 1980s, five reached or exceeded 10 yr of age whereas 12 died prior to attaining 10 yr of age (Postupalsky 1989). Thus, our rate of 13% of females exceeding 10 yr of age is lower than that of 35% in the Michigan birds, suggesting possible problems in adult survival, although data are limited.

Less than 2% of 179 fledglings that were banded in the northwestern portion of the Kawartha Lakes region from 1991–98 were later encountered breeding in the Kawartha region. After 1996, however, only the Sturgeon Lake colony was monitored intensively, so opportunities to observe banded birds were limited. Nevertheless, even at Sturgeon Lake, where 70% of all juveniles produced were banded over the 11 yr period, less than 3% of the 94 birds returned to this colony as breeders. This suggests that new breeders at the Sturgeon Lake colony arrived via dispersal from other natal sites, either in the Kawartha Lakes region or from more distant populations. The distances among the Kawartha Lakes are certainly within typical dispersal distances of Ospreys (Spitzer et al. 1983, Postupalsky 1989). That the adjusted recruitment productivity at Sturgeon Lake was highly correlated with population size, despite the fact that very few juveniles produced there actually return to the col-

ony to breed, suggests that trends in localized groups of breeders are indicative of regional population fluctuations.

The growth of the Kawartha Lakes population of Ospreys has stabilized or slightly declined in the last decade, but as rates of productivity seem to be adequate for reproduction, we do not expect further declines. Factors affecting productivity and population trends appear to operate similarly at the local and regional scale. As very few fledged juveniles returned to the Sturgeon Lake colony as breeders, yet the population was maintained, we suggest there was probably considerable dispersal of juveniles among local natal sites throughout the region. Although we did not detect nest site limitation at the Sturgeon Lake colony, it is possible that recent observed increases in the usage of utility poles in another portion of the Kawartha Lakes region may have fostered population increases throughout the region.

#### ACKNOWLEDGMENTS

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## HABITAT CHARACTERISTICS OF GREAT GRAY OWL SITES IN THE SISKIYOU MOUNTAINS OF SOUTHWESTERN OREGON

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ABSTRACT.—Great Gray Owls (*Strix nebulosa*) were located at 92 sites, 1991–97, on the northern slopes of the Siskiyou Mountains of southwestern Oregon. We characterized habitat at the 28 sites including locations with nests (10), recently-fledged young (5), pairs (10), and multiple-year observations (3). Unique to Great Gray Owl populations in western North America, 90% of the nests and 63% of all observations occurred below 915 m in elevation. Owls occurred most often in stands of mature and old growth Douglas-fir (*Pseudotsuga menziesii*) adjacent to open canopy woodland, chaparral, and natural meadows. Oregon white oak (*Quercus garryana*) was the dominant tree in the woodlands. Three species associated with woodlands and meadows, California voles (*Microtus californicus*), moles (*Scapanus* spp.), and Botta's pocket gophers (*Thomomys bottae*) comprised 73.3% of the prey items identified. While all nests were near some relatively open habitat, only four of 10 nests were within 800 m of recent clearcuts. All nests were stick platforms in live trees. The distribution of Great Gray Owls in the Siskiyou Mountains suggests that they may have a broader distribution in the western United States than previously reported, particularly at elevations below 915 m.

KEY WORDS: *Great Gray Owl*; *Strix nebulosa*; food habits; habitat selection; nest sites; Oregon; Siskiyou Mountains.

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### CARACTERÍSTICAS DEL HÁBITAT DEL GRAN BÚHO GRIS EN LAS MONTAÑAS SISKIYOU DEL SURESTE DE OREGON

RESUMEN.—Los grandes búhos grises (*Strix nebulosa*) fueron localizados en 92 sitios entre 1991–97, en las laderas septentrionales de las Montañas de Siskiyou en el sureste de Oregon. Caracterizamos el hábitat en los 28 sitios con nidos (10), volantes (5), parejas (10), y observaciones múltiples anuales (3). Solo para las poblaciones de Grandes Búhos Grises en el oeste de Norteamérica, el 90% de los nidos y el 63% de todas observaciones ocurrieron por debajo de 915 m de elevación. Los búhos se presentaron más frecuentemente en perchas en abetos de Douglas (*Pseudotsuga menziesii*) maduros y viejos, adyacentes a claros de dosel del bosque, chaparrales, y praderas naturales. El roble blanco de Oregon (*Quercus garryana*) fue el árbol dominante en los bosques. Tres especies asociadas con bosques y praderas, *Microtus californicus*, *Scapanus* spp. y *Thomomys bottae* comprendieron el 73.3% de las presas identificadas. Todos los nidos estuvieron cerca de algún hábitat relativamente abierto, sólo cuatro de 10, estaban dentro de 800 m en claros de bosque recientes. Todos los nidos eran plataformas de palos en árboles vivos. La distribución de los grandes búhos grises en las Montañas de Siskiyou sugiere que estos pueden tener una distribución más amplia en el oeste de los Estados Unidos de lo que previamente se conocía, especialmente en elevaciones por debajo de 915 m.

[Traducción de César Márquez]

In the western United States, the Great Gray Owl (*Strix nebulosa*) is a reclusive inhabitant of boreal forests (Franklin 1988). The species is considered an uncommon resident with an uneven distribution extending from the crest of the Cascade Mountains in Washington and Oregon, east to the

Rocky Mountains of western Montana and northwestern Wyoming (Bull and Duncan 1993, Duncan and Hayward 1994). These owls also occur further south in the Sierra Nevada Mountains of California (Winter 1986).

Oregon and California populations are associated with a variety of mature and old-growth coniferous forest habitats near mountain meadows above 915 m. These include true firs (*Abies* spp.), ponderosa and lodgepole pines (*Pinus ponderosa*

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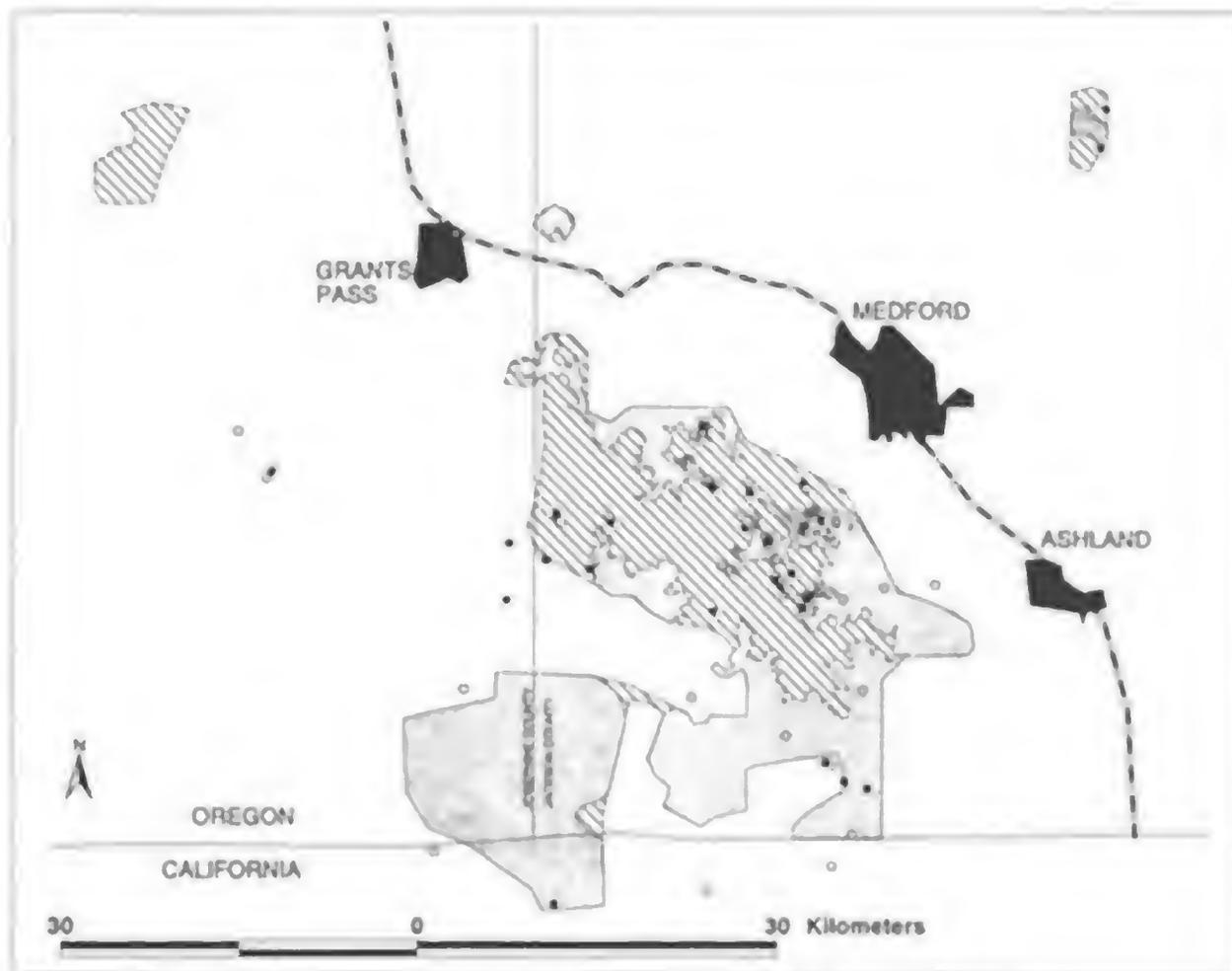


Figure 1. Distribution of Great Gray Owl sightings (1991–97) in the Siskiyou Mountains of southwestern Oregon. Solid circles represent nests, solid squares represent recently-fledged young, pairs, and multiple-year sightings, and open circles represent single observations of individuals. Shaded areas represent surveyed locations >915 m, and crosshatched areas represent surveyed locations <915 m.

and *P. contorta*), and Douglas-fir (*Pseudotsuga menziesii*; Winter 1980, 1986, Bryan and Forsman 1987, Bull et al. 1988b, Goggans and Platt 1992).

Great Gray Owls use montane meadows as preferred hunting areas (Winter 1986) as well as open stands of mature forests with grass as the dominant ground cover (Bull et al. 1988b) and clearcuts (Goggans and Platt 1992). Denser stands of trees are used less frequently.

Nests in the western United States are often in natural depressions in broken-topped snags, but stick nests and artificial platforms are also used (Bull and Henjum 1990, Bull and Duncan 1993). Typically, nests are located within 0.3 km of a meadow or clearcut (Winter 1986, Bryan and Forsman 1987).

In 1991, a population of Great Gray Owls was discovered in the Siskiyou Mountains of southwestern Oregon, west of the previously described range of the species (Johnsgard 1988, Bull and Duncan 1993). The owls in this new geographic area are distinctive because they occur below 915 m in elevation. Therefore, we set out to better understand the habitat used by these owls. In this study, we

characterize the habitat and nesting sites occupied by these Great Gray Owls, which occur 34–85 km west of the nearest previously-known nesting site.

#### STUDY AREA AND METHODS

In 1991, biologists from the Bureau of Land Management (BLM) and United States Forest Service (USFS) began to encounter Great Gray Owls in the Siskiyou Mountains, including several occupied nests. In 1996 and 1997, BLM and USFS biologists surveyed all habitat in a 117 600 ha area for this species in the Siskiyou (Fig. 1). All 75 observations in the surveyed area plus 17 additional sightings outside the survey area were recorded on 7.5 minute United States Geologic Survey (USGS) topographic maps. Surveys were conducted according to procedures outlined in Huff et al. (1996). This protocol directed that surveys be done: (1) within the range of the Spotted Owl (*Strix occidentalis*), (2) at elevations above 915 m, (3) within mature stands (80+ years) with >60% canopy cover, and (4) within 305 m of a natural meadow of more than 4 ha.

The study area includes a diverse mixture of forest types, ranging from the Interior Valley Zone at the lowest elevations (400 m) to the mountain hemlock (*Tsuga mertensiana*) zone at the highest (2300 m; Franklin and Dyrness 1973). The dominant tree species along the gradient beginning at the lowest elevations include Oregon white oak (*Quercus garryana*), ponderosa pine, Pacific ma-

drone (*Arbutus menziesii*), Douglas-fir, California black oak (*Q. kelloggii*), white fir (*Abies concolor*), red fir (*A. magnifica*), and mountain hemlock.

From the 92 locations recorded by the BLM and USFS between 1991 and 1997, we selected the 28 sites with either nests, recently-fledged young, pairs, or multiple-year observations for habitat analysis. We defined pairs as two birds in adult plumage seen in the same location at the same time simultaneously between March and August. Because the location of a nest was not identified in the latter three categories, they are referred to as activity centers without nests. "Activity centers" may be in nesting habitat, but they potentially include foraging and roosting habitat too. By excluding single observations of lone birds, we minimized the potential of including less important habitats used by transient birds. In addition to the primary study area, we included two sites, one with a nest, located below 915 m in the foothills of the Cascades about 30 km northeast of Medford, Oregon (Fig. 1). We considered these sites because the habitat was similar to that occupied by Great Gray Owls in the Siskiyou Mountains. Also, these documented use areas are important because the current federal protocol for the Northwest Forest Plan calls for surveys for Great Gray Owls above 915 m (Huff et al. 1996).

At known nest sites, we measured habitat features centered on the nest tree. The actual nest tree at one nest site could not be relocated. For this site, we sampled habitat beginning at a point equidistant between the three likely nest trees, all of which were within 10 m of one another. For activity centers, we measured habitat features centered from the point of observation indicated by government biologists on 7.5 minute USGS topographic maps, coupled with their detailed description of the location of the bird. The sampling area consisted of a 19.95-m radius circle (0.125 ha) plus four additional 19.95-m radius plots centered 100 m from the center of the central plot in each of the cardinal directions. We recorded the identity and diameter at breast height (DBH) of live trees and snags >8 cm DBH within each plot. We measured slope and aspect at the center of each plot using a clinometer and a compass. We calculated the foliage profile at the center of each plot by measuring vegetation density in each of the four cardinal directions according to MacArthur and MacArthur (1961). We visually estimated percent canopy closure.

We measured the height of each nest and the nest tree with a clinometer. In addition, tree species, DBH, nest type, and nest placement relative to the tree were recorded. We also counted the number of stick nests within 50 m of the nest tree to determine their availability and to help identify the species that may have built the nest used by the owls. For example, a large number of stick nests within a 50 m radius would tend to eliminate accipiters as the nest builders (Reynolds and Wight 1978).

Small mammal species are often restricted to particular habitats (Verts and Carraway 1998). We analyzed the contents of Great Gray Owl pellets to gain further clues to owl habitat use. We estimated the number of prey from counts of skulls and mandibles.

We used BLM and USFS aerial photos to assess habitat within 0.8 km (2.0 km<sup>2</sup>) of each nest or observation. For each site, we used aerial photos from the year that most

closely corresponded to the year of observation (1991–96). We verified with government biologists that no habitat alteration occurred in the intervening years. After determining the scale of each photo by comparison with USGS topographic maps, we placed a 67 × 67 m grid (113 points) over the nest or point of observation and assigned each grid point to one of the following vegetation categories: closed-canopy coniferous forest, open-canopy forest, chaparral, or open areas. We defined open-canopy forest as Oregon white oak and oak/madro-ne woodland and thinned stands (hereafter partial cuts) <10 yr old. Chaparral consisted of habitats dominated by wedgeleaf ceanothus (*Ceanothus cuneatus*), whiteleaf manzanita (*Arctostaphylos vicida*) and poison oak (*Toxicodendron diversiloba*). Open areas included natural meadows, clearcuts <10 yr old, agricultural fields, and roads. We also measured the distance to the nearest clearcut or partial cut <10 yr of age and meadow >2 ha in extent. We did not consider clearcuts >10 yr of age to be open areas because the growth of young conifers and shrubs approached a closed canopy condition on most sites by this time.

Level of significance for all tests was set at  $\alpha = 0.05$ . Comparison of size composition of trees at nest sites and activity centers was initially evaluated with a Multivariate Analysis of Variance (MANOVA) using Statistica for Windows (1993). However, the assumption of equal covariance matrices was violated, thus we compared the size classes independently. Due to multiple comparisons, we used a sequential Bonferroni correction (Rice 1989). When comparing the 30 habitat features between nests and activity centers (Table 2), we employed the Mann-Whitney *U*-test when the data were not normally distributed, again using a sequential Bonferroni correction. We evaluated the elevational distribution of Great Gray Owls relative to 915 m and habitat data involving percentages using a *G*-test.

## RESULTS

Great Gray Owls were detected at 92 locations, including 10 sites with nests, five sites with fledglings but no identified nests, 10 sites with pairs but no breeding confirmed, and 67 sites with single owls confirmed in one or more years (Fig. 1). Of the 92 sites, 63% were below 915 m (Fig. 2). Of the 25 sites where nests, fledglings, or pairs were confirmed, 80% were below 915 m, with a mean elevation of  $904 \pm 397$  ( $\pm$ SD). Within the area surveyed by government biologists, a similar pattern existed indicating greater abundance below 915 m ( $G_1 = 11.32$ ,  $P < 0.001$  and  $G_1 = 9.25$ ,  $P < 0.05$  for all sites and those with a nest, fledglings, and pairs; respectively). All sites were located on public lands administered by the Medford District of the Bureau of Land Management, or the Rogue River National Forest.

Ten nests were found during the study, and another was found in 1998, when conducting follow-

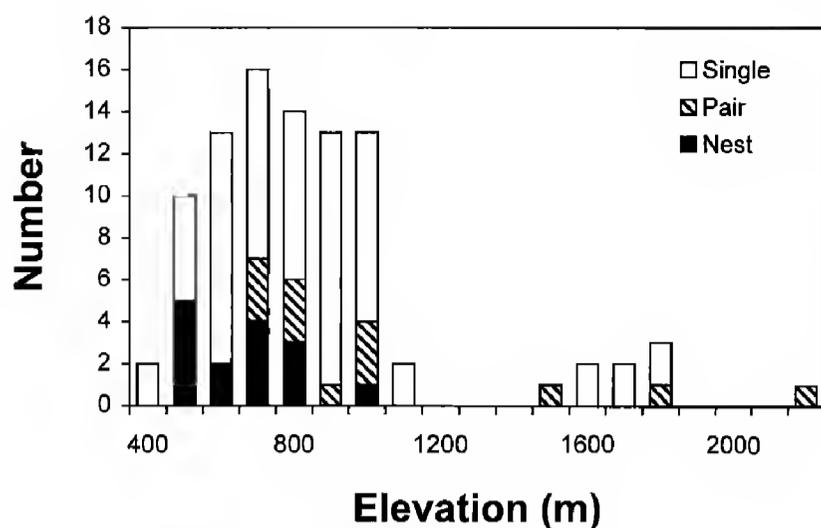


Figure 2. Elevational distribution of Great Gray Owls in the Siskiyou Mountains. "Single" represents locations of individuals observed once. "Pair" indicates locations with pairs and/or multiple-year sightings. "Nest" represents observed nests or locations of recently-fledged young.

up searches of previously used nesting sites. None of these nests were located closer than 4 km of each other. Of these, 10 were in Douglas-fir and one was in a ponderosa pine. Of 10 cases where the specific nest was identified, five were stick nests situated on lateral limbs against tree trunks, and five were stick nests in the forked top of trees. None of the nests were reused the following year. Nests were located in one of the largest trees at each site (Tables 1 and 2). The number of stick nests within 50 m of nests used by owls ranged from 2–14 ( $\bar{x} = 5.9$ ). Incubating or brooding Great Gray Owls were observed between 25 March and 12 May; the earliest fledglings were observed 14 May.

Great Gray Owls most frequently nested in late-successional stands dominated by Douglas-fir located near natural forest edges (Table 2). Nest plots typically included large trees at low densities with numerous smaller understory trees. Of the five subplots sampled at each site, basal area averaged highest on the nest plot and dropped to less than 50% on the lowest-ranked peripheral plot (Fig. 3). The decrease in Douglas-fir basal area and the corresponding increase in Oregon white oak basal area between the nest plot and the lowest-ranked peripheral plot also indicated a change in habitat within 120 m of the nest tree. Oregon white oak tends to occur in single-species or hardwood-dominated open-canopy stands and is uncommon within areas dominated by Douglas-fir.

Forest composition at nest sites was similar in most respects to activity centers without nests. The

Table 1. Characteristics of 10 nest trees used by Great Gray Owls in the Siskiyou Mountains of southwestern Oregon, 1991–97.

FEATURE	MEAN	SD	RANGE
DBH (cm) of nest tree	64.0	24.0	34–106.5
Nest height (m)	20.9	5.1	16–31
Tree height (m)	35.3	6.4	2–45
Number of stick nests within 50 m	5.9	3.6	2–14

distribution of trees in regard to six size classes did not differ ( $P \geq 0.767$ ). Of the 30 variables examined (Table 2), no significant differences between nest sites and activity centers were detected, and only two were indicated when not using the sequential Bonferroni correction (Oregon white oak density and the size of conifer snags). Because of the similarities in habitat features between nest sites and activity centers, we feel justified combining the two to characterize the habitat of Great Gray Owls in the Siskiyou Mountains (Table 2).

Both nest and all sites combined contained about 45% closed canopy forest within a 0.8 km radius (Table 3). Only four of ten nest sites and 15 of all 28 sites had clearcuts within 0.8 km. Clearcuts never accounted for more than 10% of the area at the ten nest sites.

Aspect was predominantly north, with 80% of nest sites and 67% of activity centers facing north. Overall, 20 of 28 sites had a northerly aspect ( $P < 0.02$ , binomial probability of a result this extreme or greater).

Pellets and pellet fragments were collected at 11 locations, including seven nest sites and four roosts. Of 165 prey items identified, 79.4% were located beneath nests. California voles (*Microtus californicus*) and moles (*Scapanus* spp.) were the most common prey items, comprising 63.0% of the individuals identified (Table 4). Botta's pocket gopher (*Thomomys bottae*) was the only other species accounting for >5% of the prey items.

#### DISCUSSION

Great Gray Owls of the Siskiyou Mountains most often occur in mature Douglas-fir forest close to Oregon white oak or oak/madrone woodland, chaparral, or natural meadows. The use of mature and old growth forests adjacent to open areas is similar to findings of other studies in the western United States (Forsman and Bryan 1984, Winter

Table 2. Habitat characteristics in the vicinity of 10 nest sites and 18 additional sites with fledged young, pairs, or multiple-year occupancy in the Siskiyou Mountains, 1991–97.

FEATURE	NEST SITES			ALL SITES <sup>1</sup>		
	MEAN	SD	RANGE	MEAN	SD	RANGE
Tree density (No./ha)						
Douglas-fir	251	156	75–630	274	165	0–630
White fir	20	64	0–64	25	68	0–245
Ponderosa pine	11	12	0–32	15	23	0–106
Other conifer	10	19	0–56	27	63	0–250
Pacific madrone	74	75	0–203	61	63	0–203
Oregon white oak	105	162	0–547	49	106	0–547
California black oak	32	30	0–77	29	33	0–142
Other hardwood	13	16	0–50	16	41	0–216
Total	518	254	262–1062	497	211	205–1062
Snag density (No./ha)						
Conifer	41	16	22–74	43	29	0–138
Hardwood	54	47	18–171	36	36	0–171
Total	95	55	40–219	78	52	0–219
DBH live trees (cm)						
Douglas-fir	31	10	18–47	30	10	13–53
White fir	28	0	28	21	8	10–28
Ponderosa pine	36	7	24–46	36	23	16–125
Other conifer	28	2	27–30	23	13	9–55
Pacific madrone	21	8	4–33	20	6	4–33
Oregon white oak	16	5	11–26	16	4	11–26
California black oak	20	6	14–31	19	5	12–31
Other hardwood	17	6	12–27	15	5	8–27
Total	25	5	15–32	25	5	15–32
DBH snags (cm)						
Conifer	26	7	13–37	23	7	11–37
Hardwood	15	2	13–19	15	3	10–25
Total	20	3	13–27	19	4	11–27
Distance to water (m)	405 (10) <sup>2</sup>	163	12–596	372 (28)	252	12–131
Distance to timber sale (m)	129 (4)	78	40–184	145 (18)	67	40–251
Distance to opening (m)	99 (10)	89	22–313	103 (28)	87	0–325
% canopy closure	65	14	42–81	61	19	9–87
% slope	37	16	8–59	36	15	8–65
Elevation (m)	704	161	512–1006	901	391	427–2262

<sup>1</sup> “All sites” includes nest sites and other activity centers (locations with recently-fledged young, pairs, and multiple-year observations where a nest has not been located).

<sup>2</sup> Numbers in parentheses represent the number of sites at which a feature occurred within 1 km and were used in calculating the mean.

1986, Franklin 1988, Bull and Henjum 1990, Goggans and Platt 1992), but not with respect to elevation (<915 m) and not near Oregon white oak and chaparral habitat types. This suggests a wider elevational and habitat distribution than currently understood.

The nature of the openings used by Great Gray

Owls for foraging varies throughout its range in the western United States. In California, owls foraged primarily in natural meadows (Winter 1986). In contrast, owls in the Rocky Mountains and the central Cascades of Oregon foraged primarily in clearcuts (Franklin 1988, Goggans and Platt 1992). Only in northeastern Oregon, did birds hunt reg-

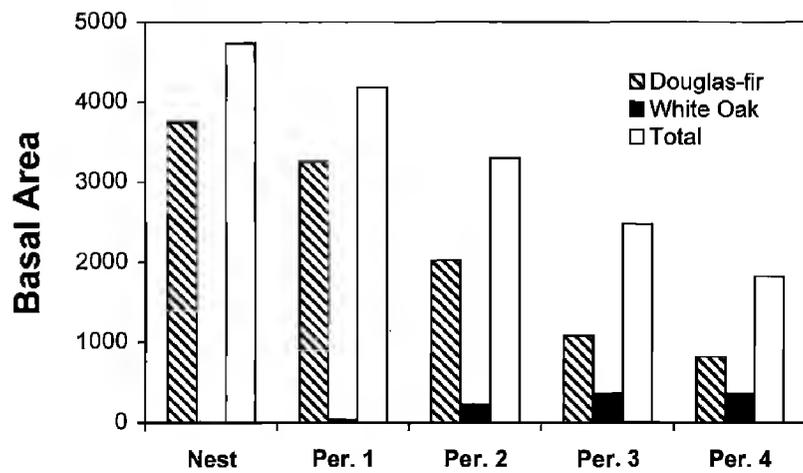


Figure 3. Basal area ( $m^2/ha$ ) of Douglas-fir, Oregon white oak, and all trees combined in the nest plot and peripheral plots (1–4) at 10 Great Gray Owl nest sites in the Siskiyou Mountains. Peripheral plots are ranked from greatest to least total basal area.

ularly in open canopy forests (Bull et al. 1988a, 1988b, 1989). In the Siskiyou, the openings in the vicinity of owl observations most often included open Oregon white oak and oak/madrone woodlands as well as chaparral and grassy meadows. Clearcuts were uncommon near owl locations, and based on prey items identified from pellets, do not appear important as foraging areas. At lower elevations in the Siskiyou Mountains, clearcuts tend

Table 3. Percent coverage of different vegetation types within a 0.8-km radius ( $2 km^2$ ) centered on the nest or point of observation of Great Gray Owls in the Siskiyou Mountains, 1991–97.

HABITAT	NEST SITES	ALL SITES <sup>1</sup>
Closed-canopy coniferous forest	45.5	45.9
Open-canopy forest		
Oak and oak/madrone woodland	23.8	17.0
Partial/selective cuts	3.8	7.0
Other	0	1.2
Total	27.6	25.2
Chaparral	12.6	10.3
Open		
Natural meadows	3.7	7.4
Recent clearcuts	3.8	6.1
Agricultural fields	4.0	2.0
Roads	2.2	2.5
Total	13.6	18.0
Other (buildings, yards, farm ponds)	0.7	0.6

<sup>1</sup> "All sites" includes nest sites and other activity centers (locations with recently-fledged young, pairs, and multiple-year observations where a nest has not been located).

Table 4. Prey items identified in the pellets of 11 pairs of Great Gray Owls inhabiting the Siskiyou Mountains, 1991–97.

SPECIES		INDIVIDUALS	PERCENT OF TOTAL ITEMS	PERCENT OCCURRENCE <sup>1</sup>
Mole	<i>Scapanus</i> sp.	49	29.7	82
Shrew-mole	<i>Neurotrichus gibbsii</i>	3	1.8	27
Shrew	<i>Sorex</i> sp.	3	1.8	18
Northern flying squirrel	<i>Glaucomys sabrinus</i>	1	0.6	9
Botta's pocket gopher	<i>Thomomys bottae</i>	16	9.7	55
Mazama pocket gopher	<i>Thomomys mazama</i>	1	0.6	9
Deer mouse	<i>Peromyscus maniculatus</i>	1	0.6	9
Mouse	<i>Peromyscus</i> sp.	2	1.2	9
Dusky-footed wood rat	<i>Neotoma fuscipes</i>	1	0.6	9
Wood rat	<i>Neotoma</i> sp.	1	0.6	9
Creeping vole	<i>Microtus oregoni</i>	3	1.8	27
California vole	<i>Microtus californicus</i>	56	33.9	82
Townsend's vole	<i>Microtus townsendii</i>	5	3.0	18
Vole	<i>Microtus</i> sp.	13	7.9	45
Red-backed vole	<i>Clethrionomys occidentalis</i>	2	1.2	9
Norway rat	<i>Rattus norvegicus</i>	1	0.6	9
Unidentified mammal		6	3.6	18
Dark-eyed junco	<i>Junco hyemalis</i>	1	0.6	9
Total		165	99.8	

<sup>1</sup> Percent occurrence represents the percentage of sites at which a given prey type was encountered.

to be dominated by brush and young conifers before a dense herbaceous layer develops. The Great Gray Owl's principal prey, California voles, moles, and Botta's pocket gophers prefer grasslands and other habitats with abundant herbaceous vegetation such as that found in Oregon white oak and oak/madrone woodlands, as well as in grassy meadows (Bailey 1936, Ingles 1965, Maser and Storm 1970).

The diet of Great Gray Owls on our study area was similar to many previous studies, in that voles, gophers, and other small terrestrial mammals were the dominant source of food (Mikkola and Sulkava 1970, Mikkola 1972, Winter 1986, Franklin 1988, Bull and Henjum 1990). The abundance of moles in the diet, however, is higher than reported in other studies. The dominance of these small terrestrial mammals in the diet suggests that the owls were foraging in the more open oak/madrone woodland and grassy areas. Because of their large size (410–465 mm wing chord; Johnsgard 1988), we believe that Great Gray Owls are unable to forage efficiently in dense forest habitats, and are best adapted to hunt edges and open forests.

The elevational distribution of Great Gray Owls in the Siskiyou Mountains is unique in the western United States. In other locations, almost all owls have been reported above 915 m (Forsman and Bryan 1984, Franklin 1988, Bull and Henjum 1990). The few observations below 915 m were attributed to dispersing juveniles, non-breeders, and wintering birds (Winter 1986, Goggans and Platt 1992). The lowest reported nesting was 912 m (Goggans and Platt 1992). By comparison, only one of the 11 nests in this study was above 825 m.

The distribution of suitable habitat in the Siskiyou may explain the large number of owls observed at low elevations. Below 915 m, mature Douglas-fir forests on north-facing slopes often occur in close proximity to Oregon white oak-dominated woodlands, chaparral and/or natural meadows on adjacent south-facing slopes. In these areas, Douglas-fir stands offer roosting and nesting sites while the open woodlands and meadows offer opportunities for foraging. Oak woodlands tend to support a dense herbaceous layer, and numerous hunting perches which are important to Great Gray Owls (Bull and Henjum 1990). Above 915 m in the Siskiyou, vegetation is dominated by coniferous forests, with few oak woodlands or open grasslands, though clearcuts are widespread.

Great Gray Owls appear to be opportunistic re-

garding nest selection, using whichever nest type is available, including artificial nest platforms (Nero 1980, Mikkola 1983, Bull and Duncan 1993, Whitfield and Gaffney 1997, Duncan 1997). At low elevations in the Siskiyou, platform nests were abundant in most stands where owls nested. Owls nested in one of the largest trees (top 10%) in each stand illustrating the importance of mature stands. The location and structure of stick nests used by the owls led us to believe that most of the nests were constructed by western gray squirrels (*Sciurus griseus*), which tend to be associated with oak woodlands or with the interface between oak woodlands and coniferous forests (Bailey 1936, Cross 1969, Maser 1998, Verts and Carraway 1998). This habitat is common below 1000 m on the northern slopes of the Siskiyou Mountains. The abundance of potential nest sites may explain why nests were not reused in subsequent years.

Another potential factor affecting the elevational distribution of Great Gray Owls is temperature. Winter (1986) speculated that Great Gray Owls do not nest at lower elevations in the southern portion of their range because they are adapted to boreal forests and are physiologically unable to occupy areas with high temperatures. Our observations suggest that the southern limits of the range may have more to do with the distribution of suitable habitat than with temperature *per se*. All nests that we located occurred at elevations where mean maximum temperatures exceeded 32°C during the summer months, and where temperatures on some summer days surpassed 40°C (Taylor 2001). Cloud cover and precipitation provide limited relief during the breeding season with less than 3 cm of precipitation in each April and May and even less in the summer months (Taylor 2001). However, all nests in the Siskiyou Mountains were shielded from direct sunlight by canopy cover, in contrast to the exposed broken-top snag nests found in California (Winter 1986).

Great Gray Owls regularly occur below 915 m in the Siskiyou Mountains where suitable habitat is available. As such, a reassessment of the current protocol used by the USFS and BLM to monitor Great Gray Owls in the Pacific Northwest in terms of conducting surveys only above 915 m (3000 ft) seems appropriate. Thus, Great Gray Owls may occur elsewhere in the western United States at elevations lower than previously recorded where shaded nesting sites are available.

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## NEST-SITE CHARACTERISTICS AND LINEAR ABUNDANCE OF CLIFF-NESTING AMERICAN KESTRELS ON SAN CLEMENTE ISLAND, CALIFORNIA

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**ABSTRACT.**—American Kestrels (*Falco sparverius*) are typically secondary-cavity nesters, and use of natural cliff cavities for nest sites is less-commonly reported. On San Clemente Island (SCI), California, however, American Kestrels nest primarily on cliffs in major canyons (93%), to a lesser extent on sea cliffs (4%), as well as in man-made structures (3%). We located and mapped 99 American Kestrel territories on SCI, and recorded 11 nest-site characteristics at 40 cliff nests during 2001–02. Nest cliffs were typically fractured igneous rock with mean height of 16.1 m  $\pm$  1.8 SE. Mean slope of nest cliffs was vertical ( $\bar{x}$  = 91°). Nest cliffs and cavities were significantly oriented to the southeast, away from the prevailing wind direction (NW). In eight canyons, where we believe that we found all occupied American Kestrel territories, the mean linear abundance was 2.1 pairs/km, greater than most published estimates. Contrary to most previous studies, no American Kestrels nested in tree cavities despite their presence in SCI canyons. The absence of cavity-excavating breeding birds from the island likely restricts kestrels to nesting in naturally-formed cavities and man-made structures.

**KEY WORDS:** *American Kestrel; Falco sparverius; California; cliff-nesting; nest abundance; nest site characteristics; San Clemente Island.*

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### CARACTERÍSTICAS DE LOS SITIOS NIDO Y ABUNDANCIA LINEAR DE CERNÍCALOS QUE ANIDAN EN ACANTILADOS DE LA ISLA SAN CLEMENTE, CALIFORNIA

**RESUMEN.**—Los cernícalos (*Falco sparverius*) anidan típicamente en cavidades usadas, el uso de cavidades naturales en los acantilados como sitios nido se reporta menos comúnmente. En la Isla de San Clemente (ISC), California, sin embargo, los cernícalos anidan primordialmente en los precipicios de los grandes cañones (93%), y en menor grado en los acantilados marinos (4%), al igual que en estructuras construidas por el hombre (3%). Localizamos y colocamos en un mapa 99 territorios de cernícalos en la ISC, y registramos 11 características de los sitios nido en 40 nidos de acantilados durante 2001–02. Los nidos de los precipicios fueron típicamente roca ígnea fracturada con una altura media de 16.1 m  $\pm$  1.8 SE. La pendiente promedio de los nidos en acantilados fue vertical ( $\bar{x}$  = 91°). Los nidos y las cavidades estuvieron significativamente orientadas hacia el suroriente, lejos de la dirección prevaleciente del viento (NW). En ocho cañones, donde creemos que encontramos todos los territorios de los cernícalos ocupados, el promedio de la abundancia lineal fue 2.1 parejas/km, mucho mayor que la mayoría de las estimaciones antes reportadas. Contrario a la mayoría de los estudios previos, ningún cernícalo anidó en las cavidades de los árboles a pesar de su presencia en los cañones de la ISC. La ausencia en la isla de aves que aniden en cavidades excavadas por ellos mismos, probablemente restringe a los cernícalos a anidar en cavidades formadas naturalmente o en estructuras construidas por el hombre.

[Traducción de César Márquez]

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American Kestrels (*Falco sparverius*) are the most widely distributed and abundant North American falcon. The species is a secondary-cavity nester typically using woodpecker-excavated or naturally formed cavities in trees, cliffs, and man-made structures (Smallwood and Bird 2002). Although numerous studies described nest box populations (e.g., Smallwood and Collopy 1991, Rohrbaugh and Yahner 1997, Smallwood and Wargo 1997), and extensive work was conducted on tree-nesting populations (Smith et al. 1972, Balgooyen 1976), few quantitative data are available regarding this species' nesting habits in natural cliff cavities (but see Smith et al. 1972, De Lucca and Saggese 1993). We measured 11 nest site characteristics for 40 confirmed American Kestrel nest cavities in natural cliffs on San Clemente Island (SCI), California. Although several studies have reported density for tree nesting and nest box populations (Craig and Trost 1979, Smallwood and Bird 2002), estimates of density and abundance for cliff-nesting American Kestrels in North America have not been previously reported. Therefore, we present mean linear abundance of breeding pairs in eight canyons on SCI.

#### METHODS

**Study Area.** San Clemente Island is the southernmost of the California Channel Islands (32°50'N, 118°30'W), located in Los Angeles County ca. 92 km west of Palos Verdes, CA. SCI is administered by the U.S. Navy and is used for active military training as part of the Southern California Offshore Range; however, the U.S. Navy has an extensive environmental program on the island for the protection and conservation of natural resources (USDN, SWDIV 2001). The island is 44 km long and 2.4–6.4 km wide, encompassing ca. 14 764 ha. Elevation ranges from sea level to 599 m. The island consists of a central plateau incised by canyons along the western and eastern slopes. The east side is a precipitous escarpment, descending from the plateau to the Pacific Ocean; the west side consists of a series of more gentle marine terraces formed as a result of periodic uplifting and erosion from wave action (Vedder and Howell 1980). Canyon bottoms typically contain little water, with continuous flow restricted to rain events. Annual temperature ranges from 10–22°C and mean annual precipitation is 15.7 cm (Olmsstead 1958, Vedder and Howell 1980).

Current vegetation on SCI includes mixed native and non-native grasslands (including *Avena*, *Bromus*, and *Stipa* spp.) on the upper terraces; maritime desert scrub containing California sagebrush (*Artemisia californica*), morning glory (*Calystegia macrostegia*), prickly pear (*Opuntia littoralis*), cholla (*O. prolifer*), snake cholla (*Berberocactus emoryi*), and wild oats (*Avena* sp.); and remnant patchy shrubland/woodland characterized by island cherry (*Prunus lyonii*), lemonade berry (*Rhus integrifolia*), island

oak (*Quercus tomentella*), canyon live oak (*Quercus chrysolepis*), and Catalina ironwood (*Lyonothamnus floribundus*) (Raven 1963, Sward and Cohen 1980, Kellogg and Kellogg 1994).

**Nest-site Characteristics.** We located and mapped 99 American Kestrel nest sites on SCI during 2001–02 (Fig 1). We surveyed the island on foot or from a boat during the breeding season (February–June). Territories were considered occupied when a courting adult pair (e.g., prey exchanges, copulation) was present. Nest sites were located by observing adult behavior. When possible, we visually located nests by following an adult flying to its cavity. Nest cavities were defined as occupied when a prey delivery was observed. Nest sites were plotted on topographic maps generated from ArcView 3.2a (Environmental Systems Research Institute, Redlands, CA). If a nesting territory was occupied in two consecutive years, only the first nest detected was included in the nest-site characteristics analysis.

Nest sites were classified as in a canyon cliff, sea cliff, tree, or artificial structure. Canyon cliffs were found within interior canyons. Sea cliffs bordered the ocean. Artificial structures included buildings, junk piles, and abandoned trailers.

We measured nest-site characteristics for all occupied nests where the exact nest cavity was directly visible ( $N = 40$ ). Standard error of the mean is reported. We measured wall height, cliff height, and distance to the canyon bottom using a laser rangefinder and clinometer from the closest accessible perpendicular vantage. Wall height and cliff height were measured on a vertical axis through the nest cavity. Wall height was defined as the area from the canyon bottom to the canyon rim or terrace edge, including vegetated slopes, sheer cliff faces, and rock outcroppings. Cliff height was defined as the facet of the wall (e.g., rock outcropping, sheer face) containing the nest cavity, from its bottom to its top. In some cases when walls were near vertical, wall height and cliff height were identical. Distance to the canyon bottom was measured from the nest to the canyon bottom.

We quantified general characteristics of nesting cliffs by estimating percent slope, cliff complexity, and cliff composition. Two to three observers visually estimated percent slope and then we averaged all estimates. In the field, nest cliffs were estimated as a percentage of vertical, where 100% = vertical. Percent slope was then transformed into degrees (100% = 90°) for analysis. The complexity of the nest cliff's surface was indexed visually and categorized as either flat or convoluted based on the presence of caves, potholes, or jutting rocks. Nest cliff composition was categorized as either solid (no holes or cracks), loose rock (loose granular, scree), fractured (cracked or segmented) or stratified (layered). The amount of vegetation obscuring the entrance of each cavity was categorized as clear of vegetation, partially obscured by vegetation, or completely obscured by vegetation.

Nest cliff aspect and nest orientation (for visible cavities) were measured using the mean compass reading of two or three independent observers. We used Program ORIANA (Kovach 1994) to calculate circular statistics. Standard error of the mean vector is reported. For cliff

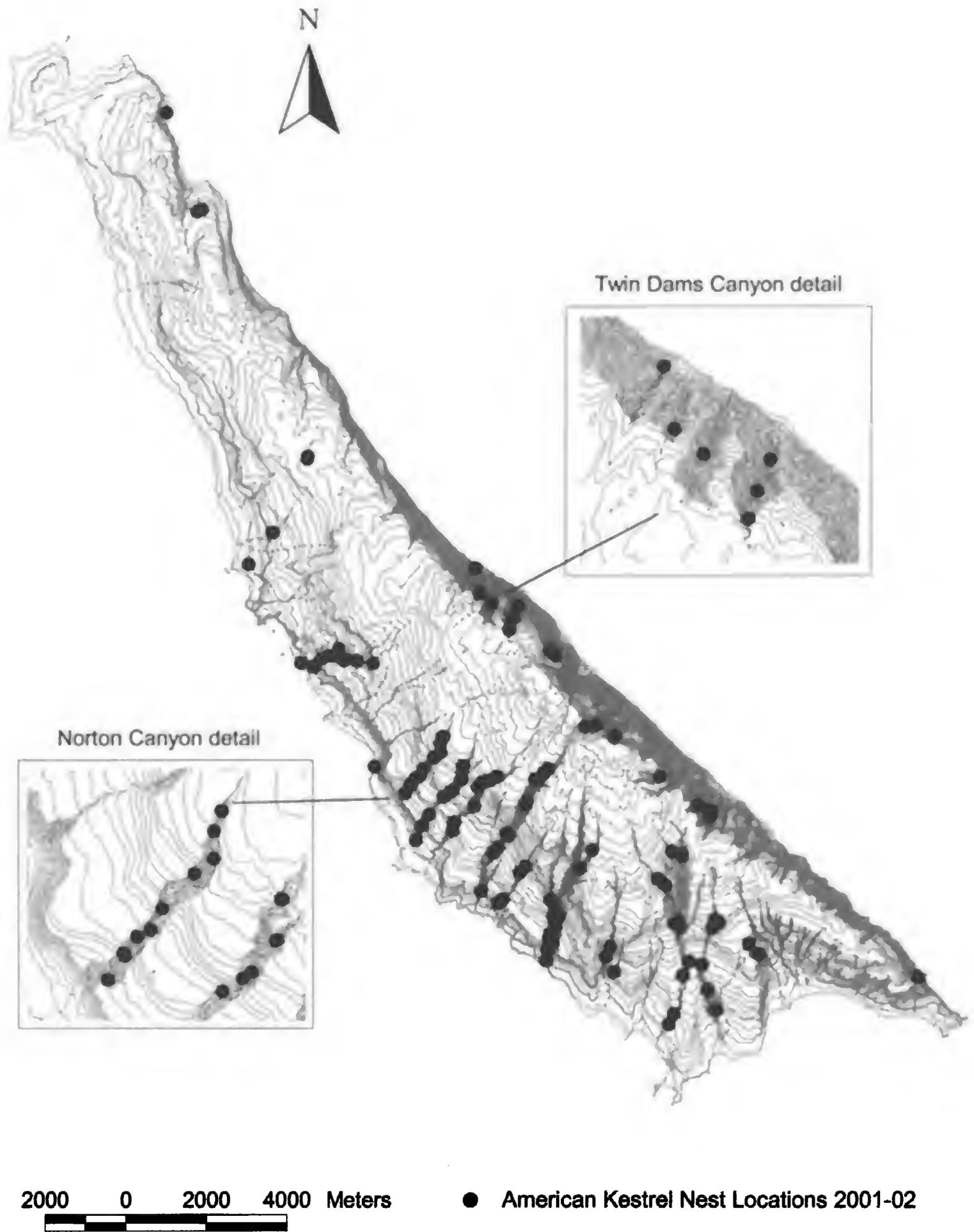


Figure 1. Map of 99 occupied American Kestrel nesting territories on San Clemente Island, CA, 2001-02.

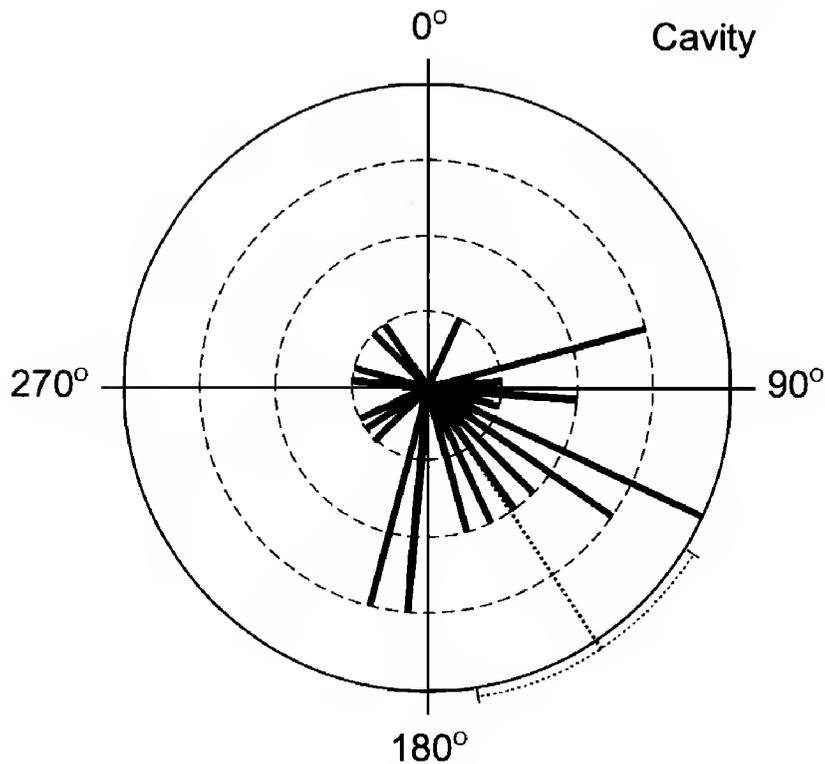


Figure 2. Mean vector analysis of American Kestrel nest cavities, San Clemente Island, CA 2001–02 (Mean vector ( $\mu$ ) =  $147^\circ \pm 13^\circ$  SE,  $P < 0.001$ ,  $N = 36$ ).

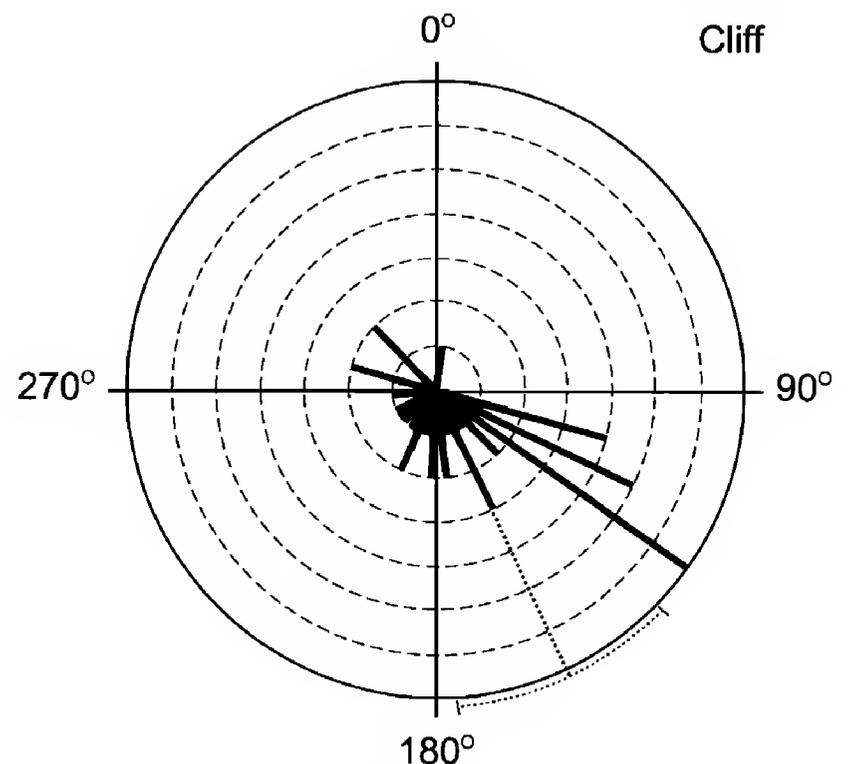


Figure 3. Mean vector analysis of American Kestrel nest cliffs, San Clemente Island, CA 2001–02 (Mean vector ( $\mu$ ) =  $155^\circ \pm 11^\circ$  SE,  $P < 0.001$ ,  $N = 40$ ).

aspect and nest orientation, Rayleigh's test of uniformity was used for mean vector analysis.

**Linear Abundance.** In each of eight canyons that were thoroughly searched, we determined the number of nesting pairs/linear km using ArcView 3.2a. The length of each canyon was calculated using the distance tool in ArcView. Canyon length was defined as the total length of each canyon from its inception on the upper terrace to its mouth along the ocean. We used a linear measure due to confounding factors involved in determining exact area of canyons. The overall mean number of pairs/linear km was then calculated for the eight canyons.

## RESULTS

American Kestrels nested primarily in canyons on SCI (Fig. 1). Of 99 territory locations, 92 were on cliffs within interior canyons, whereas four were centered on sea cliffs and three were in man-made structures. Detailed characteristics of nest sites are presented for 40 nests unless otherwise indicated. Mean wall height was  $79.4 \text{ m} \pm 7.9 \text{ SE}$ , but varied substantially at measured nests. Mean cliff height was  $16.1 \text{ m} \pm 1.8 \text{ SE}$ . Mean distance from the nest cavity to the canyon bottom was  $45.8 \text{ m} \pm 5.8 \text{ SE}$ . American Kestrels nested primarily on vertical cliffs ( $\bar{x}$  slope =  $91^\circ$ ). Only one of 38 (3%) nesting cliffs was classified as being flat whereas the remaining 37 (97%) were characterized as convoluted. Thirty-six nest cliffs (90%) were categorized as fractured and two (5%) were solid; loose rock and stratified cliffs accounted for 2.5% each. Nest-cavity entrances ranged from being completely clear of

vegetation (25; 62%) to completely obscured (4; 10%), with 11 (28%) at least partially obscured by vegetation. Vector analysis of 36 nest cavities ( $\bar{x}$  vector ( $\mu$ ) =  $147^\circ \pm 13^\circ$  SE,  $P < 0.001$ ) and 40 nest cliffs ( $\bar{x}$  vector ( $\mu$ ) =  $155^\circ \pm 11^\circ$  SE,  $P < 0.001$ ) indicated that both were oriented to the southeast (Figs. 2 and 3). Mean number of nesting pairs in eight canyons was  $2.1 \pm 0.2 \text{ SE}$  pairs/linear km (range = 1.1–3.2 pairs/km) (Table 1).

## DISCUSSION

**Nest-site Characteristics.** American Kestrels are the most widespread breeding raptor on SCI (In-

Table 1. Linear abundance of nesting American Kestrels on San Clemente Island, 2001–02.

CANYON NAME	MEAN		
	LENGTH IN KM	NUMBER OF PAIRS/YR	PAIRS/KM
Cave Canyon	5.5	11	2.0
Horse Canyon	4.5	8	1.8
Box Canyon	3	5	1.7
Norton Canyon	3.1	8	2.6
Middle Ranch Canyon	2.1	5.5	2.6
Wallrock Canyon	1.7	5.5	3.2
China Canyon	6.3	7	1.1
Horse Beach Canyon	4	6	1.5
Totals	30.2	56	2.1

stitute For Wildlife Studies unpubl. data), and are found predominately in canyons with vertical cliffs. The largely cliff-nesting habits of American Kestrels on SCI have not been commonly reported in previously-described populations. We found American Kestrels nesting in interior canyon cliffs despite the fact that trees, sea cliffs, and man-made structures were available. Although data on availability of these three cavity types were not collected, our results suggest high usage of canyon cliffs for nest sites. In central Utah, Smith et al. (1972) found only two of 41 nests (5%) in cliffs (abandoned rock quarries), the remainder being located in trees and man-made structures. While the geography of Smith et al.'s study site was clearly different from that of SCI, similar nesting resources were available for both populations. In addition, cliff nests reported in Smith et al. (1972) were located in a human-altered landscape, further confounding potential comparison with those found in naturally-formed cavities in cliffs on SCI. The high incidence of cliff use in our study indicates that American Kestrels will accept natural cliff cavities where tree cavities are not available. Several species of woodpeckers visit SCI during the non-breeding season (Jorgensen and Ferguson 1984); however, they do not construct nest cavities. Although natural-tree cavities occur on SCI, they are likely uncommon, and some features (e.g., location, cavity dimensions) may render them unsuitable for nesting American Kestrels.

American Kestrels did not use sea cliffs for nesting, despite their abundance on SCI. We hypothesize that sea cliff cavities may have undesirable moisture levels due to crashing surf and salt spray, and may be too exposed to wind. Also sea cliff nests may negatively influence fledging success due to their location over water.

Nest sites were found on a variety of wall heights, however, nest cavities were confined to sheer rock faces or rock outcroppings within these walls. The lack of American Kestrel nest cavities in anything other than fractured, convoluted rock may be an artifact of the geomorphic composition of SCI. Nest cliffs appear to be highly variable, as is evident in the large range of cliff sizes, wall heights, and elevations above the canyon bottom in our sample. Vertical cliffs seem most likely to be used for nesting on SCI and may be associated with protection from mammalian predation (e.g., black rat [*Rattus rattus*], house mouse [*Mus musculus*], deer mouse

[*Peromyscus maniculatus*], feral cat [*Felis sylvetris*], island fox [*Urocyon littoralis*]).

In contrast to other studies (Rohrbaugh and Yahner 1997, Smallwood and Bird 2002), American Kestrels on SCI occasionally nested in cavities that were partially to completely obscured by vegetation, typically cacti (e.g., *Opuntia* sp., *Bergerocactus emoryii*) or morning glory. We suggest that these sites could be selected to prevent predation by Common Ravens (*Corvus corax*). Ravens have been observed clinging to canyon walls attempting to access nests, and in some cases directly taking American Kestrel chicks on SCI (B. Sullivan pers. observ.). Use of prickly pear, cholla, and snake cholla to obscure nest entrances may be a form of nest defense on SCI.

American Kestrels typically nest in cavities with entrances facing away from the prevailing direction of seasonal storms, or those typically oriented to the east or southeast (Balgooyen 1976, 1989, Brauning 1983). West-facing nest boxes in California exhibited high failure rates (Bloom and Hawks 1983). We also found that American Kestrel nest cavities were oriented away from the prevailing northwest wind and toward the morning sun (Fig. 2). Nest-cavity selection may be especially important when driving wind and rain, associated with strong frontal systems, may make cavity microclimates on west-facing slopes undesirable.

**Linear Abundance.** American Kestrel breeding density varies with nest site and resource availability. Density estimates range from 0.11–1.74 pairs/km<sup>2</sup>, but can be as high as 24.7 pairs/km<sup>2</sup> (Bird and Palmer 1988, Smallwood and Bird 2002). In Patagonia, the density of six American Kestrel nests found in basaltic cliffs was 0.038 pairs/km<sup>2</sup> (De Lucca and Saggese 1993). In linear habitat along the Big Lost River, southeastern Idaho, Craig and Trost (1979) reported 0.5–0.9 nests/km of river. In South America, kestrel territories averaged 0.12/km<sup>2</sup> (Balgooyen 1989). Compared to the estimates in Craig and Trost (1979) and Balgooyen (1989), American Kestrels on SCI nested in higher linear abundance (2.1 pairs/km). A relatively small proportion of the island's surface is made up of canyons (ca. 9%, Institute for Wildlife Studies unpubl. data), however, even fewer tree cavities likely exist. Thus, nesting areas confined to canyons, coupled with an adequate prey base, may combine to support American Kestrel abundance to above average levels on SCI.

On SCI, considerable variation in American Kes-

rel abundance between canyons may occur due to extreme inter-canyon variation in the amount and types of cliffs available. The subsample of eight canyons used for abundance were all located on the west side of SCI, where canyons tend to be longer, shallower, and have less relief than those of the east side. Due to restricted access to the east side of SCI, a thorough census was not completed for any canyon, thus extrapolation of abundance to these areas may not be warranted.

Breeding American Kestrels on SCI are a recent phenomenon (Jorgensen and Ferguson 1984). Neither Grinnell (1897) nor Mearns (cited in Jorgensen and Ferguson 1984) detected any American Kestrels during their initial spring visits to SCI in the late 1800s. Linton (1908) reported them as "occasional" by 1907; however, he made no mention of this species breeding during an April visit to SCI. Howell (1917) considered them to be "probably resident" by 1917. Jorgensen and Ferguson (1984) counted 70 kestrels on 2 January 1980, and considered them a common resident. Currently, counts as high as 249 birds from 9–15 November 2001 (Institute for Wildlife Studies, unpubl. data) suggest a further increase in numbers.

We hypothesize that the apparent change in breeding status of this species, from unknown to abundant, may be associated with the loss of native ground cover to feral herbivores, and the subsequently ideal habitat for foraging American Kestrels. *Lavatera* reportedly constituted an "unbroken forest, extending for miles upon the high plateaus," and *Dudleya* "covered the ground throughout the entire island," prior to the introduction of feral grazers (Raven 1963). With the decimation of SCI's historic vegetation, more open habitat was created, possibly allowing American Kestrels on SCI to flourish.

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## SHORT COMMUNICATIONS

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### CUBAN RAPTOR-MIGRATION COUNTS IN 2001

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**KEY WORDS:** *Osprey*; *Pandion haliaetus*; *Cuba*; *Falconiformes*; *La Gran Piedra*; *migration*.

Scientists and conservationists often use migration counts of raptors at traditional bottlenecks and concentration points to document the status of raptor populations and to study their migratory behavior (e.g., Haugh 1986, Bednarz et al. 1990, Bildstein and Zalles 1995, Bildstein 1998). Most migration watch-sites, however, are in the northern Temperate Zone, and relatively few counts have been made in the tropics (Zalles and Bildstein 2000). For example, although islands in the Caribbean basin long have been recognized as important stopover sites and wintering grounds for many North American raptors including Ospreys (*Pandion haliaetus*), Swallow-tailed Kites (*Elanoides forficatus*), Merlins (*Falco columbarius*), and Peregrine Falcons (*Falco peregrinus*), and although many species of migrants remain threatened in the Caribbean Islands, no one has conducted season-long migration counts in the region. Consequently, our understanding of the geography and ecology of raptor migration in the area remains rudimentary, compromising our ability to protect raptors there (Poole and Agler 1987, Hoffman and Darrow 1992, Crouse and Keith 1999, Zalles and Bildstein 2000, Rodríguez et al. 2001). With this in mind, we investigated the possibility of establishing one or more raptor migration watch-sites in Cuba, the largest island in the region. Here we report the results of our first season of effort (2001) at two sites, Carahatas and La Gran Piedra, in central and southeastern Cuba, respectively.

#### STUDY AREA AND METHODS

We located our counting station at La Gran Piedra based on satellite telemetry tracking data collected by M.

Martell as part of a continental study of Osprey migration and on the observations of F. Rodríguez Santana, which together suggested the likelihood of important movements of Ospreys through eastern Cuba (Martell et al. 2001, Rodríguez et al. 2001). The La Gran Piedra watch-site (20°00'41"N, 75°37'38"W) is situated atop 1234-m La Gran Piedra, in the Sierra Maestra range. La Gran Piedra is 18 km east of Santiago de Cuba city, Santiago de Cuba province, and 9.6 km from the seacoast. Another counting station, Carahatas (22°55'02"N, 80°17'51"W), was used for 4 d. Carahatas is on the Atlantic coast in Matanzas Province, north-central Cuba.

We counted migrants for 18 d at La Gran Piedra between 27 August and 17 October 2001, and for 4 d at Carahatas between 18 and 21 September 2001. Counts at La Gran Piedra, which regularly began at 0800 H, never lasted for more than 7 hr due to afternoon rains. Counts at Carahatas took place from 0700–1400 H, except on 18 September, when the count finished at 1800 H.

We identified all migrating raptors at the count sites using 10× binoculars. Raptors were considered migrants if they appeared on the horizon to the west and flew past the watch-site using powered or soaring flight. Data were collected using the standardized daily report of the HMANA (Hawk Migration Association of North America) translated into Spanish. Wind speed and temperature were recorded using a Kestrel 2000 Pocket Weather Station (Nielsen Kellerman, Chester, PA). We also recorded barometric pressure. A mechanical counter (U.S. Government Tally Counter [Heavy Duty Model]) was used to tally the hourly count for the most numerous migrant, the Osprey, at peak hours.

#### RESULTS

We counted 1380 raptors belonging to eight species in 95 hr of observation at the two watch-sites (Tables 1 and 2). Counts at La Gran Piedra accounted for 92% of the individuals observed. Ospreys were the most common species observed at both watch-sites, accounting for 89% ( $N = 93$ ) of the migrants at Carahatas and 96% ( $N = 1223$ ) of migrants at La Gran Piedra. Other migrants included Swallow-tailed Kites, Sharp-shinned Hawks (*Accip-*

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Table 1. Mean daily counts, and minimum, maximum, and mean hourly counts of raptors seen at La Gran Piedra watch-site, Santiago de Cuba Province, southeastern Cuba, during 66 hr and 19 min of observation on 18 days (27 August–17 September), in autumn 2001.

SPECIES	MEAN DAILY COUNT	MINIMUM COUNT/hr	MAXIMUM COUNT/hr	MEAN COUNT/ hr
Osprey	67.9	0	171	18.4
Swallow-tailed Kite	0.56	0	10	0.15
Sharp-shinned Hawk	0.05	0	1	0.20
Broad-winged Hawk	0.72	0	8	0.02
Merlin	0.39	0	1	0
Peregrine Falcon	0.28	0	2	0.08
Unidentified falcon	0.28	0	2	0.08
Unidentified raptor	0.61	0	2	0.17

*iter striatus*), Broad-winged Hawks (*Buteo platypterus*), Crested Caracaras (*Caracara cheriway*), American Kestrels (*Falco sparverius*), Merlins, and Peregrine Falcons.

Ospreys passed La Gran Piedra at a rate of 18.4 birds/hr, and Carahatas at a rate of 3.3/hr. Osprey numbers at La Gran Piedra peaked at 1100–1200 H, when more than 40% of all Ospreys were counted, and the flight typically began after 0800 H. No counts occurred during the late afternoon because of precipitation. All raptors observed at this watch-site were flying from west to east, following the region's Sierra Maestra ridges, which parallel the coast of southeastern Cuba. In Carahatas, where the flight began at about 0800 H, most Ospreys passed between 0900 and 1000 H. All raptors seen migrating at this watch-site were flying from northwest to southeast along the coast.

#### DISCUSSION

Although eight species of migrants were observed, Ospreys made up the overwhelming majority of migrants counted at both watch-sites (Tables 1 and 2). Despite the fact that Ospreys typically are considered to be broad-front migrants (Poole 1989, Poole et al. 2003), our observations suggest a concentrated migration corridor for the species in southeastern Cuba. We are not certain why this is so, although most studies of Osprey migration have

been in northern temperate areas and the species may behave differently in the tropics. The number of Ospreys counted at the La Gran Piedra watch-site in autumn 2001 place it among the "top" five Osprey migration watch-sites in the world (Table 3), confirming earlier suggestions that Cuba is along an important flyway for North American Eastern Seaboard populations of the species (Hoffman and Darrow 1992, Rodríguez et al. 2001).

We observed Ospreys soaring in the thermals and moving east along the Sierra Maestra mountain ridges in flocks with as many as 43 birds, much greater than the flock of 11 at Cape May, New Jersey reported by Dunne et al. (1988). All flocks with Ospreys did not include other species, although one was preceded by a flock of 10 migrating Swallow-tailed Kites (Bildstein et al. 2002). This species has been reported flying with Ospreys at other sites in the Caribbean (Crouse and Keith 1999). Flying in flocks may allow individual Ospreys to locate thermals more efficiently (Kerlinger 1989).

Satellite-telemetry data indicate that southbound Ospreys from North America fly from the Florida Keys to northwestern Cuba, and then along the east-west axis of Cuba before crossing the Windward Passage to Hispaniola (Martell et al. 2001, Rodríguez et al. 2001). The relatively high number of Ospreys seen at La Gran Piedra lead us to conclude that this site is along the main mi-

Table 2. Mean daily counts, and minimum, maximum, and mean hourly counts of raptors seen at the Carahatas watch-site, Matanzas Province, north-central Cuba, during 28 hr and 45 min of observation on four days (18–21 September), in autumn 2001.

SPECIES	MEAN DAILY COUNT	MINIMUM COUNT/hr	MAXIMUM COUNT/hr	MEAN COUNT/ hr
Osprey	23.3	0	20	3.23
Sharp-shinned Hawk	0.25	0	1	0.03
Crested Caracara	0.72	0	3	0.10
Merlin	0.25	0	1	0.03
American Kestrel	0.75	0	3	0.24

Table 3. Annual mean, maximum annual count, maximum 1-day count, monitoring period and migration period at the world's four "top" Osprey migration watch-sites (Zalles and Bildstein 2000) compared with that at the La Gran Piedra watch-site, Santiago de Cuba Province, in southeastern Cuba, autumn 2001.

WATCH-SITE	MEAN	MAXIMUM ANNUAL COUNT	MAXIMUM 1-DAY COUNT	MONITORING PERIOD	MIGRATION PERIOD
Lighthouse Point, CT, U.S.A.	1920	4040	—	1980–91	120 days
Cape May Point, NJ, U.S.A.	1540	2940	308	1976–85	150 days
Hilton Head Island, SC, U.S.A.	1150	1560	261	1993–96	63 days
Kiptopeke, VA, U.S.A.	—	5775	1053	1977–96	150 days
La Gran Piedra, Santiago de Cuba, Cuba	—	1223	279	2001	18 days

gration corridor through the region, and may be an important monitoring point for this population. All other species of raptor migrants seen at La Gran Piedra watch-site in autumn 2001 were heading east along the southern slope of the east-west oriented Sierra Maestra range that runs parallel to the coastline of southeastern Cuba.

The Swallow-tailed Kites recorded at La Gran Piedra watch-site represented the first record of this species as a migrant in eastern Cuba (Bildstein et al. 2002). The species has been reported as an uncommon transient, and possibly a rare winter resident in Cuba by Garrido and Kirkconnell (2000). The 13 Broad-winged Hawks counted at La Gran Piedra watch-site also represent the first record of this species as a migrant in eastern Cuba. Crested Caracaras are not reported as migrants in Cuba (Garrido and García 1975, Garrido and Kirkconnell 2000), although Raffaele et al. (1998) reported them as vagrants in Jamaica Bay. We considered the three caracaras observed at the Carahatas coastal watch-site to be vagrants. All three passed the site along the seacoast in directed flight (i.e., west-to-east, straight-line flapping flight), and may have been engaged in short-distance regional movements as reported for this species in the United States (Clark and Wheeler 1987).

Our count results at two sites in Cuba, suggest that exploratory raptor-migration counts on Caribbean Islands can provide important new information regarding the routes and ecology of raptor migration in the region (Bildstein et al. 2002), as well as may provide a basis for developing long-term count efforts. Once established and working in collaboration with North American watch-sites, long-term efforts at such sites can: (1) help monitor populations of species of raptors migrating through the region, (2) study migration ecology and behavior in the tropics and compare it with that in temperate regions, and (3) introduce the regional human populations to the spectacle of raptor migration, thereby improving regional understanding of the connectivity of continental North America's and the Caribbean Island's natural resources and increasing opportunities to achieve conservation (Bildstein and Zalles 1998).

RESUMEN.—Se ofrecen los resultados del primer conteo de aves rapaces migratorias para Cuba y el Caribe insular. Se contaron 1380 rapaces de ocho especies en dos puntos de observación en el centro y sudeste de Cuba. Las Águilas Pescadoras (*Pandion haliaetus*) representaron el 89% y el 96% de las aves contadas en Carahatas y en La Gran Piedra respectivamente. Otras especies observadas durante la migración fueron el Cernícalo (*Falco sparverius*), halconcito (*Falco columbarius*), halcón peregrino (*Falco peregrinus*), caraira (*Caracara plancus cheriway*) y el gavilancito (*Accipiter striatus*), además, se reportan el gavilán bobo (*Buteo platypterus*) y el gavilán cola de tijera (*Elanoides forficatus*) por primera vez migrando en la región oriental de Cuba. Se confirma la importancia de Cuba para la migración de la población norteamericana de águila pescadora (*P. haliaetus*) durante su migración a través del Caribe.

[Traducción de los autores]

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## TERRITORY CHANGE AND NEST-SITE SWITCHING IN THE BEARDED VULTURE (*GYPÆTUS BARBATUS*)

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KEY WORDS: *Bearded Vulture*, *Gypaetus barbatus*; *conspicuous interaction*; *nest competition*; *territory change*.

The presence of old nests may indicate the suitability

of a nesting site for a given raptor species (Newton 1979, Collias and Collias 1984). The increase in the size of a species' population can cause a reduction in the availability of suitable nesting sites and lead to intraspecific and interspecific competition for those sites. In this respect, some species may adopt the strategy of occupying

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or usurping other species' nests, as has been documented in several raptors (e.g., Fernández and Donazar 1991, Margalida and García 1999). This process can lead to changes in the territory used by specific birds. In several raptor species, these changes in territories have been related to the age of the birds involved to the quality of the territory, and to mate loss (Newton 1986, Wiklund 1996, Forero et al. 1999).

The Bearded Vulture (*Gypaetus barbatus*) is a territorial, cliff-nesting vulture, whose diet mostly includes bony remains of medium-sized ungulates (Hiraldo et al. 1979, Donazar 1993), a resource which is unpredictable in time and space. In the Pyrenees (northeastern Spain), the number of occupied territories and the number of breeding pairs have shown a significant increase over the last 15 yr (annual increase of 5%; Heredia and Margalida 2001). Each pair has several nests in its territory (2–11; Heredia 1991, Margalida and García 1999), that are used on a rotational basis. These nests are on average 992 m from one another (Donazar et al. 1993). The functionality of having several nests at one territory may be to advertise the birds' territory or to avoid the accumulation of ectoparasites by changing the nest from one year to another (Newton 1979, Margalida and Bertran 2000a).

In the Bearded Vulture, both sexes participate in territorial defense (Margalida and Bertran 2000b), an activity that is generally centered within a radius of 500 m from the nest used that year (Brown 1990, Bertran and Margalida 2002). Despite this territorial behavior, the Bearded Vulture has suffered from nest stealing, mainly by the Eurasian Griffon (*Gyps fulvus*). Also, Bearded Vultures have usurped the nests of other species such as the Golden Eagle (*Aquila chrysaetos*) and the Common Raven (*Corvus corax*) (Margalida and García 1999) suggesting interspecific competition for nest sites. For some species such as the Bearded Vulture, nest building involves a considerable investment of time and energy and, both the nest location and the material it is made of can influence breeding success (Margalida and Bertran 2000a). On the other hand, the increase in density of nesting vultures can lead to a larger number of intraspecific interactions, a factor that can also influence this species' breeding success (Margalida et al. 2003). Nevertheless, probably as a result of the territoriality and the relatively great distance between neighboring nests (11 km; Donazar et al. 1993), the occupation of nests by conspecifics has not been documented in the Bearded Vulture.

Here we document the first case of a Bearded Vulture pair's change of territory and the first conspecific usurpation of a neighboring pair's nest.

#### STUDY AREA AND METHODS

The study was carried out in Catalonia (northeastern Spain). In this region, the population of Bearded Vulture increased from seven territories in 1984 to 30 in 2002 (Margalida and García 2002). Between 1992 and 2002, we intensively monitored 179 breeding attempts of 14–

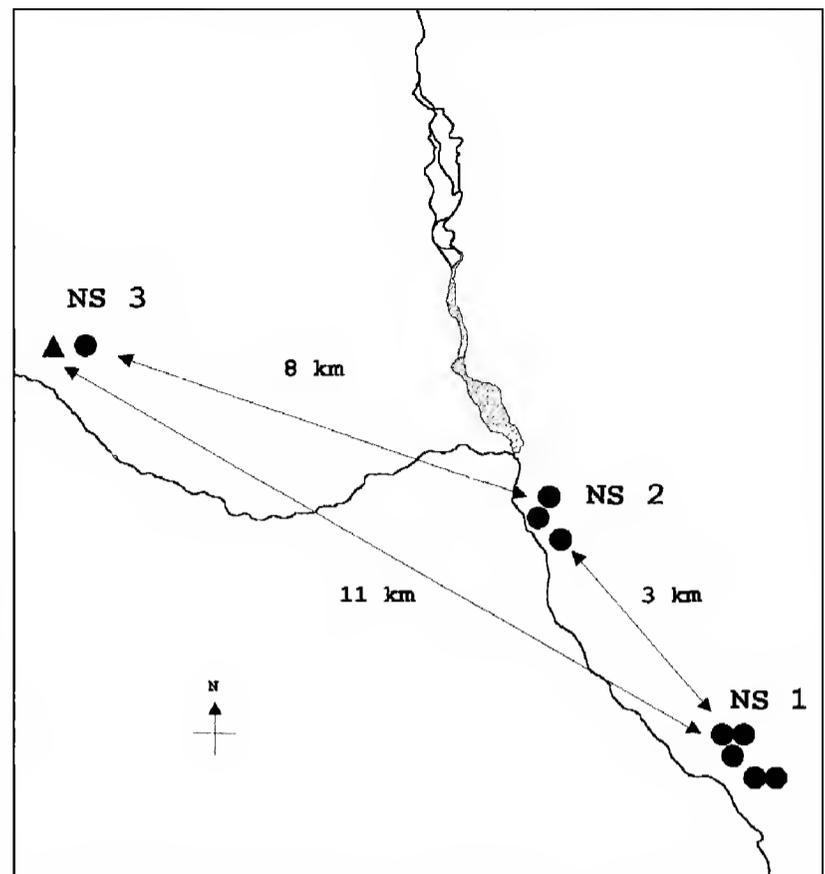


Figure 1. Location and distances between the three nesting sites occupied by the two pairs of Bearded Vultures. The circles indicate the location of the nests. The triangle in NS 3 indicates the nest shared by the two pairs, occupied by PA in 1996 and pair PB in 2001. Pair PA primarily used NS 2 (1997–2002) and pair PB primarily used NS 1 (1995–2000 and 2002).

20 focal pairs per year (García et al. 1996, Margalida et al. 2003). In this area, the mean distance between nests of the same pair was 1.7 km and the mean distance between nests of neighboring pairs was 12.7 km ( $N = 18$ ; García et al. 1996). Nests were observed throughout the breeding season at least once a week. Observations began in September, coinciding with the start of nest building (Margalida and Bertran 2000a), and ended during fledging (June–July, Margalida et al. 2003). The identification of the birds was based on comparison of the individual characteristics of their plumage, specifically the patterns on pectoral bands (black feathers in the breast and ventral region) and crowns, (see Delibes et al. 1984). The long molting process during the breeding period also facilitated identification of the birds during extensive periods of time.

#### RESULTS

Pair A: In 1996, we located this new breeding pair (PA), which occupied a vacant nesting sector (NS 3). The pair laid the eggs, incubated them, but failed during hatching. The nearest breeding pair (PB) was 11 km away in sector NS 1. After the breeding failure, PA built a second nest on the same rock face, ca. 150 m away, and it was not used again. During the pre-laying period of 1996–97, PA moved 8 km east, to an area 3 km from PB (Fig. 1). In this new sector (NS 2) PA bred in three dif-

Table 1. Summary of the breeding chronology and nest success of pairs PA and PB in the three nesting sectors studied, 1995–2002.

YEAR	NESTING SITE 3			NESTING SITE 2			NESTING SITE 1		
	PAIR	LAYING DATE <sup>a</sup>	SUCCESSFUL REPRODUCTION	PAIR	LAYING DATE <sup>a</sup>	SUCCESSFUL REPRODUCTION	PAIR	LAYING DATE <sup>a</sup>	SUCCESSFUL REPRODUCTION
1995	—	—	—	—	—	—	PB	<8 Jan	No
1996	PA	<11 Jan	No	—	—	—	PB	No eggs laid	No
1997	—	—	—	PA	?	Yes	PB	No eggs laid	No
1998	—	—	—	PA	12–13 Jan	Yes	PB	5–6 Feb	No
1999	—	—	—	PA	10 Jan	Yes	PB	5–7 Feb	No
2000	—	—	—	PA	7–10 Jan	Yes	PB	20–26 Jan	Yes
2001	PB	20–26 Jan	Yes	PA	<11 Jan	Yes	—	—	—
2002	—	—	—	PA	<8 Jan	Yes	PB	>20 Jan	No

<sup>a</sup>Jan = January, Feb = February.

ferent nests, that were built between 1996 and 2001. In 2002, PA remained in NS 2 and bred successfully.

Pair B: This pair (PB) was discovered in 1984. From 1984–94 it bred successfully each year in NS 1, using five different nests, which were grouped in a 300 m radius. In 1995, PB failed during breeding and this coincided with the presence of a third individual, who was tolerated and even entered the nest along with the two members of the pair. Between 1996 and 1999, the pair failed in its breeding attempts. In 2000, PB bred successfully in NS 1 and, during the pre-laying period of the next breeding season (2000–01), disappeared from breeding sector NS 1. That year, PA occupied one of the nests in NS 2, while another incubating pair was located in NS 3, whose phenology and plumage characteristics matched that of PB (Table 1). In 2001, PB occupied a second nest in NS 3 (built by PA in 1996) and bred successfully, having moved 11 km.

During 2002, PB rebuilt the first nest occupied by PA in NS 3. PB defended the territory and reconstructed the nest up to 19 January 2002 (the date of the last observation of the two individuals in NS 3). After this, these vultures were not seen again. During a later visit, an individual from PB was located incubating in one of the nests in NS 1.

The mean laying dates obtained between 1996 and 2002 differ significantly from one territory to another (PA: 10 January, PB: 28 January; Table 1; Mann-Whitney *U*-test,  $z = -2.74$ ,  $P = 0.0062$ ). Plumage characteristics of the individuals and egg-laying dates (the latter being very regular each year for all pairs; Margalida et al. 2003) corroborate the fact that NS 3 was shared in different years by PA and PB (Table 1).

#### DISCUSSION

In the study area, the mean distance from one nest to the next nearest nest in the same territory is 1.7 km, a

greater distance than the one obtained by Donazar et al. (1993) in the central Pyrenees (0.99 km). This fact may facilitate nest “stealing” and force the Bearded Vulture to look for other sites to breed (Margalida and García 1999). The territory change observed (moving 11 km) could be a result of nests being occupied by heterospecifics. However, PB had all its nests and had stayed in the same territory for 18 years. In many bird species, the decision to move can be based on the individual’s current breeding success or the quality of the territory (Switzer 1997). On the other hand, in several raptor species, mate loss and lack of breeding success are related to territory changes (Newton and Marquiss 1982, Wiklund 1996, Forero et al. 1999). The change in egg-laying dates (delayed 1 mo) in PB suggests that at least one of the members of the pair was replaced (probably the female), because egg-laying dates are usually very regular from one year to the next in the Bearded Vulture, with a year to year variability of  $\pm 4.2$  d (Margalida et al. 2003). Because the replacement must have taken place 5–6 yr before the territory change, this fact does not appear to explain the territory switch. Moreover, after the successful breeding in NS 3, PB returned to its original territory (NS 1) and this behavior has been observed in other pairs in which there have apparently been no replacement breeders (e.g., in territories with nests usurped by Eurasian Griffons).

Another factor, which might explain the move may be related to human disturbance, which can disrupt this species’ breeding success (Donazar et al. 1993, Margalida et al. 2003). Because human pressure is scarce in the three study sectors (pers. observ.), this factor does not appear to explain the move. Moreover, after success for breeding in NS 3, PB went back to its old breeding sector (NS 1).

Finally, the increases in the density of birds can cause interactions related to competition for the exploitation of trophic resources and nesting sites (Margalida and

Bertran 2003), which lead to the territory change. Nevertheless, during the monitoring of the pairs studied, no agonistic interactions were observed. In addition, in the study area there are pairs that coexist at lesser distances and they breed successfully and have not changed territory (Margalida and García 2002, pers. observ.).

None of the hypotheses consider appear to explain the observed nest change by Bearded Vultures. Although the specific reasons for the move are unknown, one plausible explanation would be related to the availability of trophic resources and the optimization of the time and energy this would have meant for the pair in order to exploit them. The effect of breeding success and mate loss on territory change could be a consequence of a response to local food conditions (Newton and Marquiss 1982). The presence of predictable sources of food, whether these be feeding stations or areas where farmers leave carcasses, may be favoring the settlement of new pairs (Heredia and Razin 1999, A. Margalida pers. observ.). The search for, preparation, and transportation, of food represent a significant part of the parental effort the species invests during breeding (Margalida and Bertran 2000b, Margalida and Bertran 2001). The extensive foraging areas usually exploited by a pair of Bearded Vultures measure over 600 km<sup>2</sup> and lineal movements of over 30 km can be common (pers. observ.). PB regularly exploited areas closer to NS 3 than NS 1 as a result of the abundance of Pyrenean Chamois (*Rupicapra pyrenaica*), a very important trophic resource for breeding pairs in the Pyrenees (Margalida et al. 1997, García and Margalida 1998). Movements during foraging should allow the pair to learn of potential nesting sites as well as the seasonal changes in the availability of trophic resources. A seasonal and temporary abundance in food resources may have caused this move, with the benefits (proximity of abundant food and quality territory indicated by the presence of conspecific nests) outweighing the costs of temporarily abandoning the original territory (loss of the territory and the nests due to conspecifics or heterospecifics occupying them). Nevertheless, more information is needed on territory change in this species to explain the proximate factors stimulating such action.

**RESUMEN.**—Documentamos el cambio de sector de nidificación en una pareja de quebrantahuesos (*Gypaetus barbatus*) y el primer caso de ocupación de un nido de la misma especie. En 1996 una pareja (A) colonizó el sector de nidificación NS 3. La pareja vecina más próxima (B) se encontraba a 11 km en NS 1. En 1997 la pareja A se trasladó a NS 2, situándose a tan sólo 3 km de NS 1. Tras nidificar 18 años en NS 1, en el año 2000 la pareja B se desplazó a NS 3 para ocupar el nido reconstruido por la pareja A en 1996, criando con éxito. Finalmente en 2001 la pareja B volvió a NS 1 quedando vacante de nuevo NS 3. Aunque se desconocen las causas de dicho desplazamiento, el cambio de sector de nidificación podría estar relacionado con la posible abundancia temporal de los

recursos tróficos y la optimización en la explotación de éstos, ya que NS 3 se encuentra en el área de campeo habitual de la pareja B.

[Traducción de los autores]

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## IMMATURE NORTHERN GOSHAWK CAPTURES, KILLS, AND FEEDS ON ADULT-SIZED WILD TURKEY

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KEY WORDS: *Northern Goshawk*; *Accipiter gentilis*; *Wild Turkey*; *Meleagris gallopavo*; *food-niche breadth*; *predator-prey interaction*; *sexual size dimorphism*.

The largest prey regularly taken by Northern Goshawks

(*Accipiter gentilis*) are snowshoe hares (*Lepus americanus*) (Squires and Reynolds 1997). Although remains of Wild Turkey (*Meleagris gallopavo*) polts have been reported in goshawk pellets (Bosakowski et al. 1992), we are not aware of any record of a Northern Goshawk killing a full-grown turkey. Here, we document an observation of an immature goshawk killing and feeding on a full-grown juvenile Wild Turkey in Connecticut.

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Figure 1. A female Northern Goshawk (*Accipiter gentilis*) stands above a yearling Wild Turkey (*Meleagris gallopavo*) that it killed the previous day at Lyme, Connecticut. Photograph (by H. Golet) taken with a Kodak 240 digital camera through a Bushnell Spacemaster spotting scope (20× eyepiece) from ca. 10 meters away.

#### METHODS

The initial attack sequence was observed (by A. Colton) at close range from inside the house. All subsequent observations were made outside with binoculars and a spotting scope (ca. 30 m from the carcass) aimed through a gap in a mountain laurel (*Kalmia latifolia*) thicket. Continuous observation periods were confined to the early mornings of each day, with additional checks on the carcass being opportunistic in nature. It is likely, therefore, that some feeding bouts were missed.

#### RESULTS

On 12 March 2002, at about 0930 H a Northern Goshawk (*Accipiter gentilis*) attacked and killed a Wild Turkey (*Meleagris gallopavo*) at Lyme, Connecticut. The goshawk struck the turkey while it was feeding alone in a small clearing (3 m × 12 m) beneath a backyard bird feeder situated ca. 5 m from a house. The turkey was unaware of the goshawk until the moment of the attack, when a strike on the back drove it to the ground, causing an explosion of feathers. Immediately the turkey rose and ran, head down, toward a nearby (2 m away) mountain laurel thicket that forms the border of the clearing and the adjoining oak/hickory (*Quercus/Carya*) forest. The hawk “rode” the turkey to the edge of the clearing, but then jumped off and gave pursuit by flight. The goshawk pumped its wings rapidly while flying within 1 m of the ground. Approximately 50 m from the location of the original attack the goshawk overtook the turkey, again pouncing upon its back. Meanwhile, a small flock of American Crows (*Corvus brachyrhynchos*) congregated overhead, flying in a tight circle and calling loudly. Pinned to the ground breast down, the turkey beat its wings frantically while the hawk used a kneading action,



Figure 2. The yearling Wild Turkey carcass as it appeared after the first day of being fed upon by the goshawk. Photograph by H. Golet.

repeatedly bending over and straightening up, to drive its talons into its prey. Less than 1 min elapsed from the time the hawk initially struck the turkey until it ceased flapping, unconscious on the forest floor. Shortly after the completion of the kill, the goshawk and nearby crows flew off, perhaps startled by our presence.

Within 1 hr the goshawk returned to feed on the turkey. It crouched virtually motionless for the first 5 min following its return, although at one point it spread its wings (mantling) briefly to obscure the carcass from the view of a Turkey Vulture (*Cathartes aura*) that flew overhead. The goshawk then began to pluck and eat the turkey, which it did for 20 min before flying off. The hawk made no attempt to conceal the turkey by covering it with leaves prior to its departure, although caching behavior has been observed in goshawks previously (Schnell 1958).

The hawk fed on the carcass sporadically over the next three days (Fig. 1), primarily in the early morning, but occasionally at midday and in the early afternoon. On 14 March (day 3) it arrived before 0610 H, and fed for over 1 hr. The hawk consumed much of the dorsal musculature (primarily the iliotibialis and latissimus dorsi) on the back of the turkey, partially exposing the bones of the sternum and pelvic girdle (Fig. 2). At 0615 H on 15 March (day 4), the turkey carcass was found ca. 3 m from its original position. The hawk did not return to the carcass that morning (through 0830 H at least), although it was observed feeding on it at around 1300 H. Sometime after this observation, and before 0600 H on 16 March (day 5), the carcass disappeared from the site, although there was no evidence of it being dragged through the leaves as was the case the day before.

#### DISCUSSION

Our observation suggests that prey biomass and food-niche breadth may be larger for goshawks than has been

previously recognized. We know of no previous record of a goshawk killing a full-grown Wild Turkey, although the remains of turkey poultts have been found in goshawk pellets in the New Jersey–New York Highlands (Bosakowski et al. 1992) and in prey remains on the Kaibab Plateau, Arizona (R. Reynolds pers. comm.). The immature goshawk, which we suggest was a female based on a consultation with R. Reynolds (pers. comm.), likely weighed ca. 1005 g (Mueller et al. 1976), or ca. 4× as much as the yearling female turkey (ca. 3900 g, Eaton 1992). This prey differential is 63% larger than what has been previously recorded for female goshawks (2.4× their mass) based on observations of snowshoe hare (*Lepus americanus*) predation (Squires and Reynolds 1997). The turkey weighed ca. 10× the previously calculated mean goshawk prey masses of 307 g in Oregon (Reynolds and Meslow 1984), 271 g in New Jersey (Bosakowski et al. 1992), 303 g in Connecticut (Bosakowski et al. 1992), and 413 g in Washington (Watson et al. 1998).

Also to our knowledge, this is the first record of a goshawk returning to a prey item over such an extended period (four successive days), although a mule deer (*Odocoileus hemionus*) gut pile left by hunters in Wyoming was visited by a goshawk on two subsequent days (Squires 1995).

The turkey may have been easier to kill because it was a yearling. Goshawks frequently catch young animals (Opdam et al. 1977), which may be more vulnerable to predation than adults, especially in late winter (Cresswell and Whitfield 1994). Although we cannot be certain that being alone increased the susceptibility of the turkey to attack, this is likely, as group living is advantageous in terms of predator avoidance (Pulliam and Caraco 1984).

Adult turkeys are typically preyed upon by mammalian predators (primarily coyotes, [*Canis latrans*]; Wright et al. 1996, Hubbard et al. 1999), although Great Horned Owls (*Bubo virginianus*; Hubbard et al. 1999), have been observed to take turkey hens from night roosts (Wright et al. 1996). Other raptors, including Golden Eagles (*Aquila chrysaetos*; Bent 1937) and Barred Owls (*Strix varia*; Vangilder and Kurzejeski 1995) have also been reported to prey on turkey hens, albeit infrequently.

Goshawks are sexually dimorphic with females typically weighing 20–40% more than males (Squires and Reynolds 1997). This leads to the predictions that females should have wider food niche breadth, and be better buffered against fluctuations in prey availability than males (Optimality Theory; Schoener 1971). Our observation, although of only one prey capture, suggests that female goshawks may indeed have a wider range of prey availability. This contrasts with previous studies conducted during the breeding season that found similar prey sizes among the sexes (Snyder and Wiley 1976, Widén 1989, Boal and Mannan 1996), but is in accord with a recent radiotelemetry study conducted during winter in northern Finland. In the winter study, Tornberg and Colpaert (2001) found that as forest grouse (Tetraonidae)

availability declined, female goshawks switched to preying upon hares, although males did not appear to have this option. Periods of reduced prey availability, such as winter (the season of our observation), may best reveal limits of prey capture that are imposed by body size.

RESUMEN.—En marzo 12 de 2002, cerca de las 0930 H un azor norteño (*Accipiter gentilis*) ataco, mató y seguidamente se alimento de un pavo silvestre (*Meleagris gallopavo*) de primer año, mientras este se alimentaba solo en un pequeño claro (3 m H 12 m) debajo de un alimentador para aves de jardín situado aproximadamente a 5 m de la casa. Aunque se conoce que los azores depredan sobre pavipollos, para nuestro conocimiento, este es el primer registro de que un primal sea atacado. Este encuentro es notable dada la gran diferencia en tamaño entre el depredador y la presa. Se estimó que el pavo era 4 veces mas pesado que el azor, y aun así este fue sometido con relativa facilidad. El gavilán se alimento del cadáver sucesivamente durante 4 días. Las interacciones entre estas dos especies en Nueva Inglaterra pueden ser más comunes ahora que en las décadas pasadas dado que las poblaciones tanto de pavos como de azores parecen haber aumentado.

[Traducción de César Márquez]

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## A HELPER AT THE NEST OF PEREGRINE FALCONS IN NORTHERN JAPAN

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KEY WORDS: *Peregrine Falcon*; *Falco peregrinus japonensis*; feeding; helper; juvenile; territory.

Peregrine Falcons (*Falco peregrinus*) are widely distributed throughout the world, and primarily breed in monogamous pairs that display aggressive territorial behavior around their nest sites (Cade 1982). Despite numerous studies of this species in Europe, North America, and elsewhere (e.g., Cade et al. 1988), information on the ecology of Peregrine Falcons (*F. p. japonensis*) in East Asia is very limited. Cooperative breeding is infrequent in this species (Skutch 1987) with reported exceptions in North America and in France (Spofford 1969, Monneret 1983). Here, we describe an observation of helping at the nest of a Peregrine Falcon in Hokkaido, northern Japan.

### STUDY AREA AND METHODS

We recorded observations on the behavior and the breeding status of Peregrine Falcons for about 120 hr each year from 1993–2002 at a study site on the Etomo Peninsula in Muroran Hokkaido, northern Japan (42°19'N, 140°59'E). Six pairs of non-migratory Peregrines (pairs B to G) occur on a 10-km stretch of vertical cliffs, part of which is more than 100 m in height, along the narrow Etomo Peninsula (1.0–3.5 km in width; Kumagai 1989, Ueta et al. 1995). The six sites fledged 1.1 young ( $\pm 1.1$  SD) per pair per year from 1993–2002. This peninsula is also a major landfall and point of departure for migrating songbirds and raptors, and in 1998 the banding station on the peninsula recorded 57 species of small- to medium-sized land birds (T. Banno pers. comm.), which are suitable prey for Peregrines in Japan (Yamada 2002).

We used 20× binoculars and a 77× spotting scope to make observations. Because no peregrines in Japan have been marked, we attempted to identify individual birds by their characteristic features such as malar patches, ventral marks (Enderson and Craig 1988), favorite look-out perches and behavior toward the observers.

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## RESULTS AND DISCUSSION

The territorial pairs in the study area usually showed aggression toward intruding peregrines that flew <5 m above, or close to (<100 m) the nesting cliffs. On 17 March 2001, however, we observed a Peregrine with vertically striped breast and belly perched on a crossarm of a power pylon about 3 m from an adult male in the territory of the "F" pair. The former was obviously larger than the adult male. The adult male showed no sign of aggression toward this juvenile. About 5 min later, the juvenile peregrine departed to chase a feral rock dove (*Columba livia*), but came back without the prey and probably perched out of sight on the cliff, ca. 100 m from the male. An adult pair had been seen together on the wing at this site two days earlier, on 15 March. Moreover, later on 17 March we observed a copulation between the adult pair on the nesting cliff.

On 24 May 2001, a peregrine with juvenile plumage carried a small prey item into the nest of the same pair. There were three nestlings with white down in the nest; they appeared to be about 2-wk old. The juvenile fed the nestlings for about 10 min and then flew around the point to the other side of the cliff. During this feeding bout, the adult pair were out of sight and neither called or came to chase away the juvenile. The territorial adult female glided by the nesting cliff about 20 min after this feeding. The clutch of three young fledged successfully in mid-June, when this juvenile helper was no longer observed at this site.

Peregrine Falcons have a broad range of post-nesting dependency periods, and particularly, non-migratory populations of falcons are reported to have prolonged juvenile-dependency periods (Sherrod 1983). In our study area, some juvenile peregrines were observed to stay in the natal territories long after the breeding season; five cases recorded in October (ca. 4 mo after fledging), one in December, and two in March. We assumed that these juveniles were the offspring of the territory owners for several reasons. First, they provoked no aggression from the territorial pairs. Secondly, no tolerated juveniles were observed in the territories of the pairs that failed to raise broods that year or the previous year.

Similar to our observations, Monneret (1983) reported the helping behavior of juvenile peregrines at nests of adult pairs in the 1970s and 1980s. He suggested that juveniles were tolerated in their natal territories well into the breeding season, through the period when eggs and nestlings were present. The appearance of chicks probably triggered brooding or feeding behavior by the juveniles, inducing them to become helpers at the nests. Monneret emphasized the importance of the kinship of the adult pair to the third falcon for the development of helping behavior to occur, because the parents would tolerate their own young more easily than strangers in their territory. Delayed dispersal is a common route to the evolution of cooperative breeding in many birds including raptors (Kimball et al. 2003).

Recent reviews on cooperative breeding suggest that both ecological constraints and life history traits such as low adult mortality and low dispersal rate are important factors in the evolution of cooperative breeding (Arnold and Owens 1999, Hatchwell and Komdeur 2000). Peregrines nest on cliffs over much of their range, which restricts their breeding distribution (Newton 1988). They are relatively long-living with low adult mortality (Ratcliffe 1993). Also this species is known to be philopatric (Newton and Mearns 1988). Therefore, Peregrine Falcons in Muroran Hokkaido may provide an interesting example of an early stage in the evolution of a cooperative breeding system. This sporadically-expressed behavior should be monitored and documented in other peregrine studies and observers should look for factors that correlate with and may foster this breeding system.

RESUMEN.—Lo halcones peregrinos (*Falco peregrinus*) usualmente procrean en parejas monogamas que despliegan un comportamiento territorial agresivo alrededor de sus sitios nido. A pesar de su territorialidad y agresión hacia los intrusos, una hembra juvenil fue observada en el territorio de una pareja adulta durante la estación reproductiva en un sitio de anidación del halcón peregrino (*F. p. japonensis*) en Muroran, Hokkaido, norte del Japón. Esta trajo comida y alimento a los polluelos de la pareja que estaba anidando sin que fuera expulsada. Adicionalmente a este individuo, otros juveniles fueron tolerados en sus territorios natales largo tiempo después de la estación reproductiva, lo cual puede ocasionalmente conducir a que cooperen en el nido, en la región de Hokkaido. Esta especie en el norte del Japón puede ser un ejemplo interesante de un temprano estado en la evolución de la reproducción cooperativa en rapaces.

[Traducción de César Márquez]

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## TWO CASES OF COOPERATIVE BREEDING IN EURASIAN HOBBIES

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KEY WORDS: *Eurasian Hobby*; *Falco subbuteo*; *cooperative behavior*; *polyandry*; *polygamy*.

Polygamy is a common mating system in birds (Alcock 1993), and it can involve either polygyny (a male breeding with two or more females) or polyandry (a female

breeding with two or more males). A recent review of literature shows that group breeding has been documented in 42 species of diurnal raptors (Kimball et al. 2003). Furthermore, these authors suggest that cooperative breeding is more common in terms of the number of species and in frequency than the available data indicate. In addition, they reported that 21 species of raptors showed polyandrous behavior and four other species may be polyandrous.

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Polyandry may occur by extra-pair copulations (EPCs) or be a result of two or more males breeding and maintaining a bond with one female. The first case is not commonly known in most species of raptors, in part, because the reported frequencies of mixed paternity based on DNA fingerprinting analyses are scanty (e.g., Korpimäki et al. 1996, Rohner 1997, Marks et al. 1999). The second case involves cooperative breeding behavior between two or more males and one female. Males may be related or not and the number of males that copulate with a female varies between species and sometimes only one male of the group copulates (Kimball et al. 2003).

Polygamy has been rarely documented in Eurasian Hobbies (*Falco subbuteo*). Chapman (1999) found that a stray third bird, which could have been a yearling, would consort occasionally with an established pair. Bijlsma (1980) found that out of 26 migratory arrivals of hobbies in the Netherlands, seven involved single birds, 14 involved pairs, and five a "trio" of hobbies. It is unknown whether these trios became cooperatively breeding groups or not. Cramp and Simmons (1980) described a case in which two males accompanied a female when inspecting a nest. Here, we describe two different cases of cooperative breeding by hobbies in the North of Spain.

In 2000, we began monitoring hobbies in Bizkaia, Northern Spain, and 41 pairs were found. In 2001, two of these territories were occupied by pairs, but were occupied by trios in 2002. One territory was located in a large pine (*Pinus radiata*) plantation in Muskiz and the other was in a large eucalyptus (*Eucalyptus globulus*) plantation in Derio.

The pair at Muskiz bred in a pine situated in the middle of the plantation in 2001 and they produced two fledglings. In 2002, we found the nest located only 5 m away from the previous year's nest. On 4 August 2002, we witnessed three hobbies flying together for 5 hr at the pine plantation territory. One hobby (the female) was obviously larger than the other two (the males); these size differences were evident, mainly when the falcons were flying together (Chapman 1999). The female and one male were then aged as 3+ yr and the other male was a second-year bird (Forsman 1997). They, together, mobbed two Common Buzzards (*Buteo buteo*). Later, one of the males captured a passerine and passed it to the female, who ate it while flying near the two males. Afterwards, the female and one of the males perched on the top of the trees close to the nest. Thirty minutes later, the other male approached and flew over the perched falcons. Then the two males left together, while the female went to the nest.

On 10 August, we observed one hobby 1 km away from the nest carrying a Common Swift (*Apus apus*). This falcon approached the nest after perching in a pine to pluck the bird. Two hours later, we saw one of the males trying to hunt another swift, then join the female and mob three different Common Buzzards. On 11 August, the three hobbies attacked an Eurasian Eagle-Owl (*Bubo*

*bubo*) that was 20 m from the nest. On 31 August, two chicks were flying, but always stayed close to the nest during a 5 hr period, while the female and the two males perched near the chicks. The three adult hobbies never showed signs of aggression between themselves.

In the territory at Derio in 2001, we observed a pair of hobbies, but could not find the nest because of the dense tree canopy. In 2002, we spent 3 d searching for the nest. On the first 2 d (27 and 28 July), we saw three hobbies flying together, one adult female (larger) and two males. We could not see the tail feathers of the males clearly, so we could not determine their ages (Forsman 1997). On the second day, the falcons flew together for a period of 5 hr. On the third day (1 August), under a persistent rain, we saw only one hobby and found the nest in a eucalyptus tree in the middle of the plantation. On 3 August, we climbed to the nest and found it empty, and did not see any hobbies while we were there. As we were leaving the plantation, we saw a Booted Eagle (*Hieraetus pennatus*) flying above, and the three hobbies attacked it. The falcons chased the eagle for more than 1 km, and then returned to the plantation.

Based on our observations, we could not distinguish whether these trios represented polyandrous breeding or helping behavior. It is possible that the second males in the Bizkaia trios could have been yearlings born in the same territories, who returned and then helped to defend the territory. By helping, these yearlings could gain experience that would prove useful in their third calendar year, when male hobbies usually reach sexual maturity. Most second-year hobbies behave as floaters (Chapman 1999), and they are strongly philopatric, as shown by the fact that they return from their wintering areas to within a mean of 10 km from their natal sites (Chapman 1999), which could lead to helping behavior. The age of the second male at the pine plantation nest was consistent with the interpretation that this bird was a helper. The benefits of the helping behavior for the adult pair were not clear (Donazar 1993). In the two trios monitored, reproductive success was relatively low (one of two nests fledged a total of two young), but this sample was extremely small.

RESUMEN.—En el año 2000 comenzamos un estudio sobre el alcotán europeo (*Falco subbuteo*) en Bizkaia, Norte de España. Localizamos 41 territorios. Un año después se controlaron varias parejas, a las que se realizó un estrecho seguimiento. En el año 2002 continuamos el seguimiento y encontramos que en dos de los territorios controlados un año antes, había tríos poliándricos, es decir, dos machos y una hembra compartiendo las labores de defensa de territorio y reproducción. Realizando un seguimiento más exhaustivo, comprobamos que los dos machos de cada territorio participaban en la ceba y expulsaban ferozmente a cualquier intruso que se acercase al nido. En una de las parejas pudimos comprobar que la hembra y uno de los machos eran individuos adultos,

mientras que el otro era un ejemplar nacido el año anterior, por lo que sospechamos que podría ser un pollo de la pareja que colabora con sus progenitores para sacar adelante la nidada y, de esta forma, va cogiendo experiencia mientras llega a la madurez sexual que se produce un año después.

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### FOOD HABITS OF PEREGRINE FALCONS IN KENTUCKY

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**KEY WORDS:** *Peregrine Falcon; Falco peregrinus; food habits; human-made habitat; Kentucky; monitoring.*

Many studies on Peregrine Falcons (*Falco peregrinus*) include observations of prey taken (White et al. 2002). As a consequence of the Peregrine Falcon's cosmopolitan distribution and adaptability, inferences derived from food habit studies often are limited to the study area in which they were conducted (Ratcliffe 1993, Schneider and Wilden 1994, Rejt 2001, Serra et al. 2001). In the Midwestern United States, a large proportion (0.70) of

the restored Peregrine Falcon population occupies structurally similar human-made breeding locations (Tordoff et al. 2001); however, land use adjacent to these breeding locations often is variable and may be reflected in the diet of the birds. For example, in Kentucky, three pairs of Peregrine Falcons occupy human-made breeding locations including bridges and power plants. Land use adjacent to the breeding locations varies from predominantly urban at one breeding location to predominantly rural at the other two breeding locations.

Food-habits data from this population would be useful in enhancing our understanding of prey use among habitat types within the region, monitoring potential expo-

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sure to contaminants in restored populations, and developing future management and conservation strategies. However, few studies have quantified Peregrine Falcon food habits in this region (Myers and Pease 1995). As part of a statewide effort to monitor Peregrine Falcons, our objective was to quantify and to compare the diet of Peregrine Falcons in Kentucky among habitats and seasons.

#### STUDY AREAS AND METHODS

The study area is a 175-km segment of the Ohio River Valley in the Outer Bluegrass Physiographic Region of Kentucky. Breeding locations include an interstate bridge spanning the Ohio River in Jefferson County (Louisville, KY; Urban I), and smokestacks associated with power plants in Trimble and Carroll counties (Rural I and II, respectively). The topography at these sites is nearly level to moderately sloping and dominated by upland oaks (i.e., *Quercus rubra*, *Q. alba*), hickories (*Carya* spp.), and yellow-poplar (*Liriodendron tulipifera*) on the slopes and pin oak (*Quercus palustris*), eastern cottonwood (*Populus deltoides*), silver maple (*Acer saccharinum*), and eastern sycamore (*Platanus occidentalis*) in the floodplains.

Land use adjacent to these breeding locations was the most notable difference among sites. To characterize general landscape attributes at these sites, we used a geographic information system (GIS; ArcView®, ESRI, Redlands, California) and Kentucky and Indiana GAP analysis Program data (Center for Remote Sensing and GIS 1996, Mid-American Remote Sensing Center 2001) to determine the proportion of three land use classes, including urban/developed, agriculture, and forest, within a 10-km radius of each breeding location.

We collected prey remains and pellets, and recorded visual observations of foraging Peregrine Falcons at territorial sites during March 1999–December 2001. During spring and summer, Peregrine Falcons bring much of their prey to the nest associated with courtship and the feeding of young, allowing a greater number of specimens to be collected compared to fall and winter (Ratcliffe 1993). We examined pellet contents according to the methods of Sabo and Laybourne (1994). We compared prey remains to museum specimens for identification (Oro and Tella 1995). Each species identified in a pellet or as a prey remain was considered to be one occurrence, unless multiples of the same body part were present (Mersmann et al. 1992).

We calculated and compared percent occurrence and percent biomass of prey taken over all sites, among sites and seasons, and according to seasonal status of the prey (i.e., resident, summer resident, winter resident, transient; Mengel 1965). For analyses among seasons, we classified samples as spring (March–May), summer (June–August), fall (September–November), and winter (December–February). We calculated percent biomass by multiplying the number of individuals in a species by the mean mass for that species and then divided that value by the total biomass of all species (Dunning 1984, Corser et al. 1999). We calculated Simpson's index of diversity (1-*D*) and compared diversity indices among sites and seasons (Krebs 1989). Simpson's index varies between 0 and 1, with diversity increasing as values approach one.

Table 1. Percent occurrence and percent biomass of prey taken by three pairs of Peregrine Falcons in Kentucky based on prey remains, pellets, and observations of prey captures ( $N = 465$ ), March 1999–December 2001<sup>a</sup>. Scientific names of bird species are given in Appendix

SPECIES	FREQUENCY (%)	BIOMASS (%)
European Starling <sup>b,c,d</sup>	35	14
Rock Dove <sup>b,c,d</sup>	27	70
Eastern Meadowlark <sup>b,c,d</sup>	8	3
Blue Jay <sup>b,c,d</sup>	7	3
Brown-headed Cowbird <sup>c,d</sup>	5	1
Red-winged Blackbird <sup>c,d</sup>	4	1
Northern Flicker <sup>c,d</sup>	3	2
Common Grackle <sup>b,c,d</sup>	2	1
Mourning Dove <sup>b,c,d</sup>	2	1
Northern Mockingbird <sup>c,d</sup>	2	0.4

<sup>a</sup> Species comprising <1% of observations (0.1–0.7% of total biomass) included American Robin<sup>c</sup>, Killdeer<sup>c</sup>, Red-bellied Woodpecker<sup>c</sup>, Eastern Towhee<sup>c,d</sup>, House Sparrow<sup>c</sup>, Wood Thrush<sup>d</sup>, Yellow-billed Cuckoo<sup>b</sup>, Northern Cardinal<sup>c</sup>, Scarlet Tanager<sup>c</sup>, Northern Rough-winged Swallow<sup>c</sup>, American Coot<sup>c</sup>, Pectoral Sandpiper<sup>d</sup>, Bonaparte's Gull<sup>d</sup>, Common Nighthawk<sup>d</sup>, Brown Thrasher<sup>d</sup>, White-eyed Vireo<sup>d</sup>, Cedar Waxwing<sup>d</sup>, Hairy Woodpecker<sup>d</sup>, Baltimore Oriole<sup>c</sup>, and a bat<sup>c</sup>.

<sup>b</sup> Species recorded at Jefferson County, KY (Urban I).

<sup>c</sup> Species recorded at Trimble County, KY (Rural I).

<sup>d</sup> Species recorded at Carroll County, KY (Rural II).

We derived 95% confidence intervals for Simpson's index using a jackknife resampling technique (Krebs 1989), and we report results as Simpson's index  $\pm$  95% confidence interval.

#### RESULTS

General landscape attributes differed among breeding locations. Urban I is predominantly urban/developed (0.70), with some agriculture (0.14), and little forest (0.07 of landscape within 10-km radius). Rural I and Rural II are largely agriculture (0.48 and 0.46, respectively) and forest (0.44 and 0.47, respectively), with little urbanization/development (0.02 and 0.01, respectively).

In all, we collected 465 samples ( $N = 384$  prey remains,  $N = 54$  pellets,  $N = 27$  observed prey captures). We collected 212 samples at Rural I, 192 at Rural II, and 61 at Urban I. At Urban I, many prey remains were inaccessible and likely fell into the Ohio River. We identified 21 different prey species at Rural I, 20 at Rural II, and six at Urban I (Table 1). We identified 20 different prey species during spring, 24 during summer, five during fall, and seven during winter. Across all sites and seasons combined, Rock Doves (*Columba livia*) and European Starlings (*Sturnus vulgaris*) were the most frequent prey and comprised 84% of the total biomass in the diet of Pere-

Table 2. Comparison of percent occurrence and percent biomass of prey taken among three pairs of Peregrine Falcons in Kentucky based on prey remains, pellets, and observations of prey captures ( $N = 465$ ), March 1999–December 2001.<sup>a</sup> Scientific names are given in Appendix.

SPECIES	JEFFERSON COUNTY (URBAN I; $N = 61$ )		TRIMBLE COUNTY (RURAL I; $N = 212$ )		CARROLL COUNTY (RURAL II; $N = 192$ )	
	FREQUENCY (%)	BIOMASS (%)	FREQUENCY (%)	BIOMASS (%)	FREQUENCY (%)	BIOMASS (%)
European Starling	12	2	18	8	60	32
Rock Dove	75	95	23	67	16	56
Eastern Meadowlark	0	0	13	6	3	2
Blue Jay	0	0	9	4	7	4
Brown-headed Cowbird	0	0	9	2	0	0
Red-winged Blackbird	0	0	9	3	0	0

<sup>a</sup> Percent occurrence and percent biomass of all other species was  $\leq 5\%$ .

grine Falcons in Kentucky (Table 1). Peregrine Falcons preyed heavily on resident birds ( $>97\%$ ).

At Urban I, Rock Doves and European Starlings were the most frequent prey, but Rock Doves comprised 95% of the biomass in the diet (Table 2). At Rural I, Rock Doves, European Starlings, and Eastern Meadowlarks (*Sturnella magna*) had the highest percent occurrence and together comprised 81% of the biomass in the diet (Table 2). At Rural II, European Starlings were the most frequent prey; however, Rock Doves were a majority of the biomass in the diet of falcons (Table 2). The diet of at Rural I was most diverse ( $1-D = 0.88 \pm 0.01$ ), followed by Rural 2 ( $1-D = 0.63 \pm 0.04$ ), and Urban I ( $1-D = 0.52 \pm 0.24$ ).

In spring ( $N = 112$ ), European Starlings, Eastern Meadowlarks, and Rock Doves were the most frequent prey taken and comprised the majority of the biomass in

the diet (Table 3). In summer ( $N = 319$ ), European Starlings and Rock Doves together comprised  $\geq 70\%$  occurrence and  $>90\%$  of the biomass in the diet (Table 3). During fall ( $N = 14$ ), European Starlings and Blue Jays (*Cyanocitta cristata*) were the most frequent prey and comprised the majority of the biomass in the diet (Table 3). In winter ( $N = 20$ ), European Starlings were the majority of the diet in terms of occurrence and biomass (Table 3). The diet of Peregrine Falcons was most diverse in spring ( $1-D = 0.88 \pm 0.01$ ), slightly less diverse in fall and summer ( $1-D = 0.80 \pm 0.06$ , and  $1-D = 0.75 \pm 0.01$ , respectively), and least diverse in winter ( $1-D = 0.62 \pm 0.15$ ).

#### DISCUSSION

Results of this study are consistent with other research that identified Rock Doves as a primary prey item in the

Table 3. Comparison of percent occurrence and percent biomass of prey taken by Peregrine Falcons among seasons in Kentucky based on prey remains, pellets, and observations of prey captures (by/  $N = 465$ ), March 1999–December 2001.<sup>a</sup> Scientific names of bird species are given in Appendix.

SPECIES	SPRING ( $N = 112$ )		SUMMER ( $N = 319$ )		FALL ( $N = 14$ )		WINTER ( $N = 20$ )	
	FREQ. (%)	BIOMASS (%)	FREQ. (%)	BIOMASS (%)	FREQ. (%)	BIOMASS (%)	FREQ. (%)	BIOMASS (%)
European Starling	22	12	37	13	36	19	65	40
Rock Dove	14	51	33	78	14	50	5	20
Eastern Meadowlark	18	10	0	0	0	0	5	3
Blue Jay	13	7	0	0	29	16	10	6
Common Grackle	7	5	0	0	0	0	0	0
Northern Flicker	5	5	0	0	14	12	5	5
Brown-headed Cowbird	0	0	6	1	7	2	0	0
Red-winged Blackbird	0	0	5	1	0	0	0	0
Red-bellied Woodpecker	0	0	0	0	0	0	5	2
American Coot	0	0	0	0	0	0	5	24

<sup>a</sup> Percent occurrence and percent biomass of all other species was  $\leq 5\%$ .

diet of Peregrine Falcons in urban and industrial habitats (Barber and Barber 1988, Myers and Pease 1995, Cade et al. 1996a). Worldwide, Peregrine Falcons consistently rely on Columbidae as their main prey (Ratcliffe 1993, White et al. 2002). Other species recorded in the diet of urban Peregrine Falcons represented locally abundant and resident species (Barber and Barber 1983, 1988, Bell et al. 1996, Rejt 2001). Peregrine Falcons in Kentucky exhibited similar tendencies; however, the percent occurrence and biomass of European Starlings in the diet of Peregrine Falcons at Rural I and II were higher than in other studies of urban and non-urban areas of (4.1–11.2% occurrence and 3–5% biomass) in North America (Barber and Barber 1988, Myers and Pease 1995, Corser et al. 1999, Rejt 2001). Herbert and Herbert (1965) reported Peregrine Falcons feeding regularly on European Starlings when they nested near roosting colonies. We observed roosting colonies of European Starlings at both power plants in this study, which may have accounted for their frequent occurrence in the diet of these pairs.

Throughout their range, Peregrine Falcons have been documented to prey on a diverse assemblage of taxa (Ratcliffe 1993, White et al. 2002); however, few studies have compared the diversity of prey in the diets of Peregrine Falcons among territorial pairs. Based on description data, Bell et al. (1996) suggested that one pair of Peregrine Falcons appeared to exhibit a more diverse diet than two other pairs that nested on bridges in the vicinity of San Francisco Bay, CA. Similarly, in our study Peregrine Falcons at Rural I had a more diverse diet compared to the other rural pair. The proximity of Rural I and Rural II to each other and their similarities in breeding structure type (i.e., power plants) would imply similar availability of prey assemblages. The differences in diversity we observed between these two pairs may be attributed to a difference in preference. Several studies examining the diet of Peregrine Falcons have demonstrated individual preferences for a specific prey type. For example, Cade et al. (1996a) surveyed Midwestern Peregrine Falcon food habits and identified cases in which some birds concentrated on single species such as Common Nighthawks (*Chordeiles minor*) in Minnesota or cuckoos (*Cuculus* sp.) in Wisconsin. Nonetheless, information on prey species abundance would be necessary to determine whether differences were attributed to availability of prey or to preference.

Seasonal variation in the diet of Peregrine Falcons could likely represent differences in prey availability (Cade et al. 1996a, Serra et al. 2001). For example, migratory species increase in frequency in the diets of Peregrine Falcons during spring and fall, whereas the percentage of resident birds decreases (Ratcliffe 1993, Cade et al. 1996a, Rejt 2001, Serra et al. 2001). Results of this study reflected the lowest frequency of Rock Doves and European Starlings in the spring coinciding with the presence of migratory species such as Common Nighthawks and Scarlet Tanagers (*Piranga olivacea*). Also, the

presence of migratory species during spring coincided with an increase in diversity in the diet. Some species may be more conspicuous to Peregrine Falcons in the spring because of their mating displays. For example, the increase in the frequency of Eastern Meadowlarks in this study may be attributed to flashy mating displays exhibited in the spring.

Monitoring efforts are important to endangered species recovery (Cade et al. 1996b). Continued vigilance is necessary to assure long-term success of restored populations. For example, although there are no data linking use of urban or industrial breeding locations to reproductive maladies, potential contaminants and possible routes of exposure through prey should be examined (e.g., Fimreite et al. 1970, DeMent et al. 1986, Cade and Bird 1990, Mora et al. 2002). Monitoring food habits and other detailed aspects of feeding ecology in restored Peregrine Falcon populations can be useful in detecting long-term population exposure to food-related threats and in developing proactive management strategies.

RESUMEN.—Colectamos restos de presas y egagropilas de sitios nido, e hicimos observaciones visuales de captura de presas ( $N = 465$ ) para tres parejas de halcón peregrino (*Falco peregrinus*) que anidaron en Kentucky de marzo a diciembre de 2001. Las aves residentes dan cuenta del 97% de la dieta del halcón, y específicamente la paloma zorita (*Columba livia*) y el estornino europeo (*Sturnus vulgaris*) abarcan el 62% de las presas tomadas. La dieta de una pareja rural del halcón peregrino fue más diversa (índice de diversidad =  $0.88 \pm 0.01$ ) que la de otras parejas. La dieta fue más diversa en primavera (índice de diversidad =  $0.88 \pm 0.01$ ) comparada con otras estaciones. A pesar de las diferencias en el hábitat (urbano versus rural), los halcones peregrinos consistentemente hicieron presa sobre estorninos europeos y palomas zoritas mas que sobre otras especies. Los esfuerzos de monitoreo para restaurar las poblaciones de halcones peregrinos pueden beneficiarse de estudios cuantitativos de los hábitos alimenticios, especialmente donde las especies de presa consumidas pueden ser indicadores de la calidad ambiental.

[Traducción de César Márquez]

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Appendix. Common and scientific names of prey taken by three pairs of Peregrine Falcons nesting in Kentucky based on observations of prey remains, pellets, and prey captures ( $N = 465$ ), March 1999–December 2001.

FAMILY	SCIENTIFIC NAME	COMMON NAME
Rallidae	<i>Fulica americana</i>	American Coot
Charadriidae	<i>Charadrius vociferus</i>	Killdeer
Scolopacidae	<i>Calidris melanotos</i>	Pectoral Sandpiper
Laridae	<i>Larus philadelphia</i>	Bonaparte's Gull
Columbidae	<i>Columba livia</i>	Rock Dove
	<i>Zenaidura macroura</i>	Mourning Dove
Cuculidae	<i>Coccyzus americanus</i>	Yellow-billed Cuckoo
Caprimulgidae	<i>Chordeiles minor</i>	Common Nighthawk
Picidae	<i>Picoides villosus</i>	Hairy Woodpecker
	<i>Melanerpes carolinus</i>	Red-bellied Woodpecker
	<i>Colaptes auratus</i>	Northern Flicker
Vireonidae	<i>Vireo griseus</i>	White-eyed vireo
Corvidae	<i>Cyanocitta cristata</i>	Blue Jay
Hirundinidae	<i>Stelgidopteryx serripennis</i>	Northern Rough-winged Swallow
Turdidae	<i>Hylocichla mustelina</i>	Wood Thrush
	<i>Turdus migratorius</i>	American Robin
Mimidae	<i>Mimus polyglottos</i>	Northern Mockingbird
	<i>Toxostoma rufum</i>	Brown Thrasher
Sturnidae	<i>Sturnus vulgaris</i>	European Starling
Bombycillidae	<i>Bombycilla cedrorum</i>	Cedar Waxwing
Thraupidae	<i>Piranga olivacea</i>	Scarlet Tanager
Emberizidae	<i>Pipilo erythrophthalmus</i>	Rufous-sided Towhee
Cardinalidae	<i>Cardinalis cardinalis</i>	Northern Cardinal
Icteridae	<i>Sturnella magna</i>	Eastern Meadowlark
	<i>Agelaius phoeniceus</i>	Red-winged Blackbird
	<i>Quiscalus quiscula</i>	Common Grackle
	<i>Molothrus ater</i>	Brown-headed Cowbird
	<i>Icterus galbula</i>	Baltimore Oriole
Passeridae	<i>Passer domesticus</i>	House Sparrow
Vespertilionidae	<i>Myotis sp.</i>	Bat

## LETTER

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### HARRIS'S HAWK DIES AFTER COLLIDING WITH MOTOR VEHICLE

The extent to which raptors are killed on roads may depend on a variety of biotic and abiotic factors (e.g., hunting habits, dispersal behavior, vehicular traffic, motor speed). Vehicle mortality is a main source of mortality in populations of Barn Owl, *Tyto alba*, (Newton et al. 1999, Mortality causes in British Barn Owls (*Tyto alba*), based on 1101 carcasses examined during 1963–1996. Pages 299–307 in J.R. Duncan, D.H. Johnson, and T.H. Nicholls [EDS.], *Biology and Conservation of Owls of the Northern Hemisphere*, Second International Symposium, U.S. Department of Agriculture, St. Paul, MN U.S.A.) and Red-tailed Hawk, *Buteo jamaicensis* (Keran 1981, *Raptor Res.* 15:108–112). Motor collision deaths have also been reported in large, diurnal raptors such as the Ferruginous Hawk, *B. regalis*, (Bechard and Schmutz 1995, *In* A. Poole and F. Gill, [EDS.], *The birds of North America*, No. 172. The Academy of Natural Sciences, Philadelphia, PA and the American Ornithologists' Union, Washington, DC U.S.A.), Red-shouldered Hawk, *B. lineatus*, (Keran 1981), and Broad-winged Hawk, *B. platypterus* (Loos and Kerlinger 1993, *J. Raptor Res.* 27:210–213). Nocturnal hunting habits may increase some species' risk of colliding with motor vehicles. For example, bright headlights may temporarily blind owls hunting at night. In support of this idea, the frequency of road mortality in the Little Owl (*Athene noctua*) was slightly lower (2.3%) along illuminated roads (Hernandez 1988, *J. Raptor Res.* 22: 81–84). Species that habitually scavenge road-killed animals are more likely to die due to collisions (Brown and Amadon 1968, *Eagles, hawks, and falcons of the world*, Vol. I. McGraw-Hill Book Co., New York, NY U.S.A.). Within species, certain age and gender classes may be more prone to this type of mortality; specifically, young birds undergoing post-fledging dispersal may be more susceptible to vehicular collisions (Hernandez 1988, Loos and Kerlinger 1993, Massemin et al. 1998, *Ibis* 140:70–75). Poor body condition may increase the likelihood of being struck by a vehicle. However, the body condition of road-killed carcasses compared to captive immature Barn Owls fed *ad libitum* differed only for mature females (Massemin et al. 1998). The type of roadside vegetative cover and available perch height may also contribute to the likelihood that a raptor may be killed on the road (Hernandez 1988). Newton et al. (1999) documented an increase in the proportion of mortality due to collision with vehicles over time in the Barn Owl. They attributed this increase to the construction of more roads, increased traffic volume, and higher vehicle speeds. The extent to which the susceptible raptor species are killed by vehicles may warrant further study where highway infrastructure is expanding.

Quantitative data on the causes for mortality in the Harris's Hawk (*Parabuteo unicinctus*) are lacking (Bednarz 1995, *In* A. Poole and F. Gill [EDS.], *The birds of North America*, No. 146. The Academy of Natural Sciences, Philadelphia, PA, and the American Ornithologists' Union, Washington, DC U.S.A.). Documented cases of mortality in fledged and adult Harris's Hawks include deaths by shooting, Great Horned Owl (*Bubo virginianus*) predation, illegal trapping (Bednarz 1995), electrocution, probable drowning in livestock water tanks, and injuries sustained from cholla cactus (*Opuntia* sp.) that probably would have been fatal had not the researcher intervened (Whaley 1986, *Raptor Res.* 20: 1–15). We provide the first report of a Harris's Hawk dying from collision with a motor vehicle.

On 22 November 2002, we found a dead adult female Harris's Hawk on the ground, 10 m from U.S. Highway 287, Electra, Wichita County, Texas. The hawk had been dead for ca. 3 d. The body lay prostrate, with the head facing away from the road and the tail pointing upward, emerging above the grass, and directed toward the road. The bird was at least 3 yr old judging from its plumage of only adult feathers from two different years (some rectrices and remiges were extremely worn and faded while others were bright and intact). We examined the specimen and found a compound fracture at the proximal end of the left humerus; the humerus was broken in two pieces. The bird appeared to have been in very fit condition. Pectoral muscles protruded on either side of the keel, and a layer of subcutaneous fat covered much of the abdomen. The location of the bird and the nature of its injury implied that it died after colliding with a motor vehicle; there was no other structure nearby with which the bird could have collided.

Also of interest is the location of this specimen outside the reported breeding range. The nearest published population of Harris's Hawks is ca. 290 km away (Bednarz 1995). We did not search the area for additional Harris's Hawks. The habitat consisted of 3–4 m high mesquite (*Prosopis glandulosa*) and appeared to be suitable for the species (Bednarz 1995).

The frequency of Harris's Hawk deaths from collisions with motor vehicles is probably very low (J. Bednarz and

W. Whaley pers. comm.). Harris's Hawks rarely feed on road-killed animals. During his 10-yr study in southeastern New Mexico, J. Bednarz observed only two instances of scavenging, and both involved immature birds in late winter or early spring. The Harris's Hawk's apparent avoidance of road killed animals probably reduces its risk of motor vehicle collision (J. Bednarz pers. comm.). Nonetheless, Harris's Hawks often perch on utility poles and wires, fence posts, and signs along roads. Typical quarry are ground-dwelling mammals and generally low-flying birds associated with scrub (Bednarz 1995). Thus, collision with motor vehicles seems a likely source of mortality for populations of Harris's Hawks living around humans. Moreover, we predict that the frequency of such deaths will increase as Harris's Hawk habitats become more developed by humans.

We thank J. Bednarz, M. Patten, W. Whaley, and H. Woodward for helpful suggestions that improved this manuscript.—**Jennifer O. Coulson, Department of Ecology and Evolutionary Biology, Tulane University, 310 Dinwiddie Hall, New Orleans, LA 70118 U.S.A.; e-mail address: jacoulson@aol.com; and Thomas D. Coulson, Department of Biochemistry, Louisiana State University Medical Center, 1100 Florida Avenue, New Orleans, LA 70119 U.S.A.**

## BOOK REVIEW

*J Raptor Res.* 37(4):352–353

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**North American Owls: Biology and Natural History, Second Edition.** By Paul A. Johnsgard. 2002. Smithsonian Institution Press, Washington, DC. xiii + 298 pp., 42 color plates, 12 tables, 76 text figures, 3 appendices. ISBN 1-56098-939-4. Cloth, \$49.95.—The first edition of this book, published in 1988, was widely acclaimed as one of the finest books on North American owls. I described it as a well-balanced coverage of the biology and natural history of the North American species (*Wilson Bull.* 102:192–193, 1990). Now, 14 years later, we see a revised edition, and I will focus primarily on how the new book differs from the first edition.

The new edition is essentially the same size as the first and only three pages longer despite adding coverage of 12 Mexican species, 24 new figures, 12 new color plates, about 15% more text, and some Internet sources on owls. Packing all this new information in without increasing the size of the book was accomplished by eliminating detailed plumage descriptions and making some changes in the format: double-column format versus single column, smaller font size, and reducing the size of figures and tables. Johnsgard also nearly doubled the number of references, but admits that some of the new references are not cited in the text, a mistake in my opinion because important new information has not been integrated into the book.

Organization of the second edition, like the first, consists of two parts. Part one is a general review of owl evolution and systematics, ecology, morphology, physiology, behavior, reproduction, and folklore, and part two contains detailed natural histories of the 31 species.

How well did Johnsgard pull off this revision? Unevenly, I think, is the best word to describe it. On the one hand, large sections in the first part are unaltered other than some minor editing. For example, the sections on vocalizations and mythology are essentially unchanged. On the other hand, the section on morphological characteristics has several new figures and a whole new paragraph on leg and foot musculature, but its source is not cit-

ed. I am left wondering if the asymmetrical updating was based on the relative amounts of new material in the literature or some other criterion.

The same imbalance of revision applies to the species accounts. The Spotted Owl (*Strix occidentalis*) account contains much new material, reflecting the intense research on it since the first edition came out. The Eastern Screech-Owl (*Otus asio*) account is also heavily updated with information from many new sources, but the Great Horned Owl (*Bubo virginianus*) account is not. Only a few new sources are cited adding very little new information. The Barn Owl (*Tyto alba*), the species I am most familiar with, was updated primarily from Taylor's *Barn Owls: Predator-Prey Relations and Conservation* (Cambridge Univ. Press, Cambridge, UK, 1994); although an excellent book, it is concerned mostly with European birds. I was disappointed to find that several of my recent publications on Barn Owl reproduction, mate choice, and morphology (*Auk* 107:246–254, 1990; *Condor* 96:468–484, 1994; *Auk* 114:581–592, 1997) were not even listed in the reference section; others were listed but not cited in text.

The selection process for the color images remains as elusive as it was in the first edition. Of the 31 species covered in the book, 26 are represented by either a painting or color photograph (all good to excellent in quality) or both, but eight species are represented by both a painting and a photograph. Five species, all Mexican, are not illustrated in color, and, unfortunately, the Barn, Snowy (*Nyctea scandiaca*), and Burrowing (*Athene cunicularia*) owls are shown only in Fuertes paintings. In my opinion, Fuertes paintings are not the best to represent a species in a scientific book. Finally, why did Johnsgard use three photos of a Western Screech-Owl (*Otus kennicottii*) and two each of six other species?

We have come to expect a lot from Johnsgard's books. Unfortunately, this one does not quite live up to those expectations. Although it is still one of the best sources of information on North American owls, especially with the addition of the Mexican species, it could have been so much more. Did Johnsgard, perhaps, anticipate this view? Why else did he pen the second sentence of the Preface:

“Raptoriphiles, and especially strigiphiles, are so attached to their subjects that few books escape the wrath of reviewers who feel that their particular species or subject area has been slighted or badly dealt with.”

Who should buy this book? Libraries need a copy and those individuals interested in owls, who

do not own a copy of the first edition, will surely want one. People who have the first edition have a harder problem—does the second edition have enough new information? The answer to that will have to be a personal one.—**Carl D. Marti, Raptor Research Center, Boise State University, Boise, ID 83725 U.S.A.**

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BY JAIMIE VARNER

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## 2004 ANNUAL MEETING

The Raptor Research Foundation, Inc. 2004 annual meeting will be held on 9–13 November 2004 in Bakersfield, California. For information about the meeting see the following website: [http://www.calhawkingclub.org/field\\_meet/34th\\_annual/](http://www.calhawkingclub.org/field_meet/34th_annual/) or contact Rick Holderman (parabuteo1@cox.net).

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

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### RAPTOR RESEARCH FOUNDATION, INC., AWARDS

#### Lifetime Achievement Awards

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

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

#### Student Recognition and Travel Assistance Awards

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

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

#### Grants

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

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

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

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