Appendix F – Species Accounts

ECOLOGY OF RED-TAILED HAWK IN CALIFORNIA

By Carie Battistone
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The red-tailed hawk (*Buteo jamaicensis*) is a robust, broad-winged hawk that resembles most other hawk species in shape and form. There are 12 subspecies of red-tailed hawks. As is typical for most raptors, red-tailed hawks exhibit distinctive color variations, making differentiation of subspecies difficult. Of the two most extreme are the Harlan's hawk (*B. j. harlani*) and the Krider's hawk. The Krider's does not have its own distinct range and therefore has not been given it own subspecies designation. This hawk is a white variation differentiated by a mostly white tail, white face, pale underwings (Preston and Beane 2009). The Harlan's hawk was formerly considered a separate species, but is now considered a dark variation of the red-tailed hawk, with black above and below, some white speckling intermixed throughout, and white-gray tail feather mottled or streaked with black and tinged rufous or light brown (Collins and Bloom 2000, Preston and Beane 2009).

Other subspecies share key distinguishing features to some extent, such as the rusty tail feathers of the adults, a dark bar on the underwing between the shoulder and wrist (patagial bar), and a banding pattern across the abdomen (belly band) (Palmer 1988, Preston and Beane 2009).

Light variations of red-tailed hawk adults exhibit lighter coloration and a dark head, red tail and patagial bar, with little to no markings on the chin, breast, throat, and thighs, with the exception of dark streaks on the belly and flanks (Bent 1937, Palmer 1988). Light morphs from western North America exhibit more ventral rufous streaking than red-tailed hawks in other parts of North America (Clark and Wheeler 2001, Preston and Beane 2009). Dark morph red-tailed hawks are common in Canada and the western United States, and are typically chocolate brown throughout, with some dark morphs having a heavy rufous pattern (Preston and Beane 2009). The patagial bars are obscured by the dark pigmentation; however the rufous tail feathers can still be seen in adults.

Juvenile red-tailed hawks have a heavily barred tail that lacks the distinctive rusty tone of the adult, with undertail feathers more faintly barred (Bent 1946, Clark and Wheeler 2001). The most distinguishing feature for juveniles is the two-toned upper wings, with primary flight feathers showing a lighter brown then secondary flight feather (Clark and Wheeler 2001). Juvenile plumage is retained until they reach the breeding age of 2 years.

Male and female red-tailed hawks are similar in appearance but express sexual dimorphism, meaning females are generally larger then males. Adult males have a body length of 45 to 56 cm (17.7-22 in), and weigh between 690 to 1300 g (1.5-2.9 lbs); whereas females range from 50 to 65 cm (19.7-25.6 in) in length, and weigh 900 to 1,460 g (2.0-3.3 lbs) (Palmer 1988, Farmer et al. 2009, Preston and Beane 2009).

The oldest known wild red-tailed hawk is 30 years and 8 months (Lutmerding 2011); however, most do not attain such an age in the wild. The Bird Banding Lab (BBL) records through 1994 showed out of 5,194 red-tailed hawks recovered, only 31 survived longer then 17 years in the wild, and of these, 11 survived longer than 20 years (Soucy 1995).

The red-tailed hawk is widely used in falconry due to their even temperament and their ability to hunt and obtain sizable game (Palmer 1988). Falconry take data shows 223 red-tailed hawks were obtained from the wild from 2006 to 2010, 41% of all raptors obtained from the wild during this time frame (CDFG 2012). Of the 223, 5 were adults, 185 were immature, 9 were nestlings, and sex was undetermined for 24.

Habitat

Red-tailed hawks occupy a variety of habitats. Generally, they prefer open habitat, such as sparse woodlands, grasslands, agricultural regions, and urban areas (Grinnell and Miller 1944, Howell et al. 1978, Palmer 1988, Preston and Beane 2009). They are often found in areas with nearly equal proportions of woodland, pasture and cropland habitat (Bednarz and Dinsmore 1982). Densely vegetated habitat or areas devoid of trees or perches are generally avoided (Bednarz and Dinsmore 1982, Farmer et al. 2009). The highest densities of red-tailed hawks were reported in the oak woodlands of California and Colorado (Fitch et al. 1946). Red-tailed hawks often frequent urban and suburban areas, which likely provide sufficient prey and nest sites, as indicated by higher productivity sometimes found from hawks inhabiting here (Stout et al. 2006).

Nests are commonly located in patchy woodlands and riparian corridors adjacent to foraging sites, especially those near agricultural fields (Bent 1937, Fitch et al. 1946, Bednarz and Dinsmore 1982, Palmer 1988, Moorman and Chapman 1996, Tietje et al. 1997). Often, nests are found in the upper quarter of trees with moderate canopy cover and are placed at the crotch of main branches and limbs (Fitch et al. 1946, Bednarz and Dinsmore 1982). A variety of tall trees are used for nesting, including oak, sycamore and eucalyptus (Tyler 1913, Fitch et al. 1946, Palmer 1988, Speiser and Bosakowski 1988). Mader (1978) found red-tailed hawks nesting on a saguaro cactus in Arizona's Sonoran desert. Red-tailed hawks also nest on cliffs (Bond 1939). Artificial nesting sites, such as transmission towers, billboards and ledges of buildings, are utilized regularly (Stout et al. 1996, Stout et al. 2006). Accessibility to edge habitat may play a role in nest site selection (Bednarz and Dinsmore 1982, Speiser and Bosakowski 1988, Moorman and Chapman 1996, Preston and Beane 2009).

During the winter months, red-tailed hawks can be found in much of the same habitat as during the breeding season. Foraging habitats include grasslands, parks, vacant lots, and agricultural areas (Fitch et al. 1946, Palmer 1988, Preston 1990, Leyhe and Ritchison 2004, Preston and Beane 2009). Distribution across a landscape is a function of prey and perch site availability (Preston 1990). Alterations to the landscape (e.g. forest clearing, fire suppression and agricultural development) likely have benefited the species distribution due to the increased availability of open land with abundant prey (LaSorte et al. 2004, Farmer et al. 2009).

Migration

Red-tailed hawks are considered partially migratory, although typically in California, the species is a year-round resident and remains in or near the breeding territory (Fitch et al. 1946, Gates 1972, Farmer et al. 2009, Preston and Beane 2009). Hawks in the northernmost range (portions of Alaska and Canada) will migrate south, while hawks in the mid-latitudes (45°–50°N) are typically resident year-round (Preston and Beane 2009). Those that do migrate usually only do so at distances of less than 1,500 km, and will rarely cross bodies of water more than 25 km across (Preston and Beane 2009).

Satellite tracking and band return data shows that young hawks from southern California and northern Baja will migrate longer distances into Nevada, Montana, and Wyoming (Farmer et al. 2009). Red-tailed hawks practice “leap-frog migration”; that is, some northern birds winter farther south then southern birds (Farmer et al. 2009). Abundant prey may alter or delay migratory patterns (Craighead and Craighead 1956). For instance, red-tailed hawks have been known to “short-stop” (overwinter farther north then in previous years) during migration, a behavior adaptation likely due to long-term trends for milder climatic conditions and increased prey availability at bird feeders (Goodrich and Smith 2008).

Timing of migration is determined by weather patterns and prey availability (Farmer et al. 2009, Preston and Beane 2009). For northern breeders, spring migration initiates in February to March, with migrating individuals arriving at the breeding sites as late as June (Preston and Beane 2009). For mid-latitude breeders migration is typically shorter, with small north-south movements, altitudinal movements, or little to no migration.

The Golden Gate Raptor Observatory (GGRO) in the Marin Headlands, California, is one of the “hotspots” for migrating diurnal raptors in California since it is along one of the major migratory bird pathways along the Pacific Coast. Red-tailed hawks are one of the most observed raptors at GGRO. Sixty-six percent of all sightings in 2009 were red-tailed hawks (n=8297) and turkey vultures (n=8060) (Elliot and Fish 2010). Hawk sightings during the Fall/Winter migration season are illustrated in Figure 1.

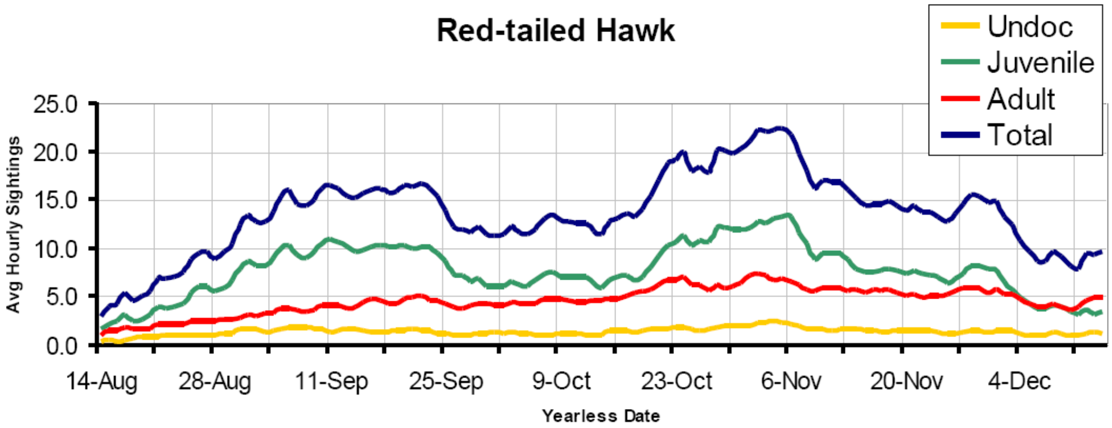

Figure 1. Red-tailed hawk migration profile at GGRO, 1986-2009 (Elliot and Fish 2010)

Food Habits

Red-tailed hawks consume a diverse array of prey, including mammals, snakes, small to medium sized birds, reptiles, amphibians, and insects; however, small mammals, reptiles and birds are most common (Gates 1972, Palmer 1988, Steenhof and Kochert 1988, Marti and Kochert 1995, Farmer et al. 2009, Preston and Beane 2009). Diets vary seasonally depending on prey availability and abundance (Peterson 1979, Steenhof and Kochert 1988). In California, Fitch et al. (1946) documented a spring and summer diet consisting of (in order of frequency) ground squirrels, pocket gophers, and rabbits, whereas fall and winter diets consisted of (in order of frequency) pocket gophers, ground squirrels, and rabbits (Fitch et al. 1946). Red-tailed hawks prey on ducks in the Prairie Pothole Region, and scavenge when carrion is available, especially in winter months when live resources are less available (Palmer 1988, Sargeant et al. 1993, Langley 2001).

Most hunting is done in a sit-and-wait style; that is, where hawks wait at a perch site until prey is detected and then swoops downward to capture the prey (Bent 1937, Fitch et al. 1946, Palmer 1988, Leyhe and Ritchison 2004). Red-tailed hawks have also been known to hunt on-the-wing during low flight, even capturing some prey from the ground on foot (Fitch et al. 1946, Preston and Beane 2009). Sit-and-wait hunting is more energetically beneficial as it requires less energy for the amount of prey obtained (higher success-to-cost ratio), and this likely explains the preference for this method (Ballam 1984).

Breeding

Red-tailed hawks generally mate for life or until one of the pair dies (Craighead and Craighead 1956, Palmer 1988, Goodrich and Smith 2008, Preston and Beane 2009). Age of first breeding is typically at 2 years, however breeding attempts at one year have been documented (Luttich et al. 1971, Peterson 1979, Palmer 1988). Courtship displays occur throughout the year with both the male and female soaring in wide circles, at times coming very close together. The male often flies with legs extended, and the display concludes with long vertical dives by both birds (Fitch et al. 1946, Palmer 1988). An established pair may visit several nest sites in their territory, including ones previously used, before selecting a site (Fitch et al. 1946). Both male and female are active in the nest site selection and construction process, and often continue to adorn the nest with green twigs throughout the nesting season (Fitch et al. 1946). During initial nest construction, the pair will act discreet to avoid attracting attention (Fitch et al. 1946). The pair is particularly sensitive to disturbance during nest building, and may abandon the site if disturbed (Bent 1937).

The breeding season in Californa initiates in February and March, but has been noted to occur as early as December, and as late as April in some parts of California (Tyler 1913, Grinnell and Wythe 1927, Johnson 1975, Mader 1978, Palmer 1988). Egg laying is initiated between mid- to late-March with the last egg in a clutch being laid 2 to 5 days after the first (Willet 1933, Johnson 1975, Preston and Beane 2009). Clutch size ranges from 1 to 4 eggs, but is more typically 2 to 3 (Bent 1937, Mader 1978, Palmer 1988, Preston and Beane 2009). The breeding pair will have one clutch per season, although if nesting fails on this first attempt, the pair may attempt a second clutch if failure occurs early enough in the breeding season (Palmer 1988). Incubation is performed by both male and female, and lasts 28 to 35 days, at which time hatching takes place over 2 to 4 days (Bent 1937, Fitch et al. 1946, Johnson 1975, Preston and Beane 2009). Luttich et al. (1971) asserted that nests are particularly sensitive to disturbance during incubation, and documented abandonment after nests were accessed by researchers.

Males do the majority of the hunting and prey delivery while the females feed the young (Fitch et al. 1946, Peterson 1979). Cooperative breeding where 3 adults attend the same nest is relatively rare (Wiley 1975a, Santana 1986).

Young generally fledge from the nest by early- to mid-July, when they are 40 to 48 days old (Wiley 1975b, Mader 1978, Palmer 1988). After fledging, young will stay relatively close to their parent's territory for up to six months, depending on how fast they learn to hunt on their own and how often parents continue to feed them (Fitch et al. 1946, Johnson 1973, Preston and Beane 2009). Once fledglings do leave parental territory, they may disperse long distances or stay in the margins of parental territory (Fitch et al. 1946). In Montana, juvenile mortality was low 18 to 25 days after fledging (Johnson 1973). However, Henny and Wight (1970) reported an average mortality rate from fledging to one year old at 54 percent.

Numerous studies have investigated productivity of red-tailed hawks across their range. Henny and Wight (1970) determined that in order to maintain a stable population, productivity should be between 1.33 and 1.38 young produced (fledged) per breeding attempt. Productivity of red-tailed hawks varied widely among studies at between 0.4 to 2.1 young per nest, and was dependent on factors such as prey availability, disturbance, and predation (Petersen 1979, Seidensticker and Reynolds 1971, Gates 1972, Wiley 1975b, Howell et al. 1978, Mader 1978). In Wisconsin, Wiley (1975b) noted that red-tailed hawk nests in areas with higher indirect (e.g. road noise) or direct (e.g. harvest of nestlings, predation) disturbance produced 0.95 young per nest, whereas areas with little disturbance produced 2.06 young per nest. Stout et al. (2006) also found higher productivity in high-density urban-suburban areas then in non-urban areas, suggesting that urban areas may be have high production rates, thus supplying other red-tailed hawk populations with potential breeders. The differences in reproductive rates in the various studies may be due to how acclimated the individual hawks are to human disturbance.

Unsuccessful nesting rates vary among red-tailed hawk populations, with between 17 and 41 percent of the nests that initiated incubation failing between the egg and early nestling phases (Fitch et al. 1946, Luttich et al. 1971). Johnson (1975) indicates that many nest failures occurred shortly after the young hatch, with an 8 to 11% nestling mortality rate; similarly Luttich et al. (1971) reported nestling mortality between the ages of 2 to 3 weeks was 11%. Potential causes for nest failures include level of disturbance and predation, but may also be related to clutch size and inexperienced breeders (Luttich et al 1971).

Home Range and Territory

The home range size (broad area which all activity is confined to) is dependent on many factors including topography, habitat, food availability and season. Petersen (1979) documented home range size of male red-tailed hawks in Wisconsin between 3.90 km^2 (1.5 mi^2) in the fall, 1.57 km^2 (0.61 mi^2) in the winter, 1.63 km^2 (0.63 mi^2) in the spring, and 1.17 km^2 (0.45 mi^2) in the summer; an artifact likely due to prey availability.

Across its range, territory size (the area that an animal defends against intruders) varies from approximately 1.25 km^2 to more than 2.5 km^2 (0.48-0.97 mi^2), with a reported inter-nest distance of around 0.2 miles (0.32 km) (Preston and Beane 2009). Fitch (1946) determined territory size for red-tailed hawks in California was between 0.1 and 0.3 mi^2 (e.g. 0.26-0.78 km^2), with 2.0 breeding pairs per mi^2 (0.8 pairs per km^2).

Historical and Current Distribution

Historically, the red-tailed hawk was common throughout California, from deserts, islands, low elevation valleys, and high elevation mountains (Grinnell 1915). Grinnell (1915) noted this species exhibited a tolerance for a wide array of climatic conditions.

Willet (1933) labeled this hawk as common from "lowlands to at least 9000 feet". Red-tailed hawks were found at Tule Lake National Refuge and Lava Beds National Monument, Siskiyou County (Bond 1939). Grinnell and Miller (1944) remarked that the red-tailed hawk was a permanent "common and widespread" resident. The species was also documented at several islands off the southern California coast: Anacapa Island; Santa Catalina Island; Santa Cruz Island; and Santa Barbara Island (Howell 1917, Willet 1933, Grinnell and Miller 1944). In the early 1900s, the red-tailed hawk was also documented on the Farallon Islands during April and May (Grinnell and Wythe 1927).

Currently, the distribution of the red-tailed hawk is much the same as the historic period. In North America breeding range extends from coastal Alaska south to the Baja peninsula, and winter distribution includes southern portions of Canada, all of the United States, and portions of Mexico and Central America (Preston and Beane 2009). In California, this hawk is found most anywhere suitable foraging and nesting habitat exists; from the coast and Central Valley to the east side of the Sierra Nevada, and in all valleys, desserts and mountainous regions.

Historical and Current Abundance

Grinnell (1902) recorded the red-tailed hawk as an "abundant resident almost throughout the state". Willet (1912) noted the species bred commonly in the foothill regions of Southern California. In Fresno County, Tyler (1913) remarked that this bird was most numerous in the foothill regions and along the San Joaquin River, and in winter months was well distributed across vineyard and orchards. In the San Francisco Bay area, red-tailed hawks were considered the most numerous large hawk, inhabiting parks, hills, canyons, and mountain ridges (Grinnell and Wythe 1927). Grinnell and Miller (1944) recorded the species as "common and widespread; numbers holding up, probably close to normal save in lowland areas thickly settled by humans, where marked reduction or even, locally, elimination has taken place."

Today, the red-tailed hawk is one of the few raptors in North America that has maintained a stable or increasing population. Partners in Flight (PIF), using Breeding Bird Survey (BBS) data, estimated a population size of nearly two million red-tailed hawks in North America, representing 89% of the global population; the population trend was considered stable or possibly increasing (Rich et al. 2004). The population of red-tailed hawks in California was estimated at 160,000 individuals, or 7.2% of the estimated global population (Rich et al. 2004). Data quality was rated as "good" in both North America and California estimates due to level of species coverage in BBS surveys.

Newton (1976) related density to availability of prey. Density of breeding pairs varies from two pairs per 1 mi^2 (2.6 km^2) in California, one pair per 3.1 mi^2 (8.0 km^2) in Montana, one pair per 2.7 mi^2 (7.0 km^2) in Alberta, and one pair per 1.6 mi^2 (4.1 km^2) in Wisconsin (Fitch et al. 1946, Luttich et al. 1971, Johnson 1975, Peterson 1979). General density varies from 0.37-0.76 birds per 1 mi^2 in the upper Midwest (Craighead and Craighead 1956, Orians and Kuhlman 1956, Kabat and Thompson 1963, Gates 1972).

The Christmas Bird Count (CBC) and BBS are both long-term data sets collected by volunteers across North America. CBC indicates red-tailed hawks as moderately abundant and widespread in North America throughout the winter months (Figure 2 and Figure 3). According to BBS data, summer distribuition is widespread across North America (Figure 4). In California, several areas of higher winter and summer abundance exist in the northeast portion of the state, and througout the Sacramento and San Joaquin Delta.

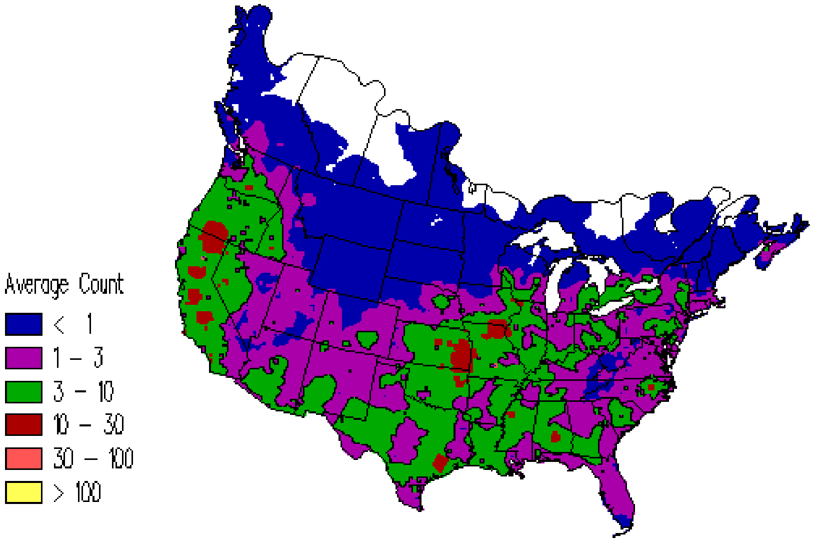

Figure 2. This map shows an index of abundance that predicts the average number of red-tailed hawks that could be seen in about 100 party-hours of bird watching in a CBC circle. (Saurer et al. 1996)

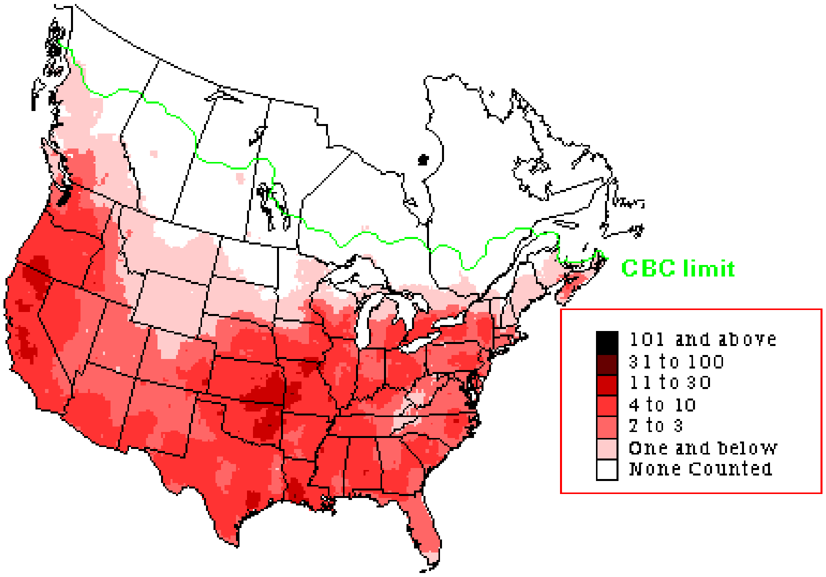

Figure 3. Red-tailed hawk wintering distribution map based on average effort-adjusted counts from CBC, 1959 – 1988 (Saurer et al. 1996)

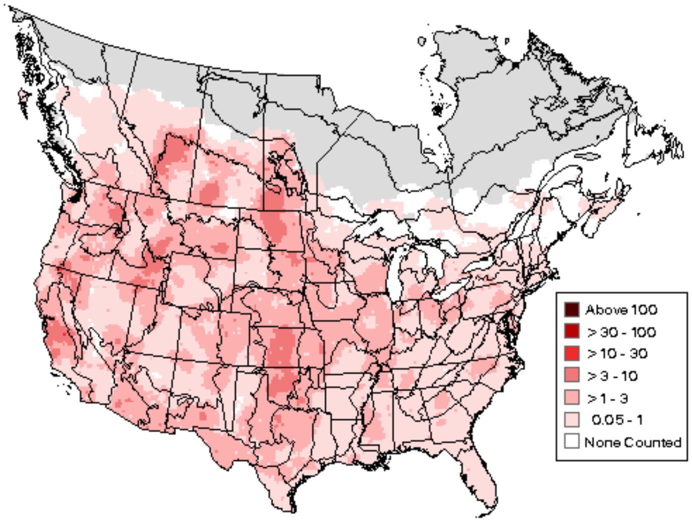

Figure 4. BBS summer distribution map. This map indicates the number of birds on BBS routes, grouped into categories of relative abundance. The maps predict the average number of birds of the species that could be seen in about 2.5 hours of birdwatching along roadsides, and are based on mean counts on BBS routes over the interval 2006 – 2010.

Population Trend

Determining population trend throughout the red-tailed hawks distribution is complicated to interpret due to the fluxes observed at various hawk watch sites throughout North America. Some migration counts suggest populations are stable or decreasing, whereas CBC and BBS data suggests increasing trend; an apparent conflict likely due to shifting migratory-residency behaviors and weather patterns (Farmer et al. 2009). For example, between 1976 and 2004, trend for red-tailed hawks at Hawk Mountain, Pennsylvania, showed a decline (-1.9%), and the same was true for Cape May Bird Observatory in New Jersey (-1.8%) (Preston and Beane 2009). However, during the same time period, Lighthouse Point, Connecticut, showed an increase of 3.1% (Preston and Beane 2009).

Generally, data suggests that populations have increased in the northeast (from 1974 to 2004), intermountain west (since the early 1980s), and the southern Rocky Mountains (since 1995); have decreased in the southern Rocky Mountains (since 1995); and have remained stable in the Pacific Northwest (since 1995) and around the Gulf of Mexico (Farmer et al. 2009).

The Golden Gate Raptor Observatory (GGRO) a the Marin Headlands in California documented 8,297 red-tailed hawk sightings in 2009 during fall migration counts, making it one of the most numerous species detected (Elliot and Fish 2010). From 1986 to 2009 red-tailed hawk have demonstrated an increasing trend in California based on migration counts (Figure 5).

The number of red-tailed hawks reported on surveys, such as the CBC and BBS, has been stable or increasing over time. BBS trends showed general increases over time from 1966 to 2010, with a significant increasing trend value of 0.8 (p value less than

0.05) for California (Sauer et al. 2011). The majority of California shows a strong increasing trend, with the exception of southern portions of the state, including much of the desert region, and portions of the north coast and the upper Sacramento Valley (Figure 6). CBC data shows similar upward trends for red-tailed hawks in California (Figure 7).

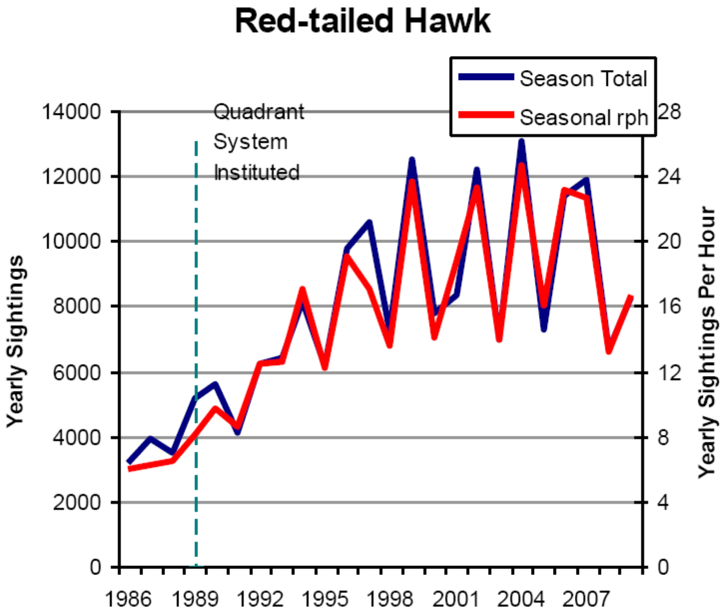

Figure 5. GGRO yearly trends in red-tailed hawk sightings, 1986-2009 (Elliot and Fish 2010)

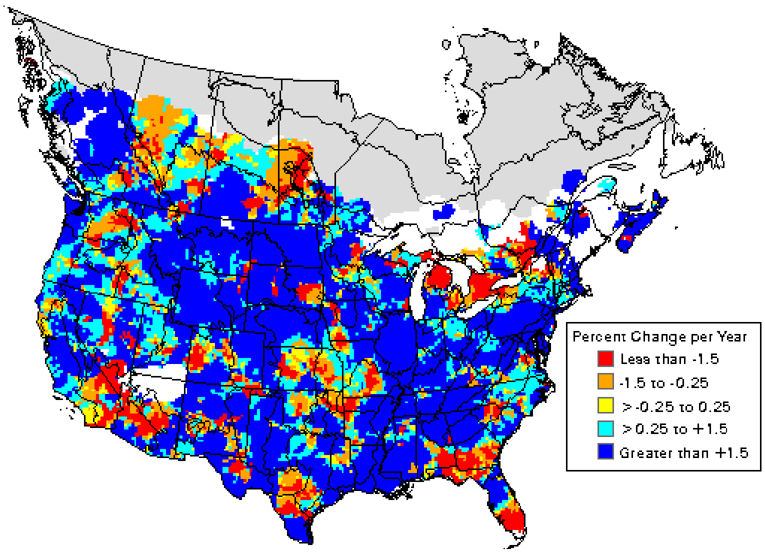

Figure 6. BBS trend map for red-tailed hawks in North America, 1966-2010. Note: areas of yellow, turquoise, and blue indicate increasing trend, whereas areas of orange and red indicate decreasing trend. (Sauer et al. 2011)

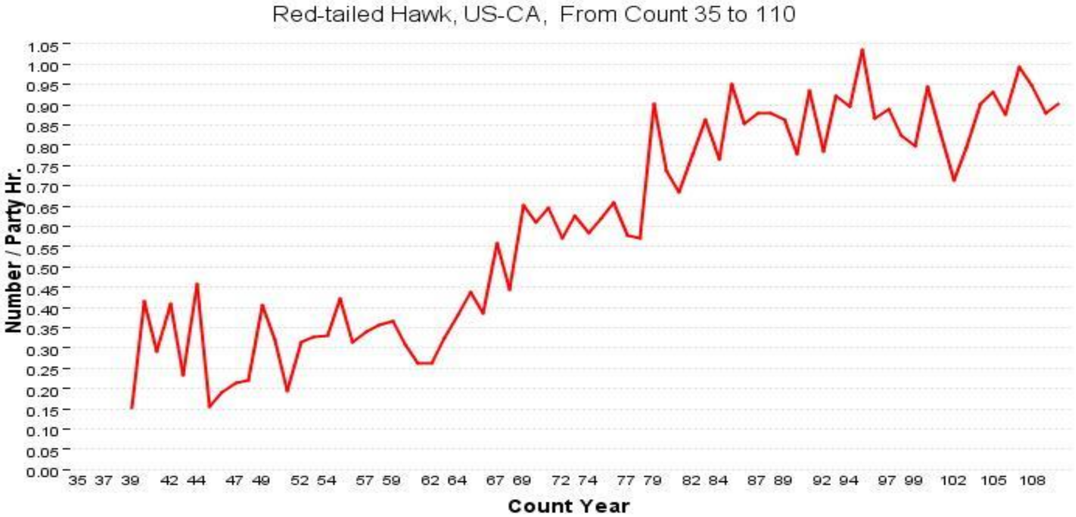

Figure 7. CBC counts for red-tailed hawks in California from 1934 to 2010 (National Audubon Society 2010)

Threats

Red-tailed hawks face a number of threats, including disease, parasites, and predation, as well as human-related threats such as poisoning, disturbance, and collisions.

Great-horned owls have been documented as the most frequent predator on red-tailed hawk nestlings, but they rarely takes adults (Seindensticker and Reynolds 1971, Houston 1975, Petersen 1979, Preston and Beane 2009). Other nest predators include crows and mammals. Johnson (1975) noted that the majority of nestling mortalities in a two-year study in southwestern Montana was from mammalian predators.

Some studies have shown that breeding pairs are particularly sensitive to disturbance during nest building and incubation, and may abandon the site if humans or other threats are detected (Bent 1937, Fitch et al. 1946, Luttich et al. 1971, Seidensticker and Reynolds 1971). Wiley (1975b) reported 9 out of 14 “nest failures” in a Orange County study site were attributed to falconers accessing nests to obtain a nestling during 1973 (January through July).

Aircraft, power line, and wind turbine collisions are documented causes of red-tailed hawk mortality. Aircraft strikes of red-tailed hawks account for 18% (637/3510) of all raptor strikes in the United States from 1990 to 2005 by civil aircraft, and 32% of all raptor strikes by the U.S. Air Force (Kelly 1999, Cleary et al. 2006). Wind turbines pose an increasing threat to raptors, with a significant numbers of red-tailed hawk fatalities reported (Hoover and Morrison 2005). At the Altamont Pass Wind Resource Area 316 raptor mortalities were documented between 1998 and 2000, 96 of these were red-tailed hawks (Rugge 2001). At a wind facility in Washington, golden eagles and red-tailed hawks had the highest exposure rate to turbine blades (Erickson et al. 1999). At the Tehachapi Pass Wind Resource Area, red-tailed hawks, along with great horned owls and American kestrels, had the highest risk of collision with turbine blades (Anderson et al. 2004). Electrocution from contact with power lines was a threat for red-tailed hawks, especially in places where natural perches are lacking (Lehman and Barrett 2002).

Diseases found in red-tailed hawks include West Nile virus (WNV), Aspergillosis, and Chlamydiosis, all of which can be fatal. Transmission of WNV is transmitted by infected mosquitos and possibly through the consumption of infected prey (Garmendia et al. 2000, Nemeth et al. 2006). Aspergillosis commonly affects red-tailed hawks in captivity and is caused by the fungus *Aspergillus* (Abundis-Santamaria 2003). Chlamydiosis, a respiratory disease, has been known to cause deaths in hawks (Mirande et al. 1992). Parasites also can have a negative effect on wild hawks. In Wyoming, blackflies (*Simulium canonicolum*) were responsible for nestling mortality through harassment, transmission and infection of blood protozoan, and direct blood loss (Palmer 1988, Smith at al. 1998). In California, mortality caused by blood-sucking flies (*Simulium sp.*) has also been documented during wet years (Fitch et al. 1946).

Pesticide contamination is a potential threat and his been documented in red-tailed hawks and other raptor species in the Columbia Basin, Oregon (Henny et al. 1984). Between 1985 and 1995, 255 raptors died due to pesticides in the United States, with most raptors likely dying from secondary poisoning after consuming poisoned prey (Mineau et al. 1999). In California, Fry et al. (1998) documented 34 cases of red-tailed hawk poisoning by a pesticide applied to almond orchards. In Canada, of all raptors, red-tailed hawks and bald eagles were more often associated with pesticide-related mortality (Elliot et al. 1996, Mineau et al. 1999). A study in the mid-1960s showed thinning red-tailed hawk egg shells, suggestive of organochlorine pesticide contamination (e.g. DDT) (Seidensticker and Reynolds 1971). Red-tailed hawk mortality associated with secondary poisoning from anticoagulant rodenticides is a concern as well (Stone et al. 1999). Lethal and sub-lethal effects of lead poisoning has also been documented in raptors, especially those who have access to prey species that are targeted for removal (e.g. ground squirrels) or those that utilize carrion as a food source (Redig et al. 1991, Fisher et al. 2006, Pauli and Buskirk 2007).

Impacts of climate change are largely unknown for this species. Because it relies on a wide variety of habitat and food resources, the red-tailed hawk may be more resistant to changes in temperature and precipitation then other raptor species. Potential impacts include, but are not limited to, shifts in preferred prey abundance, level of competition with other raptor species, and distribution across its range, and migration patterns; all of which may impact survival and productivity of this species.

Data gaps and monitoring needs

Research is needed to help determine the effects of drought on red-tailed hawk populations. This is especially relevant as potential climate change impacts are on the horizon. In addition, a better understanding of the effects of anthropogenic activities (e.g. forest thinning and burning, grazing, urbanization, agricultural practices, etc.) on red-tail populations will help implement responsible management decisions.

The increasing use of wind energy presents a significant threat to raptor conservation. A better understanding of how these facilities impact raptors populations is needed, as well as determining proper placement of wind turbines to reduce bird mortality (Hoover and Morrison 2005).

Red-tailed hawks can be used as a sentinel species to help inform overall impacts of climate change to other more sensitive raptor species by assessing long-term changes in productivity, behavior, migration, prey availability and abundance, and range shifts.

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ECOLOGY OF AMERICAN KESTREL IN CALIFORNIA

By Justin D. Garcia

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The American kestrel (*Falco sparverius*) is a small, widespread falcon ranging from the northern treeline in North America, down through much of Central and South America (Smallwood and Bird 2002). There are seventeen classified subspecies, with *F. s. sparverius* being the only subspecies resident to California (Smallwood and Bird 2002). American kestrels use a variety of open to semiopen habitats throughout their life cycle; and are characterized as secondary cavity-nesters (using old woodpecker cavities), requiring large trees in areas lacking a thick forest canopy (Grinnell and Miller 1944, Roberson 1993, Smallwood and Bird 2002, Cooke and Hannon 2011). Prey consists of insects, birds, mammals, reptiles, and amphibians (Bent 1937). Due to this specific diet of agricultural pests, this bird has been considered a friend of the farmer for some time (Cooper 1870). The kestrel was known as the "sparrow hawk" by early Europeans because of its misrepresented preference for sparrow-sized birds (Bent 1937, Ryser 1985).

This raptor is sexually dimorphic in size and plumage (Smallwood and Bird 2002, Wheeler 2003). Male kestrels average 102.7 g, are 20 to 25 cm in length, and have a wingspan of 51 to 56 cm; while females average 116.5 g are 23-28 cm in length, and have a wingspan of 53 to 61 cm (Roest 1957, Wheeler 2003). The American kestrel has sexually dimorphic plumage, meaning males and females have different markings (Roest 1957, Smallwood and Bird 2002, Wheeler 2003). Adult males are characterized by brown-orange colored breast and flanks, white belly, blue-gray wings, and a rufous tail with a black band (Smallwood and Bird 2002, Wheeler 2003). Adult females have rufous colored wings, back, and tail with black bars (Smallwood and Bird 2002, Wheeler 2003). Both sexes have two narrow black vertical markings on the head, two black spots on each side of the nape, long narrow/pointed wing tips in flight, and a characteristic tail pump when perched (Wheeler 2003).

Range-wide declines in North America were first noted at the Raptor Research Foundation Meeting in 2004, and have continued to be a concern (Bird 2009, Fish 2011). Various studies have reached similar conclusions that this once common raptor may be moving in the direction of scarce abundance (Bird 2009, Farmer and Smith 2009, Smallwood et al. 2009a, Fish 2011). Recently the American kestrel has been considered a Bird of Conservation Concern (BCC) in Bird Conservation Regions (BCRs) 25, 27, 31 and U.S. Fish and Wildlife Service region 4 (USFWS 2008). While no formal listing exists in California, the Golden Gate Raptor Observatory has observed a long term decline of American kestrel at fall and winter raptor migration counts (Elliot and Fish 2010, Fish 2011).

Life span for a wild kestrel is difficult to determine. The oldest known banded kestrel found dead in the wild was almost 15 years old, but this is considered quite rare (Lutermerding and Love 2011). The average life expectancy is 2 years, 9 months (Henny 1972). Survivorship and population abundance in California is poorly understood.

According to the Federal Environmental Assessment for falconry, approximately 100 kestrels per year were reported harvested from the wild throughout North America (Millsap and Allen 2006). In California, American kestrel is a commonly used in falconry, with 70 individuals harvested from the wild in 2006 through 2010 (CDFG 2012). Of the 70 individuals, 17 were adults, 44 immature, 1 nestling, and 8 were of unknown age. Harvest was well dispersed throughout the State, with no one county where harvest dominated. While in captivity, 16 kestrels died during the period 2006 to 2010. Twenty-eight kestrels were released back into the wild from 2006 through 2010, and 8 were lost.

Habitat

The American kestrel occupies a wide spectrum of habitat types across its range in the Americas, as well as in California, and generally prefers semi-open terrain containing short vegetation, an adequate supply of prey, hunting perches, and suitable nesting cavities during the breeding season (Bird and Palmer 1988, Smallwood and Bird 2002). This small raptor is found in grasslands, meadows, oak woodland, pinyon-juniper woodland, riparian woodland, alpine tundra, subalpine forest, montane forests, low elevation forests, coastal forest, early succession burned forest, marshland, desert scrub, sagebrush flats, dry chaparral, lake shores, islands, and coastal habitats (Grinnell and Miller 1944, Miller 1951, Brown and Amadon 1968, Cade 1982, Bird and Palmer 1988, Small 1994, Unitt 1984, Roberson 1993, Rudesill 1995, Peeters and Peeters 2005). In addition, kestrels will use agricultural fields, orchards, ranches, parks, and other open suburban/urban settings (Cade 1982, Ryser 1985, Johnsgard 1990, Berry et al. 1998, Unitt 2004). Suitable perches, include but are not limited to power poles, power lines, fence posts, dead snags, shrubs, tree branches, and rocks.(Grinnell and Miller 1944, Brown and Amadon 1968, Wheeler 2003). Edge habitats are also important for the kestrel, allowing for accessible forage opportunities during the breeding season (Balgooyen 1976).

Breeding habitat for the kestrel can include many of the habitat conditions described above, though generally includes patches of low vegetation for foraging and tree cavities for nesting (Grinnell and Miller 1944, Balgooyen 1976, Smallwood 1987, Smallwood and Wargo 1997). During the breeding season, the availability of nesting cavities is an important limiting factor for kestrels (Cade 1982, Smallwood et al. 2009). They do not build their own nest, but rather, are considered secondary-cavity nesters (Bird and Palmer 1988, Shuford 1993, Smallwood and Bird 2002). This falcon relies on the natural decomposition of trees from fire, natural branch breakage, lightning damage to trees, wood-boring insects holes, and subsequent cavity creation by woodpeckers to create optimal nesting habitat (Balgooyen 1976, Bloom and Hawks 1983, Bousman 2007). Varying intensities of forest fire create a mosaic of successional habitat with many edges, less competition for prey, and plenty of dead snags for perching and nesting (Shorger 1917, Balgooyen 1976, Hoffman and Collopy 1987, Greenwood and Dawson 2011b). Nest cavities selected by kestrels are generally located 2.06 to 24.38m (~7.78m) high (Balgooyen 1976).

The primary nest excavator the kestrel relies on is the northern flicker (*Colaptes auritus*); but other birds such as black-backed woodpecker (*Picoides articus*), hairy woodpecker (*Picoides villosis*), downy woodpecker (*Picoides pubescens*), white-headed woodpecker (*Picoides albolarvatus*) are also important (Balgooyen 1976, Johnsgard 1990). Old growth forest habitat is very important to the kestrel since primary-cavity excavators (woodpecker) generally do not use young forests to nest in (Bloom and Hawks 1983, Roberson 1993, Hunter et al. 2005).

In some regions kestrels will use atypical nest sites, such as the undersides of dead palm leaves, steep cliffs or crevices in river banks of riparian habitat, crevices in desert bluffs, and old magpie nests (Roest 1957, Roberson 1993, Unitt 2004). In habitats that lack primary-cavity excavators, such as San Clemente Island, kestrels will use natural cliff cavities in canyons (Sullivan et al. 2003). They will also use man-made structures such as holes in telephone poles, cavities in buildings, and power transmission towers, and nest boxes (Brown and Amadon 1968, Bloom and Hawks 1983, Bousman 2007, Lambrechts 2012). Nest-boxes are readily used in areas of good foraging habitat (Bird and Palmer 1988, Bloom and Hawks 1983). Nest-box programs were designed to create easy access for research and to increase nesting densities of kestrel (Nagy 1963). Nest-box introduction also provides a useful long-term monitoring tool for assessing trend and productivity (Katzner et al. 2005, Smallwood et al. 2009).

Post-breeding habitat requirements are similar to the above, except that nesting cavities are not needed (Johnsgard 1990). Widespread research in North America has documented winter habitat segregation between males and females (Koplin 1973, Mills 1976, Smallwood 1987, Ardia and Bildstein 1997, Ardia and Bildstein 2001, Ardia 2002, Smallwood and Bird 2002). Generally females occupy larger open habitats, with a higher abundance of mammals and birds, while males occupy habitat with a higher amount of vegetation (Meyer and Balgooyen 1987). This segregation is thought to reduce intersexual competition for prey, with females forcing males into less optimal habitats (Koplin 1973, Mills 1976). Smallwood (1988) suggested the early arrival of females at winter territories explains the domination of higher quality habitat.

Pandolfino et al. (2011a) found a significant positive association of kestrels wintering in the Central Valley with alfalfa, other irrigated crops (e.g., winter wheat), rice fields, irrigated pasture, and grassland. Significant negative associations were found for oak savannah, orchards, spring row crops, vineyards, open water, fallow fields, riparian-dominated areas (Pandolfino et al. 2011a). Females frequently used open alfalfa fields and other irrigated crops, whereas males more frequently used oak savanna, wetlands, and urban areas (Pandolfino et al. 2011a). While males show a positive association with grasslands, females are present in grasslands at higher densities during the winter (Pandolfino et al. 2011a). Pandolfino et al. (2011b) also found no significant difference between usage of grazed and ungrazed grasslands during winter.

Migration

American kestrels exhibit differential migration patterns, depending on variation of latitude, elevation, climate, and prey availability (Roest 1957, Cade 1982). According to Cade (1982), kestrels north of ~45º (northern Oregon) are highly migratory, kestrels south of ~35º (southern California) are year-round residents, while kestrels in between are partially migratory. California’s resident kestrel population becomes supplemented with northern birds and migrants from higher elevations in the winter, a phenomenon that explains why the kestrel is considered to be abundant in California during winter (Grinnell and Miller 1944, Cade 1982, Small 1994, Peeters and Peeters 2005). Some individuals migrate down-slope from higher elevations, beginning mid-August, to areas with more mild climates and suitable foraging, such as: agricultural areas, urban areas, the southeastern deserts, the Central Valley, and coastal areas like the Marin Headlands (Grinnell and Miller 1944, Mills 1976, Bird and Palmer 1988, Small 1994, Pandolfino et al. 2011a). In California, both hatch year juveniles and adults begin fall migration in early September where the winter climate becomes unfavorable; and hatch year females have been shown to leave earlier than their male siblings (Balgooyen 1976, Hull et al. 2012). Kestrels start spring migration to their breeding areas mid-February through April (Bird and Palmer 1988, Small 1994, Fish and Hull 2010).

After nestlings fledge, adults and juvenile often move upslope to areas with abundant prey (e.g. grasshoppers), or to the northeastern Modoc plateau, prior to the onset of winter weather patterns (Grinnell and Miller 1944, Bird and Palmer 1988).This movement is thought to begin the mid-August migration. During the fall/winter migration period, kestrels may migrate alone or in groups (Smallwood and Bird 2002). Large congregations of migrating kestrels, numbering in the thousands, have been documented in other States; with 25,000 kestrels over one day in New Jersey, being the largest movement ever reported (Law 1915, Choate 1972).

When talking about migration, it is important to note that the only Raptor Migration Count (RMC) location in the State of California is at the Marin Headlands, and is administered by the Golden Gate Raptor Observatory (GGRO). GGRO bands between 30 and 90 kestrels each year (1983-present); and observes between 300 and 800 kestrels per year during the migration period (Elliot and Fish 2010, Fish and Hull 2010, Figure 1).

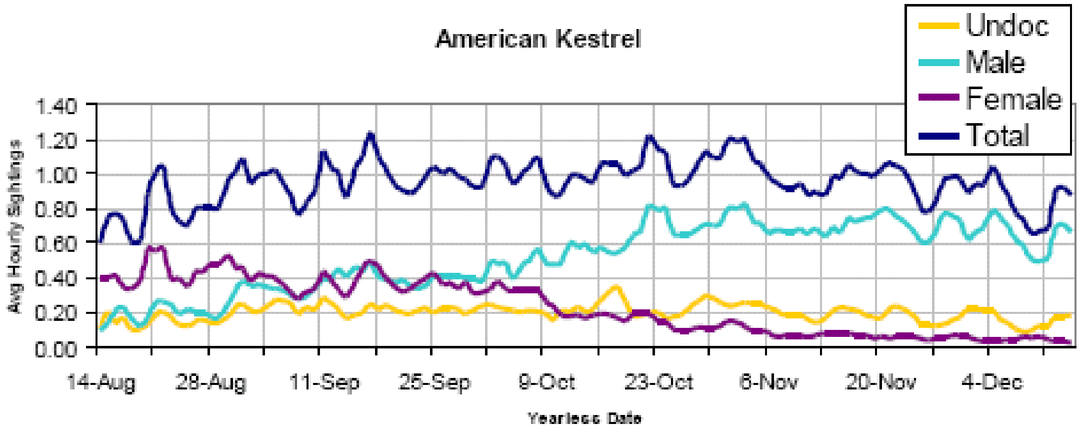

Figure 1. Migration profile from 1986 to 2009 (Elliot and Fish 2010).

Food Habits

Many studies have documented the diet, foraging methods, and forage habitat requirements of the kestrel. Hunting techniques vary and are comprised of sit-and-wait, soaring, hovering, coursing, ground foraging, aerial attacks, nest robbing, caching, and carrion eating (Balgooyen 1976, Collopy 1977, Collopy and Koplin 1983, Bird and Palmer 1988). A Sierra Nevada study by Balgooyen (1976) notes the importance of perches, as 97% of foraging attempts are classified as sit-and-wait. Prey includes insects, small mammals, small birds, reptiles, amphibians, and bats (Sherrod 1978, Mueller 1987, Bird and Palmer 1988, Bird and Smallwood 2002). Insects are eaten regularly, and grasshoppers constitute the preferred prey (Brown and Amadon 1968, Cade 1982, Bird and Palmer 1988, Capinera 2010). Mueller (1987) found that long-term preferences for prey are positively associated with a high probability of capture. In addition, as a particular prey species abundance changes, so does the kestrels preferences for that prey (Bent 1938). For example, only when grasshopper abundance plummets below a viable level of sustainability, does the kestrel shift hunting towards an alternate prey source (Fisher 1893).

Secondary to grasshoppers, prey items of importance include ground dwelling beetles, spiders, caterpillars, and small mammals (Capinera 2010). Small mammals taken include mice, voles, small squirrels, shrews, wood rats, pocket gophers, bats, and even rabbits (Sherrod 1978, Cade 1982, Ryser 1985, Mueller 1987, Capinera 2010). To a lesser degree, kestrel also preys upon small reptiles (lizards and snakes), dragonflies, earthworms, scorpions, amphibians, and a large variety of birds (Brown and Amadon 1968, Cade 1982, Capinera 2010). Average individual prey weight is about 15 grams, ranging from 0.05 to 89 grams (Balgooyen 1976, Johnsgard 1990). In a study by Lenihan (2007) in Contra Costa and Alameda Counties, American kestrel was more commonly found at California ground squirrel colony sites, than off site, where increased availability of birds and invertebrates preferred by kestrel may be found.

Breeding

Depending on latitude and elevation, the breeding season for American kestrel begins in late February to late May (Wheeler 2003, Peeters and Peeters 2005). In southern California where kestrel is non-migratory, courtship begins as early as December (Cade 1955). In the higher evaluations of the Sierra Nevada, where severe weather and low prey availability prevents winter residency, kestrels arrive at their respective breeding territories in early to mid April (Balgooyen 1976). After arrival at a territory, there is period (1 to 2 weeks) of promiscuity and courtship during nest site selection, until monogamous pairs are formed (Cade 1955, Roest 1957, Willoughby and Cade 1964, Balgooyen 1976, Cade 1982). Kestrels do not build their own nest, therefore final selection of a nest initiates the nestling period (Johnsgard 1990, Wheeler 2003). Natural cavities and woodpecker holes are preferred; however artificial nest sites mimicking natural conditions are readily accepted (Roest 1957, Brown and Amadon 1968, Balgooyen 1976, Cade 1982, Ryser 1985, Bird and Palmer 1988, Johnsgard 1990, Smallwood and Bird 2002, Hunter et al. 2005, Bousman 2007).

Studies in California show American kestrel lay eggs between late May and early June (Balgooyen 1976, Bloom and Hawks 1983). The average clutch size is between 4 and 5 eggs, with a range of 3 to 7, and incubation lasts 29 to 30 days (Balgooyen 1976, Cade 982, Bird and Palmer 1988, Wheeler 2003, Peeters and Peeters 2005). Replacement clutches are not uncommon when a nest attempt fails, especially in more temperate areas like southern California (Cade 1982, Bird and Palmer 1998). Both sexes develop brood patches on their breast during this period and both participate in incubation (Willoughby and Cade 1964).

After hatching, young kestrels weigh about 10 to 12 grams and are fed about 2 to 4 grams of food per day (Balgooyen 1976). In a week's time, the young consume about 25.5 grams of food per day, the same food intake as an adult; at this time both sexes must hunt and provide food for the hungry nestlings (Balgooyen 1976). Within two weeks the young weigh as much as an adult and fledge at about one month old (Craighead and Craighead 1956, Balgooyen 1976, Wheeler 2003). Once fledged, young are still dependent on the parents for food from one to three weeks (Cade 1955, Balgooyen 1976, Wheeler 2003). In this time the young observe the hunting behavior of the parents, until the first successful kill marks the end of the fledgling phase (Balgooyen 1976). Recently fledged juveniles have been observed in flocks up to 20 individuals during summer (Wheeler 2003).

Home Range

American kestrel home range and density of individuals depend upon the season and prey demands. Kestrels generally return to the same breeding territories year after year, hence, familiarity with a past home range may provide returning kestrels advantages over intruding conspecifics (Cade 1955, Balgooyen 1976). Within larger patches of suitable breeding habitat, male kestrels will confine activities to small defendable home range patches (Balgooyen 1976). Smallwood et al. (2009) suggested the small home range of kestrel may rely upon larger contiguous patches of suitable foraging habitat during the breeding season. Females on the other hand move freely between male home ranges during courtship; they are initially promiscuous, until selecting a mate that has a defended territory and established a nest site (Balgooyen 1976).

Average home range size is estimated to be between 1.094 and 1.296 km^2 (Balgooyen 1976, Craighead and Craighead 1956). Santolo and Yamamoto (2009) found kestrels foraging from 0 to 3.4 km away from the nest location in Merced County. There are many factors that determine boundaries of home ranges, including but not limited to: vegetation type and density, canopy characteristics, food availability, low ground cover, topography, potential cavities, food availability, and availability of hunting perches (Balgooyen 1976, Greenwood and Dawson 2011). Kestrel preferentially selects for large patches of habitat that meet their needs, thus allowing for densities as high as 24.7 pairs/km^2 and higher rates of occupancy than small, fragmented habitat patches (Bird and Palmer 1988, Smallwood et al 2009). While still largely misunderstood, it is hypothesized that larger habitat patches are essential for allowing home range overlap and social interaction among breeding American kestrels (Smallwood et al. 2009).

Historical and Current Distribution

Grinnell and Miller (1944) documented resident kestrels throughout almost the entire State of California. The historical literature includes elements of occurrence in nearly every County in the State (Bond 1943, Grinnell and Miller 1944). Breeding pairs were reported from the "Lower Sonoran" up to the "Hudsonian", in every region of California where their preferred habitat occurs; except the treeless mountain tops of the Sierra Nevada and Cascade mountain ranges (Grinnell 1915, Grinnell and Miller 1944). Historically kestrels were found on all of the coastal California islands, but only reported breeding occurred on Santa Cruz Island (Dawson 1924, Grinnell and Miller 1944). Winter snow shifted kestrel distribution down into the lower open valleys of California and expands in the southeastern deserts (Grinnell and Miller 1944). Altitudinal distribution varied from 70 m below sea level in Death Valley to 4,000 m on Mount Shasta (Fisher et al. 1893, Merriam 1899).

Today, the distribution of kestrel in California remains relatively unchanged (Peeters and Peeters 2005). Some shifts have occurred as historical habitat is removed or altered, however this loss is often replaced with other suitable habitat. For instance, the Sacramento and San Joaquin Valleys (and other interior valleys of California) historically consisted of extensive prairie grasslands, oak woodlands, and the extensive riparian forests associated with tributaries to the Delta (Tyler 1923, Thompson 1961). Much of this native habitat has been changed drastically due to increasing urban and agricultural development; thus forcing kestrel to use nest boxes and other man-made structures for breeding and agriculture fields for foraging (Thompson 1961, Allen-Diaz 2007).

Kestrels have recently established breeding populations on Santa Barbara, San Nicolas, and San Clemente Islands (Garrett and Dunn 1981, Sullivan et al. 2003). This expansion in breeding distribution has been attributed to eradication of the native flora (e.g., *Dudleya spp.*), thus creating more open hunting ground (Sullivan et al. 2003).

Historical and Current Abundance

The relative abundance of American kestrel in California varies seasonally, annually, and locally (Figures 2, 3, and 4). Historical records note this falcon as common throughout the State during summer, aside from the southeastern deserts and the northwestern humid forest, and abundant during the winter in the lower valleys and grasslands, with a widespread distribution (Grinnell and Miller 1944).

Varying degrees of suitable breeding habitat that existed across the State explained the differences in local abundance. Early accounts reported kestrels as an abundant resident in the coastal slope from San Diego County up to Santa Barbara County (Grinnell 1893, Willet 1912, Willet 1933). Kestrels in the San Jacinto area were fairly common (Grinnell and Swarth 1913). Areas of scarcity included the San Bernardino Mountains and the desert regions (Grinnell 1908, Grinnell and Swarth 1913). Exceptions include regions containing suitable nesting cavities such as the giant saguaro cactus in the Lower Colorado River Valley and Joshua tree woodlands in the Mojave Desert (Fisher et al. 1893, Grinnell 1914, Johnson et al. 1948). Surveys in the Yosemite region, the San Joaquin Valley and to the west in the Coast Ranges, in the Salinas Valley, and in the San Francisco bay area region all reported kestrels as common (Fisher et al. 1893, Grinnell and Storer 1924, Grinnell and Wyeth 1927, Willet 1908). In the more wooded areas of the Lassen transect, residents were scattered from Eagle Lake to Mt. Lassen to Red Bluff; and more numerous at lower elevations (Grinnell et al. 1930).

Today, kestrel abundance is in flux across North America (Bird 2009). Small (1994) considers kestrels in California as a “fairly common resident and uncommon coastal fall transient”; with seasonal movements remaining the same. Inzunza (2007) analyzed a “Raptor Population Index” at various migration sites, and has suggested widespread declines of abundance of kestrel in the eastern United States. This same decline was later suggested for the Pacific Northwest region (Farmer and Smith 2009, Fish 2011).

According to the Breeding Bird Survey (BBS) the highest levels of abundance from 1994 to 2003 occurred in the Central Valley (concentrated around the Bay Area), extending down to the southern San Joaquin Valley and through the coastal ranges down to Point Conception, and also in southern California near the inland areas of agriculture (Figure 3). BBS data from 2006 to 2010 shows a marked decrease in abundance, with the area from Santa Barbara County south to Los Angeles County showing no abundance data (Figure 4). Christmas Bird Count (CBC) data from 1959-1988 show kestrel as abundant in the Central Valley (concentrated around the Bay Area) and common in the rest of the State besides the Great Basin and high Sierras (Figure 2); but there has not been a CBC distribution/relative abundance map produced since 1996.

In Santa Clara County, Bousman (2007) reported kestrel as still common in the Diablo Range, the eastern foothills, and the valley floor; despite declines noted in CBC data. Kestrel is also reported common in Napa County, and fairly common but increasingly numerous during winter in Sonoma County (Berner et al. 2003, Rudesill 1995). In San Diego County kestrel is considered an uncommon breeder, but more numerous in winter (Unitt 2004). In Orange County, kestrel is considered common in open grasslands (Hamilton and Willick 1996). In Santa Barbara County kestrels are common near the coast, inland, and winter valleys; while uncommon in the mountains (Lehman 1994).

In contrast, surveys in Santa Cruz County confirm an uncommon declining breeding population (Suddjian 2009). Only two nests (from 2001 to 2003) were found in San Francisco County (Anonymous 2003). Roberson (1993) considered kestrel a common permanent breeder in the open oak savanna, but noted a population decline around urban areas and in the Salinas Valley. This study notes a possible correlation between the reduction in oaks and declines in kestrel abundance (Roberson 1993).

The Glass Mountain Breeding Bird Atlas (BBA) project on the eastern slope of the Sierras, below Mono Lake. found kestrel to be an uncommon breeder (Shuford and Metropulos 1996). Note: other BBA’s are either not available or still in progress, and to properly assess statewide abundance of kestrel using the BBA, all States would need to be well represented.

The Partners in Flight (PIF) used BBS data to model population estimates throughout North America. PIF population estimate for kestrels in California is 240,000 individuals; which is 5.6% of the estimated North American population of 4.3 million individuals (Rich et al. 2004). Population estimates for Bird Conservation Regions (BCR) in California are: 150,000 individuals in Coastal California (inclusive of the entire Central Valley, surrounding foothills, and coastal counties from northern Bay Area to the border of Mexico); 6,000 for the northwestern coniferous forest; 8,000 for the Sierra Nevada; 50,000 for the southeastern deserts; and 150,000 for the Great Basin. Cade (1982) estimated 2.4 million individuals in North America based on extrapolating nesting densities of one pair per 100 km^2 north of 45ºN and one pair per 5km^2 south of 45ºN (Balgooyen 1976, Craighead and Craighead 1956). Johnsgard (1990) gave a conservative estimate of 236,000 individuals in the U.S. and Canada, based on 1986 CBC data. Using survivorship and productivity values, Millsap and Allen (2006) derived an estimate of 2,175,000 kestrels per year in North America; of which 60% are juveniles (1,305,000 juveniles/ 870,000 adults).

There are obvious conflicts when taking all of these abundance estimates into account. Much of the variation is likely due to sources of data used in the estimate and the variation in modeling techniques. It does not seem likely kestrels would have increased from 2.4 million individuals in 1982, to 4.3 million in 2004, to 2.2 million in 2006, especially when trends indicate a declining abundance in some portions of North America (see Population Trend section below for more information). For California, the issue is further complicated by the 240,000 estimate for California and the 236,000 estimate for the U.S. and Canada; an almost identical estimate between two very different spatial areas.

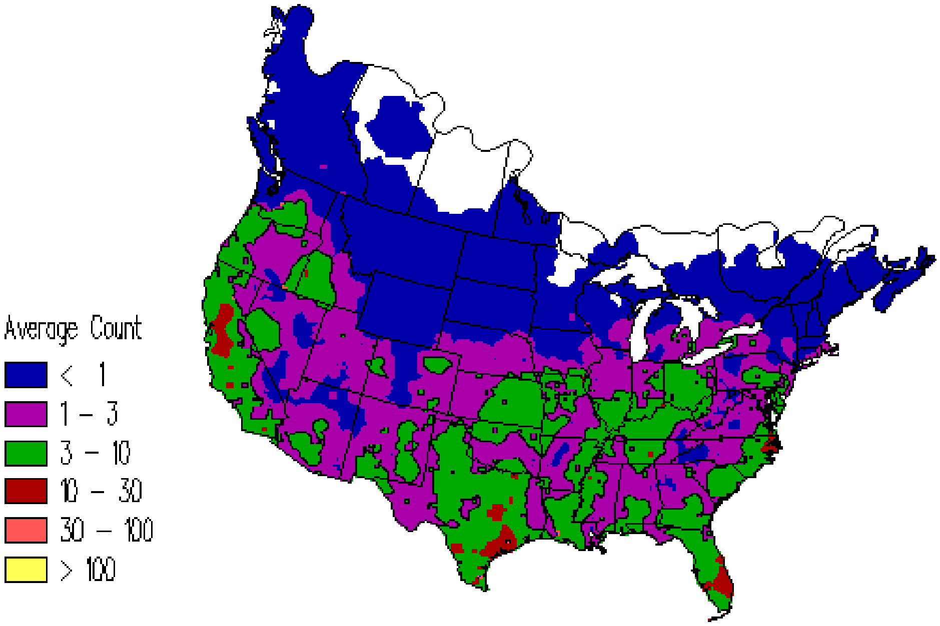

Figure 2. Winter distribution and relative abundance 1959 to1988 (Sauer et al. 1996)

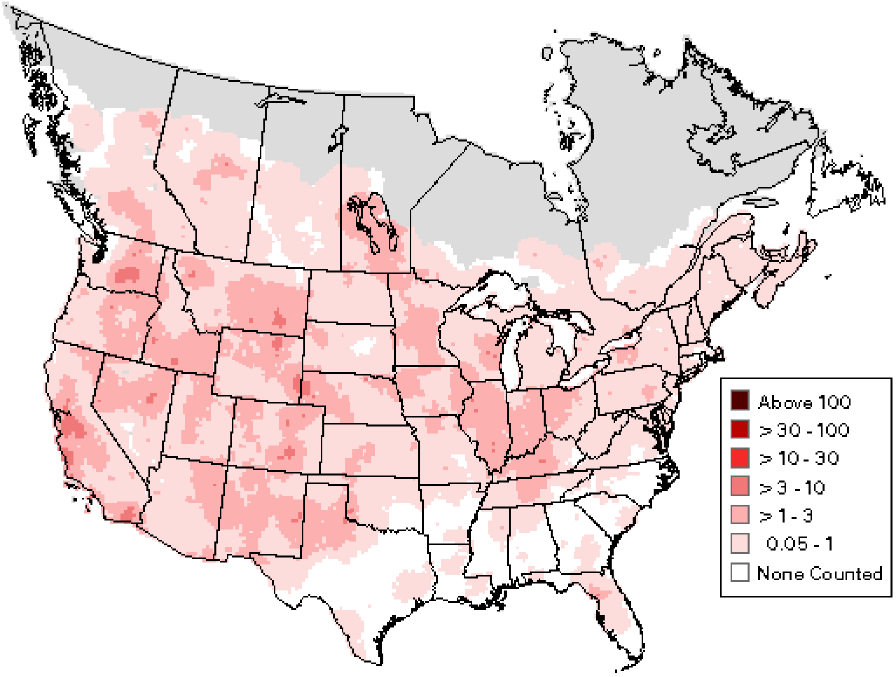

Figure 3. BBS summer distribution and relative abundance 1994-2003 (Sauer et al. 2008)

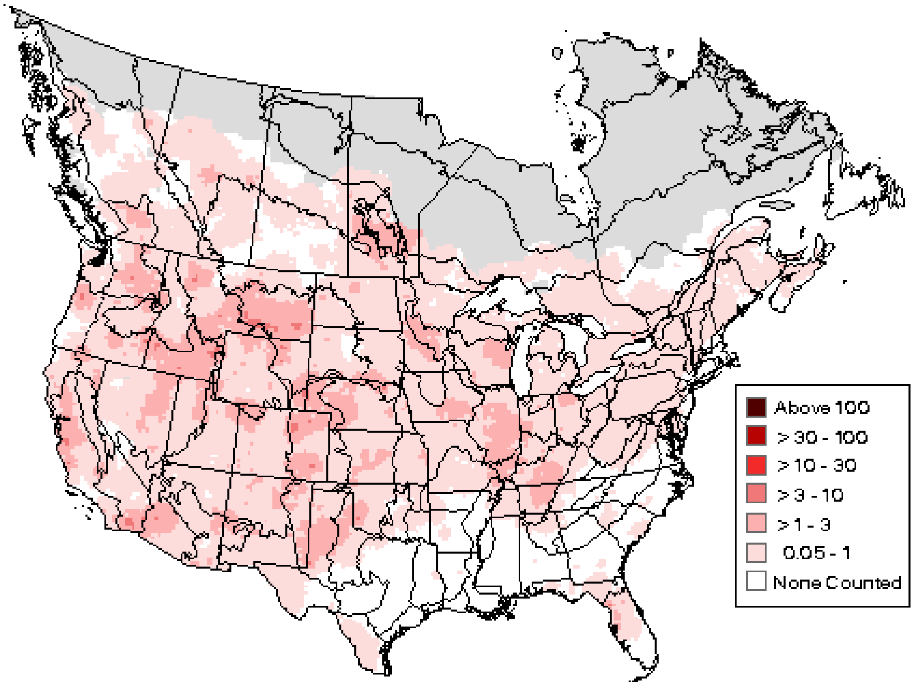

Figure 4. BBS summer distribution and relative abundance 2006-2010 (Sauer et al. 2011)

Population Trend

Analyzing long-term trend from nest-box studies in eastern North America have determined significant declines in resident and migratory kestrel populations of between, 10% and 40% (Smallwood et al. 2009, Steenhof and Peterson 2009). Nest-box studies in California were not geared towards a long term analysis, and no trends could be determined using this method (Bloom and Hawks 1983).

RMC trends are useful in supplementing BBS trend data; trends of higher precision may be obtained when using both monitoring methods in concert, since both reflect true changes in population size for kestrel (Farmer et al. 2007). Until recently, there had not been an attempt to determine the rate of population change of kestrel throughout North America. Farmer and Smith (2009) analyzed twenty different raptor migration sites across the continent that showed moderate to strong declines in kestrel populations (4 to 12% decline per year) from 1995-2005 in the western United States (except Manzano Mountains, New Mexico). The GGRO in California, showed a significant decline in kestrel observations of 2% per year since1989 (Figure 5 and 6), a result that supports the nationwide decline of about 5% per year (Elliot and Fish 2010, Fish 2011). According to the GGRO website current kestrel migration counts are below average for 2011, at only 358 individuals by the end of the fall migration count on December 4th (Golden Gate Raptor Observatory 2011).

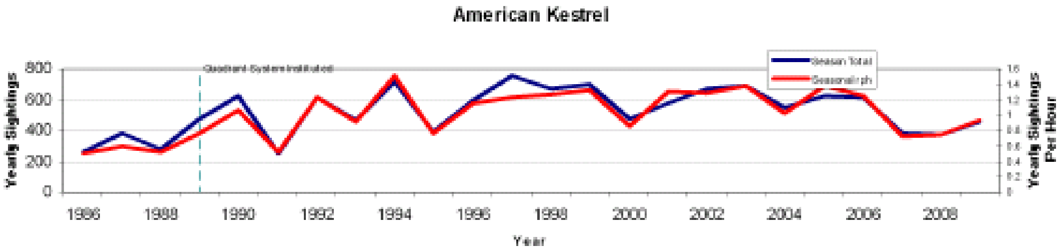

Figure 5. Since quadrant system instituted in 1989, an average 2% decline/year was noted - linear regression ANNOVA p=0.013 (Elliot and Fish 2010)

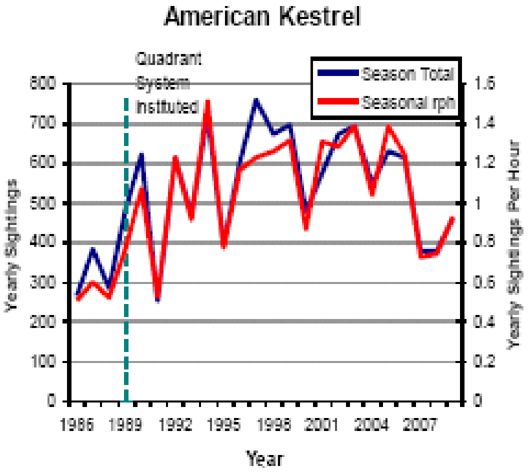

Figure 6. Yearly trends in sightings at Marin Headlands from 1986 to 2009 (Elliot and Fish 2010)

The latest trend data from the BBS is available for the period of 1966-2010 (Sauer et al. 2011). The overall BBS trend for the State of California shows a decline of 1.9%, with most of the

BCRs showing a decline as well (Figures 7-13). The Sonoran and Mojave Desert BCR (Figure 11) represents the only positive trend in California from 1966-2010, although this trend is not statistically significant (Sauer et al. 2011). The most alarming negative trend from 2000-2010 has occurred in the Coastal California BCR (-3.6%), which contains approximately 50% of all California BBS routes (n=107/215) (Figure 10). The overall California trend from 2000 to 2010 shows a steeper decline, -2.1%, than the period from 1966 to 2010 (Figure 7).

Figures 7-13: American kestrel BBS trend data from all Bird Conservation Regions (BCRs) in California from 1966 to 2010 (Sauer et al. 2011).

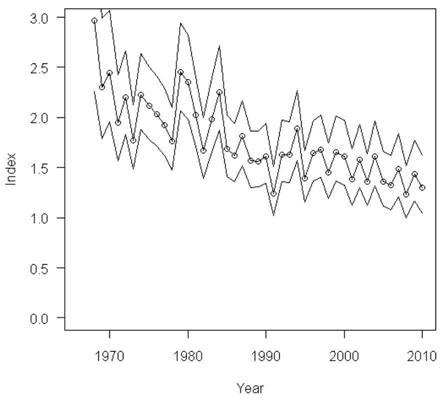

Figure 7. Entire State of California

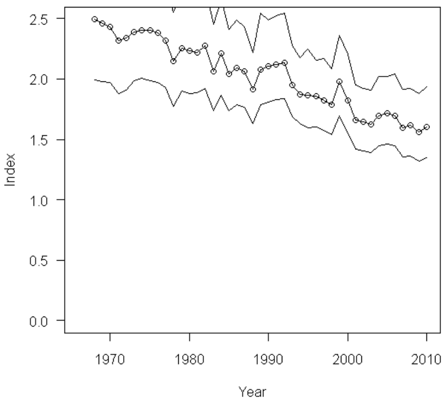

Figure 8. Great Basin BCR

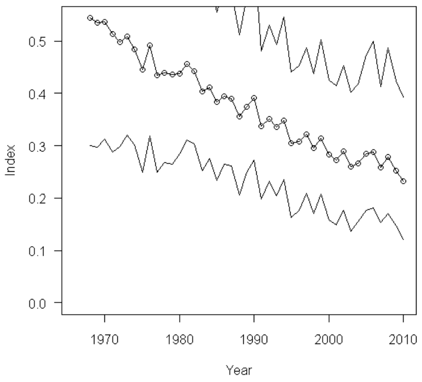

Figure 9. Sierra Nevada BCR

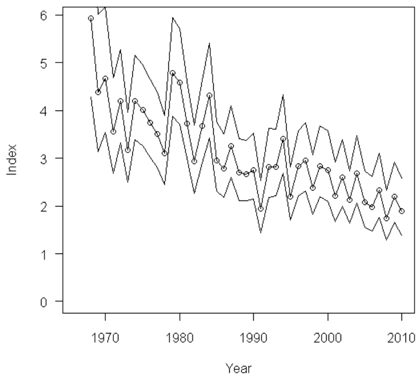

Figure 10. Coastal California BCR

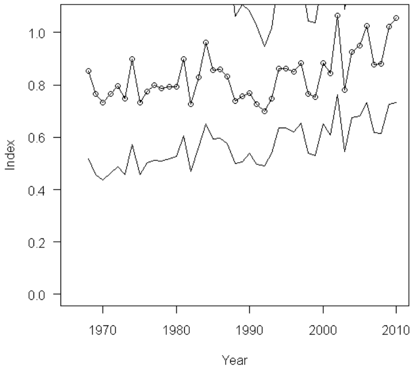

Figure 11. Sonoran/Mohave Desert BCR

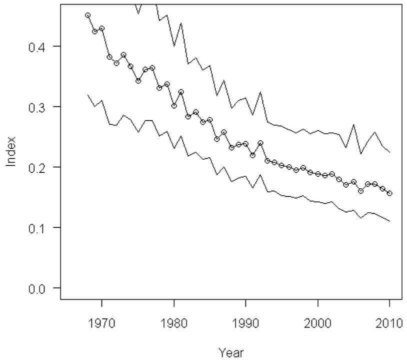

Figure 12. Northern Pacific Rainforest BCR

CBC data is also useful in assessing changes in population trend for kestrel. A recent CBC trend analysis, using log-linear regression, found population declines from 1983 to 2005 (-1.3% per year) and from 1995 to 2005 (-2.3% per year) in Western North America; these declines closely resemble BBS trend data for the same time periods (-1.7% and -2.7% per year, respectively) (Farmer and Smith 2009). Although there has not been a published trend analysis since Farmer and Smith (2009), it is apparent that the number of kestrels recorded via CBC has been declining in California, with a low of 0.3046 kestrels/per hour in 2008 (Figure 14 and 15). Besides a few locations in California that show an increasing trend (e.g., Salton Sea), wintering kestrels have experienced a long shallow decline since the 1960s, close to -1% per year (A. Fish, Golden Gate Raptor Observatory, personal communication).

GGRO migration counts, BBS data, and CBC data all point to declines in trend for kestrels in California since the 1980s (Elliot and Fish 2010, National Audubon Society 2010, Fish 2011, Sauer et al. 2011, A. Fish, personal communication).

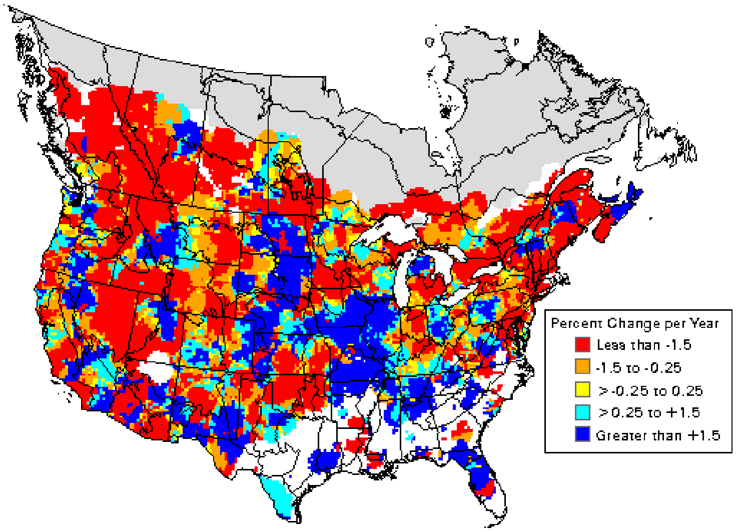

Figure 13. Population trend in North America from 1966 to 2010 (Sauer et al. 2011)

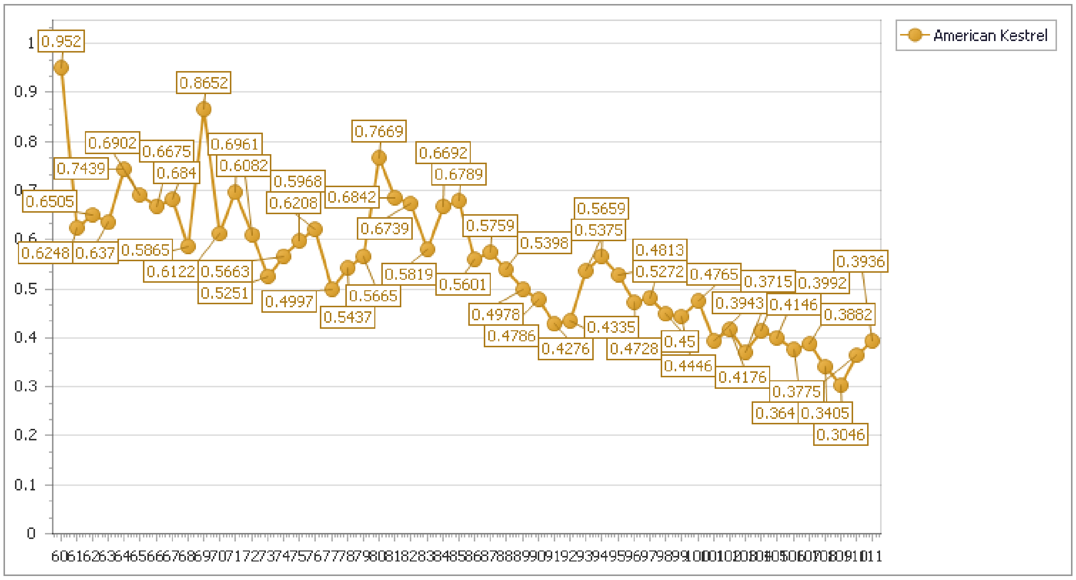

Figure 14. CBC in California from1959 to 2010 (National Audubon Society 2010) [# kestrels/party hour]

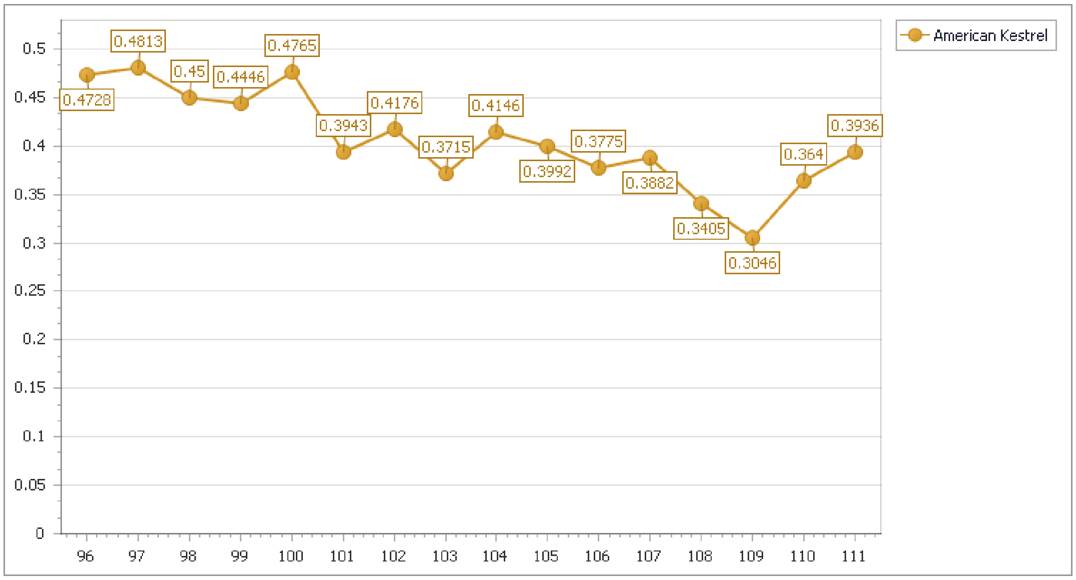

Figure 15. CBC in California from 1995 to 2010, # kestrels/party hour (National Audubon Society 2010)

Threats

Many of the threats that can negatively impact American kestrel are caused by human activity, including shooting, pesticide exposure, collisions with vehicles and aircraft, nest site disturbance, research impacts, and loss or degradation of habitat (Smallwood and Bird 2002). Uncertainty remains regarding the main factor contributing to the observed long-term decline in kestrel, but changes in trend are speculated to be caused by a culmination of multiple threats (Bird 2009, Smallwood et al. 2009, Farmer and Smith 2009, Fish 2011).

Habitat alteration, degradation, and destruction may be the worse threat to American kestrel (Smallwood and Bird 2002, Bildstein 2006). . Major losses of old-growth forest to agriculture have already occurred in the Salinas Valley, Santa Clara Valley, the lower Colorado River Valley, the Santa Rosa Plain, and the great Central Valley, among other areas (Thompson 1961, Rosenburg et al. 1991, Roberson 1993, Bousman 2007, Cumberland 2009). Although some agriculture practices benefit kestrels by creating foraging habitat, intense agriculture (vineyards and orchards) can result in unsuitable habitat (Pandolfino et al. 2011b). Increasing urbanization may also be detrimental to kestrels as foraging areas become less abundant (Unitt 2004, Bousman 2007, California Department of Conservation 2011, Pandolfino et al. 2011b). For long-lived species, such as the valley oak, regenerating to historical levels in the interior valleys and foothills of California may be difficult (Allen-Diaz 2007, Sawyer et al. 2008). These natural, old large trees (mainly oaks) and the primary cavity nesters (e.g. woodpeckers) that create nesting sites are essential to American kestrel, and should be conserved (Roberson 1993, Shuford 1993, Rudesill 1995, Unitt 2004, Hunter et al. 2005, Bousman 2007, A. Fish, personal communication). Fire suppression may prove detrimental as well, as many tree cavities are located in dead or burned trees (Balgooyen 1976). A new potential threat to American kestrel is the impact Sudden Oak Death (*Phytophthora ramorum*) may have on California’s landscape.

The kestrel was an early test species to determine the effects of pesticides on raptors (Nichols et al. 2010). Early toxicology studies focused on the relationship between eggshell thinning/breakage and organochlorine insecticides such as DDT (Porter and Weimeyer 1969, Lincer 1975). Studies since then have focused on the impacts of other toxic substances such as insecticides, lead, sodium fluoride, mercury, selenium, and fire retardants, among others (Pattee 1984, Henny et al. 1993, MacLellan et al.1996, Martin and Barrett 2001, Smallwood and Bird 2002, Fernie et al. 2009, French et al. 2010, Santolo and Yamamoto 2009).

The toxic substances can make it into the kestrel diet via secondary poisoning (e.g. consuming contaminated prey) or direct contact with contaminated sources in the natural environment (Smallwood and Bird 2002). For instance, when the toxins are consumed by the targeted pest, such as sparrows, kestrels have been shown to actively select for these contaminated prey, and mortality can result from this single dose (Hunt et al. 1991, Hunt et al. 1992). Dicofol, one of the last insecticides to be deemed illegal, has been linked to negative imapcts on kestrel reproduction including feminization of male embyos, reduced viability of hatched chicks, and negative alteration in reproductive behavior (MacLellan et al.1996, 1997). Other pollutants, such as selenium and methylmercury, have also been shown to negatively impact reproduction (Hoffman et al. 1998, Fernie et al. 2000, Fernie et al. 2001, Fernie et al. 2009, Fernie et al. 2011, Marteinson et al. 2011a, Marteinson 2011b, Santolo et al. 1999, Ohlendorf 2002, Albers et al. 2007). Santolo and Yamamoto (2009) has shown kestrel to be a valuable indicator species for the presence of selenium (and possibly other contaminants in the environment), that build up in lethal amounts in adult raptors and theiroffspring. Immunosuppression from methylmercury may have negative impacts on gene expression and immune system functioning, making nestlings and juveniles vulnerable to pathogens during the early stages of development (Fallacara 2011). Anthropogenic flame-retardent toxic chemicals have become globally persistent in the environment, and may play a role in the ongoing decline of kestrel (Bird 2009, Fernie et al. 2009, Marteinson et al. 2011a).

As some kestrels are found in close association with agricultural and urban habitat, the use of anticoagulant rodenticide in farmland and urban settings may also adversely impact American kestrel by sublethal and lethal bioaccumulation in their diet (Smallwood and Bird 2002, Lima and Salmon 2010, Pandolfino et al. 2011a). Research specific to the toxicity of the pesticide diaphacinone found kestrel to be 20 times more sensitive to the pesticide than mallard or northern bobwhite in previous studies (Rattner et al. 2011).

The indirect effects of using pesticide control on grasshoppers and other insect prey of American kestrel may be worse than actual biomagnification and direct poisoning (Capinera 2010). This loss of insect prey from broad-spectrum insecticides undoubtably impacts the food sources for kestel, as well as other insectivorous birds (Capinera 2010, Fish 2011). Ultimately, pesticides threaten productivity during the breeding season by reducing available prey for hungry chicks and by introducing harmful chemicals into the diets.

WNV is another possible source of kestrel decline (Smallwood et al. 2009, Bird 2009, Farmer and Smith 2009). West Nile Virus (WNV) was initially detected near the Salton Sea in 2003 and spread throughout California by 2004 (Reisen et al. 2004, Hom et al. 2005). The highest rates of WNV mortalities in California have been found in blue oak and valley oak woodlands of the Central Valley and surrounding foothills and valleys, an important breeding and wintering habitat for kestrel (Scott et al. 2008, Roberson 1993, Pandolfino et al. 2011a). High detection of WNV antibodies in kestrels from southern California and other areas may have a large impact on the population abundance in California (Dusek et al. 2010, Smallwood et al. 2009, Medica et al. 2007, Dube et al. 2010).

An increase in predators can also negatively impact kestrel abundance (Fish 2011). Increases in numbers of Cooper's hawk, merlin, and peregrine falcon may impact kestrels due to the potential increase in predation, displacement, and competition (Sullivan and Wood 2005, Farmer et al. 2006, Smallwood et al. 2009, Farmer and Smith 2009, Fish 2011). Farmer et al. (2006) found predation by Cooper's hawk in open patches of vegetation surrounded by woodlands, the preffered edge habitat of the kestrel. No correlation has been found between kestrel abundance declines and an increase in Cooper's hawk abundance, however increased predation may be one small factor immersed among a plethora of other factors contributing to kestrel decline across its range (Smallwood et al. 2009a).

As the number of wind farms increase in California, the combined potential threat to kestrel will also be increased by collisions with turbines and disturbance in open grassland habitat (Bilstein 2006). There has been much research focused on the Altamont Pass Wind Resource Area (APWRA), which has been documented to kill more than 2,000 raptors each year, including but not limited to burrowing owl, golden eagle, red-tailed hawk, and American kestrel (Smallwood et al. 2009c, Smallwood 2010). Initially, Smallwood and Thelander (2008) estimated between 54.9 and 347.6 kestrel deaths occured annually at the APWRA. More recent estimates were refined to predict mean annual kestrel deaths to be between 475.3 and 477.2 (Smallwood 2007, Smallwood et al. 2010). Fatality of American kestrels are correlated with the frequency and close proximity of flights to wind turbines (Smallwood et al. 2009c). Turbines are more likely to be active during high wind speeds when kestrels are more likely to be hover hunting, and kestrels may lose track of spinning turbine blades while actively foraging and interacting with other birds (Balgooyen 1976, Smallwood et al. 2009c).

Aircraft also pose a potential threat to birds and other wildlife, as well as posing a significant safety hazard to humans. Many airports are placed in grassland ecosystems, where grasshopper outbreaks can supply an abundant supply of prey (Garland et al. 2009). Airport lights and traffic signs offer excellent perches for hunting, and the runways allow for easy capture of grasshoppers, and available cavities offer adequate nesting sites (Garland et al. 2009). From 1990 to 2009 2,019 kestrels were struck by aircraft throughout the United States, representing 35% of all raptor strikes (Garland et al. 2009, Dolbeer et al. 2011). Out of those collisions only 19 caused damage to aircraft, with a total cost of $1.4 million (Dolbeer et al. 2011).

Other threats include vehicle collisions near roads, as utility lines and poles are widely used by kestrel during foraging (Smallwood and Bird 2002). Open pipes used across the landscape (e.g. those used at mining claims) also pose a threat to kestrels and other cavity nesters when birds investigate the pipe as a potential nest site, get trapped in the pipe, and subsequently starve (Audubon California 2012).

The impacts of climate change on American kestrel are generally unknown. With ever rising levels of carbon dioxide and increasing global temperatures, direct effects on plants and their herbivorous insect predators may have indirect consequences on kestrel (Krebs 2009, NOAA 2012). Changes in plant community structure, weather patterns (e.g., altered rainfall), timing of insect hatching, and changes in prey availability may be a cumulative threat to current and future declines in American kestrel abundance and reproductive success (Krebs 2009).

As kestrel abundance continues to decline, all sources of mortality could become increasingly problematic (Bird 2009, Farmer and Smith 2009, Garland et al. 2009, Smallwood et al. 2009a).

Data Gaps and Monitoring Needs

Multiple threats to kestrel have been documented and hypothesized; however there is not one clear cause for the decline in trend seen in recent years (Bird 2009). There is still much more research and monitoring needed to determine causes for the decline of American kestrel in California.

Further understanding of the importance of old growth, ancient oaks and other tree species commonly used by flickers and woodpeckers, and consequently kestrels, should be determined. Understanding kestrel territory requirments (breeding and foraging habitat) will help inform landsacpe management in California, especially considering the widespread loss of habtiat due to agriculture and urban developement.

The only long-term method of monitoring missing in California is a statewide nest-box program (Katzner et al. 2005, A. Fish, personal communication). Using artificial nest-boxes can be vital in kestrel conservation, temporarily stopping population declines in areas lacking adequate nesting trees while providing a means to monitor populations (Roberson 1993, Lambrechts 2012). Construction, placement, and maintenance of nest-boxes should be considered and included in all subsequent scientific journal publications and management projects (Lambrechts 2012). Placement of nest boxes next to urban and abandoned areas may provide habitat for sink populations, especially if there is detrimental exposure to insecticides, rodenticides, and other biocides (A. Fish, personal communication). Large-scale banding and biological sampling programs in concert with a nest-box program would help determine migratory movements, residency patterns, home range variations, and effects of toxic substance exposure (A. Fish, personal communication).

Establishing Breeding Bird Atlas projects through out each County in California, especially in the Central Valley region where there has been the greatest amount of old growth valley oak deforestation in the State, would aid in population monitoring for kestrels as well as other avian species (Thompson 1961).

While much study has been focused on chemicals toxic to American kestrels since the early 21st century, much more research is warranted. As long as rodenticides are used to control rodent populations, kestrel will likely be a nontarget species impacted, and may go unnoticed without further study (Rattner et al. 2011). Pesticides used on grasshopper control should also be studied further to determine the direct and indirect affects on kestrels. American kestrel may ultimately serve as an environmental barometer of system health, indicating pollution that can affect other wildlife species, as well as humans.

While kestrel did not receive a high "climate vulnerability score" by Gardali et al. (2012), closely monitoring the effects from climate change is warranted. Using vegetation models in combination with climate change models can give land managers an idea where the highest quality habitat will be for American kestrel in the future. This proactive approach to bridging the data gap of climate change impacts may difficult, but is integral to the conservation of California's raptor biodiversity.

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ECOLOGY OF THE COOPER'S HAWK

By Kevin P. Cahill
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Cooper's hawk (*Accipiter cooperii)* is a sexually dimorphic accipiter that utilizes a variety of habitat types ranging from urban to rural, dry upland to moist lowland, and from large mixed coniferous groves to narrow deciduous riparian strips (Wheeler 2003). It breeds from near sea-level to upwards of 2896 meters (m) (9,500 feet (ft)) throughout much of the United States (U.S.), Southern Canada, and Northern Mexico. In many parts of California, Cooper's hawk can be a year-round resident, seasonal breeder, or passing migrant (Grinnell and Miller 1944, Shuford and Metropulos 1996, Rosenfield and Bielefeldt 1993). Migratory Cooper's hawks winter from California and the rest of the contemporaneous U.S., south to Mexico, Central America, Costa Rica, and as far south as Columbia, South America (Grinnell and Miller 1944, Reynolds 1989). Medium-sized birds such as pigeons (*Columba* spp.), jays (*Cyanocitta* spp.), and robins (*Turdus migratorius*) make up a large portion of its prey base which also includes small mammals, reptiles, and occasionally insects and fish (Bent 1961, Fitch et al. 1946, Rosenfield and Bielefeldt 1993, Wheeler 2003).

Cooper's hawk was named by Charles L. Bonaparte after William C. Cooper founder of the New York Lyceum of Natural History and father of James G. Cooper for whom the Cooper Ornithological Society is named (Bousman 2007). Along with other accipiters such as the northern goshawk (*Accipiter gentilis*) and sharp-shinned hawk (*Accipiter striatus*), Cooper's hawk was demonized in the early twentieth century by poultry farmers and described by Bent in 1937 as a blood-thirsty villain and essentially *the* chicken hawk (Bent 1961, Bildstein 2001). Shooting and trapping were historically significant to populations in the east until the 1960's when they began to receive legal protection (Bildstein, 2001, Rosenfield and Bielefeldt 1993). Population declines in the 1940's and 50's are attributed to the wide use of organochlorine pesticides including dichlorodiphenyltrichloroethane (DDT) and its metabolite dichlorodiphenylethylene (DDE) which caused eggshell thinning (Anderson and Hickey 1972). Whereas Grinnell and Miller (1944) considered the Cooper's hawk as "common" to "abundant" in favorable territory, Gaines (1974) was only able to identify one in his 1973 Sacramento Valley survey. However, Walton's 1972-1975 breeding survey in the Central Valley suggested that Cooper's hawks were reproducing at record levels (Walton, as cited in Asay 1980). Currently, California populations of Cooper's hawks seem to be increasing, especially in urban settings (Bousman 2007, Unitt 2004).

Cooper's hawks have long tarsi and toes, and broad rounded wings adapted for quick, agile flight through dense woodland (Wheeler 2003). Their tail is proportionately the longest of the three North American accipiters (i.e. northern goshawk and sharp-shinned hawk) and has three to four equal-width black bands and a rounded tip with a broad, white terminal band. The ventral bands on the shorter outermost rectrices do not properly align giving the tail a zig-zag appearance (Wheeler 2003). Pale auriculars and nape distinguish the cap from the rest of the upper parts (Wheeler 2003). Ventrally, the white axillaries, breast, belly, flanks and leg feathers are rufous barred while white underwing coverts are spotted and barred rufous (Wheeler 2003). Barring of flight feathers varies from prominent on the outer primaries to variable on the inner primaries (Wheeler 2003).

Cooper's hawk exhibits reversed sexual dimorphism with females ranging from five to twelve centimeters longer than males and wingspans ranging from 8 to 14 centimeters (cm) (3.1 to 5.5 inches (in)) wider (Wheeler 2003). Both males and females are grayish to bluish-grey dorsally with a dark gray or black cap and hackles that can be raised giving the head a square-like appearance; however, females are distinctly grayer than the male's grayish blue (Wheeler 2003). Adult females can uncommonly be brown-backed with a dark brown cap and brownish-gray dorsal plumage (Wheeler 2003). Adult iris color is orange to red and gets darker with age, with the iris color of the male getting much darker than the female (Wheeler 2003).

Juveniles are tawny or brown dorsally and often appear to be hooded (Wheeler 2003). The iris is pale gray, grayish-green or lemony-yellow (Wheeler 2003). Ventrally, they are white with narrow brown streaking on breast and belly tapering toward the lower belly, which may be unmarked (Wheeler 2003). Flanks can be narrowly streaked with brown or have narrow brown diamond shapes or bars (Wheeler 2003). Under-wing coverts are brown streaked while dark barring occurs on flight feathers and retrices (Wheeler 2003). Sub-adults are difficult to distinguish from adults, yet subtle differences exist. The iris of sub-adults is yellowish-orange to orange, and the auriculars and nape are rufous colored (Wheeler 2003). Some scattered brown feathers on the rump and upperwing coverts of sub-adults may be retained from juvenile plumage (Wheeler 2003).

Cooper's hawks are often confused with the smaller sharp-shinned hawk (Clark and Wheeler 1987, Wheeler 2003). However, in flight the head appears larger, protruding beyond the leading edge of their wings. This distinguishing field characteristic is important for differentiating from the sharp-shinned hawk whose wrists protrude forward causing the illusion of a small head (Clark and Wheeler 1987, Dunne et al.1988).

Cooper's hawks can be held in captivity but are considered by some to be temperamental and difficult to maintain (Rosenfield and Bielefeldt 1993). Fitch et al. (1946) provided a good description of a Cooper's hawk that was successfully reared in captivity. Here, the hawk became more hostile as it gained independence and eventually escaped only to be habitually trapped in a quail trap warranting euthanization (Fitch et al. 1946). Bent (1961) describes Cooper's hawks as wild and untamable after his attempts to rear them in captivity but, includes an instance where a Cooper's hawk was successfully raised in captivity, only to be shot by a hunter before further data could be collected. On the other hand, Glasier (1978) suggests that Cooper's hawks are extremely useful birds in falconry, are relatively easy to train, and their size makes it easier to judge how much to feed them compared to smaller falconry species.

Cooper's hawks were the third most popular raptor harvested for falconry in California, with eighty individuals captured from 2006 to 2010 (CDFG 2012). Of the eighty hawks harvested, one was an adult, forty were immature, thirty-six were nestlings, and three were of unknown age. Harvest was spread relatively evenly throughout the state. Of the nestlings harvested, just under half were taken from Riverside, Orange and Los Angeles counties. Seven mortalities and four escapes were reported from 2006 to 2010. Forty Cooper's hawks were released back into the wild during the same period.

Habitat

Cooper's hawks appear less tied to extensive tracts of forest, are less frequent in high-mountain habitats, and more commonly nest in riparian or lowland woodland settings than goshawks or sharp-shinned hawks (Hunter et al. 2005). Grinnell and Miller (1944) described their habitat as chiefly open, interrupted or marginal woodland with nesting sites in riparian deciduous trees, particularly live oaks (*Quercus* spp.). True to this description, 75 of 77 nests (97%) studied in Central California and the Northern San Diego foothills were in live oaks (*Q. agrifolia* and *Q. wislizenii*) and 16 of 20 (80%) nesting pairs chose coast live oaks (*Q. agrifolia*) in Pinnacles National Monument (Asay 1987, Fletcher 2003). However, Cooper's hawks in largely human altered landscapes choose a variety of tree species. Berkeley hawks utilized 13 different tree species, 85 % of which were non-native (Fish et al. 2008). In San Diego County, oaks were commonly used but twice as many nests were found in eucalyptus trees (Unitt 2004). Throughout California, Cooper's hawks also utilize other native evergreen species such as coast redwoods (*Sequoia sempervirens*), ponderosa (*Pinus ponderosa*) and Monterey (*P. radiate*) pines, and deciduous natives such as Fremont cottonwood (*Populus fremontii*) and willow (*Salix* spp.), as well as a variety of exotics such as avocado (*Persea americana*), and blackwood acacia (*Acacia melanoxylon*) (Bousman 2007, Hunter et al. 2005, Palladini 2007, Fish et al. 2008).

Nest are usually located high in the tree, typically 6 to 18 m (20 to 60 ft) from the ground, but have been found up to 41 m (135 ft) high (Wheeler 2003). Berkeley birds nested well below the tops of the crowns, ranging from 12 to 38 m (38 to 125 ft) from the ground (Pericoli and Fish 2004). In Arizona, nest trees were taller for urban nesters than their rural counterparts (Boal and Mannan 1998). In the San Joaquin Experimental Range, Madera County, a pair of Cooper's hawks nested approximately 9 m (30 feet) from the ground near the top of a dense live oak (Fitch 1946). Cooper's hawks at Pinnacles National Monument tended to nest in live oaks that were 20 m (65 feet) tall (Fletcher 2003). Nest trees average 61 cm (24 in) diameter at breast height (DBH) (Felcher 2002).

In California oak woodlands, Cooper's hawks typically nest in uniform groves with low ground cover in close proximity to ephemeral streams or water sources (Asay 1987, Felcher 2002). These birds select stands of six or more trees, at least two of which are growing close enough to form a continuous canopy (Asay 1987). Trees in nest stands are evenly distributed around the nest tree forming a dense uniform closure (84% canopy closure) in all directions (Felcher 2002). Ground cover is either absent or comprises only short grass or poison oak (Asay 1987, Felcher 2002). The understory is relatively open, comprising mostly trunks and large branches (Asay 1987). Areas surrounding a nest stand vary greatly from dense oak woodlands to rolling grasslands (Asay 1987, Fletcher 2002).

In Pinnacles National Monument, Cooper's hawks nested in close proximity to streams averaging 16.7 m (55 ft) from the channel; however, as streams dried up during the breeding season the average distance from water increased to 137.8 m (452 ft) (Fletcher 2002). Twenty-six percent of urban nests in Berkeley, California were within 50 m (164 ft) of water (Fish et al. 2008). As ephemeral streams dry up during the nesting season and prey species concentrate at isolated pools, they provide advantageous foraging opportunities for Cooper's hawks (Fletcher 2003).

Cooper's hawks are increasingly urban residents and can reach higher nest densities in urban versus exurban settings (Boal and Mannan 1998, Unitt 2004, Pericoli and Fish 2004, Bousman 2007). In San Diego, California, the importation of water and trees, especially eucalyptus (*Eucalyptus globulus*), has created ideal environments for species such as the Cooper's hawk in areas that were formerly uninhabitable (Unitt 2004). Urban nests are typically located in the upper third of nest trees which are taller than randomly available trees (Pericoli and Fish 2004).

Disturbance levels at nest site have no detectable influence on nest height; however, Cooper's hawks avoid land use areas focused on agricultural, industrial, and commercial activities (Boal and Mannan 1998, Pericoli and Fish 2004). Fletcher (2002) found that nest sites in close proximity to intense recreational use have greater canopy closure and are less visible from the ground. Urban nests may occur in single-family residential, high-density residential or high-use recreational areas such as city parks (Pericoli and Fish 2004). Nests in Berkeley, California, were concentrated in areas with mature trees and 30% were within 50 m (164 ft) of a park (Fish et al. 2008).

Migration

Most Cooper's hawks from Canada and northern portions of the U.S. are migratory, wintering mainly from Oregon south to Mexico in the west, and as far south as Columbia (Palmer 1988). Local migration occurs in more southerly hawks as they move from snow-covered higher altitudes into valleys and lowlands during the winter months (Rosenfield 1988b). Still, others remain on their breeding grounds throughout the year (Boal and Mannan 1999). In California, juveniles migrate earlier than adults, and females migrate significantly earlier than males (Hull et al. 2012). This is consistent with migration patterns recorded for Cooper's hawks at other western migration points (DeLong and Hoffman 1999).

Detection of Cooper's hawks at the Golden Gate Raptor Observatory (GGRO) in the Marin Headlands begins in mid-August and peaks around September 25 tapering off into mid-December (Figure 1, Elliott and Fish 2010). This peak occurs approximately a week earlier than recorded in the east at Hawk Mountain Sanctuary in Pennsylvania, where the peak occurs in October (HMS 2012). In Southern California, arrival and departure of migrants is hard to determine due to large resident populations but small numbers of migrants are occasionally seen over Point Loma and through the Cuyamaca Mountains, San Diego County. Winter residents concentrate at low elevations and developed areas but are also widespread over the southern coastal slope (Unitt 2004). Cooper's hawk is widely reported by California bird atlases as common during the winter (Lehman 1994, Hamilton and Willick 1996, Berner et al. 2003, Unitt 2004, Hunter et al. 2005, Bousman 2007).

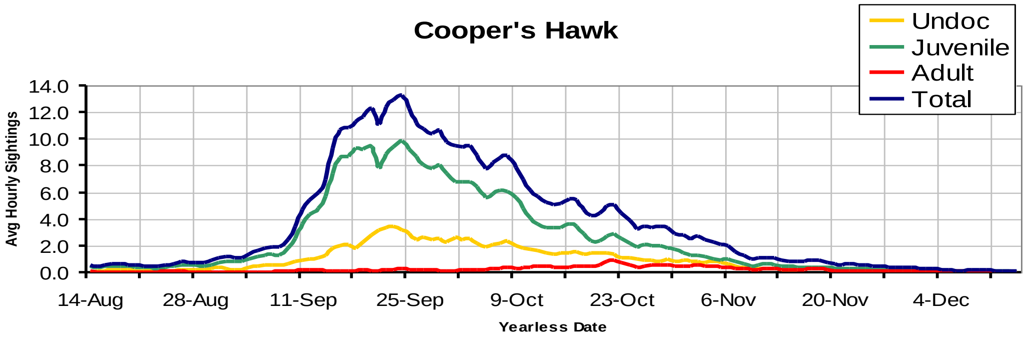

Figure 1. Migration profile for Cooper's hawk from August through December on the California coast. Data is from more than 55,000 sightings of Cooper's Hawks over 25 years at the Marin Headlands, California Golden Gate Raptor Observatory (Elliott and Fish 2010).

Spring migration occurs from mid- to late-March into May and takes place in the reverse sequential order of fall migration (Bent 1961, Rosenfield 1988b). Larger older males migrate first to establish breeding territories followed by adult females. Yearling males and then yearling females take up the rear (Rosenfield 1988b).

Food Habits

Cooper's hawks are both sit-and-wait and aerial hunters, but will often wait under cover of foliage for surprise attacks on prey species (Wheeler 2003). These hawks are hunting zone generalists, hunting in the ground-shrub, shrub-canopy, and canopy zones (Reynolds 1989).

Backyard bird feeders provide concentrations of prey and are frequented by Cooper's hawk (Dunn and Tessaglia 1994, Boal and Mannan 1998). Cooper's hawks have also been observed stooping in a falcon-like attack on pigeons in open habitat near bird feeders (Mead 1963, Clark 1977, Dunn and Tessaglia 1994).

A kneading motion of the feet kills the prey and is initiated directly after the strike which can reach velocities of up to 1,140 centimeters per second (cm/s) (Goslow 1971, Rosenfield and Bielefeldt 1993). Plucking posts within nesting territories are often used during the breeding season but prey is consumed at point of capture, including on the ground, at all other times (Wheeler 2003). Avian prey is plucked before consumption (Clark 1977, Wheeler 2003). Cooper's hawk is also known to pursue its prey on foot and drown prey by holding it under water (Bent 1961).

Foraging activity coincides with the timing of prey activity. Wintering Cooper's hawks leave the roost site just before sunrise, with peak hunting occurring just before sunrise and just prior to sunset before hawks return to the roost (Roth and Lima 2007). The hunting strategy for Cooper's hawks during the winter is balanced between surprise attacks and open attacks for both urban and rural hawks (Roth and Lima 2006). In an urban winter telemetry study conducted in Indiana, 59 of 120 attacks by Cooper's hawks were classified as surprise attacks and 61 of 120 attacks were classified as open attacks. Attacks were made both on solitary individuals and large flocks; however attacks on solitary individuals were about one-third more successful (Roth and Lima 2003).

Cooper's hawks have a variable diet based on prey species abundance and availability (Rosenfield 1988a). Preferred prey is medium sized birds and mammals, though reptiles, insects, and even fish are also consumed (Bent 1961, Fitch et al. 1946, Bielefeldt et al 1992, Rosenfield and Bielefeldt 1993, Wheeler 2003). Birds typically represent the largest percentage of the Copper's hawk diet, followed by mammalian prey (Rosenfield 1988a, Rosenfield and Bielefeldt 1993). Mammalian prey deliveries may increase during nestling stages as females begin to hunt (Millsap 1981, Rosenfield 1988a, Bielefeldt et al 1992). Common avian prey include: American robin (*Turdus migratorius*), jays (*Cyanocitta spp., Aphelocoma spp*.), northern flicker (Coaptes auratus), European starling (*Sturnus vulgaris*), and pigeons (*Columbia livia*). Mammalian prey include: hares (*Lepus* spp.), chipmunks (*Tamias* spp*., Eutamias* spp.), gray squirrels (*Sciurus carolinensis*), deer mice (*Peromyscys* spp.), ground squirrels (*Ammospermophilus* spp*., Spermophilus* spp.), tree squirrels (*Sciurus* spp*., Tamiasciurus* spp.), and bats (*Tadarida* spp.) (Rosenfield and Bielefeldt 1993).

In the Southern Sierra Nevada of Tulare County, California, Cooper's hawks preyed on 21 different bird species, 70% of which were passerine. Of their total diet, 82% was avian, 15% mammalian and 3% reptilian (Carlson 2006). In a Sierra Nevada Foothill study in San Joaquin County, Fitch et al. (1946) found that 68% of a breeding pair's diet consisted of reptilian prey, 26% avian, and 6% mammalian, with whiptail lizards (*Teiidae spp.*) as the most common prey item consumed. In the Klamath Mountains in Northern California mammals represented 65 to 71% of prey deliveries, birds 16 to 23%, and lizards only 1%, with the remaining prey items being unidentifiable (Palladini 2004).

Breeding

Breeding displays begin as early as January 9 and include male sky-dancing, a succession of high altitude, deep close-winged dives ending in a sharp upswing; accentuated slow and deliberate wing-flapping with wrists raised high above body on upstroke; high circling with undertail coverts retracted out beyond the closed tail; and male pursuit flight of the female (Meng and Rosenfield 1988, Pericoli and Fish 2004, Wheeler 2003).

Cooper’s hawks exhibit strong nest site fidelity (Wheeler 2003, Mannan 2010). Males frequently refurbish the previous year’s nest, choose an alternate nest in close proximity to the previous year’s nest or build a new nest in a different tree in the same territory (Wheeler 2003). Cooper’s hawks in the Sacramento Valley foothills had an 80% nesting area re-occupancy rate and 32% reused the previous year’s nests (Asay 1987). In Berkeley, California, ten of twelve Cooper’s hawks nested in the previous year’s nesting territories. Of those breeding attempts, four used the previous years nest and six were located within 121 m (397 ft) (Pericoli and Fish 2004). In Tucson, Arizona, breeding males showed 96.6% site fidelity and females showed 90.6% site fidelity once a breeding territory was established (Mannan 2010).

Initial nest site selection has a distinct order, with a post-fledging home range selected first, then a nesting area within the post-fledging area second, and finally a nest site within the nest area (Fletcher 2003). Non-migratory Cooper’s hawks establish a home range during their first winter in which they will later establish a nest site or wait for one to become available (Mannan et al. 2007, Mannan 2010). Older males tend to occupy the nest sites which are most desirable and most attractive to females. Non-breeding floaters restrict activity to a limited area established during their first year and become familiar with potential nest sites that they can procure from older males once available (Mannan 2010)

Nesting occurs from March through June in southern latitudes and from April through July in northern latitudes (Wheeler 2003). Males do most of the nest building and copulation commonly occurs in the morning about 20-100 m (65.6-328 ft) from the nest after it is inspected by the female (Meng and Rosenfield 1988). Copulation occurs frequently throughout the nest building and egg laying period. Both sexes are vocal throughout copulation, and perching and preening follow (Meng and Rosenfield 1988).

Eggs are laid on alternate days and incubation begins after the third egg is laid (Meng and Rosenfield 1988). Clutch sizes range from 3 to 7 eggs but are commonly 4 to 5 (Meng and Rosenfield 1988, Wheeler 2003). Reported clutches in California follow this pattern. Fitch et al. (1946) reported a clutch of four in the Southern Sierra Nevada Foothills. Walton (as cited in Asay 1980) reported a mean clutch size of 4.26, and Asay (1987) a mean of 4.0.

In Berkeley and the Central Valley, incubation started between late-March and early-May and ranged from between 29 to 42 days (Pericoli and Fish 2004, Asay 1987). The female incubates the egg and only leaves the nest when the male brings her food (Wheeler 2003). Males deliver food 2 to 3 times daily to the female in a nearby tree and incubate while the female eats (Meng and Rosenfield 1988). Eggs hatch asynchronously at the same intervals as they were laid and shells are removed by adults (Rosenfield and Bielefeldt 1993).

Males and females communicate with each other through a series of calls during the pre-incubation and nesting period including “duets” of a variety of notes lasting as long as an hour. The larger females have a wider repertoire than males, call more frequently, and during a wider array of activities (Meng and Rosenfield 1988). Rosenfield and Bielefeldt (1991) suggested that the females influence male-female interactions with each different vocalization. Males are known to consistently use particular routes upon leaving and returning to the nest, and give an alert call when approaching with food (Murphy et. al, 1988, Rosenfield and Bielefeldt 1991). One route is used primarily for prey delivery while another is used for departure. Specific flight routes suggest the importance of nest site familiarity and are one explanation for site fidelity in male Cooper's hawks (Murphy et al. 1988).

The female feeds the young from prey delivered by the male. Prey delivery rates peak during the late nestling period and females take over foraging when young near fledging (Fitch et al. 1946, Murphy et al. 1988). Nestlings begin preening and flapping after 17 days and can dismember prey by day 18 (Meng and Rosenfield 1988). Male nestlings typically depart from the nest to nearby branches after 30 days and females follow after 34 (Meng and Rosenfield 1988, Murphy et al. 1988). Fledglings learn how to hunt by trial and error but prey is delivered up to 10 days after they depart from the nest (Fitch et al. 1946, Meng and Rosenfield 1988). They stay near the nest and perch on nearby branches into late-June, and by mid- to late-July fledglings leave the nest area (Pericoli and Fish 2004). Most mortality, especially from predation by great-horned owls, occurred after nestlings left the nest in the Sacramento Valley (Walton, as cited in Asay 1980). Survival rates dramatically increased from around 9% to about 75% once adulthood was reached in Midwest populations (Roth et al. 2005).

In Berkeley, California, Cooper's hawks averaged 3.7 to 4.0 young per nest. In 2003, 89% of successful nests had 4 fledglings (Pericoli and Fish 2004). This was a relatively high rate compared to Asay (1987) who recorded 2.3 young per active nest and 2.7 young per successful nest from the Sacramento area to Southern California. Nest productivity in Washington and the Northeastern U.S. ranged from 2.67 to 3.6 young per successful nest (Henny and Wight 1972, Kennedy and Johnson 1986).

Cooper's hawks breed in immature plumage but breeders are typically greater than two years old (Rosenfield 1982, Henny et la. 1985, Asay 1987). Over an eleven-year period in Milwaukee, Wisconsin, 85% of breeding males and 78% of breeding females had gray adult plumage indicating they were at least two years of age or older. The remaining breeders were in one-year-old brown juvenile plumage (Stout et al. 2007). It is rare for two juveniles to form a breeding pair however a juvenile male and female pair was reported in California (Kline 1976, Wheeler 2003).

Cooper's hawks are known to occasionally inbreed, possibly due to density and site fidelity (Stewart et al. 2007). Also, a case of polygyny was documented in North Dakota when a marked male successfully fledged young at two nests (Rosenfield et al. 2007). Evidence of surrogacy was documented when two consecutive females took over brooding duties after their predecessors were shot (Bent 1961).

Home Range

Home range of Coorper's hawks depends on habitat and geographical features. Size estimates from observational studies range from 173 hectares (ha) to 1,590 ha (427.5 to 3,929 acres (ac)) throughout the U.S. (Reynolds 1989). Radio telemetry studies are rare however, nine urban radio-tracked Cooper's hawk in Arizona had relatively small home ranges 13.3 to 130.6 ha (32.9 to322.7 ac), with a 65.5 ha (161.9 ac) average (Mannan and Boal 2000). Radio-tracked males in Santa Fe National Forest, New Mexico averaged 1206 ha (2,980 ac) (Kennedy 1989). A radio tracked urban male in Wisconsin had a home range of approximately 784 ha (1,937.3 ac), in which oak-pine woods and shrub savannah were preferred and residential, business and open areas avoided (Murphy et al. 1988). The majority of its time was spent in only 12% of this area and maximum distance traveled away from the nest averaged 1 to 2 km (0.6 to 1.2 mi); a distance typical of and consistent with nest spacing, even in the densest recorded population (Meng and Rosenfield 1988, Murphy et al. 1988, Pericoli and Fish 2004).

Non-breeding behavior has not been studied thoroughly but Cooper’s hawks are generally believed to be solitary during winter (Rosenfield and Bielefeldt 1993). However, banded pairs in Arizona were regularly observed on their territories during the non-breeding season perching near each other, and displaying occasional pair-bond behaviors suggesting pair-bond maintenance throughout the year (Boal and Mannan 1999). Some non-migratory Cooper’s hawks establish well-defined home ranges during their first winter after natal dispersal (Mannan et al. 2004, Mannan et al. 2007, Mannan 2010). Winter dispersal ranges of banded and radio-tracked Tucson, Arizona hawks averaged 771 ha (1,905.2 ac) and nest sites were usually established within or near these territories (Mannan et al. 2004) Fall/winter home ranges for first year Cooper’s hawks in Tucson were on average 11 times larger than the home ranges of breeding males (Mannan et al. 2000)

Cooper’s hawks can attain higher nest densities in urban areas than exurban areas (Rosenfield et al. 1996, Pericoli and Fish 2004). An urban population in Wisconsin had a density of 1 nest per 272 ha (672.12 ac) (Rosenfield et al. 1996). Another study, in California documented the highest density of breeding Cooper’s hawks recorded, with 1 nest per 232 ha (573.3 ac), ranging from 0.8 to 1.11 kilometers (km) (0.5 to 0.7 mi) apart (Pericoli and Fish 2004). Rural nests were as close as 1.2km (0.75 mi) in Pinnacles National Monument, California, and averaged 1.6 km and 2.1 km (1.0 and 1.3 mi) in the Sacramento Valley and Southern California foothills respectively (Asay 1987, Fletcher 2003). In the Southern Sierra Nevada 0.09 to 0.26 Cooper’s hawks per 100 hectare (257.1 ac) were documented (Carlson 2006). Nesting areas in Pinnacles National Monument were comparable in size to Oregon nesting areas at approximately 6.0 ha (14.8 ac) (Reynolds and Wight 1978, Fletcher 2003).

Historical and Current Distribution

Bent (1961) described Cooper’s hawk distribution as North and Central America from British Columbia to New Brunswick, including most of the U.S. (except for the central plains) and Mexico. Bent’s (1961) description included Southern California, the Channel Islands, Central Coast, southern end of the Central Valley, and near Mt. Lassen. According to Cooper (1870), Cooper’s hawks were common during the winter months in all wooded portions of California but retreated to the high mountains and northern portions of the state during the spring. Grinnell and Miller (1944) further described the geographic range as practically the entire area of California below the Canadian life-zone, including the Farallones and most of the Channel Islands. The breeding range in California was chiefly within Upper Sonoran and Transition life-zones from the most northern record near Kangaroo Creek, Siskiyou County to the southernmost records in Poway, San Diego County, and Picacho on the Colorado River, Imperial County (Grinnell and Miller 1944).

The current distribution in California is consistent with the historical accounts described above, with breeding records from Del Norte and Siskiyou Counties, south to San Diego County, and from the coast in Humboldt County to 9500 ft in the White-Inyo Range, Inyo County (Harris 1991, Shuford and Metropulos 1996, Hunter et al. 2005, Unitt 2004, Palladini 2007). The Cooper’s hawk is widely reported by California bird atlases as common during the winter (Lehman 1994, Hamilton and Willick 1996, Berner et al. 2003, Unitt 2004, Hunter et al. 2005, Bousman 2007). Observations were reported throughout the year statewide from 2002 to 2012 in eBird, a citizen science reporting tool, with concentrations in the Central Valley, the entire Pacific Coast, Coastal Ranges and Transverse Ranges as well as the eastern and western slopes of the Sierra Nevada Range (eBird 2012). Observations in the desert region, southern end of the Central Valley, and interior Trinity and Six Rivers National Forests were minimal (eBird 2012).

Historical and Current Abundance/Population Trends

According to Bent in 1937, Cooper's hawk was one of the most common hawks in nearly all parts of the U.S. (Bent 1961). Grinnell and Miller (1944) reported Cooper's hawks in the fall as varyingly common to abundant in favorable habitat in California. In winter Cooper's hawks were more widespread and concentrated in the southern portions of California due to a migratory drift from territories north of California and movement of individuals from areas of heavy snow to lower elevations (Grinnell and Miller 1944).

In the 1970s and 1980s Cooper's hawk populations were thought to be in decline throughout North America. However, winter counts in California were increasing (Brown 1973). Regardless of California's increasing wintering population, the Cooper's hawk was placed on the National Audubon Society's "blue list" from 1972 to 1981 due to overall indications of population declines, and again returned to the "blue list" in 1986 (Anonymous 1971, Arbib 1977, Tate 1986). In addition, Cooper's hawk was included on California's 1978 and 1992 Bird Species of Special Concern (BSSC) lists (third priority) due to declines in many portions of the state, including Santa Clara and San Diego Counties, and destruction of lowland riparian habitat (Remsen 1978, California Dept. of Fish & Game 1992, unpublished list). Cooper's hawk was not included in the 2008 list of California BSSC list due to recent population increases in California, including Santa Clara and San Diego Counties (Unitt 2004, Bousman 2007, Shuford and Gardali 2008).

Recently, Cooper's hawk numbers seem to be stable or increasing, though data is scarce for this secretive breeder and few recent studies have been conducted in California. The Breeding Bird Survey (BBS) data indicates a slightly increasing population trend (0.4 % 1966-2000; 1.2 % 2000-2010); however, this data show an important deficiency for Cooper's hawk due to the low number of encounters (e.g. low abundance) per route and small sample size, both leading to imprecise trend results (Sauer et. al, 2011).

The Partners in Flight (PIF) estimated population size for Cooper's hawk across North America. These estimates were based on BBS detections from 1990-1999 and were extrapolated from relative abundance per route. PIF indicates approximately 31,000 Cooper's hawks in California (Rich et al. 2004). As previously noted, a certain level of uncertainty and bias is associated with BBS data for species that are not always detected during roadside surveys (Blancher et al. 2007).

Data from daily hawk counts at the GGRO (2012) from 1986-2011 show four phases over the last 25 years: (1) 1986-1993 increasing; (2) 1994-2003 static; (3) 2004-2007 increasing; and (4) 2008-2011 decreasing (Figure 2, Elliot and Fish 2010). It is unknown why a sharp decline in observations is seen over the last few years in California, and this trend should be monitored closely (A. Fish, 2012 pers. comm.). However, the net result of GGRO data show a slight increase since the mid-1980s. Similarly, an increase can be seen in sightings per day from 1966-2003 in the east at Hawk Mountain Sanctuary, Pa (Figure 3, HMS 2012).

Christmas Bird Count (CBC) survey data for the non-breeding population also indicate a slightly increasing population trend (1.0 %) in California. Again, caution should be noted with interpreting results due to low detection and abundance of this species on CBC routes (Figure 4, NAS 2010).

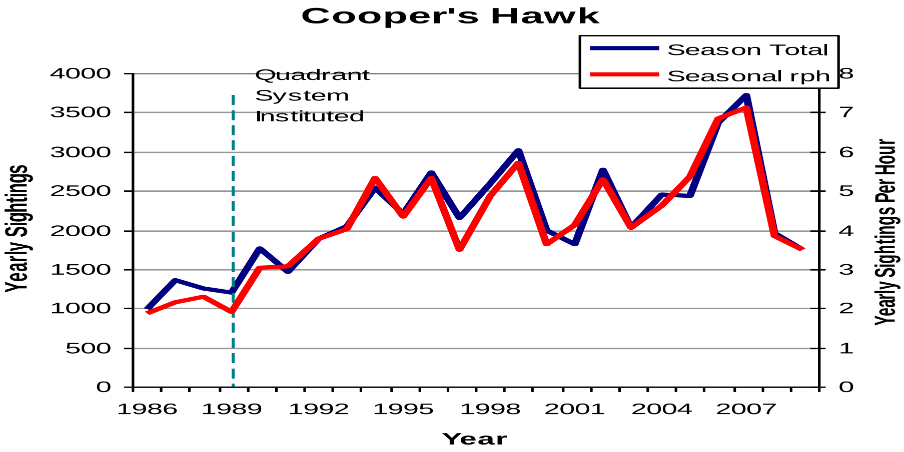

Figure 2. Yearly sightings and yearly sightings per hour of Cooper's hawk at the Golden Gate Raptor Observatory, Marin County, CA 1986 to 2010. The more careful "Quadrant System" count methodology was instituted in 1989 to be more consistent over time (Elliott and Fish 2010).

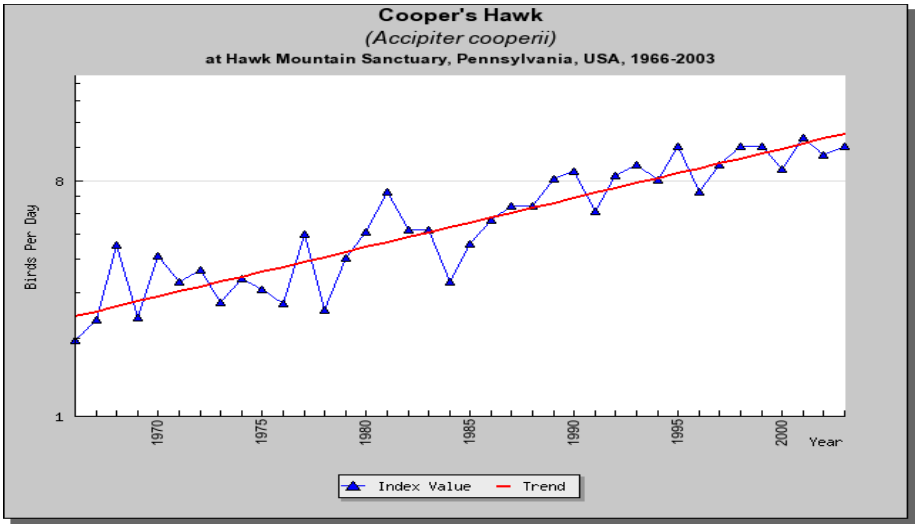

Figure 3. Sightings per day of Cooper's hawk at Hawk Mountain Sanctuary, PA (HMS 2012).

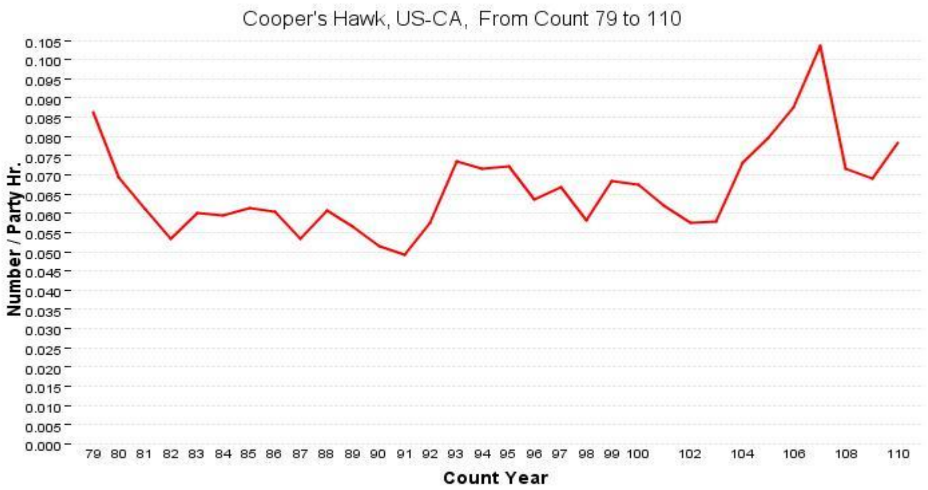

Figure 4. Number of Cooper's hawks detected per party-hour on Audubon Christmas Bird Counts in California from 1978 to 2009 (NAS 2010).

Cooper's hawk continues to be rare in Yosemite National Park and breeding continues to decline in Santa Barbara County (Remsen 1978, Lehman 1994, NPS 2012). However, urban breeding populations in California appear to be dense in areas such as the City of San Diego and Berkeley, as well as Palo Alto, Sunnyvale, and San Jose (Unitt 2004, Pericoli and Fish 2004, Bousman 2007).

Threats

Early in the 20th century, declines in Cooper's hawk numbers were attributed to heavy gun pressure used to control accipiters as threats to poultry and game (Henny and Wight 1972, Bildstein 2001). Legal protection in the 1960's reduced mortality rates from shootings to insignificant levels in the U.S. (Evans 1982, Bednarz et al. 1990). Still, shooting remains a threat to migratory populations that winter in Mexico (Rosenfield and Bielefeldt 1993).

Eggshell thinning caused by bioaccumulation of organochlorine contaminants such as DDT and its metabolite DDE are credited with population declines in the 1950's and 60's but their regulation in the 1970's reduced residues below levels believed to cause thinning (Rosenfield and Bielefeldt 1993).

Today, bioaccumulation resulting in secondary poisoning from rodenticides poses a risk to Cooper's hawks and warrants further study (A. Fish, 2012 pers. comm.). In California, Lima and Salmon (2010) detected residues from retail anticoagulant rodent poisons, including Brodifacoum, Bromadiolone, and Difethialon, in 92% of hawks tested from San Diego and 69% of hawks tested from the Central Valley. Twelve of the ninety-six birds tested were Cooper's hawks (Lima and Salmon 2010).

Human-caused "accidental" mortalities such as collisions with automobiles, windows, or power lines may become increasingly common as urban populations of Cooper's hawks increase (Roth et. al 2005). In Arizona, 69.8% of urban mortalities was caused by collisions particularly with windows, and in California, one third of deaths for all banding recoveries from 1983 to 1995 were due to window collisions (Boal and Mannan 1999, Fish et al. 2008).

Mourning doves (*Zenaida macroura*) and Inca doves (*Columbina inca*), which carry the parasitic protozoan *Trichomonas gallinae,* pose a threat to urban Cooper's hawks that rely heavily on these species as a food source (Boal and Mannan 1999). The protozoan causes a deadly infection called trichomoniasis in chicks. Mourning doves and Inca doves represented 84% of the diet for urban Cooper's hawks in Arizona compared to 4 % in exurban populations (Boal and Mannan 1999, Mannan et al. 2008). Eighty percent of urban nestling mortality was attributed to *T. gallinae* whereas relatively few exurban nestlings were exposed (Boal and Mannan 1999). Another study from three widely dispersed urban North American populations that did not rely heavily on Columbids as a food source detected *T. gallinae* in only 2.7 % of the nestlings (Rosenfield et al. 2002). In Berkeley, California, doves were an important prey species comprising 41% of Cooper's hawk's diet, however the effects of *T. gallinae* have yet to be explored in that population (Pericoli and Fish 2004). Practices such as maintaining clean bird feeders and the use of "dove proof" feeders may reduce transmission between free-ranging doves (Boal and Mannan 1999).

Loss of both suitable wintering and breeding habitat due to logging and urban expansion may threaten populations on local and regional scales (Rosenfield and Bielefeldt 1993). Palladini's (2007) telemetry studies showed that Cooper's hawks in a managed pine forest in the Klamath Mountains preferred to nest in dense untreated stands but foraged in areas where selective thinning and prescribed burning were practiced.

Data Gaps and Monitoring Needs

Few in-depth breeding and wintering habitat use studies have been conducted in California. While Breeding Bird Atlases from throughout the state provide valuable information on distribution, breeding confirmation, density, and habitat association data, our knowledge of the life history of Cooper's hawk in California is minimal and the major sources of reference data are 25 years old or more.

Master's theses from Clay Fletcher (2002) and Jennifer Carlson (2006) of California Polytechnic University, San Luis Obispo, and Michael Palladini (2007) of Humboldt State University have furthered our knowledge of nest site preferences, habitat use, and survey cost and feasibility. Additionally, intensive studies performed by Pericoli and Fish (2004) and the GGRO (2012) have broadened our understanding of urban breeding biology and migration patterns. These studies are important at a time when many rural habitats used by Cooper's hawks and their prey are becoming urban (Pericoli and Fish 2004).

Continued monitoring of breeding populations in high use recreation areas, particularly noting how patterns of nest site re-occupancy relate to recreational intensity, will become increasingly important as recreational levels increase with human population growth (Fletcher 2003). Further studies in urban areas will provide data on how Cooper's hawks adjust to the conversion of rural habitat and the opportunity to examine information such as habitat needs, ecological requirements, and human tolerances (Pericoli and Fish 2004) This information is critical for developing plans and environmental analyses.

Telemetry studies and banding efforts throughout the state will increase our understanding of Cooper's hawks' use of space (Reynolds 1989). More data is also needed to understand the predator-prey dynamics. The effects of rodenticides have not been thoroughly addressed and a better understanding of natural pathogens such as the Trichomoniasis in doves is needed (Wheeler 2003, A. Fish, 2012 pers. comm.). Mortality studies for migratory Cooper's hawks should also be a priority to understand how activities such as pesticide use and gun mortality affect populations in Mexico and Central America (Reynolds 1989).

To limit further habitat loss from logging activities, Reynolds (1983) recommends nest stands and surrounding habitat in coniferous forests be protected, and at least two replacement sites at least 4.7 km apart be provided throughout a management area. Palladini (2007) suggests that alternative management strategies that promote biodiversity through structural heterogeneity may benefit Cooper's hawks that utilize pine forest timber land.

Urban planners should consider including settings such as parks or stands of trees to offer nesting opportunities for Cooper's hawks and other avian species (Unitt 2004). As urban trees mature and human activity attracts opportunistic species such as pigeons, mourning doves, and western scrub jays (*Aphelocoma californica*) and mammals such as California ground squirrels (*Spermophilus beecheyi*), these settings become attractive nesting and foraging areas (Unitt 2004). Biotype mapping within urban areas can help planners develop basic ecological data which can be used in collaboration with wildlife experts to create ideal habitat for native plant and animal species, promoting biodiversity and ecological stability (Niemelä 1999). Inclusion of habitat corridors linking open spaces may help maintain species flow and reduce the effects of fragmentation due to development (Soule1991).

Park operators should consider regulation of recreational use intensity in or near potential nesting habitat, implementation of buffer zones, and education of visitors about disturbance factors such as noise (Fletcher 2003). Trail plans should try to avoid high quality nesting and foraging habitat or high frequency stream crossings to minimize disturbance impacts (Fletcher 2003). Monitoring prey species population levels to better understand community dynamics in conjunction with adaptive management could benefit predator-prey relationships and trophic pathways (Fletcher 2003).

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ECOLOGY OF FERRUGINOUS HAWK IN CALIFORNIA

By Justin D. Garcia

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The ferruginous hawk (*Buteo regalis*) is the largest, most powerful member of its genus in North America (Bent 1937, Bechard and Schmutz 1995). The direct translation of the Latin name is " king hawk" or "royal hawk", which is fitting due to the striking similarities in morphology and behavior with the golden eagle (*Aquila chrysaetos*) (Bent 1937, Borror 1960). This regal predator of the prairie is endemic to the western portion of the North American continent, from southern Canada to northern Mexico, filling its niche as a small mammal pradator in the open landscapes of grasslands, shrubsteppe, and desert (Bechard and Schmutz 1995). In California, it was historically considered to be an abundant winter visitant, but currently is a fairly common to uncommon winter visitor throughout most of the state in areas of suitable habitat and prey (Grinnell and Miller 1944, Small 1994, Polite and Pratt 1999, Peeters and Peeters 2005). Ferruginous hawk has been confirmed to be a scarce and irregular breeder in extreme northern and northeastern California (Harlow and Bloom 1989, CDFG 2012, B. Woodbridge U.S. Fish and Wildlife Service personal communication 2012). There are no recognized subspecies of ferruginous hawk (Gosset 1993, Bechard and Schmutz 1995).

In 1991, the ferruginous hawk was petioned to be listed throughout its range in the United States under the federal Endangered Species Act (ESA), due to population declines and range retraction allegedly as a result of habitat loss (Ure et al. 1991). The petition was rejected due to lack of data to confim long-term declining trends that would warrant listing as threatened or endangered (U.S. Fish and Wildlife Service 1992). Formal status is granted to ferruginous hawk (threatened) in the state of Washington (Richardson 1996). The ferruginous hawk is currently listed as threatened throughout its range Canada, endangered in Alberta, and threatened in Manitoba (COSEWIC 2008, Alberta Ferruginous Hawk Recovery Team 2009)

The ferruginous hawk was designated a California Species of Special Conern in 1992 but was removed from the list in 2008 as there was no documented evicence of substantial decline for this species (CDFG 1992, Shuford and Gardali 2008). A few other states consider ferruginous hawk to be a Species of Special Concern (SSC), and the U.S. Fish and Wildlife Service considers it a Bird of Conservation Concern (BCC) (Bechard and Schmutz 1995, USFWS 2008). One of the Bird Conservation Regions (BCR) where the ferruginous hawk is considered a BCC is the Great Basin (BCR), which encompasses the east side of the Sierra Nevada escarpment to the Modoc Plateau in California (USFWS 2008)

Lastly, the ferruginous hawk is considered a species of "least concern" by the International Union for Conservation of Nature (IUCN 2011). Range-wide, Nature Serve (2012) lists this species as "apparently secure-G4" but "vulnerable-S3" in California.

The ferruginous hawk typically weighs between 977 to 2,074 g, and measures 51 to 66 cm in length with a wingspan of 135 to 168 cm, with males smaller in size (Wheeler 2003). They are distinguished by a black bill, large yellow cere and gape, feathered tarsi (similar to golden eagle), thick toes, small black tips on the outer five ventral primaries (visable in flight), and moderately broad wings that taper to a point (Bent 1937, Bernard and Schmutz 1995, Wheeler 2003). Adults have uniformly white ventral tail feathers, medium grey dorsal secondaries crossed with narrow black bars, and dorsal primaries with a pale grey outer webb and white inner web. There are four different adult plumage variations including light, intermediate (rufous), dark intermediate (dark rufous), and dark, all with varying degrees of markings (Wheeler 2003). Juveniles generally have long tail feathers, making them look longer than adults (Wheeler 2003).

The maximum recorded age for a banded ferruginous hawk recovered from the wild is 23 years and 8 months (Lutmerding and Love 2011). Using mortality data from 2,444 wild banded birds in southeastern Alberta, 89% (n=80) were recovered in their first five years (Schmutz and Fyfe 1987),.Mortality rates for first year and adults are estimated around 65% and 25% (Schmutz and Fyfe 1987, Woffiden and Murphy 1989). Both mortality rates could be overestimates, when trying to determine average life expectancy of an individual hawk (Bechard and Schmutz 1995).

When used for falconry, the ferruginous hawk captures prey from small birds to rabbit-sized mammals (Glasier 1978). Being an open rangeland specialist, individuals used for falconry fly poorly in enclosed areas, an unnatural habitat (Peeters and Peeters 2005). Millsap and Allen (2006) found it is one of the least commonly used falconry species, probably for that purpose, in North America There are only a few records of harvest in California, possibly due to the scarcity of breeding pairs within the state or the difficulty in training (CDFG 2012, J. Alleveneda, falconer, personal communication). From 2006 to 2010, one ferruginous hawk nestling was harvested (CDFG 2012). Two other records exist for wild take of nestlings, one in 1996 and one in 2003 (CDFG 2012).

Habitat

The ferruginous hawk prefers expansive open rangeland, consisting of prairies and shrubsteppe, where high abundances of ground squirrels exist (Grinnell and Wythe 1927, Grinnell and Miller 1944, Schmutz 1989, Peeters and Peeters 2005). In California, it is generally a bird of the interior valleys, plains, foothills, ranches, some moderate agriculture (e.g. pastures), select coastal prairies, and desert edges (Small 1994, Peeters and Peeters 2005). Grasslands, sagebrush (*Artemisia spp.*), salt bush (*Atriplex spp.*), desert scrub with creosote bush (*Larrea tridentata*), pinyon-juniper forest edges, open riparian forest, and open woodland edges are all important habitat components for foraging and nesting (Bent 1937, Cotrell 1981, Ryser 1985, Palmer and Smith 1988, Roth and Marzluff 1989, Watson 2003, Unitt 2004, Peeters and Peeters 2005). Dense forest, narrow canyons, high elevation mountain ranges, pure desert scrub, urban development, and heavy agriculture are generally avoided (Palmer and Smith 1988, Bechard et al. 1990, Berry et al. 1998, Schmutz 1984, 1987, 1989, Small 1994, Pandolfino et al. 2011).

Habitat requirements are very similar between the non-breeding and breeding seasons (Bernard and Scmutz 1995). One caveat, the majority of the ferruginous hawk nesting habitat is in the Great Plains, and limited breeding in California has resulted in a lack of research (Peeters and Peeters 2005). Generally the preferred nesting substrates are large, isolated trees within small groves of trees that are in close proximity to foraging habitat (Fisher 1893, Bent 1937, Ryser 1985, Olendorff 1993). While Bent (1937) noticed no particular preference in tree species, others found that juniper trees (*Juniperus spp.*) are the most commonly used tree, especially at the edge of juniper forest/shrubsteppe vegetation in the Great Basin (Ryser 1985, Olendorff 1993, Collins and Reynolds 2005). Ferruginous hawks will nest in other sites as well, such as buttes, hillsides, cliffs, pinnacle outcroppings, cutbanks, boulders, creek banks, knolls, large shrubs, abandoned buildings (roof, chimney, etc.), haystacks, straw piles, old wind mills, utility poles, power-line structures, and on the ground (Fisher 1893, Bent 1937, Ryser 1985, Palmer and Smith 1988, Roth and Marzluff 1989, Olendorff 1993, Bernard and Schmutz 1995). Nest height is usually between 2 and 8 meters (6.5 to 26 ft) high, and range from ground level to greater than 20 m (65.6 ft) (Bechard et al. 1990, Restani 1991, Bechard and Schmutz 1995). Prey availability and abundance may be a more important determining factor than vegetation regarding nest site selection (Bent 1937, Cook et al. 2003).

The broad connection of expansive prairies and shrubsteppe landscapes appear to be important to ferruginous hawk (Schmutz and Fyfe 1987). Winter observational counts, such as the Christmas Bird Count, have noted the highest numbers of ferruginous hawks in open habitat, while dense coniferous forest and deciduous woodland show lower numbers of detections (Schmutz and Fyfe 1987, Garrison 1990, Bechard and Schmutz 1995). In a study on winter habitat use in California's Central Valley, the ferruginous hawk exhibited a strong association with grasslands, while avoiding urban areas, row crops, rice fields, vineyards, and orchards (Pandolfino et al 2011). Grazed grasslands showed significantly higher abundances of ferruginous hawks compared to un-grazed grassland (Pandolfino et al. 2011). In a study of the California ground squirrel, Lenihan (2007) only observed ferruginous hawks hunting at grazed squirrel colony sites, and never observed a ferruginous hawk offsite. In the Great Plains, another ecosystem engineer that modifies grasslands and supplies ferruginous hawk with a critical prey base are prairie dogs (Schmutz and Fyfe 1987). Both the prairie dog and ground squirrel are integral in the health of grassland and prairie habitats, in which ferruginous hawk specializes (Lenihan 2007, Kotliar 2000, Giovanni et al. 2007, Pandolfino et al. 2011). Human-altered fragmentation of the landscape may be tolerated, as long as the preferred prey sources remain available (Plumpton and Anderson 1997, 1998).

During the non-nesting season, ferruginous hawks have been found to roost communally in San Diego (Lake Henshaw), San Bernardino (Harper Dry Lake), and elsewhere (Steenhof 1984, King et al. 1988, Peeters and Peeters 2005).

Migration

Populations of ferruginous hawk breeding in Canada and the northernmost portions of the United States are completely migratory, while patterns for the remaining populations are largely unknown (Bechard and Schmutz 1995). Specific migration routes in California have not been studied in detail, and what is known regarding seasonal movements comes from opportunistic observations, annual surveys, and various bird atlas accounts. Early observations in California noted an influx of fall migrants in October, followed by an emigration in March (Grinnell and Miller 1944). More recent observations suggest ferruginous hawks arrive mid-September and leave California in early April (Small 1994, Peeters and Peeters 2005, Figure 6). Seasonal movements are probably tied to availability and abundance of prey, such as California ground squirrel (Garrett and Dunn 1981, Wheeler 2003).

Band return and telemetry evidence suggests migrants in California come from northern populations in Oregon and Washington and/or from populations east of the Continental Divide/Great Plains (Garrison 1990, Gossett 1993, Watson 2003). Winter band recoveries of ferruginous hawks in California from 1939 to 1986 are few (n=15) but reveal that migrants from both subpopulations on the eastern and western sides of the

Rocky Mountains travel to California, with most coming from west of the Rockies (Garrison 1990).

More recently, satellite telemetry was used to track the migrations/movements of ferruginous hawks from Washington (Watson 2003). While adult migration routes spent time at fall use areas, adults spent 74% of the total time away from breeding territories at winter use areas in California's Central Valley (Watson 2003). Adults traveled an average of 3882 ± 336 km during a year's migration and average 179 ± 16 days away from nesting territories (Watson 2003). Adult ferruginous hawks showed a high degree of winter range philopatry, and juveniles may learn migration routes by following and communally roosting with adults (Watson 2003). All winter use areas in California had a high abundance of California ground squirrels (Watson 2003).

Movements of California's small breeding population are unknown at this time.

The Golden Gate Raptor Observatory (GGRO) at the Marin Headlands in California has only banded two individual ferruginous hawks since 1983 (Fish and Hull 2010). However, observational records by the GGRO tracked long-term migration patterns along the coast, despite low levels of detection (Figure 1, Elliot and Fish 2010).

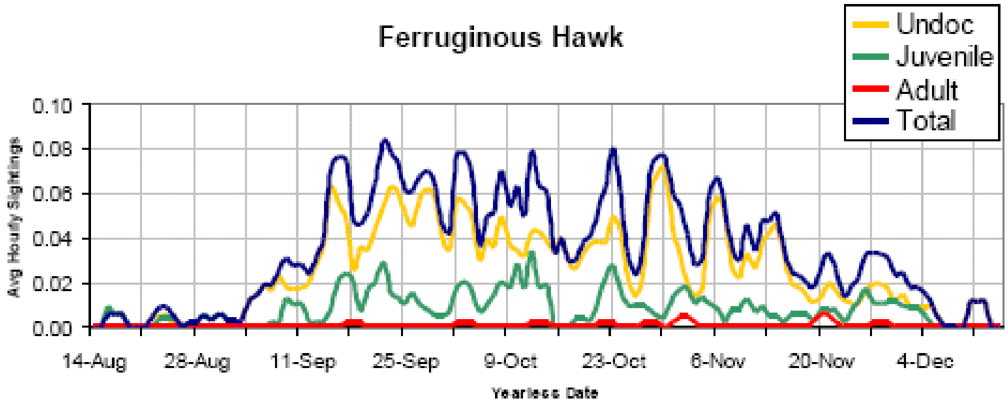

Figure 1. Migration Profile at the Marin Headlands, California, from 1986 to 2009 (Elliot and Fish 2010)

Food Habits

Ferruginous hawk is a fierce predator adapted to select a small niche of squirrels and rabbits (95% of diet biomass) that inhabit the grassland and shrub-steppe ecosystems of western North America (Cottrell 1981, Steenhof and Kochert 1985, Olendorff 1993, Bechard and Schmutz 1995, Giovanni et al. 2007, Schmutz et al. 2008). Prairie dogs and ground squirrels are critical east of the Continental Divide; whereas ground squirrels, jackrabbits, and cottontail rabbits are a critical component to ferruginous hawk diets west of the Continental Divide (Olendorff 1993, Bechard and Schmutz 1995, Peeters and Peeters 2005). Historical records report ground squirrels as the main prey item in California (Cooper 1870, Grinnell and Storer 1924, Bent 1937, Grinnell and Wythe 1927, Grinnell and Miller 1944). Belding's ground squirrel (*Urocitellus beldingi*) is documented as prey for breeding individuals in Oregon (Cottrell 1981). Other small to medium-sized mammals (pocket gophers, kangaroo rats, mice, etc.), birds, reptiles, amphibians, and Orthopteran insects also comprise smaller percentages of the overall diet biomass across the range of the ferruginous hawk (Palmer and Smith 1988, Olendorff 1993).

Ferruginous hawk employs a few different hunting strategies in the pursuit of prey items, relying upon vast landscapes devoid of tree cover and dominated by short vegetation (Peeters and Peeters 2005). The ferruginous hawk spends most of its foraging time scanning the open terrain from a perch or on the ground (Ryser 1985, Palmer and Smith 1988, Wheeler 2003). Wait perches consist of trees, rock outcrops, or some other elevated location with a good view of the landscape (Palmer and Smith1988). Most prey capture results from a relatively short flight, less than 100 m (328 ft) from the perch (Wakeley 1974). Another commonly used hunting strategy is ground hunting; initially, the hawk scans for active mammal burrows (ground squirrel, prairie dog, rabbit, gopher, etc.) aerially, followed by waiting at the selected prey's burrow entrance (Wakeley 1974, Palmer and Smith 1988). In California, Coues (1874) noted ferruginous hawks were almost always found in the vicinity of California ground squirrel colonies, waiting at the burrow entrance for prey to emerge. Strategies used less frequently consist of low level flight (4.6 to 15.2 m; 15 to 50 ft) in close proximity to hillsides and slopes, and soaring and hovering in strong winds (Wakeley 1974, Palmer and Smith1988, Peeters and Peeters 2005).

Throughout the entire range, trends in ferruginous hawk distribution and abundance are shown to be highly correlated to prey abundance and availability (Schmutz and Hungle 1989, Woffinden and Murphy 1989, Schmutz et al. 2008). Biological responses to fluctuations in abundance of preferred small mammal prey populations (i.e. ground squirrels, prairie dogs, rabbits) are complex, and influenced by many biotic and abiotic factors (Schmutz et al. 2008). This relationship becomes especially important when linked to nesting productivity (Schmutz et al. 2008). In Utah, a population of ferruginous hawk almost became extirpated after declines in black-tailed jackrabbit, due to limited availability of other prey sources (Woffinden and Murphy 1989). When the jackrabbit population recovered, response in hawk productivity was extremely slow in recovery (Woffinden and Murphy 1989). A study in Canada documented ferruginous hawk occupancy and clutch size closely related to increased squirrel survival during winter, and hawk nestling survival may have been dependent on abundance of young squirrels (Schmutz et al. 2008). Variations in prey abundance and availability drastically impact the number of occupied nests and the number of young fledged (Peeters and Peeter 2005).

Breeding

There are many studies regarding the breeding phenology and behavior of the ferruginous hawk, however data are lacking for California (Dechant et al. 2002). Initiation of the breeding season starts at the arrival at nesting sites, sometime in late-February through late-March, and proceeds through mid-June to late-August (Wheeler 2003). Courtship includes "high-soaring", "leg-dangling", "cartwheeling", "sky dancing", and "high perching" (Wheeler 2003). During courtship, pairs select a nest site within a breeding territory (Wheeler 2003, Collins and Reynolds 2005). Lone trees are used preferably for nesting, especially in the pinyon-juniper and shrub/steppe ecotone bordering open valleys with plentiful grasslands (Ryser 1985, Olendorff 1993, Sawyer et al. 2008). Juniper species are used for nesting in the Modoc Plateau (Ryser 1985, Sawyer et al. 2008, J. Alleveneda, personal communication). Ferruginous hawk can be opportunistic in use of other nesting locations, but they tend to select sites inaccessible to human disturbance and agricultural cultivation (Gilmer and Stewart 1983, Roth and Marzluff 1989, Collins and Reynolds 2005). Nesting within 8km (24ft) to good foraging habitat is also integral to selecting a nest site (Roth and Marzluff 1989).

After a nest site is selected, a ferruginous hawk pair may either build a nest or refurbish an old nest (Wheeler 2003). The abandoned nests of red-tailed hawk (*Buteo jamaicensis*), Swainson's hawk (*Buteo swainsoni*), and black-billed magpie (*Pica hudsonia*) may be utilized (Wheeler 2003). Various components are used from the surrounding habitat, including sticks, bones, shrub stems, grasses, moss, debris, cow dung, and soft material for the lining (Heerman 1859 in Cooper 1870, Fisher 1893, Bent 1937, Bernard and Schmutz 1995). Historically bones and wool from bison were integral components of the ferruginous hawk nest substrate and the prairie ecosystem, and likely reflects this hawks habitat niche (Bendire 1892, Fisher 1893, Cameron 1914, Bechard and Schmutz 1995); but bison had already severely declined in distribution and abundance by the late 1800s (Hornaday 1890). Interestingly, the historic range of the bison included the same open plains of the Modoc Plateau where a few ferruginous hawks are found to breed in California (Merriam 1924). Today, livestock and pronghorn antelope (*Antilocarpa americana*) bones are commonly used in nest construction (Ryser 1985). Nests are large, averaging about 2 to 4 feet in diameter, considerably deep, and comparable in size to golden eagle nests (Bent 1937, Wheeler 2003). Ferruginous hawk nests may be reused for many years, often by the same pair, and may become massive towers over 4 m (12 ft) tall (Palmer and Smith 1988, Bechard and Schmutz 1995, Peeters and Peeters 2005). One nest in Saskatchewan fledged juveniles for thirty two years in a row, by a succession of different pairs (Wheeler 2003).

Eggs are laid from March to May; depending on latitude, weather, and the presence of prey (Palmer and Smith 1988, Olendorff 1993). The average size of a ferruginous hawk clutch is usually between 2 and 4 eggs, however anywhere from 1 to 5 eggs are laid, and rarely up to 8 eggs (Bent 1937, Smith et al. 1981, Palmer and Smith 1988, Wheeler 2003, Peeters and Peeters 2005). Both males and females incubate. Eggs hatch after 32 to 33 days, coinciding with the availability of prey such as ground squirrels (Schmutz et al. 1980, Palmer and Smith 1988, Olendorff 1993, Wheeler 2003). Nestling activity increases at 23 days old; they are more aggressive towards food items and begin wing flapping behavior at that age (Powers 1981). Fledging occurs at 38 to 50 days, but the fledglings stay close to the nest territory (Powers 1981, Wheeler 2003). After about a month of practicing hunting skills, young ferruginous hawks finally leave the nesting territory and are no longer dependent on the adults for survival (Blair and Schitoskey 1982).

During the nesting season, ferruginous hawks defend their territories and nestlings aggressively from predators, including but not limited to: great horned owls (*Bubo virginianus*), crows and ravens (*Corvus spp.*), coyotes (*Canis latrans*), badgers (*Taxidea taxus*), foxes (*Vuples spp.*), and humans (*Homo sapiens sapiens*) (Bechard and Schmutz 1995, Keeley and Bechard 2011). Ferruginous hawks defend their nest territory with similar intensity as shown by red-tailed hawks, with females behaving more aggressively than males (Keeley and Bechard 2011). This parental aggression decreases with nestling age and the distance of the intruder is from a nest (Keeley and Bechard 2011).

Home Range

While studies of ferruginous hawk home range have not been conducted in California, there have been some studies in other western states, including Washington, Idaho, Montana, Oregon, Utah, and Wyoming. The size of the home range depends on habitat quality, and the distribution and abundance of prey populations (Leary et al. 1998). A few studies estimated home range size indicated areas between 3.4 km^2 (1.3 mi^2) and 21.7km^2 (8.4 mi^2) (Smith and Murphy 1973, Platt 1984, Janes 1985, McAnnis 1990, Harmata 1991). Leary et al. (1998) found home range size in Washington to average 90.3 km^2 (34.9 mi^2), using radio telemetry, possibly indicating ferruginous hawks will expand foraging range when prey abundance is low. While sample size was low (n=6), using the minimum convex polygon (MCP) method, southeastern Washington home ranges found by Leary et al. (1998) were significantly larger than previous radio-telemetry studies (Platt 1984, McAnnis 1990, Harmata 1991). The other reported home range studies did not use radio-telemetry (Smith and Murphy 1973, Wakeley 1978, Janes 1985). There have been no studies of ferruginous hawk home range since Leary et al. (1998). Numerous studies of ferruginous hawk found that the average distance between nests was 3.4 km (range 0.8-7.2km) (*in* Olendorff 1993).

Data on winter home range size are lacking as well, but hunting territories during the winter may expand to a much larger area (Smith and Murphy 1973). Up to 24 birds have been found communally roosting during winter in California (*in* Olendorff 1993). Bechard and Schmutz (1995) suggest individuals may roost less than one meter (3.3 ft) apart.

Historic and Current Distribution

Grinnell and Miller (1944) described the range of the ferruginous hawk during the non-breeding season as occurring from the border of Mexico (mainly western half of southern California), north throughout the Central Valley and the larger rangelands in the foothills, plains, and valleys surrounding the Central Valley north to the Oregon border (Grinnell 1915, Grinnell and Storer 1924, Grinnell and Miller 1944). On the coastal slope,, ferruginous hawks ranged from the southern California pacific slope in San Diego north to the greater San Francisco Bay area, and the Santa Rosa Plain, Sonoma County (Willet 1912, Grinnell 1915, Grinnell and Wythe 1927, Willett 1933). The species was not recorded in the dense forests and woodlands of northwestern California (Grinnell and Miller 1944). It was recorded on the east side of the Sierra Nevada, from Death Valley north to the Modoc Plateau, Modoc County (Fisher et al. 1893, Grinnell 1915, Mailliard 1927, Gilman 1935, Bond 1939, Grinnell and Miller 1944). The species was recorded between 1885 and 1887 on the Farallon Islands (Bryant 1888).

Cooper (1870) noted the ferruginous hawk “abounds in spring and fall in the southwestern parts of California, migrating in summer through the interior plains at least as far north [to the Columbia River in Oregon]”.

Historic records of ferruginous hawk occurrences in California during the summer months (breeding season) are scarce. Grinnell (1898) noted that in Los Angeles County, “a few undoubtedly breed, though I have failed to learn of any authentic nesting data.” Mailliard (1924) documented some individual ferruginous hawk records in the extreme northeastern corner of the Modoc Plateau during the breeding season; suggesting they may nest there. Bent (1937) reported the breeding range of the ferruginous hawk as including the Central California near the Cosumnes River and Jess Valley, Modoc County. Grinnell and Miller (1944), however, regarded the Central Valley nesting record near the Cosumnes River (Heerman 1859, Bent 1937) as “not altogether trustworthy” and “dubious.” Heerman (1859) claimed in 1851: “While climbing a tree to examine some magpie’s nest, the hawk in flying from her own betrayed her retreat. It was placed in the centre of a bunch of mistletoe spring from the forks of the oak, and was composed of course twigs, lined with grasses and moss; the eggs, two in number being white, marked with faint brown dashes.”

Currently, ferruginous hawks are found throughout the state from mid-September to early April, and have been recorded in every county (Hunting 1998, Peeters and Peeters 2005). In southern California, notable local concentrations of ferruginous hawks during the winter months have been found at the Carrizo Plain, San Luis Obispo County, Cuyama Valley, San Luis Obispo and Santa Barbara counties, Santa Maria Valley, Santa Barbara County, the foothills surrounding the southern San Joaquin Valley, Kern County, Antelope Valley, Los Angeles and Kern counties, Lucerne Valley and Harper Dry Lake area San Bernardino County,, Perris Valley, San Jacinto Valley, and the foothills of Lake Matthews, Riverside County, Lake Henshaw and Borrego Valley, San Diego County, and the lower Colorado River Valley, San Bernardino, Riverside and Imperial counties, and parts of the Mohave Desert (Garrett and Dunn 1981, Rosenburg et al. 1991, Small 1994, Watson 2003, Unitt 2004, Peeters and Peeters 2005).

The species is found in the Central Valley, Salinas Valley (Monterey County), and other arid coast range valleys south of Mendocino County (Small 1994, Watson 2003, Pandolfino et al. 2011). In the Bay Area Region, individuals have been reported in Petaluma, Sonoma County, Stockton, San Joaquin County, Livermore Valley and Altamont Pass, Alameda County, and even some southern San Mateo agricultural valleys (Watson 2003, Peeters and Peeters 2005).In northwestern California, ferruginous hawks are regularly found each winter in scattered locations, including: Eel River bottoms, Arcata bottoms, Bear River, Cape Mendocino Ridges, Table Bluff, and the Blue Lake area, Humboldt County (Harris 1991). On the eastern side of the Sierra Nevada, ferruginous hawks winter in Hamilton Valley and Owens Valley, Inyo County and Fish Lake Valley and Hammil Valley, Mono County and north in the Modoc Pleateau (Small 1994, Garrett and Dunn 1981, Gaines 1988, Peeters and Peeters 2005). The CBC winter distribution map (Figure 5), confirms wintering ferruginous hawks likely occupy much of their widespread historical winter range described by Grinnell and Miller (1944).

Breeding in California was first documented in the Madeline Plains, Lassen County in 1987 (Harlow and Bloom 1989). Three Breeding Bird Survey (BBS) routes in Modoc and Lassen counties reported ferruginous hawks from 1990 to 2008, suggestive of breeding although no nests were documented (Figures 2, 3, and 4). Data from falconry harvest indicated one ferruginous hawk eyas was taken from a nest in 2003 and another one in 2006, both from Lassen County near the Madeline Plains (CDFG 2012).

Ferruginous hawk was also first documented breeding in Butte Valley, Siskiyou County, in 2006 (B. Woodbridge, personal communication). There are now 5 to 6 breeding sites in this area (B. Woodbridge, personal communication).

Additionally, BBS reported a ferruginous hawk near Barstow in 1983; this record was considered indicative of a small breeding population in that area, but no nest has been documented in that region (Figure 2, Dechant et al. 2002). Hammil Valley, Mono County, was reported as the site of take of an eyas ferruginous hawk in 1995 (CDFG 2012).

Using older BBS data (Figure 2), the occurrence of ferruginous hawk in the western Mojave Desert is apparent. The more recent breeding distribution maps (Figures 2 & 3) do not show ferruginous hawk occurrence in the desert region.

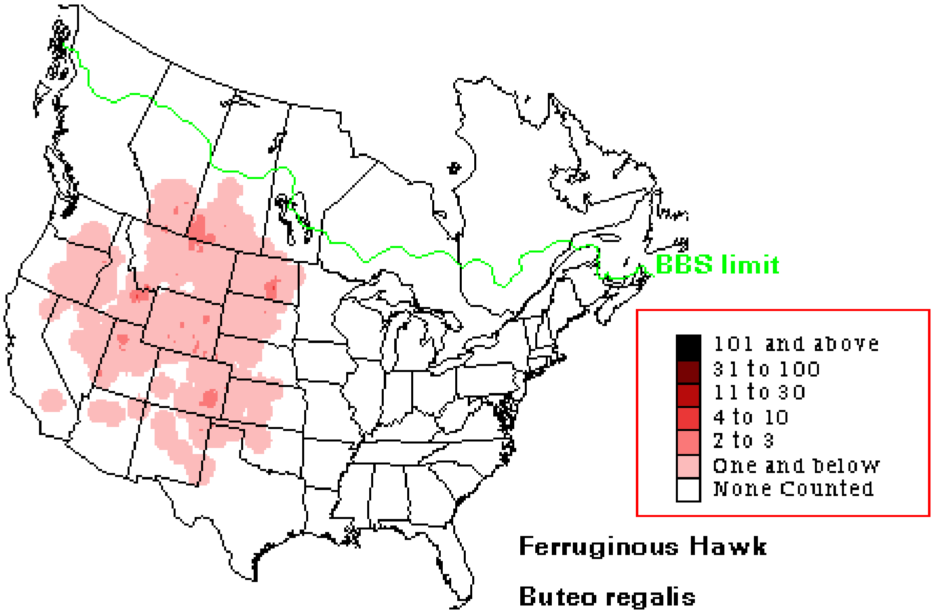

Figure 2. Ferruginous hawk summer distribution based in BBS survey data from 1966 to1996 (Sauer et al. 1997)

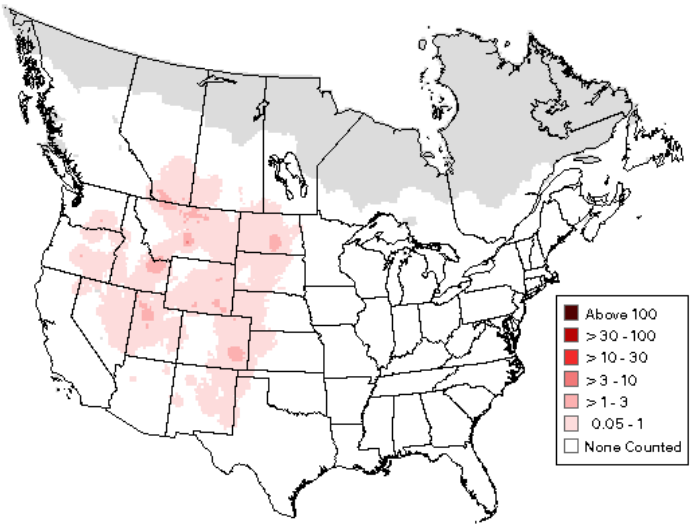

Figure 3. Ferruginous hawk summer distribution based in Breeding Bird Survey data from 1994 to 2003 (Sauer et al. 2008)

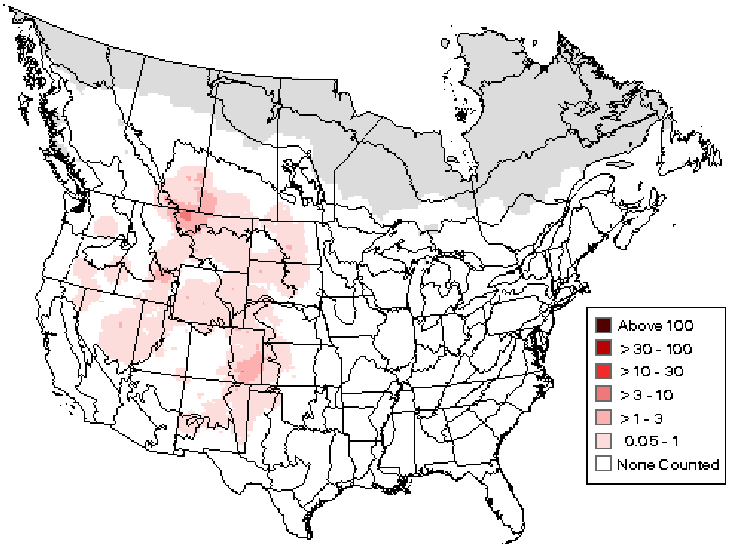

Figure 4. Ferruginous hawk summer distribution based on Breeding Bird Survey data from 2006 to 2010 (Sauer et al. 2011).

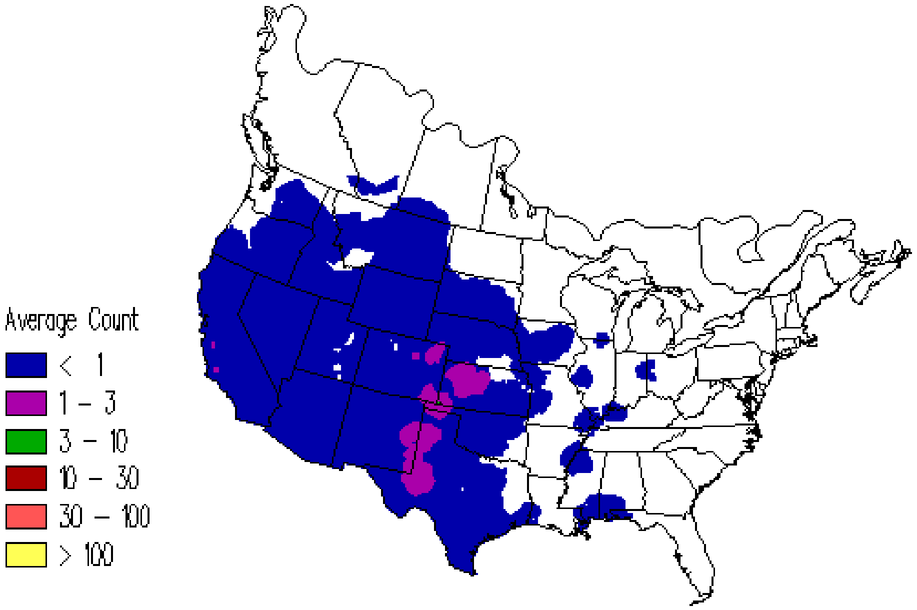

Figure 5. Ferruginous hawk winter distribution/relative abundance based in Christmas Bird Count from 1959 to 1988 (Sauer et al. 1996).

Historical and Current Abundance

Knowledge of ferruginous hawk abundance comes from the historical records compiled by Grinnell and Miller (1944) and their predecessors. The ferruginous hawk was considered “formerly common as a transient and winter visitant” throughout its range, but “notably scarce or altogether wanting in all regions from which previous reports have been made” (Grinnell 1915). Grinnell and Miller (1944) regarded ferruginous hawk as “formerly abundant” during winter; but only common locally throughout its distribution. An older report of ferruginous hawk in California stated “…it is the most characteristic and one of the most abundant species of its family” (Coues 1874).

Willet (1933) considered the ferruginous hawk to be “formerly common in fall, winter, and early spring, from the lowlands well up into the mountains; now rather rare, except in portions of San Diego and Riverside Counties.” The ferruginous hawk was a “rare and irregular winter visitant” in the San Francisco Bay area and “is, or was under original conditions, a common winter visitant to the plains of central California and to the larger open tracts in the foothill country” (Grinnell and Storer 1924, Grinnell and Wythe 1927, Bent 1937); it was “sparse” on the west side of the Sierra Nevada mountain range (Grinnell and Storer 1924). Ferruginous hawk was not historically recorded in the northwestern corner of California (Grinnell and Miller 1944).

In southern California, the ferruginous hawk is currently considered a “fairly common winter visitant” in the desert grasslands and agricultural regions and interior coastal slope valleys (Garrett and Dunn 1981). Along the coast, ferruginous hawk is considered “rare to uncommon” (Garrett and Dunn 1981, Lehman 1994, Hamilton and Willick 1996, Unitt 2004). The Santa Maria River Plain was the only coastal area Garrett and Dunn (1981) considered ferruginous is found to be “fairly common”, but later considered “uncommon” by Lehman (1994). The southern coast is the only area in California where Hunting (1998) saw declines in CBC abundance, attributed to habitat loss. “Rare to uncommon” abundance was also noted in the agricultural areas surrounding the Salton Sea and along the Colorado River (Garrett and Dunn 1981, Rosenburg et al. 1991).

In the west slope of the Sierra Nevada, ferruginous hawks are considered a “rare transient” and locally “rare to uncommon winter resident in Mariposa region” (Gaines 1988). East of the Sierra Nevada escarpment, they are an “irregularly/scarce fall transient”, but “uncommon winter residents” in the Hammil Valley (Gaines 1988). In the Modoc Plateau, “they can appear numerous in some years” (Peeters and Peeters 2005). Ferruginous hawk are now considered a “regular rare winter visitor” in river bottoms of the northwest, and was probably overlooked before the 1980s (Harris 1991).

Hunting (1998) estimated “the average California population could be between 400 and 500 individuals,” based on average of 2.5 individuals seen for all CBC circles from 1990-1997. There is no other population estimate for California’s wintering ferruginous hawk population, aside from the current CBC data.

Partner’s in Flight (PIF) has estimated California’s breeding population to be 50 individuals, comprising 0.2% of the North American population (Rich et al. 2004). Using BBS extrapolated data, the total North American population size is estimated to be 25,000 individuals by Rich et al. (2004) and 11,500 (30% juveniles, 70% adults) by Millsap and Allen (2006)..Earlier population estimations by Schmutz (1984) were 3,000 to 4,000 breeding pairs in North America; 5,842 to 11,330 individuals in North America (Olendorff 1993); and 14,000 individuals were estimated just in the Great Plains (Schmutz et al. 1992). Ferruginous hawk breeding population size is unknown in California at this time.

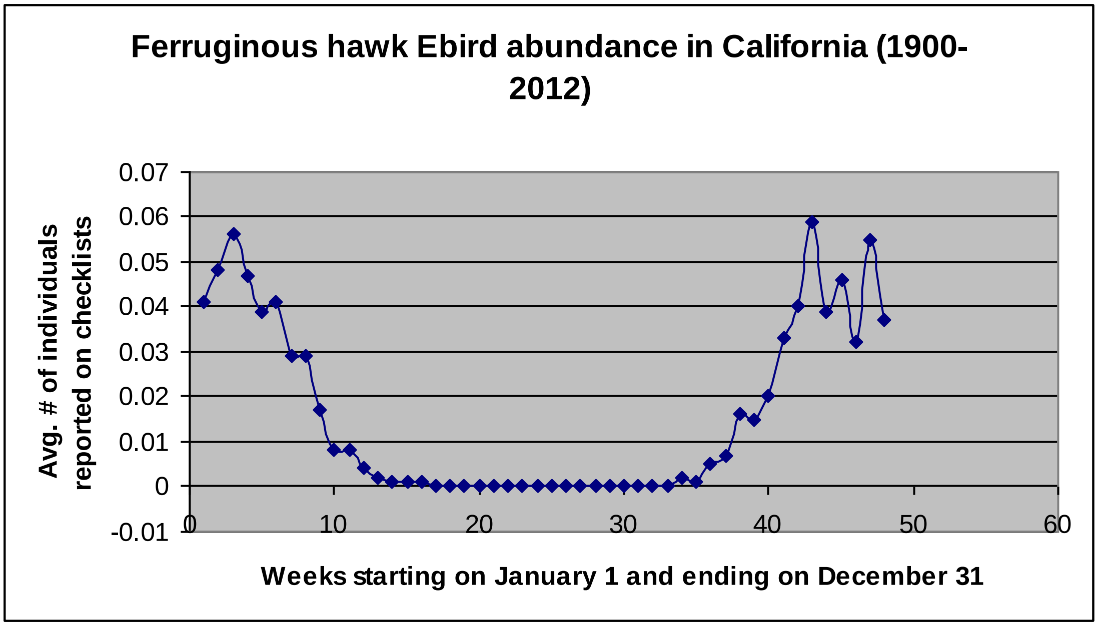

Figure 6. Ferruginous hawk abundance from the citizen science reporting database, Ebird, 2002 to 2012 (Sullivan et al. 2009).

Population Trend

Past evaluation of CBC trends regarding distribution and abundance (as recent as 1987-1988) indicated significant increases in the California wintering population of ferruginous hawk (Figure 7, Garrison 1990, Sauer et al. 1997). It should be noted that from 1951 to 1987 there was a 5-fold increase in count circles, from 20 to 100, and most of the old count circles were in urbanized areas where ferruginous hawk are scarce (Garrison 1990, Berry et al. 1998). Newer count circles were founded in rural areas with open rangeland where ferruginous hawks are more likely to be encountered (Garrison 1990). Caution should be exercised when interpreting CBC data (Garrison 1990, Hunting 1998).

Overall, winter the population may have leveled out since the late 1980s (Figure 8).From 1988 to 2010, an average of 68 CBC count circles reported a combined average of 309 ferruginous hawks per year (Figure 8). Number of ferruginous hawks observed per party hour from 1988 to 2010 was 0.040, and from 2001 to 2010 it was 0.039 (National Audubon Society 2010). No recent trend analysis of winter abundance and population size of California's ferruginous hawk in California are available.

Migration data for the ferruginous hawk are available from the Marin Headlands Golden Gate Raptor Observatory (GGRO) for the period from 1986 to 2009 (Figure 9). Sample sizes are low and sharp variations from year-to-year are evident with a range of 6 to 37 (or 0.01–0.07 yearly sightings per hour) individual sightings/year by GGRO (Figure 9).

BBS data are inadequate to determine breeding trend for the ferruginous hawk in California (Sauer et al. 2011). However, Figure 9 does show a geographically small, increasing population trend in the extreme northeastern corner of the state.

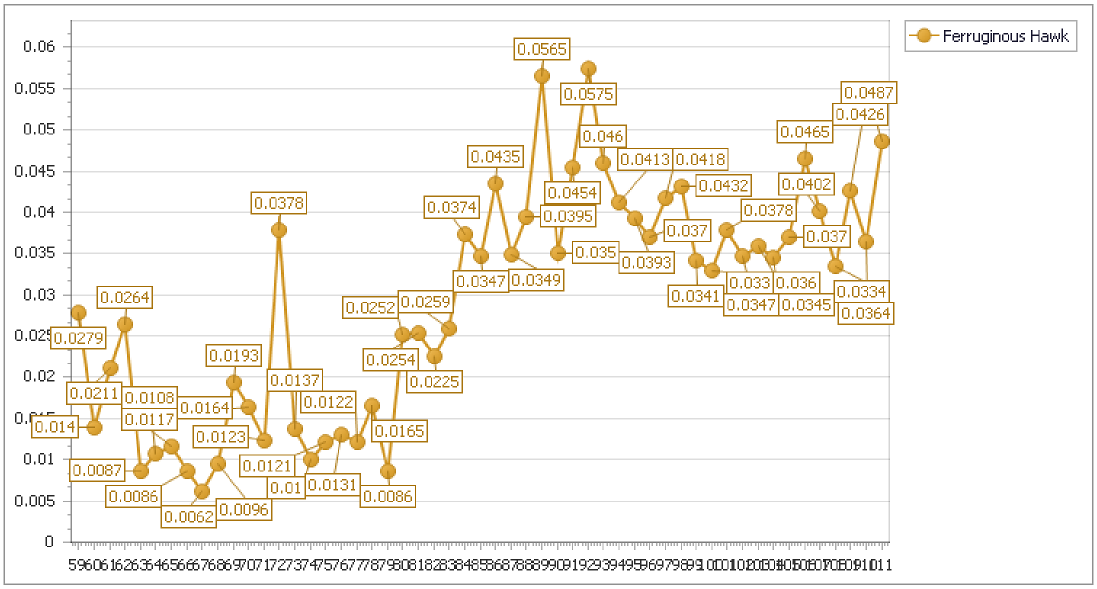

Figure 7. Ferruginous hawk observations (#/party hour) during California Christmas Bird Count, from 1958-2010 (National Audubon Society 2010).

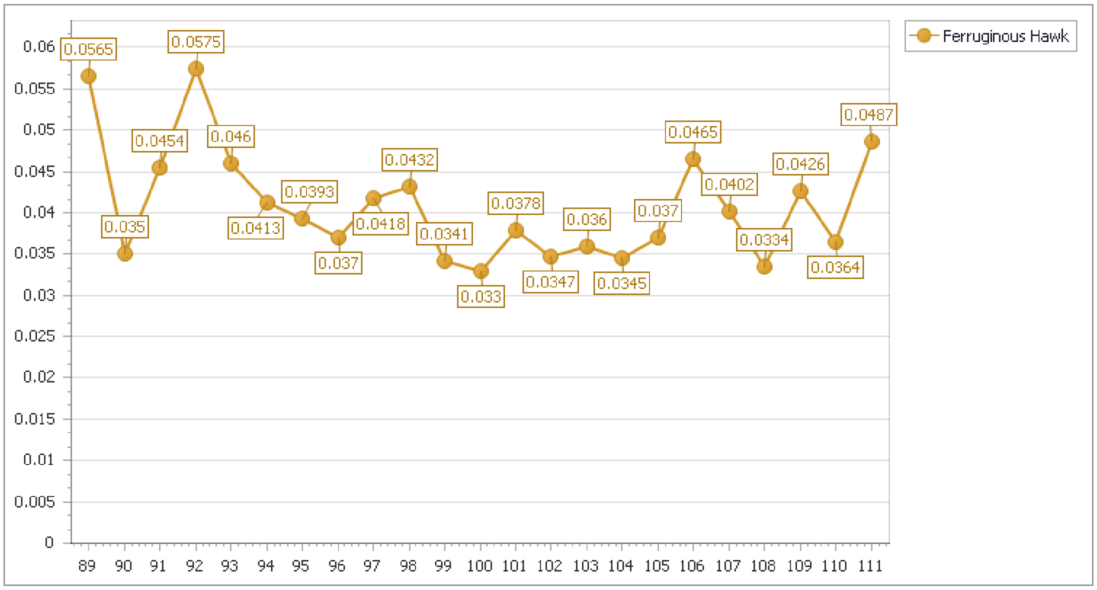

Figure 8. Ferruginous hawk observations (#/party hour) during California Christmas Bird Count, from 1988-2010 (National Audubon Society 2010).

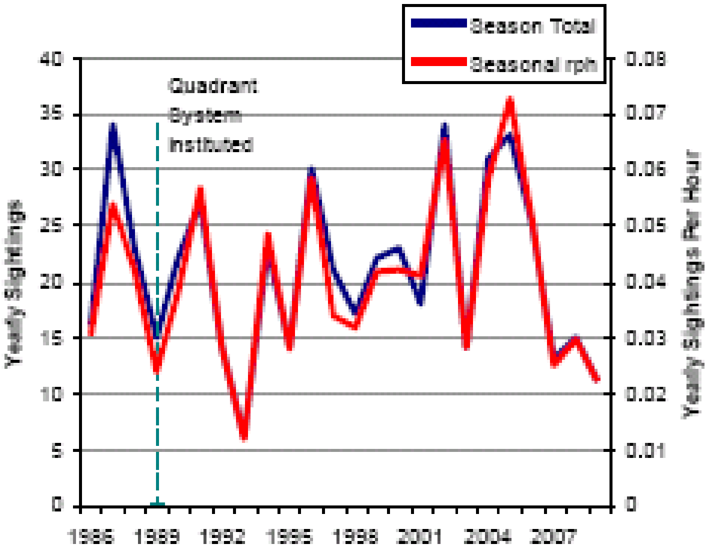

Figure 9. Yearly trend in ferruginous hawk observations at the Marin Headlands, from 1986 to 2009 (Elliot and Fish 2010)

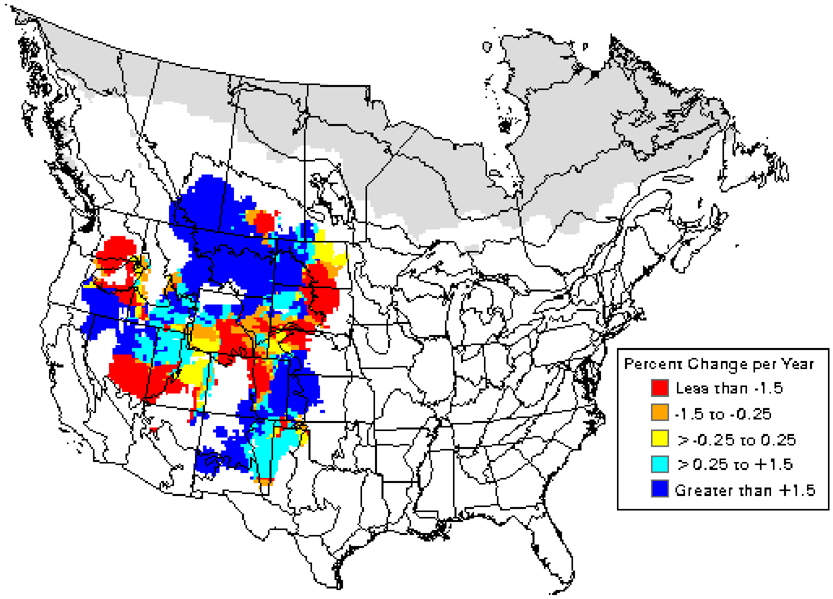

Figure 9. Breeding Bird Survey trend map for ferruginous hawks, from 1966 to 2010 (Sauer et al. 2011)

Threats

There are many threats, both natural and anthropogenic, that may potentially reduce the reproductive success and survival of ferruginous hawk. Human disturbance, direct mortality, reduction in nest site availability, reduction/extirpation of preferred prey base, and habitat loss/alteration may all have negative impacts throughout this raptors entire distribution (Olendorff 1993). For ferruginous hawks wintering in California, the single most significant threat is habitat alteration, degradation, fragmentation, and loss (Watson

2003). Conversion of native grassland and shrubteppe habitat into intensive agriculture and urban development has indirect cascading consequences, ultimately resulting in the decrease or removal of prey, winter roosting sites, and nest sites (Watson 2003, Unitt 2004, Alberta Ferruginous Hawk Recovery Team 2009). Threats to California's small breeding populations have not been previously studied, but habitat loss is probably significant.

Much of California's native grasslands (bunchgrass and other annuals), riparian forest, oak savanna, and wetlands have been replaced by agriculture, urban areas, rural subdivisions, and ranchettes (Thompson 1961, Bartolome et al. 2007, Vaghti and Greci 2007, Pandolfino et al. 2011). The pre-European landscape of California consisted of about 15 to 20 million acres of grasslands (~20% Central Valley) and a more recent estimate found less than 10% of the Central Valley grasslands remained (Huenneke 1989, CDF 2002). These grasslands are continually lost to agriculture and urban development (Ewing et al. 1988). Furthermore, more than one million acres of agricultural land have been converted to urban land use around the perimeter of cities located in California's agricultural valleys and coastal areas since 1984 (California Department of Conservation 2011). Decline in the abundance of ferruginous hawks on the coastal slope of southern California is attributed to loss of coastal prairie and grasslands (and even agriculture) to increasing urban expansion (Lehman 1994, Hunting 1998, Unitt 2004). Intense agriculture, such as vineyards and orchards, has been increasing as well (Volpe et al. 2010). A continuation of habitat loss and fragmentation will leave little wintering and breeding habitat for historically sustainable levels ferruginous hawk and their prey.

The California ground squirrel was considered the most serious pest to California rangelands and agricultural lands since the Spanish mission era (Marsh 1998). Ground squirrel control efforts may have repercussions on ferruginous hawk populations. Secondary poisoning of ground squirrels via bioaccumulation of rodenticides, strychnine, and other forms of small mammal poisons has potential direct impacts on ferruginous hawk reproduction and survival (Olendorff 1993, Lima and Salmon 2010, Proulx 2011). Indirect effects to ferruginous hawks include a loss of prey base, which may result in local extirpations of breeding birds, and loss of sutiable foraging habitat (Schmutz and Hungle 1989, Woffinden and Murphy1989, Bak et al. 2001, Cook et al. 2003, Unitt 2004, Schmutz et al. 2008). Conversion of traditional winter grasslands in the southern California coastal prairie, the Central Valley, and other interior valleys has resulted in loss of ground squirrel colonies, and therefore historical foraging habitat (Hunting 1998, Watson 2003). Maintaining abundant prey populations (i.e. California ground squirrels) in remaining foraging habitat is important to California's small breeding population and for individuals with winter ranges in California (Suter and Jones 1981, Watson 2003).

While ferruginous hawk do not have a positive association with orchards, row crops, vineyards, and alfalfa fields; they do occur in these areas of California's Central Valley (Pandolfino et al. 2011). Winter ferruginous hawk may be exposed to dormant organophosphates and other pesticides used in valley agriculture (Hooper et al. 1989, Hosea 1996). The same may apply to breeding ferruginous hawks using agricultural lands. However, there are no studies linking pesticide poisoning to mortalities of ferruginous hawk.

Alteration of fire regime may adversely impact plant community structure and composition, and ultimately ferruginous hawk breeding habitat. Fire suppression, in combination with the introduction of livestock, has lead to dense stands of juniper tree encroachment into the valley bottoms of the Modoc Plateau, outcompeting grassland and sagebrush vegetation (Young et al. 2007, Sawyer et al. 2008). Stands of ponderosa pine are becoming extensive and are mixing with the juniper stands (Riegel et al. 2006). Alterations in fire regime may also lead to conversion of shrub-steppe habitat into non-native annual grasslands (e.g., cheatgrass, medusahead), which do not supply an adequate food supply for the prey of ferruginous hawk (Olendorff 1993, Collins and Reynolds 2005, Young et al. 2007). Fire was and is an important factor in the landscape of northeastern California (Riegel et al. 2006).

While nest site disturbance has not been studied in California, it has been studied in other states. Reproductive success may be impacted by causing adults to leave the nest during the sensitive egg incubation or early nestling phase, or cause premature fledgling (White and Thurow 1985, Roth and Marzluff 1989, Bechard et al. 1990, Collins and Reynolds 2005). Sensitivity to disturbance may be magnified during periods of low prey availability and in urban areas where people recreate (e.g. hiking, mountain biking, dog walking, off-road vehicle use), or conduct agricultural activities close to nest sites (White and Thurow 1985, Berry et al. 1998, Ward 2001, Keeley and Bechard 2011). Recent estimates recommend a spatial buffer of 650 m or greater around nest sites, which should prevent 95% of ferruginous hawks from flushing to protect their eggs or nestlings (Keeley and Bechard 2011). Continuous disturbances (such as mineral, gas, coal, geothermal, and other construction activities) should allow for greater than a 1 km buffer (Suter and Joness 1981, Smith et al. 2007).

Historically shooting, trapping, and general persecution has been a threat to the species (Couse 1874, Bendire 1892, Merriam 1924, Bent 1937). Shooting may still be a recurring problem today, with at least one band recovery reported due to shooting in California since the passage of the Migratory Bird Treaty Act in 1972 (Gilmer et al. 1985, Gosset 1993). Direct collision with power lines (electrocution) and automobiles are another source of mortality, especially in southern California where human population density is high (Harmata et al. 2001).

A new emerging threat to ferruginous hawk is renewable energy. While estimates of direct mortality with turbine blades are considerably smaller than other raptors (red-tailed hawk, American kestrel, burrowing owl, golden eagle), Smallwood (2010) estimated between 2.8 (range -0.6 to 6.1) and 4.7 (range -1.0 to 10.3) mortalities per year at Altamont Pass Wind Resources Area (APWRA). Ferruginous hawks that winter near APWRA, have been observed to fly more often ≤50 m from wind turbines, when compared to all other bird species (Smallwood et al. 2009). Wind energy in other regions, including the desert, may have cumulative impacts greater than reported from APWRA. Increasing solar power developments may also pose a threat to ferruginous hawks inhabiting the California deserts, but in the form of habitat loss and fragmentation (Leitner 2009).

Data Gaps and Monitoring Needs

Indirect and direct effects of habitat loss is one of the more concerning risks to this species, as the habitat ferruginous hawk evolved in also serves as prime land for agricultural, urban, and renewable energy development. Data gathered on ferruginous hawks occupying California may be important in assisting land managers in land use decision making. This is especially important considering the ferruginous hawk is threatened in Canada and the state of Washington, due to drastic declines in distribution and abundance as a breeder, and many of these threatened birds may spend their winter in California (Salt 1939, Garrison 1990, Harmata et al. 2001, Watson 2003, COSEWIC 2008). Protection of these threatened migratory populations will require interstate and international participation in monitoring (Watson 2003).

Monitoring of the small breeding population in the extreme northern and northeastern California should be a high priority for future research in order to determine the breeding population size, home range size, habitat use, concentration, productivity, site fidelity, predator-prey interactions, and the significance of various threats. Further investigation of potential breeding populations in the western Mojave Desert and the Hammil Valley may be warranted. Understanding nest-site availability is important in determining how this factor may be limiting distribution, and if artificial nesting platforms would be beneficial substitutes in areas where substrate is lacking, similar to the nest-box for American kestrels (*Falco sparverius*) (Bloom and Hawks 1983, Schmutz et al. 1988). Other limiting factors, such as prey base, should be monitored where ferruginous hawks breed. Overall, the breeding ferruginous hawks in California may be a small population vulnerable to extirpation, therefore any and all information dealing with their ecology is important.

While wintering ferruginous hawk abundance is probably well monitored by CBC, species-specific studies and in depth monitoring is warranted. Identification, monitoring, and research of key wintering locales throughout the state, home range size, and population dynamics are integral steps in understanding the ecology of this species (Hunting 1998). Determining all source populations of winter visitors and limiting factors that affect their habitat use is also important (Hunting 1998).

Understanding the seasonal movements to and from California is warranted for both breeding and wintering ferruginous hawks. Satellite and radio telemetry, and banding studies, will help determine year-round behavior. Satellite telemetry studies of breeding adults and dispersing juveniles from Washington’s threatened population have shown the Central Valley of California to be a critical winter region, where preferred habitat contains abundant prey (Watson 2003). Telemetry data would ultimately be integral in determining the locations of habitats that should be conserved for ferruginous hawks, thus benefiting a plethora of other grassland/shrub-steppe species. Where key foraging areas have been solidified as integral to ferruginous hawk habitat use, monitoring of prey base should be important.

Monitoring the direct and indirect consequences of wind and solar energy development near vital breeding and wintering ferruginous hawk areas will also be increasingly important. Winter use areas in California have been reported near wind farms, such as Altamont Pass Wind Resource Area (Watson 2003, Peeters and Peeters 2005, Smallwood and Thelander 2008, Smallwood et al. 2009, Smallwood et al. 2010). Monitoring current renewable energy facilities will yield vital data for future project placement and help to reduce negative impacts that lead to lower survivorship, occupancy, and/or local extirpation.

Lastly, trying to understand and prepare for the future effects of climate change on all aspects of ferruginous hawk ecology is vital to preserving this predator of the grasslands, considering this is a guild of birds undergoing concurrent declines across the continent. A transboundary approach, across the three North American countries, is needed for developing a uniform protocol to monitoring, and for directions of future research with regards to the conservation of ferruginous hawk and its preferred habitat (Commission for Environmental Cooperation 2005). Currently, there is a tri-national approach to determine migratory behavior and winter range (Watson and Banasch 2005). California may play a vital role in helping to determine impacts to various breeding populations (from both inside and outside the state) and contribute to the evaluation of the metapopulation status of the species (Watson 2003). Ferruginous hawk may ultimately serve as an indicator species for grassland/shrub-steppe ecosystem health.

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ECOLOGY OF THE GREAT HORNED OWL IN CALIFORNIA

By Madeleine Wieland

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The great horned owl (*Bubo virginianus*) is a large owl found throughout North America and much of South America. This species has been nicknamed the “Hoot Owl” for its distinctive call, “hoo-hoo-hoo”, heard from the end of fall through spring. This species is also recognizable by its large body size and prominent ear tufts. The great horned owl is highly adaptable to most habitats and can be found in almost any ecoregion, with the exception of artic-alpine. It has one of the broadest diets of any raptor, along with wide nest site variation, and is remarkably adaptable to human disturbance and habitat change (Baumgartner 1938, Marti and Kochert 1996, Rohner 1997).

The International Union for the Conservation of Nature (IUCN) described the great horned owl as species of least concern on the 2009 Red List due to its large range and high, stable population numbers. It is not listed as threatened or endangered under either the federal Endangered Species Act (ESA) or the California Endangered Species Act (CESA), or identified as a sensitive species. Though not considered to be highly migratory, this species is protected by the Migratory Bird Treaty Act (MBTA) and it is unlawful to capture, kill or possess this owl or any of its parts including feathers, nests and eggs without the proper permits for research, education or falconry purposes.

The great horned owl is one of the largest owls in North America measuring 46 to 63 centimeters (18 to 25 inches) in length and weighing 0.91 to 2.5 kilograms (2.0 to 5.5 pounds) (Houston et al. 1998). Like many birds of prey, the females are generally larger than males but there are no significant differences between sexes in plumage and coloration. This bird is typically mottled or lined brown, orange and black with a distinct white patch on the throat. Large, yellow eyes are set in a brown or orange facial disc outlined in black. The “horns” which give the great horned owl its name are actually feathers, known as ear tufts, which can stand upright or lie flat on the head. The exact purpose of these tufts is unknown but current theories suggest they may help camouflage owls sleeping in trees during the day (Perrone 1981). With its horns extended, an owl resembles a broken tree stub and is much harder to detect.

Newly hatched birds are completely white and slowly gain darker feathers as they get older. Juvenile birds look much like adults but can be distinguished by some remaining white or grey in contour feathers that have not finished developing along with a shorter tail and smaller ear tufts (Peeters 2007). Neither juveniles nor adults show much variation between sexes. Like most raptors, females are often larger than the males but in great horned owls the difference is not very pronounced. The great grey owl (*Strix nebulosa*) is similar in size and color but it lacks ear tufts and has smaller eyes. The long eared owl (*Asio otus*) has long ear tufts, but is much smaller in size.

The current longevity of this species has not been well researched. In one study (Stewart 1969), only 5% of the owls banded lived past 8 years old. However, with adequate food and protection great horned owls have been known to live for more than 25 years in captivity (Bent 1938).

From 2006 to 2010, 13 great horned owls were legally captured from the wild for falconry purposes: 1 adult, 5 immature, 7 nestlings. This harvest was distributed mostly through central and southern California counties. During this same period, three owls were released back into the wild, and 3 escaped. No falconry related deaths were reported during 2006 to 2010 (CDFG 2012).

Habitat

The great horned owl can occupy and forage in most habitat types, but they prefer to live in fragmented areas with both wooded and open habitat (Johnson 1993). This territorial owl often makes its home near fields and forest edges with ample foraging habitat within its defended territory (Morrell and Yahner 1994). Smith et al. (1999) suggested that fragmented habitat may be preferred because it provides greater visibility of the surrounding area for nest defense than thick forest would. Open habitats also provide easy access to the nest and increased sunlight for the owlets as they develop (Smith et al.1999).

Great horned owls living in California can be found in a wide range of habitats. In Mono county, a study of the habitat of great horned owls (n=45) found most of the owls in pinyon (*Pinus* spp.) woodland or Jeffrey pine (*Pinus jeffreyi*) forest (Shuford and Fitton 1998). Other habitats these birds inhabit in California include sagebrush scrub, sandflats, lodgepole pine (*Pinus contorta*) forest and areas of mixed pines (Shuford and Fitton 1998). Bennet and Bloom (2005) found that oak/sycamore woodland and ruderal grasslands were the dominant habitats in southern California where great horned owls were found. In the foothills of the Sierra Nevada mountains, Fitch (1947) found these owls nesting in mixed grassland including scattered trees, chaparral and boulders. In deserts, including California's Mojave Desert, great horned owls nest in stands of juniper (*Juniperus* spp.) trees or large yuccas (*Yucca* spp.), on cliff ledges, in caves and in abandoned quarries (Bent 1938, Smith 1969).

Although open forests are the main habitat for these owls, they are highly adaptable and can live almost anywhere. They have been found inhabiting woodlots (Bohm 1980) and college campuses (Franks and Warnock 1969). This species is only absent when habitat conditions become more extreme, such as dense rainforest or elevations above 3,658 meters (12,000 feet). Wintering habitat is generally similar to breeding habitat because this species is non-migratory (Houston et al. 1998) (see Migration section, below).

Migration

The great horned owl is non-migratory and occupies the same range throughout the year (Houston et al. 1998). Birds living in the boreal forest sometimes travel south but this is considered to be a relocation event rather than a true migration. This relocation occurs particularly when a major food source, such as the snowshoe hare (*Lepus americanus*), becomes scarce (Adamcik et al. 1978). In years when snowshoe hares are abundant, great horned owl populations in the boreal forest stay fairly stable and relocations do not occur. The longest known distance that a great horned owl has traveled was 2,058 kilometers (1,279 miles) from Alberta, Canada to Illinois (Houston et al. 1998). It is not known whether the owls return to the same location when prey populations increase.

Food Habits

The great horned owl is primarily nocturnal and usually hunts from a perch, sometimes as high as 90 m (295 ft), overlooking an open area (Petersen 1979). Peak foraging times are shortly after sunset and again around the first light of dawn (Marti 1974). Their large, yellow eyes have numerous rods that allow the owl to spot potential food from great distances and in low light (Marti 1974, Ault 1984, Ault and House 1987). This owl also possesses acute hearing, and the ability to swivel its head 180 degrees. The feathers of the great horned owl are particularly soft allowing it to hunt silently while providing exceptional insulation at colder temperatures. The species has short, wide wings that provide maneuverability between trees as it leaps from its perch to pounce on prey. It captures prey with its strong talons. One study determined that the clenched talon of a great horned owl requires 13 kg (29 lbs) of force to open (Marti 1974). Smaller prey items are swallowed whole while larger animals are dismembered and decapitated before feeding (Peeters 2007).

The great horned owl has the broadest diet of any North American owl (Marti and Kochert 1996). Its primary diet consists of rabbits, hares, mice, and waterfowl. However, it may also consume reptiles, insects, scorpions, and other birds. In a study by Jaksic and Yañez (1980) of great horned owls in California, the mean prey size was 98 grams (3.5 ounces) for 1,471 prey items. Prey range in size from insects up to weasels and skunks (Errington 1932). This bird has even been known to eat domestic cats (*Felis domesticus*) and porcupines (*Erethizon dorsatum*) (Bent 1938). The largest component of the great horned owl's prey base is typically mammals (90%) (Seidensticker 1968). There is, however, regional variation in diet. Birds living on islands in Mono Lake, California subsist primarily on deer mice (*Peromyscus* spp.) and montane voles (*Microtus montanus*) (Aigner et al 1994). In the boreal forest, when snowshoe hare populations peak, they make up 81% of the great horned owl's diet with the rest of the diet being comprised of other mammals (8%) and birds (11%) (Adamcik et al. 1978). In Baja California, the owl's diet consists of only 24.3% rabbits, 22.7% rodents, 15.7% reptiles, and 16.5% insects (Llinas-Gutiérrez et al. 1991). In contrast, studies of these owls living in the deciduous forests of the eastern United States showed that rodents and non-game birds were the primary prey while rabbits were only a minor contingent of the diet (Bosakowski et al. 1989).

While birds generally make up only a small portion of the diet in comparison to mammals, they can be an important source of food for the great horned owl. Rusch et al. (1972) found that in Alberta, Canada, waterfowl was the next greatest portion of this owl's diet after snowshoe hares. Boal et al. (2005) reported the predation of two northern goshawks (*Accipiter gentilis*) by great horned owls in Minnesota. A study on Protection Island, Washington, revealed a unique population of great horned owls that fed solely on birds, with rhinoceros auklets (*Cerorhinca monocerata*) being the main source of food (Hayward et al. 1993).

Breeding

Great horned owls utilize a wider variety of nest sites than any bird in the Americas (Baumgartner 1938). This owl does not build its own nest but lays its eggs in nests that were constructed by other birds or squirrels in previous years (Bosakowski et al. 1989). A scrape on a cliff side may be used instead of a traditional nest in a tree if it provides better isolation from predators (Dixon 1914, Baumgartner 1938). Other locations where great horned owls have been known to lay their eggs include snags, badger (*Taxidea taxus*) dens, man-made platforms, empty buildings and on the open ground (Baumgartner 1938, Bent 1938, Bohm 1980). In Baja California, great horned owls have been known to nest in cardon cactus (*Pachycereus pringlei*) (Llinas-Gutiérrez et al. 1991).

Great horned owls are monogamous for multiple breeding seasons and may mate for life (Houston et al. 1998). Mated pairs are territorial and will occupy the same territory year round. Birds that have not yet found or taken over a territory are "floaters" and it is suspected that they do not mate (Rohner 1997). In the non-breeding seasons, the male and female owls will often roost separately but remain within the same territory. About a month before breeding, the hooting of the male owl becomes more frequent and noticeable as he calls to his mate and challenges other male owls (Baumgartner 1938).

Great horned owls breed primarily between January and March with eggs laid later in higher latitudes (Baumgartner 1938, Bosakowski et al. 1989, Rohner 1996). Eggs are laid 1 to 7 days apart and most often only two or three are laid (Dixon 1914, Baumgartner 1938, Bent 1938). In California, the size of eggs averaged 4.51 to 5.34 cm (1.78 to 2.10 in) (Bent 1938). These owls are generally single brooders, but they may produce a second set of eggs if their first clutch is lost (Marti 1969).

The female begins incubation after the first egg is laid and incubation lasts about 30 days (Hoffmeister and Setzer 1947). Usually, the female does all of the incubating, however, Baumgartner (1938) witnessed a male replace his mate as she left to go hunting. Normally, the main duty of the male during incubation is to provide food for the female and protect the nest from harassment (Baumgartner 1938). The parent birds have been known to attack humans climbing the nest trees (Keyes 1911, Murie 1922, Bent 1938). They may also try to protect the nest by using deception to lead predators away. While searching for its nest, Murie (1922) followed a great horned owl that appeared to be injured only to have it fly off in perfect health once it had led him away from the nest.

After the eggs hatch, the female spends most of her time brooding her young but will leave the nest to hunt if the hatchlings require more food than the male bird provides (Rohner and Smith 1996). The young fledge at around 45 days old and move out of the nest to nearby trees. Only short, hopping flights are possible when the fledglings first leave the nest and they are only about three-quarters fully grown (Dixon 1914, Hoffmeister and Setzer 1947). Young great horned owls are capable of full flight at about 9 or 10 weeks at which time their parents stop bringing them food (Bent 1938). Young owls may stay in the territory in which they were born for several months before dispersing (Bent 1938).

Stewart (1969) found that the average lifespan of this owl in the wild was often less than ten years. In a study of 97 birds banded as nestlings and juveniles, 46% died in their first year and all but 5 birds had perished by 8 years old. One bird did survive long enough to reach the age of 13 (Stewart 1969). However, the primary cause of death in these birds was gunshot (52%). Since then, hunting of the great horned owl has been prohibited by the Migratory Bird Treaty Act and these numbers may no longer be accurate. There is little information on the current average lifespan for the great horned owl. In captivity these birds can live quite long. Bent (1938) reported one great horned owl that lived in captivity for more than 29 years. Survival rate among juveniles seems to vary upon prey availability. In the Yukon Territory, mean survival of these birds from the time of fledging to 20 weeks of age dropped from 80.0% to 23.2% after 3 years of prey decline (Rohner 1996). Evidence from this study may be biased due to the inclusion of banded birds only.

Home Range

Great horned owls are highly territorial, and their range is maintained through hooting. Studies in the Yukon Territory show that dispersal from the breeding site increased from 0.6 to 0.8 km (0.4 to 0.5 mi) in September, and from 16 to 35 km (10 to 22 mi) in October of the first year (Rohner 1996). Dispersal will also depend on the prey availability in respective habitats. There is little data on movements of adults between breeding seasons.

Home range size varies greatly as exhibited by great horned owls (n=3 pairs) in Utah that were estimated to have home range sizes of 70, 96, and 152 hectares (173, 237, 376 acres) (Smith 1969). In this same study, Smith (1969) reported that the great horned owl nests were evenly spread out with an average distance of 1.6 km (1.0 mi) between each nest. Owls in the Yukon

Territory were monitored daily and their home ranges averaged 483 ha (1,194 ac) (Rohner 1997). The large difference in home range size between Smith (1969) and Rohner (1997) is due to the inclusion of non-territorial floaters that tended to travel further, in the latter study.

Historical and Current Distribution

Information that documents the historical range of the great horned owl is limited. Grinnell and Miller (1944) noted four subspecies for California, broadly distributed throughout the state. The "Pacific horned owl" (*B. v. pacificus*) occupied the largest geographic area, and was noted over an elevation range of near sea level up to 2,134 m (7,000 ft). One study (Brodkorb 1971) reported that great horned owl bones had been discovered at various fossil localities in the United States that date back to the Pleistocene. Also, remains of the great horned owl ancestor, *Bubo sinclairi,* have been found in various caves in California (Miller 1911).

The great horned owl is a habitat generalist found in many different regions across both North and South America. It can be found across the North American continent excepting the northernmost parts of Canada and Alaska (Bent 1938). In California, this bird is found throughout the state and is only absent at the highest areas of the Sierra Nevada mountain range (Peeters 2007). Although widely distributed in North America, it is not an abundant species when compared to other raptors, such as the red-tailed hawk (*Buteo jamaicensis*).

Historical and Current Abundance

Early records show that the great horned owl was a regular sight in California. Cooper (1870) suggested that the great horned owl was probably the most common owl in California after the burrowing owl (*Athene cunicularia*). The great horned owl was considered to be present "in fair numbers" in the San Francisco region (Grinnell and Wythe 1927). Adams (1909) described this bird to be a "fairly common resident" of the wooded areas in Placer County. McGregor (1901) called the great horned owl "a very common breeder" in Santa Cruz County. In the foothills and canyons of the Sierra Nevada mountains, this owl was considered to be "a fairly common resident" (Sumner and Dixon 1953). However, in Fresno County, Tyler (1913) wrote that the great horned owl was "rapidly becoming rare." Willet (1933) described this owl as "fairly common" in the foothills and mesas of southern California, but also noted that it was much rarer than it used to be.

Currently, the great horned owl is still considered to be a common species in California if not as common as it once was. Shuford (1993) described the great horned owl as "the most widespread breeding owl in Marin County." Additionally, the owl was noted as "fairly common" in both Yosemite and the Colorado River Valley, and "common" in Santa Barbara County (Gaines 1988, Rosenberg et al. 1991, Lehman 1994). Harris (1991) reported the great horned owl as a "common resident and breeder" in northwestern California. Even though this species has been known to breed in the Imperial Valley, Small (1994) stated that it is now only a "rare visitor" there. Elsewhere in California, Small (1994) reported the great horned owl as "fairly common" and "widespread." Unitt (2004) noted the species as an "uncommon resident" in San Diego County.

There is little information about historical population levels of the great horned owl. However, the number of these owls reported on surveys, such as the Audubon Christmas Bird Count, has increased over time. It is unclear as to whether this is due to increasing population levels or increased knowledge in bird identification. It was reported that as hunting increased, great horned owls became less common in Michigan (Barrows 1912). Year-round take has not been allowed in the United States for great horned owls since 1970, except in the case of owls in the process of killing private poultry (Belcher 1996).

Breeding Bird Survey data from 2006 to 2010 indicate that the great horned owl is widespread, but thinly distributed (Figure 1). Studies show that the raptor was observed on at least 848 out of 1,832 North American Breeding Bird Survey routes during 1965-1979, with an average of 0.2 sightings per route per year (Robbins et al. 1986). According to the United States Breeding Bird Atlas, the highest densities of great horned owls are in Florida, Kentucky, Louisiana, and Virginia. Studies documented great horned owl presence in 78% of 10 km^2 (3.9 mi^2) blocks in these states (Gauthier and Aubry 1996). In contrast, the raptor is the most common owl in Quebec, yet was present in only 17% of 10 km^2 (3.9 mi^2) blocks.

The great horned owl is present across North America and is most abundant in the United States. Between the years 1966 and 2009, there were 2,121 great horned owls identified in the United States on survey counts, as opposed to only 319 in the same time span in Canada (Sauer at al. 2011). The areas with the highest density in one study by Root (1988), were aspen (*Populus* spp.) parklands and grasslands of Alberta and Saskatchewan, eastern Colorado, Nebraska, Kansas, and northern Oklahoma. Using data that has been rated as green or good quality, Partners in Flight has estimated that California has about 110,000 individuals, projected to be 2.1% of the world population (Ruth 2006).

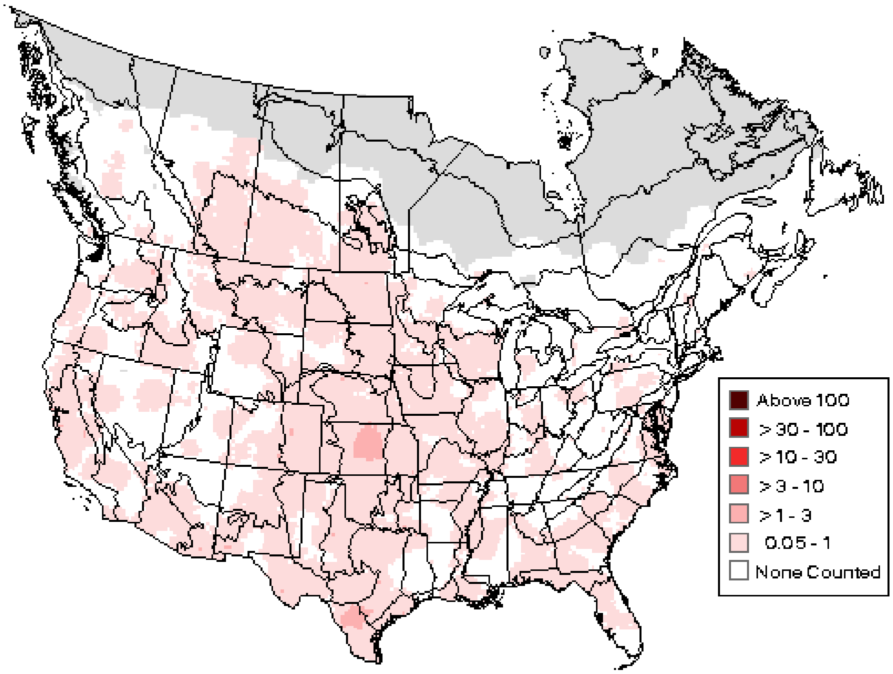

Figure 1. Great horned owl summer distribution based on Breeding Bird Survey data from 2006 to 2010 (Sauer et al. 2011).

Population Trend

According to the International Union for the Conservation of Nature the population of the great horned owl is relatively stable in the United States, and it is not a species of concern (IUCN, Birdlife International 2009). Breeding Bird Survey results show an increase in great horned owl numbers with a trend index of 0.2 (n = 2121, CI = -0.3, 0.6) between 1966 and 2010 in the United States and Canada (Figure 2) (Sauer et al. 2011). The index value is a measure of percent change per year, and in this case is listed as a “significant" increase.

In California, both Christmas Bird Counts and Breeding Bird Surveys have indicated overall increases in great horned owl populations. Christmas Bird Counts (Figure 3) from 1960 through 1981 show a steady rise in the number of great horned owls counted per party hour. After 1981, count numbers became fairly stable, fluctuating around 0.085 birds counted per party hour (NAS 2010). According to the Breeding Bird Survey (Figure 4), the trend index for this species between 1966 and 2010 is 0.1 (n = 146, CI = -1.0, 1.1) showing a small increase in great horned owl numbers in California (Sauer et al. 2011).

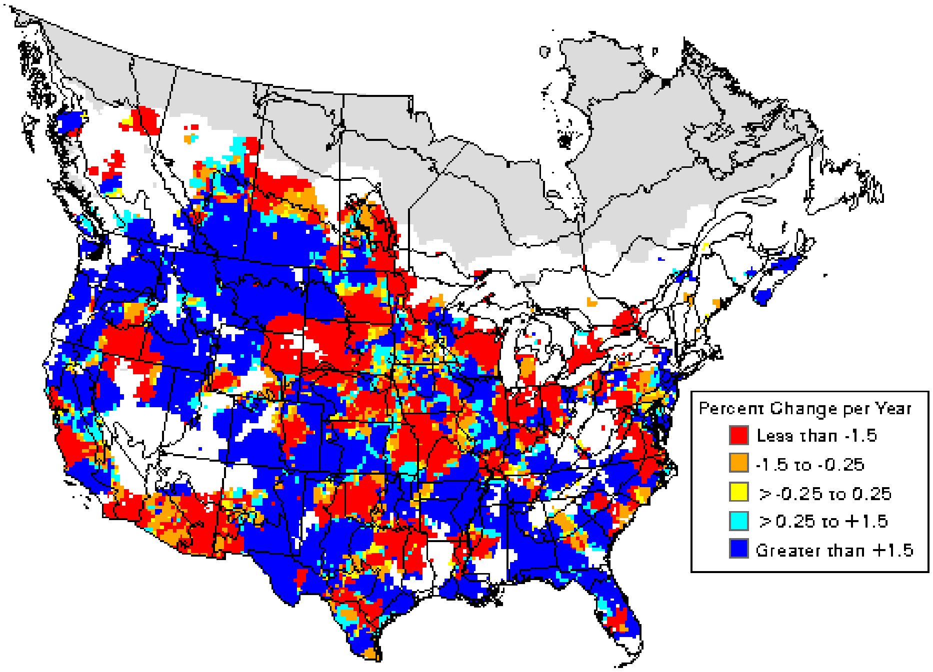

Figure 2. Great horned owl population trend map from 1966-2010 from Breeding Bird Survey Counts (Sauer et al. 2011).

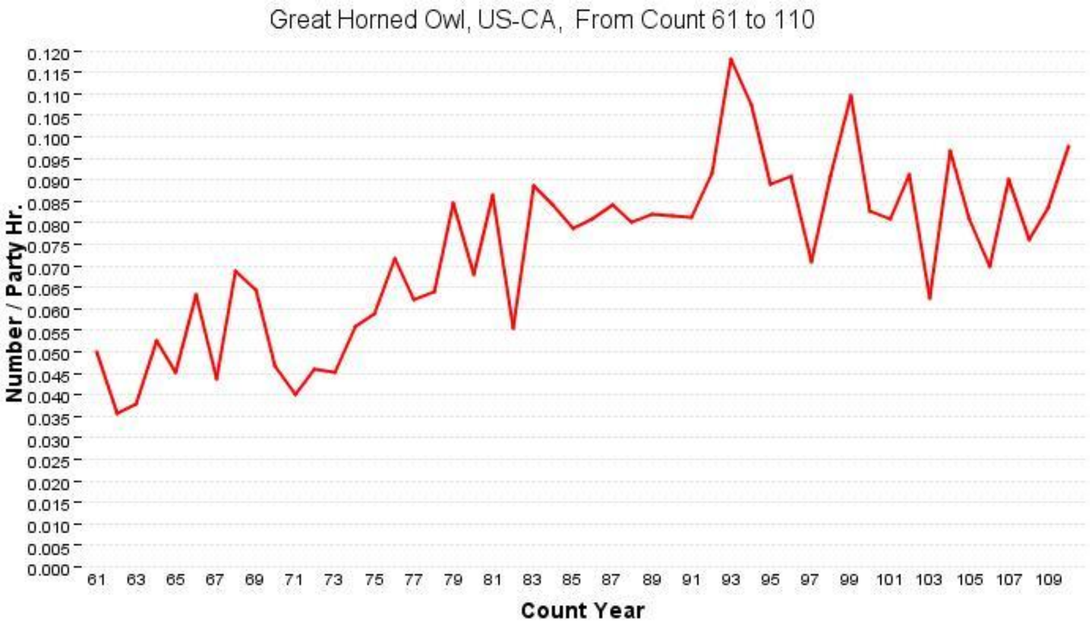

Figure 3. The number of great horned owls counted per party hour on Audubon Christmas Bird Counts in California from 1960 to 2010. Yearly changes in the number of circles or surveyors in a party are accounted for by dividing the number of birds seen by the number of hours spent surveying (NAS 2010).

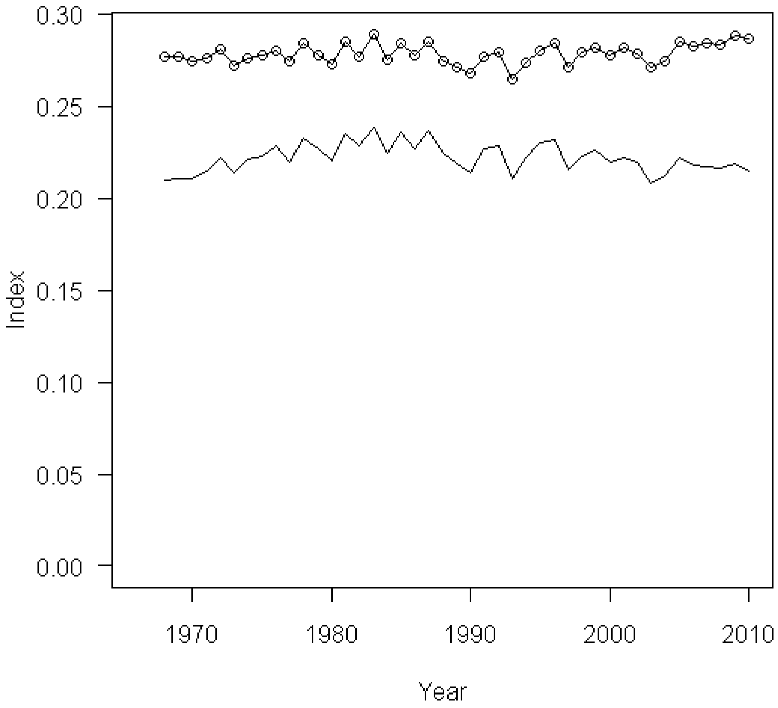

Figure 4. Breeding bird survey trend estimate of great horned owls in California from 1966-2010 (Sauer et al. 2011).

Threats

The great horned owl is thought to be a robust species and not in need of intensive conservation measures, at this time. Though the species is not considered to be in jeopardy, these owls are vulnerable to wind turbine mortality based on information specific to two California wind farms (Anderson et al. 2000), and other threats. Previously, one of the main causes of mortality in these birds was humans. In a study on mortality in great horned owls, Stewart (1969) found that, out of 372 dead birds, 52% had been shot. However, this amount may have been biased since the study analyzed only birds reported by humans. Since the addition of great horned owls to the Migratory Bird Treaty Act in the early 1970's the percentage of reported gunshot kills has decreased significantly. Franson and Little (1996) found that only 9% of 132 great horned owls carcasses submitted to the National Wildlife Health Center between 1975 and 1993 were the results of gunshot wounds.

Great horned owls can be susceptible to diseases. One female survived the wound caused by a pellet from a shotgun only to succumb to pneumonia brought on by an infection of the mold *Aspergillus niger* (Wobeser and Saunders 1975). After necropsy of a dead male owl in Colorado, Lairmore et al. (1985) found numerous nodules containing the bacterium responsible for avian tuberculosis, (*Mycobacterium* spp.). They concluded that tuberculosis could be a significant cause of death among great horned owls and red tailed hawks in Colorado after a red tailed hawk was also diagnosed with avian tuberculosis in the state 26 months later (Lairmore at al. 1985). In New York, a case of bumblefoot was documented in a great horned owl. Pus gathered in the feet of the owl presumably after an injury making it unable to hunt (Stoner and Stoner 1945).

Another disease that can be fatal to the great horned owl is West Nile Virus (*Flavivirus* spp.). The Minnesota Department of Health reported that 25 great horned owls submitted between 2002 and 2003 tested positive for West Nile Virus (Wünschmann et al. 2005). After assessing the health of each of the birds they determined that West Nile Virus resulted in disease and death in great horned owls. In California, 8 great horned owls were reported to have died from West Nile Virus in 2008 which appears to be the largest number for the reporting period available (CDPH, http://www.westnile.ca.gov/).

An analysis of the carcasses of twenty-eight fledgling great horned owls in the Yukon Territory showed that black flies (Diptera: Simuliidae) were a major threat to young owls in this region (Hunter et al. 1997). Out of 28 birds, 13 died of severe anemia and dehydration caused by blood loss to black flies. Nine of those thirteen were also infected with the blood parasite *Leucocytozoon* spp. Four of the birds were road-kills, four others died of unspecified trauma, one succumbed to an infection from a wound and one died of cloacal impaction. The cause of death could not be determined for the remaining five birds (Hunter et al. 1997).

Eggs and immature great horned owls are not usually vulnerable to predation unless the nest is left unattended by the parents. In this case, nestlings and eggs may be consumed by foxes (*Vulpes vulpes*), coyotes (*Canis latrans*), and raccoons (*Procyon* spp.). Immature owls can be killed by bobcats (*Lynx rufus*) or other raptors including other great horned owls (Rohner and Hunter 1996, Hunter at al 1998). Nestlings are occasionally killed by their parents or siblings in times of food shortage (Errington et al. 1940). Other natural cases of injury possibly causing mortality have been porcupine quills (Bent 1938) and blindness caused by the spray of skunks (Mephitidae)(Garcelon 1981).

Secondary poisoning threats to great horned owl populations include accumulation of rodenticides, strychnine, organophosphates, organochlorines, and PCBs in prey items (Seidensticker and Reynolds 1971, Mendenhall and Pank 1980, Sundlof et al. 1986, Cheney et al. 1987, Henny et al. 1987, Blus 1996). In a study by Blus (1996), PCB's accounted for 32 direct owl deaths. In another study by Houston and Francis (1993,) 16.% and 9.1% of 481 of banded great horned owls were found dead from vehicle collision and electrocution, respectively.

Data Gaps and Monitoring Needs

A considerable number of publications have been issued on this raptor. However, few documents have been published that describe the species as a whole in terms of systematic, ecological relationships. The most detailed studies relate to great horned owl populations in the boreal forest. Although the information is useful, there is a void in data on the species as a whole. The ecology of the boreal forest is cyclical, so this information does not directly relate to great horned owl populations in California. An example of how this information does not relate elsewhere is in food scarcity, which is common in northern populations, but rare in southern populations (Houston et al. 1998).

There are also few valid statewide population estimates on great horned owls, chiefly because there are many individuals that "float" from one state to another due to food scarcity and territorial changes. Also, little is known about whether home range size changes in relation to food scarcity, hooting patterns, breeding, and territorial defensiveness. Finally, more research is needed about the effects of the great horned owl on other owl populations (Houston et al. 1998).

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ECOLOGY OF THE NORTHERN GOSHAWK

By Kevin P. Cahill
July 5, 2012

The Northern Goshawk (*Accipiter gentilis*) (hereafter goshawk) occurs throughout the Holarctic region in forest and woodland environments feeding primarily on birds and mammals and often nesting and hunting in old-growth forests (Kennedy 1997, Morrison et al. 2011). Goshawks occur from sea level to alpine in most forest types throughout their range (Squires and Reynolds 1997). Two subspecies of goshawk are recognized in Western North America, *A. g. atricapillus* and *A.g. laingi* (AOU 1957). *A. g. atricapillus* breeds throughout North America including California. *A.g. laingi* breeds from the Queen Charlotte Islands and Vancouver Island north through insular British Columbia and insular and coastal mainland Alaska to Icy Strait and Lynn Canal. A third, currently debated subspecies, *A.g. apache*, resides from Southern Arizona south through Mexico to Jalisco (Squires and Reynolds 1997). In California goshawks occur at the greatest densities in the Cascades, Sierras, and Northeastern forests, although they also occur at lower densities in north coastal mountains and infrequently in the higher mountains of Southern California (Dan Applebee, CA. Dept. of Fish & Game, personal communication, 2012).

A.g. atricapillus is the largest and least sexually dimorphic *Accipiter* in North America and the sexes have nearly identical plumages (Clark and Wheeler 1987, Palmer 1988). Females however are noticeably larger than males (Clark and Wheeler 1987). Morphological features such as relatively short, broad wings and a long tail facilitate maneuverability for hunting in mature forests (Bent 1961). The head has a slate-black cap with a thick, white superciliary line and greenish-yellow cere (Clark and Wheeler 1987, Palmer 1988). Iris color is deep red to mahogany (Clark and Wheeler 1987). The back and upperwing coverts are slate blue and underparts are grey with dark barring and vertical black streaks which are more accentuated in females. Flight feathers are light grey with light barring and some streaking ventrally. Undertail coverts are plain white and fluffy and the tail is grey with 3 or 4 dark bars and a white terminal strip that is often worn (Clark and Wheeler 1987, Palmer 1988). Legs and feet are yellow with black talons (Palmer 1988).

A.g laingi is similar to *A.g. atricapillus* but is darker in color with a much narrower superciliary line which can be heavily streaked with black and may only appear as a white patch above the eyes (Wheeler 2003). *A.g. laingi* is smaller in size with a proportionately longer tail (Wheeler 2003).

Immature iris color is pale greenish-yellow to light brown and the head is brown with a thin white superciliary line (Clark and Wheeler 1987). The back and upperwing coverts are brown with white mottling and the cream colored underside is heavily streaked. Flight feathers have dark barring underneath and the wavy dark and light brown bands on the tail are accentuated by thin white highlights and a thick white terminal band. Undertail coverts are streaked and are not fluffy (Clark and Wheeler 1987).

Sub-adults retain some brown feathers for up to two years, are more heavily barred, and have more streaking than adults but are otherwise similar in plumage. Iris color is orange to red (Clark and Wheeler 1987).

The association with valuable old growth timber resources has led to considerable debate over habitat loss including proposals to the US Fish and Wildlife Service (USFWS) for listing goshawk as a threatened or endangered species west of the 100th meridian in 1991 and 1994. USFWS determined in 1992 and 1996 that listing was not warranted on taxonomic grounds (Squires and Kennedy 2006, Richter 2005). Furthermore, the conclusion of the 1998 status review of the species by the USFWS found insufficient evidence to support listing the goshawk under the Endangered Species Act (USFWS 1998, Squires and Kennedy 2006). However, the USFWS 2008 Birds of Conservation Concern list included goshawk in Bird Conservation Region (BCR) 5 (Northern Pacific Forest) and goshawk was also included on the US Forest Service (USFS) Sensitive Animal Species list (unpublished 2007) for the Pacific Southwest Region. It was also listed as a 3rd priority Bird Species of Special Concern in California in 1978, 1992, and 2008 (CDFG unpublished list 1992, Remsen 1978, Keane 2008). Keane (2008) stated that degradation and loss of habitat are the primary concerns, including the uncertainties surrounding forest management practices, and the development and recreational activities associated with human population growth and expansion, combined with the lack of knowledge about California population trends (Keane 2008).

In California, falconers harvested 46 goshawks from the wild from 2006 to 2010, 34 of which were nestlings, 9 immature, 1 adult, and 2 of unknown age. Of the 34 nestlings, nearly half (n=18) were harvested from Mono and Modoc counties. Eighteen goshawks mortalities and 4 losses were reported 2006 to 2010. Eleven goshawks were released back into the wild during the same period (CDFG 2012). Statewide, populations in California appear to be unaffected by falconry harvest, however negative effects may occur at the local level if local populations are repeatedly harvested (Keane 2008).

Habitat

Goshawks nest in mature old-growth forests composed of large trees, with 50-90% canopy closures and an open understory (Anderson et al. 2005, Reynolds et. al 1982, Woodbridge and Dietrich 1994). In California, nest stands occur in mature and old-growth forest stands and consist of larger trees, greater canopy cover, and relatively more open understories than stands without nests (Saunders 1982, Hall 1984, Hargis et al. 1994, Keane 1999, Maurer 2000, Keane 2008). Nest stands may include single or multi-story canopy structure and a broad range of conifer and conifer-hardwood types (Reynolds et al. 1982, Keane 2008). Pacific Ponderosa (*Pinus ponderosa*), Jeffrey (*P. jeffreyi*), and Lodgepole (*P. contorta*) pine, mixed conifer, White (*Abies concolor*) and California Red (*A. magnifica*) fir, Douglas-fir (*Pseudotsuga menziestii*), mixed Redwood (*Sequoiadendron giganteum*)-Douglas-fir-hardwood, mature Quaking Aspen (*Populus tremuloides*) stands within aspen-shrub steppe vegetation east of the Cascade-Sierra axis, and (rarely) pinyon-juniper (*P. monopylla, Juniperus* spp.) are preferred forest types; however, structural characteristics of the nest tree and surrounding vegetation may be more important than tree species (Anderson et al. 2005, Keane 2008). Foraging occurs in a variety of habitats including early- to late-successional forest as well as open habitats (Anderson et al. 2004).

Close to the nest, dense, late seral stage structure is an important feature for goshawks but decreases in relative abundance with distance from the nest (DeStefano et al. 2006). Eastern California nest sites had greater basal area, more canopy cover, and more trees in the largest diameter classes than random plots (Hargis et al. 1994). Several studies throughout the United States found that nesting areas were generally near the bottom of moderate slopes in areas of sparse ground cover (Reynolds et al. 1982, Moore and Henry 1983, Speiser and Bosakowski 1987, Hayward and Escano 1989, Ingraldi and MacVean 1994, Squires and Ruggiero 1996). In California, an average of one forest opening was present within 15 meters (m) (49 feet (ft)) of goshawk nests, and nest sites were located on north-east slopes, near streams, and closer to meadows than random sites (Allison 1996).

In California, canopy cover around the nest tree is usually >60% to 100% and is 25% to 40% in Pacific Ponderosa and Jeffery pine forests east of the Cascade-Sierra axis (Keane 2008). Nest sites are often used over multiple years but goshawks will alternate nests every one to five years (Reynolds and Wight 1978, Wheeler 2003). Typically, in conifers, nests are constructed in the lower third of the largest tree in the stand against the trunk or on large limbs up to 4 m (13 ft) from the trunk (Reynolds et al 1982). 45% of nests (N=157) sampled in the Southern Cascades and Western Sierra Nevada of California were placed in a deformed top or mistletoe clump, 43% against the trunk, and 12% out on a limb (Richter 2005, Keane 2008). Nests are placed in primary or secondary forks in deciduous trees. Nests are 5 to 23 m (15-75 ft.) high but can reach up to 27 m (90 ft) in the Pacific Northwest (Wheeler 2003).

Non-breeding habitat has not been thoroughly studied; however annual and between-sex variations exist in non-breeding habitat use (Anderson et al. 2004). For example, Males in northern Arizona moved from ponderosa pine forest to pinyon-juniper forests while females stayed in the ponderosa pine (Drennan and Beier 2003). In the Yukon and Alaska, more goshawks remained on breeding territories during the winter in years with high prey numbers but moved elsewhere during low prey years (McGowan 1975, Doyle and Smith 1994).

Winter habitat use is also likely influenced by migratory status, with goshawks that migrate longer distances having more variable winter habitat use than those who migrate short distances. Radio tracked migratory birds in Wyoming for example used a variety of habitats ranging from cottonwood (*Populus* spp.) groves surrounded by sagebrush to mixed aspen and lodgepole pine to spruce-fir and lodgepole pine (Squires and Ruggiero 1995). Non-migratory residents however, may stay on breeding home ranges or move only short distances to better foraging habitats (Squires and Kennedy 2006). In managed timberlands of Central and Northern California, eighty-seven percent of telemetry points (n=333) recorded by Richter (2005) were less than 16 km (9.94 mi) from known nesting locations during both breeding and non-breeding seasons. Range expansion may also occur during winter months. Year-round residents in the Lake Tahoe basin remained in the area of their breeding territories during the non-breeding period but increased their range by approximately 275% for females and 350% for males (Keane 1999).

Food Habits

Goshawk hunting strategies include short flight excursions combined with sit-and-wait searching from elevated perches, rapid flights along forest edges and across openings, and surprise attacks through dense vegetation (Squires and Reynolds 1997, Wheeler 2003). Goshawks are foraging zone generalist taking prey from the ground-shrub, shrub-canopy, and canopy layers (Reynolds and Meslow 1984). Open space between the bottom of the canopy and the top of the shrub layer is characteristic of foraging sites and is possibly utilized as a flight path (Anderson et al. 2004). Goshawks will also stalk and track prey on the ground using the surrounding vegetation and topography for cover, occasionally hopping through thicker brush or running to flush a victim (Bent 1961, Palmer 1988, Squires and Reynolds 1997). Schnell (1958) documented a female goshawk wading into shallow water to obtain prey. More typical attacks are made suddenly at high speed including reckless crashes through bushes (Palmer 1988). Food caching (storing food for later consumption) by the female occurred in Nevada County, California when nestlings were too young to consume large amounts of food at single feedings. Food was also cached prior to nest defense when human intruders were detected by the female (Schnell 1958). Goshawks also cache prey in the winter if it is too large to be completely consumed or removed to a safer location (Drennan and Beier 2003).

Goshawks are generalist species and take a wide variety of prey including ground and tree squirrels, rabbits and hares, large passerines, woodpeckers, game birds, and corvids (Squires and Kennedy 2006). Out of 30 North American raptor species, goshawks have the fourth highest ranked dietary diversity (Drennan 2006). There are certain taxa that are especially prevalent in most diets, and include; chipmunks (*Eutamias sp*), cottontail (*Sylvilagus sp*), snowshoe hare (*Lepus arcticus*), Douglas squirrel (*Tamiasciurus douglasi*), red squirrel (*T. hudsonicus*), golden-mantled ground squirrel (*Citellus lateralis*), grey squirrel (*Sciurus sp*), northern flying squirrel (*Glaucomys sabrinus*), American robin (*Turdus migratorius*), blue jay (*Cyanocitta cristata*), Stellar's jay (*C. stellari*), ruffed (*Bonasa umbellus*) and blue grouse (*Dendragapus obscurus)*, common crow (*Corvus sp*), domestic pigeons (*Columba livia*), and northern flicker (*Colaptes auratus*). (Drennan 2006, Squires and Reynolds 1997)

Squirrels, especially Douglas squirrels and red squirrels, are widely distributed and abundant throughout goshawk range and are an important prey species especially during winter (Squires and Kennedy 2006). Even though goshawk diet is diverse, seasonal shifts in key prey abundance (e.g. squirrels) can affect goshawk breeding success (Salafsky et al. 2007). Variation in Douglas squirrel populations was a contributing factor to annual variation in the proportion of goshawk pairs nesting, the number of active and successful nests, and the number of young fledged in the Sierra Nevada in California (Keane 1999, Keane et al. 2006). Likewise, in Arizona, variation in red squirrel density explained more annual variation in goshawk reproduction than any other species, illustrating their importance as prey (Salafskyi et al. 2007).

Available data indicates that diets may change considerably due to shifts in prey species availability throughout the year (Drennan and Beier 2003, Widén 1987, Rickman 2005). For example, Swedish goshawks shifted from 89% avian prey in the spring and summer to 79% squirrels in the winter (Widén 1987). Factors contributing to prey availability include prey migration, estivation, feeding and hibernation behaviors (Squires and Kennedy 2006). Variable migration and shifts in habitat use by goshawks during winter may also be linked to prey availability (Drennan and Beier 2003, Rickman et al. 2005, Keane et al. 2006). In northeastern California, goshawks shifted roost locations from ponderosa pine where hibernating prey species such as golden-mantled ground squirrel and chipmunk were available in the spring and summer, to mixed-conifer and fir forests where non-hibernating species such as Douglas' squirrel and snowshoe hare were still active (Rickman et al. 2005). Drennan and Beier (2003) found that while most females inhabited ponderosa pine forests during the winter in Arizona, most males shifted to lower-elevation pinyon-juniper forest. Individuals in ponderosa pine forest took Albert's squirrels and cottontails, while those in pinyon-juniper were only known to take cottontails (Drennan and Beier 2003).

Breeding

The breeding season typically begins when pairs return to nesting territories between late March and early April, although some individuals may spend the whole winter at the nesting site (Reynolds et al. 1994, Squires and Reynolds 1997). Breeding activity lasts until fledglings disperse in mid-August (Squires and Reynolds 1997). Goshawks pairs are monogamous choosing the same mate year after year (Moller 1987, Montgomerie and Thornhill 1989). In California, Detrich and Woodbridge (1994) reported that 72 percent of pairs in the Southern Cascades were with the previous years mate. The remaining 28 percent were explained by new mate selection in some cases resulting from mate mortality.

Courtship displays include high-circling, where one or both members of the pair circle above the canopy with tail partially spread and under-tail coverts alternately flared and retracted as well as paired slow-flapping flight in which male and female fly close together, wings held above the body dihedral. The male often dives toward the female during this time and performs "sky-dancing", in which the he conducts a series of high altitude climbs above the canopy followed by a series of shallow undulations and high velocity dives with wings tucked plunging down through the canopy. Pairs may be silent or highly vocal with the female's voice distinguishably lower (Palmer 1988, Squires and Reynolds 1997, Wheeler 2003).

Goshawks copulate approximately 518 times during courtship; among the most frequent for birds (Moller 1987, Palmer 1988, Squires and Reynolds 1997). The female often interrupts nest building for copulation. Two peaks in copulation occur, one 30-40 days before egg laying and one just before and during egg laying (Palmer 1988, Squires and Reynolds 1997).

During the breeding season, goshawks can be categorized as: subadults (1-2yr) with primarily juvenile feathers, young adults (2-3 yr) with primarily adult plumage and some juvenile feathers, and (>3 yr) with full adult plumage. Females may nest as subadults but males, not having completely developed testicles may not be able to reproduce as subadults (Palmer 1988, Squires and Kennedy 2006). In Arizona, mean age at first breeding was 4.2 years for females and 3.9 years for males (Wiens et al. 2006a).

Goshawks have high site fidelity and will often use the previous year's nest adding fresh green branches and foliage (Palmer 1988, Wheeler 2003). Nest areas are occupied from early March until late September and include one or more forest stands of large, old trees with dense canopy cover containing several nests. They are usually 8 to 10 hectares (ha) (20 to 25 acres (ac)) but size and shape are dictated by topography and the availability of patches of dense, large trees (Reynolds et al. 1992). Mean distance between alternate nests was 273 m (± 68.6 SE) (896 ft ± 225) in the Southern Cascades of California (Woodbridge and Detrich 1994).

Female goshawks are responsible for nest construction (Palmer 1988, Squires and Reynolds 1997). Usually one clutch is laid per year but re-nesting may occur in rare occasions if the first attempt fails during early incubation. Clutch size is usually two to four eggs, and rarely consists of only one and up to five (Squires and Kennedy 2006). Average clutch size in North America was 2.7 ± 0.88 eggs (Apfelbaum and Seelbach 1983). Palmer (1988) indicates that clutch size may be limited by food availability. Eggs are laid asynchronously two to three days apart and incubation lasts 32 to 34 days until hatching occurs (Palmer 1988). Females are the primary incubator; however, males will occasionally incubate for brief periods, especially during early incubation while the female feeds. The male provisions the female and will pluck the prey on a stump, snag, old nest, or downed log before delivery (Palmer 1988, Bull and Hohmann 1994). In California, these “plucking posts” averaged 68.5 m (225 ft.) from the nest (Schnell 1958).

The female broods the nestlings continuously for four to seven days post-hatching, and starts actively defending the nest around nine to twelve days (Squires and Reynolds 1997). She will aggressively defend the nest area from other raptors and attack humans approaching the nest (Dixon and Dixon 1938, Zirrer 1947, Craighead and Craighead 1969). Once the young are a few days old the female does not allow the male to linger near the nest after prey delivery (Palmer 1988). The young are fed by the female until they are about three weeks old and are able to dismember prey on their own. Both adults then deliver prey to the nest (Palmer 1988). Siblicide, where the largest nestling aggressively pecks at the head or plucks the down of one or both of its siblings until they are forced from the nest to their death, occurs rarely when food is scarce or when the female fails to return to the nest (Schnell 1958, Boal and Bacorn 1994, Estes et al. 1999).

Fledging occurs between late-June and mid-August (Wiens et al. 2006a). Young leave the nest to perch on nearby branches at about 34 to 35 days and males fledge first after 35 to 36 days, the larger females around 40 to 42 days (Johnsgard 1990). Fledglings stay within 300 m (984.3 ft) of the nest for approximately 65 days until their flight feathers harden and they are capable of extended flight, dispersing after 65 to 90 days (Kenward et al. 1993). In Northwestern Arizona 89 radio-tagged individuals initiated dispersal from 71 to 103 days post-hatching between July 30 and September 15. Fledglings remained in the natal territory for 33 to 66 days before dispersal. Twenty-eight percent of the tagged juveniles dispersed from 0.1 to 72.3 km (0.6 to 45 mi) with a median dispersal distance of 15 km (9.3 mi). The other 72% dispersed >80 km (>49.7 mi). The main factors associated with dispersal dates were prey abundance and weather (Wiens et al. 2006a).

In Northwestern Arizona, mortality of recent fledglings was highest for the first three to ten days but decreased to nearly zero after week three as flight skills were honed (Wiens et al. 2006b). Wiens et al. (2006b) documented that prey abundance affected survival immediately after fledging and immediately after dispersal. During years with lower red squirrel abundance, fledgling mortality increased to 12%, whereas years with peak prey numbers showed a 39% higher chance of fledgling survival during the first twelve weeks (Wiens et al. 2006b).

The percentage of active and successful nests in a given year is variable and can be influenced by weather and prey availability (Bloom et al. 1986, Keane 1999). In the Lake Tahoe Region of California 82.4% to 100% of surveyed territories were active and nest success ranged from 36.8% to 82.4% from 1992-1995 (Keane 1999). Bloom et al. (1986) discovered considerable variation in the proportion of active nests throughout California's mountain regions. In total 61% of the territories examined had active nests and variation was highest in the Southern Sierra Nevada-White Mountains (27% to 86% from 1981 to 1983). Only 14% were active in the lowest year East of the Sierra crest where snow can cover nests in the spring (Bloom et al. 1986).

In eighteen Western North America studies productivity ranged from 1.2 to 2.0 young fledged per active nest and from 1.4 to 2.7 young fledged per successful nest (Squires and Kennedy 2006). Bloom et al. (1986) documented 1.71 young fledged per active nest and 1.89 estimated young fledged per successful nest in California. Austin (1993) documented 1.39 young fledged per active nest and 1.77 fledged per successful nest in the Southern Cascades of California. Detrich and Woodbridge (1994) found that frequently occupied territories had 0.5 more fledglings than infrequently used territories in the Southern Cascades, and an estimated 1.93 fledglings were produced per active nest and 2.22 per successful nest (Detrich and Woodbridge 1994). In the California Tahoe region, Keane (1999) found that mean fledged young per territory ranged from 0.7 to 2.0, and ranged from 1.6 to 2.4 per successful nest over a four year period. Richter (2005) found an average of 1.4 young fledged per active nest and 1.7 fledged per successful nest on actively managed timberlands in fourteen California counties.

Home Range

Goshawks have relatively large spatial requirements and home range varies with landscape patterns and habitat quality (Hargis et al. 1994, Keane 2008). Variation might also exist due to experience and foraging ability of individual birds (Keene 1999). Home ranges may overlap with adjacent pairs while smaller nesting territories are defended (Squires and Reynolds 1997). Core areas include the nest and represent approximately 32% of the home range area (Kennedy et al. 1994).

Based on telemetry data, home ranges in the Inyo National Forest, California averaged 2,400 ha (5930.5 ac) for males and 1,790 ha (4,423.2 ac) for females during the breeding season, with ranges expanding significantly after August 1st (Hargis et al. 1994). Nest sites were significantly closer to water than random points but water sources as far away as 3.5 km (2.17 mi) were incorporated into home ranges where permanent sources were scarce. During the nestling-phase, home ranges incorporated a greater number of vegetation types than the post-fledging phase and included vegetation types and seral stages that were not present around the nest. Although old-growth habitat was available around the nest, areas that were disjunct from the nest, vegetatively diverse, and up to 3.5 km away were utilized (Hargis et al. 1994).

Telemetry studies in Lake Tahoe, California, found home ranges during the breeding season ranged from 877 to 4,404 ha (2167.1 to 10,882.5 ac) for females, and 817 to 6,264 ha (2,018.9 to 15,478.7 ac) for males (Keane 1999). Average home ranges increased for the non-breeding period to 5,555 ha for females (13,726.7 ac) and 8,193 ha (20,245.3 ac) for males. Differences in home range between seasons may be due to a combination of exploration and reduction in prey species over winter months (Keane 1999). Telemetry-tracked home ranges in the Southern Cascades of Northern California averaged 3,774 ha (9325.8 ac) for females, and 2,425 ha (5,992.3 ac) for males (Austin 1993).

Defended nesting territories in the Southern Cascades ranged from 1.3 to 6.1 km (0.8 to 3.8 mi) apart, with reduced spacing around meadows and riparian systems, and contained forest stands 4.1 to 115 ha (10.1 to 284.2 ac) in size and 0.7 to 1.8 km (0.43 to 1.12 mi) apart (Woodbridge and Detrich 1994). These Southern Cascades territories contained 3 to 9 alternate nests which ranged from 30 m to 2.1 km (98.4 ft to 1.3 mi) apart, however, nests were typically found in close clusters. Nest cluster stands ranged from 10.5 to 114 ha (25.9 to 281.7 ac) in size. Nesting density in the Sierra Montane Forest was 1.07 pairs per 1,000 ha (2,471.05 ac), and 0.575 pairs per 1,000 ha for Upper Montane Forest (Woodbridge and Detrich 1994).

Migration

Goshawks are partial migrants. Some individuals maintain year-round occupancy of nest territories while others undergo seasonal movements to wintering areas (Squires and Kennedy 2006). Keane (1999) and Palmer (1988) noted that when prey was abundant, territorial birds remained in their home range year-round, or only made small local shifts. Little information exists on migratory timing and patterns in Western North America (Palmer 1988). Fledglings are believed to migrate first followed by older birds although timing overlaps (Mueller and Berger 1968, Palmer 1988, Squires and Reynolds 1997). Migrants probably follow low-pressure fronts and topographic features such as mountains that promote gliding and will migrate through extensive non-forested habitat. They will also migrate into extreme headwinds and during winter storms (Squires and Reynolds 1997). Western populations migrate across a broad front and may follow the Rocky Mountains (Squires and Reynolds 1997).

Movement is typically restricted to dispersal and short distance migration however, about every 10 years, goshawks from the northern part of the range migrate en masse much farther south, as far as San Diego County, in California, due to crashes of available prey such as the snowshoe hare (Mueller and Berger 1968, Mueller et al. 1977, Hoffman et al. 2002, Unitt 2004). During these occurrences, sex ratios typically fluctuate, for instance during an incursion in 1992-1993, more females were present than males (Meuller and Berger 1977, Squires and Kennedy 1997).

During fall migration counts in the Marin Headlands on the California coast, only ten individual goshawks were counted from 2001 to 2009 at the Golden Gate Raptor observatory and only five were banded from 1983 to 2010 (Fish and Hull 2010). Binford (1979) considered goshawk an occasional fall transient in the Marin Headlands and along the coast of Northern California and suggested that they were the latest migrants with average dates in November. Rowe and Gallion (1996) documented the first record of migrating goshawks in Kern County, California in the southern terminus of the Sierra Nevada Range at 900 m (2,950 ft) in elevation.

Historical and Current Distribution

Grinnell and Miller (1944) described goshawks as uncommon permanent resident in higher altitudes of the northern third of California, and south along the Coastal Ranges to Laytonville and Mount Sanhedrin, Mendocino County. They also included the Warner Mountains, Modoc County and south from Mount Shasta along the Sierra Nevada to Kings Canyon national Park and Whitaker's Forest, Tulare County. Known nesting occurred from 1,524 m (5,000 ft) in Trinity County to 2,743.2 m (9,000 ft) in Mono County (Grinnell and Miller 1944). Actual nesting records were reported near Yosemite Valley, Lake Tahoe, and Virginia Lakes and June Lake in mono County (Grinnell and Miller 1944). Some of the earliest records of goshawks in southern California were from Mount Pinos, in the Los Padres National Forest, in 1904 and 1919 (Kiff and Paulson 1997). Kiff and Paulson (1997) also report eggs collected in 1937 at 1,524 m (5,000 ft) in the Cuyamaca Mountains, San Diego County. The earliest record of goshawk in the Sierra Nevada is from Cooper (1870), who reported a bird shot at Michigan Bluff, Placer County. Multiple records of goshawk in the Central Sierras exist from the 1910's and 1920's (Ray 1926, Schnell 1958).

According to Dunk et al. (in prep. a) the current distribution of the species occurs throughout the historical range, with the largest concentrations occurring in the Northern and Central Sierra Nevada, Southern Cascades, Modoc Plateau, and Warner Mountains. Breeding range extends from about 305 m to 3,290 m (1,000 to 10,800 ft) in elevation and occurs throughout the Sierra Nevada as far south as Isabella Reservoir in the Tehachapi Mountains, Kern County and east to Glass Mountain and the White-Inyo Ranges in Mono County (Keane 2008, Dunk et al. in prep. a). Distribution in the northwest follows the montane forest zones extending south through the Coast Range of Mendocino County; however it is largely restricted to drier forest types on the slopes of the Klamath Mountains and northern Coast Ranges (Dunk et al. in prep. a).

From 1997 to 2003, 79 nest sites were documented in Plumas National Forest in northeastern California (Dunk et al. in prep. b). Nesting also occurs throughout the Lake Tahoe Basin (Keane et al. 2006, Slauson et al. 2008, S. Zannetti, USFS 2012 pers. comm.). Small (1994) reported goshawk as a rare to uncommon breeder in the mountains of Western Siskiyou County and Northern Coast Range. Harris (1996) reported breeding in Humboldt County and in some years the presence of dispersing juveniles in the coastal lowlands. Goshawks were also reported in the south-central portion and far eastern portion of Humboldt County on Humboldt Redwood Company timberlands and from the Del Norte County line to Dinsmore (Hunter et al. 2005). Three breeding attempts were confirmed in Humboldt County near the Trinity and Mad Rivers (Hunter et al. 2005).

Rare occurrences in the lowlands bordering the northern mountains and the Sacramento Valley have been reported (Small 1994). Goshawks were considered rare in Yosemite National Park, however nesting has been documented from 1,646 m to 2,621 m (5,400 ft to 8,600 ft) (Gaines 1988). They were casual fall transients and winter visitors to the Lower Colorado River Valley and an irregular breeder on high mountains in Ventura and Kern Counties (Rosenberg et al. 1991, Lehman 1994). Kiff and Paulson (1997) reported nesting in Ventura County on Mt Abel in 1989 and 1990 and Mt Pinos in 1989.

eBird (2012) lists a sighting in the Santa Cruz Mountains in 1985 and several sightings in the South Coastal Ranges including one near Salinas in 1992, and Hollister in 2006, and two east of highway 33 near San Benito Mountain Natural Area and Mercey Hot Springs in 1999 and 2011. During the summer, adults are found on Pine Mountain, Santa Barbara County; San Bernardino Mountains, and Clark Mountain, San Bernardino County; Peninsular Ranges near San Jacinto, Riverside County; and Inyo Mountains, Inyo County (Small 1994).

Historical and Current Abundance

Historical documentation of the goshawk in California is minimal and no quantitative studies of nesting densities or estimates of populations were conducted historically (Bloom et al. 1986). Grinnell and Miller (1944) considered goshawks uncommon and Ray (1926) described them as “very scarce” in the Tahoe region. In an eight year nesting study in Mono County, Dixon and Dixon (1938) only observed three goshawks other than the pair they were focused on however they described goshawks as a common winter visitant to San Diego County. Based on historically available habitat and estimated habitat loss from forest management activity, Bloom et al. (1986) suggested that the historical population was most likely 25% to 50% larger than their 1986 estimate of 805 breeding pairs.

Bloom et al.’s (1986) estimate of abundance was the first attempt in California and was based on extrapolations from “sample tracts” surveyed in defined geographic regions of the state and information gathered from observations by falconers, USFS, Bureau of Land Management (BLM), and National Parks Service (NPS). Bloom et al. (1986) estimated 1,320 breeding territories in California of which the approximated 805 were active each year with 733 producing at least one fledgling. Total estimated breeding territories by region summarized in Bloom et al. (1986) include: Southern Transverse Range, 5; Mojave-Colorado Desert, 5; Northern Coast Range-Klamath Mountains, 275; Southern Sierra Nevada-White Mountains, 515; Northern Sierra Nevada-Cascade Ranges, 425; and Great Basin, 95. A more recent synthesis of breeding territory records from federal and state agencies, private land management companies, museums, and published literature from 1970 to 2001 reduced the number of estimated territories to 1,000 (Keane 2008).

By modeling suitable habitat for breeding goshawks with GIS information, Dunk et al. (in prep. a) estimated 144 to 1,922 breeding territories statewide, with more than half of these territories occurring in the Sierra Nevada-Cascade region. In assessing the modeling results however, Dunk et al. (in prep. a) caution that data over ten years old for the Northern Coast bioregion may be unreliable due to intensive forest management which may have altered suitable habitat predicted by their model. In addition one should consider that the Dunk et al. (in prep. a) estimate predicts the total number of potential territories based on habitat suitability, and not on observed occupied territories or breeding pairs.

The Partners in Flight (PIF) landbird population estimate for goshawk in California is 3000 (95% confidence limits within 50% of the regional average) (Rich et al. 2004). The PIF estimate was based on Breeding Bird Survey (BBS) detections from 1990 to 1999 and was extrapolated from relative abundance at the route level. According to BBS criteria, goshawk data contains important deficiencies which make it imprecise (Sauer et al. 2011). A certain level of bias is associated with BBS roadside surveys since secretive species like the goshawk are not always detectable (Blancher et al. 2007). Goshawk was detected on only 9 of 200 BBS routes used to evaluate the California population estimate (Rich et al. 2004).

588 known territories were reported in the Sierra Nevada Forest Plan Amendment from ten Sierra Nevada National Forests and Management Unitis including: 130 in Modoc National Forest (NF), 113 in Lassen NF, 75 in Plumas NF, 52 in Tahoe NF, 69 in

Eldorado NF, 22 in the Lake Tahoe Basin, 34 in Inyo NF, 53 in Stanislaus NF, 12 in Sierra NF, 17 in Sequoia NF, and 11 Outside USFS boundaries (USFSa 2001).

Slauson et al. (2008) compiled goshawk records from 1977 to 2007 for the Lake Tahoe Basin (LTB) from CA state parks, Nevada Division of Wildlife, U.S. Forest Service (USFS) Lake Tahoe Basin Management Unit (LTBMU), and private consultants and found that 92 nests and 400 goshawk detections were recorded throughout the LTB over the 30 years examined. Records were concentrated in the mid and lower elevations and sparse on the west-central and southwest portions of the LTB possibly due to habitat suitability as well as survey effort (Slauson et al. 2008). Slauson et al. (2008) conclude that insufficient data exists for population trend analysis but goshawks are still relatively well distributed in the LTB. From 2009 to 2011 in the LTBMU, 12 separate pairs were reported as reproductively active in at least one of the three years and an additional 6 pairs were reproductively active at least once from 2002 to 2008. Four nonbreeding individuals were also reported (S. Zannetti, USFS 2012 pers. comm.).

Based on the rarity of breeding records, the potential for extirpation exists in the Coastal Redwood zone (Keane 2008). Weber (2006) found no evidence of nesting in eight historic nesting sites and many other potential sites in old-growth forests, and no nesting was documented in the Redwood zone proper. One active nest was found in a historical site at Angelo Coast Range Reserve, Mendocino County at the transition from Redwood (*Sequoia sempervirens*) to xeric Douglas fir and mixed–oak (*Quercus* spp.) habitat (Weber 2006).

Extensive railroad logging in the Eastside pine zone eliminated goshawks from vast areas of pine forest in this region however, Dunk et al. (in prep. a) and Keane (2008) maintain that breeding goshawks are still abundant in much of Northeastern California. In the Westside Ponderosa Pine zone on the entire lower west slope of the Sierra Nevada, logging has eliminated 90% to 95% of the original old-growth Pacific Ponderosa Pine forests causing the likelihood of goshawk numbers reduction (Franklin and Fites-Kaufmann 1996, Keane 2008). The occurrence of only 25 to 30 recent nest records causes need for concern in the Southern Sierra Nevada south of Yosemite National Park; however, this may represent low breeding density, suboptimal habitat at the edge of goshawk range, or limited survey efforts in this area (Keane 2008, Dunk et al. in prep. a).

Finally, nest records in the Southern California mountain ranges are few and of the 25 adult sightings recorded during the breeding season over the last 50 years, half occurred in the San Jacinto Mountains (Keane 2008). From 1987 to 2003 however, only two records in the San Jacinto Mountains, one active nest and an aggressive pair of adults, were reported during the breeding season of 1990 to 1991 despite extensive surveys for California Spotted Owl which use similar habitat (Kiff and Paulson 1997, Keane 2008). Active nests were also reported on Mt. Abel (1989 to 1990), Mount Pinos (1989), and the Tecuya Range (1991) in Northern Ventura County (Keane 2008).

Population Trend

Population trends for goshawk in California are difficult to determine due to the species' elusive nature and distribution across broad, rugged landscapes at low breeding densities (Bloom et al. 1986, Squires and Kennedy 2006). A complete multi-year census of all breeding and non-breeding individuals would be difficult, if not impossible, due to low detection rates, undetected non-breeding birds, and large home ranges. There are currently no effective methods for surveying non-breeders, although they may constitute a significant portion of the population (Squires and Kennedy 2006). Studies in Sweden and Finland suggested that one third to half of the population were non-breeders (Widén 1985, Lindén and Wikman 1983). Another compounding factor is that density fluctuates annually and spatially within populations (Kennedy 1997, Squires and Reynolds 1997).

Few long-term studies designed to determine population changes over time have been conducted in California. Abundance estimates available for California all rely on considerable extrapolation to arrive at the results, and therefore some error is associated. Regardless, looking at these estimates, the goshawk population has remained relatively stable or has slightly increased over the last 20 years. Other studies that have been conducted in California may be used as models for estimating statewide populations but because they focus on individual populations, they may represent ecological density rather than crude density. Therefore, broad scale application may result in inaccurate estimates (Hall 1984, Woodbridge and Detrich 1994, Hargis 1994, Keane 1999, Richter 2005, Squires and Kenned 2006).

BBS data indicates an increasing California population (1.7 % 1966 to 2000; 3.7% 2000 to 2010) (Figure 1, Sauer et al. 2011, NAS 2011). However, BBS data credibility show an important deficiency for goshawk due to the low number of encounters (e.g. low abundance) per route and small sample size, both leading to imprecise trend results. Christmas Bird Count (CBC) data indicate a stable nonbreeding population (±0.2%) in California, with a slight decline in detections in recent years (Figure 2). CBC and BBS data rely heavily on vocal and roadside surveys which may misrepresent secretive species such as goshawk that occur in forested montane (Blancher et al. 2007, Rosenberg 2004). For instance, high elevations may be underrepresented by road surveys because roads tend to follow valley bottoms in mountainous regions (Rosenberg 2004).

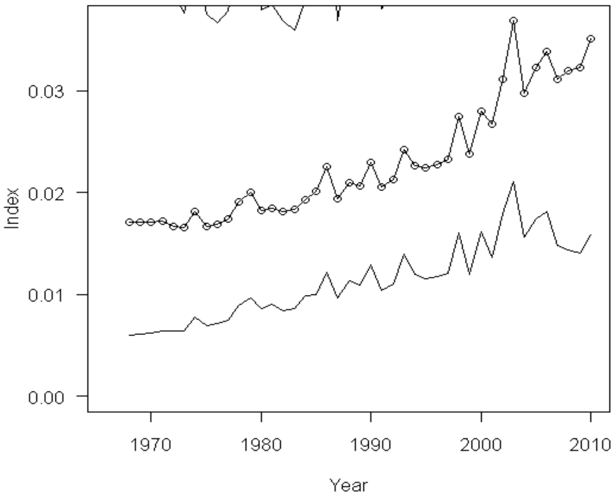

Figure 1. Breeding Bird Survey trend estimate index presented as percent change per year (middle line) for Northern Goshawk in California from 1966 to 2010. 2.5% and

97.5% posterior distribution of trend estimates represented by top and bottom line (Sauer et al. 2011).

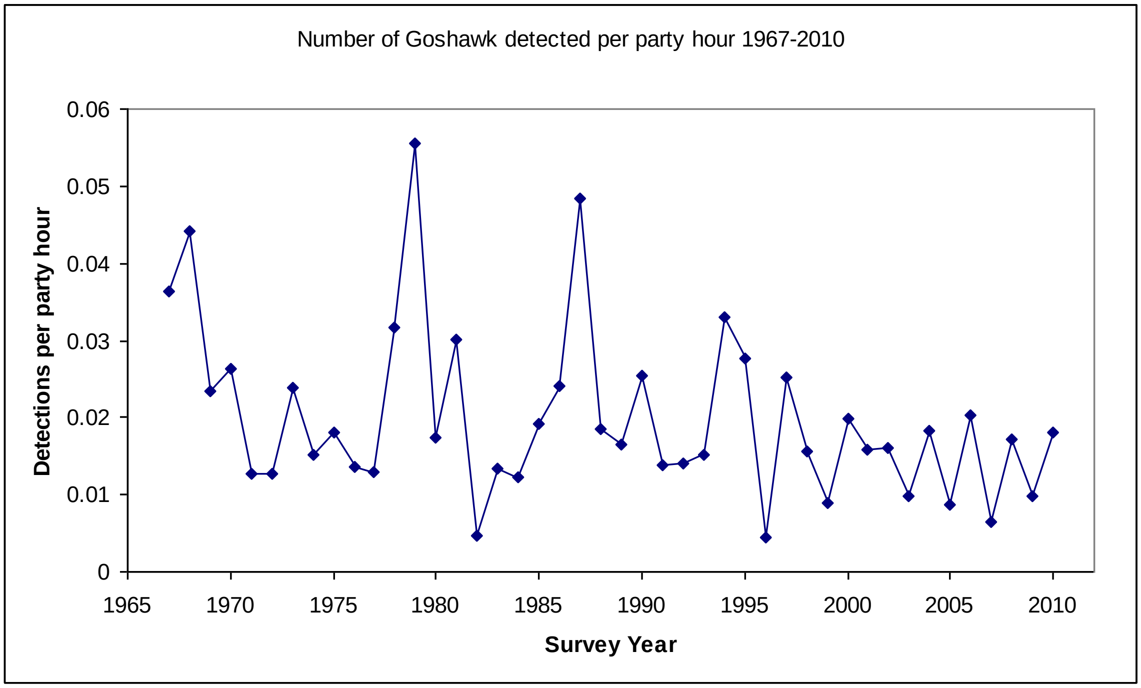

Figure 2. Number of Northern Goshawk detected per party hour on Audubon Christmas Bird Counts in California from 1967 to 2010 (NAS 2011).

Threats

Habitat alteration due to timber harvest and fire management is the primary threat to goshawk populations (Squires and Kennedy 2006). The alteration of historic, natural disturbance regimes as a result of timber harvest and fire suppression policies over the last century has changed the structure, composition, and function of contemporary California forests (Keane 2008). Forest management activities and outcomes, such as fragmentation, reduction of old-growth forest stands, removal of snags, downed trees, canopy and mature trees, increased understory density, and shifting tree species composition (e.g. broad-scale reduction of pine in forest stands) all degrade goshawk habitat (Keane 2008, Squires and Kennedy 2006). Fragmentation and reduction of old-growth may also increase threats from competitors and predators such as red-tailed hawk, long-eared and great-horned owl (Woodbridge and Detrich 1994, La Sorte et al. 2004, Squires and Kennedy 2006). In Arizona, following logging activity, long-eared owl, great horned owl, and red-tailed hawk occupied former goshawk nests. None of these three species was observed within one km of any goshawk nest tree in control sites with no logging activity (Crocker-Bedford 1990). Boal and Mannon (1994) found that Great Horned Owl predation was the greatest contributing factor to nestling mortality in Arizona.

Today, fire management standards focus on the nest stand and a surrounding buffer zone (CalFIRE 2012). Management practices around goshawk territories vary and buffers range widely (Keane 2008). The SNFP as amended in 2004 requires protection of at least 81 ha (20 ac) of the best available forested habitat in the largest contiguous patches around known and newly discovered breeding territories, while California Forest

Practice Rules (CFPR) (2012) require a minimum of 5 acres (~2 ha), unless justification is provided to do otherwise, with a maximum buffer zone of 20 acres (~8.09 ha) (USFSa 2001, USFSb 2004). However, logging outside of buffer zones in surrounding forests may not only reduce prey but result in a less suitable nesting and foraging habitat (Crocker-Bedford 1990). Logging operations in the vicinity of nests can also cause nest failure (Boal and Mannon 1994, Squires and Reynolds 1997). Richter (2005) found that after four years, more than half (54% N=106) of territories monitored on managed timberlands did not meet CFPR criteria for having an active nest site; 80% of these had minimal buffers from 2 ha to 8 ha (Richter 2005). USFWS (1998) also concluded that management in California focused on 2 ha to 20 ha (4.9 to 49.4 ac) buffers around nests are inadequate.

Stand replacement fires in forests altered by management and fire suppression can result in the complete loss of goshawk nest stands (Keane 2008). Likewise, increased understory density due to fire suppression reduces foraging quality and can increase fuel load and amplify natural fire intensity (Gruell 2001, Keane 2008). On the other hand, some fire management may be beneficial to goshawks and their prey by opening up dense understory vegetation, creating snags, downed logs, and other woody debris (Reynolds et al. 1992, Graham et al. 1999, Squires and Kennedy 2006). Effects of forest-management practices on goshawk habitat at the home range and landscape scale are still under investigation and a level of uncertainty remains (Squires and Kennedy 2006, Keane 2008).

Negative effects on goshawk populations from recreational activities have been documented. For example, camping near nests can lead to nest failure (Speiser 1992). In Plumas National Forest, northeastern California, female goshawks spent more time off the nest during intensive all terrain vehicle (ATV) experimental treatments than in control treatments and had similar responses when nests were directly approached by hikers, especially 2 hikers (Dunk et al. in prep. b). Fledglings were also found to use areas with lower habitat suitability during sustained-ATV treatments (Dunk et al. in prep. b).

Coupled with increased recreation is human population increase and expansion of development, especially on the west slope of the Sierra Nevada (Keane 2008). As forests are converted to urban and exurban developments habitat quality is degraded, direct loss of habitat may occur, and the risk of human induced fires is increased (Stein et al. 2005). Indirect effects of urbanization include increased numbers of humans in forested habitats and higher traffic along urban fringe that can disturb foraging and nesting goshawks (Slauson et al. 2008). In the Lake Tahoe Basin, Morrison et al. (2011) found that infrequently occupied territories had 75% more human activity than frequently occupied territories and reproductive success was significantly higher in frequently occupied territories with 0.5 more fledglings per nest. Furthermore, the extent of all roads and trails was twice as great in infrequently occupied territories than frequently occupied (Morrison et al. 2011).

Large mammals such as marten pose a predatory threat to goshawk as do eagles, great horned owl, spotted owl, and red tailed hawk (Paragi and Wholecheese 1994, Boal and Mannon 1994, Woodbridge and Detrich 1994, Doyle 1995, Squires and Ruggiero 1995, Woodbridge and Detrich 1994, La Sorte et al. 2004).

Weather patterns may indirectly affect goshawks by causing fluctuations in primary production and prey species abundance. Weather driven changes in food supply from climatic patters like El Niňo Southern Oscillation limit prey species abundance especially through influence on the survival of young (White 2008). Rutz and Bijlsma (2006) and Wiens et al. (2006a) showed that goshawk reproduction is directly linked to prey abundance. In the Lake Tahoe region of the Sierra Nevada, Keane et al. (2006) found that goshawk reproduction was associated with Douglas squirrel abundance as well as late-winter and early-spring temperature fluctuation. Goshawk reproduction was reduced following years with lower temperatures in late-winter and early-spring which were associated with low Douglas fir cone crop production and lower winter and spring Douglas squirrel abundance (Keane et al. 2006).

Several diseases and parasites affect goshawks including: tuberculosis (*Mycobacterium avium*), erysipelas (*Ersipelas insidiosa*), heart failure caused by *Chlamydia tsittaci* and *E. coli*, and *Aspergillus* (Redig et al. 1980, Lumeij et al. 1981, Schröder 1981, Ward and Kennedy 1996). Ectoparasites and lice occur in goshawks (Keymer 1972). Blood parasites including *Leucocytozoon*, *Haemoproteus*, *Trypanosoma*, and *microfilariae*, were present in approximately 56% (n=36) of North American goshawks (Greiner et al. 1975). The protozoan parasite *Trichomonas gallinae* transmitted through predation on pigeons can cause death and reduced productivity (Cooper and Petty 1988). West Nile Virus is also capable of causing fatal heart and central nervous system lesions in goshawks (Wünschmann et al. 2005).

Captive goshawks are at risk of contracting the fungal respiratory disease Aspergillosis which is caused by the inhalation of spores produced from items such as moldy hay, and is aggravated by stress (Abundis-Santamaria 2003). Poor sanitation or ventilation conditions, shipping, heat, capture, changes in management, and lowered immune response increase risk of contraction. The acute form which is seen most often in wild birds rapidly leads to death in less than one week. Prolonged treatment for the chronic form is generally ineffective. Prevention includes good ventilation in housing facility, daily bedding changes, and general good management of birds. Vaccines exist but little research has been conducted as to their effects (Abundis-Santamaria 2003).

Goshawks have been used for falconry for 2,000 years and were more prized than any other bird from Turkey to Japan however in North America they were historically considered unfavorable (Squires and Reynolds 1997, Palmer 1988). Impacts from falconry are unknown but do not appear to pose a great threat to California's statewide goshawk population (Squires and Reynolds 1997, Squires and Kennedy 2006, Keane 2008). However, over-harvest and excessive disturbance may negatively impact local populations by causing nest failure and decreasing productivity below sustainable levels (Keane 2008). The cumulative effects of illegal goshawk take for falconry purposes have not been analyzed. It is reported that illegal take occurs by unlicensed falconers, and illegal removal of all nestlings also occurs. These activities may be most harmful when concentrated in frequently harvested areas, e.g. Mono, Inyo, and Modoc Counties (CDFG 2012).

Data gaps and monitoring

Keane (2008) outlines several research and management recommendations, including: developing site-specific protection around nest and alternate nest trees; incorporation of new information into management standards and guidelines; establishment of a statewide nesting record database; specialized inventories in poorly known areas; further radio telemetry research to understand foraging habitat and prey use; continued demographic research to understand factors affecting survival and reproduction; and collaboration between researchers and managers in an adaptive management framework to assess the effects of forest and fuels management policies with special focus on appropriate spatial scales.

Monitoring needs recommended by Keene (2008) include standardization of protocols to assess population distribution, status, and trends at the appropriate scale; using US Forest Service monitoring design to assess trends associated with broad-scale habitat changes (Hargis and Woodbridge 2006); utilizing empirical models to monitor changes in habitat at home-range and landscape scales; and monitoring project-level responses to management treatments of nesting goshawks. Richter (2005) recommends an adaptive management approach coupled with extensive monitoring to determine the ability of mitigation to negate impacts and examine the issue of territory viability on long-term, large-scale managed versus unmanaged timberlands. Studies of goshawk habitat relationships in forests that are not managed for timber such as protected areas are needed and can provide baseline data for comparisons (Squires and Kennedy 2006; See also Bosakowski 1999 and Squires and Kennedy 2006 for extensive recommendations and expansions on these themes.).

Non-breeder contribution to population dynamics warrants further investigation as these individuals may buffer populations during times of stress, stabilize breeding population abundance by filling gaps caused by mortality to breeders, and serve to quickly increase breeding density in years with ample food supply. However, no effective survey methods currently exist for assessing the density of non-breeders (Squires and Kennedy 2006). Winter densities are also currently unavailable (Squires and Reynolds 1997, Squires and Kennedy 2006).

Potential impacts of diseases such as West Nile virus and other emerging infectious diseases are also poorly understood and require further investigation as they could play a role in regulating some goshawk populations (Squires and Kennedy 2006).

Prey habitat requirements and use patterns, prey demographic response in survival and reproduction to variation in forest structure and composition, and the direct affects of prey dynamics on goshawk fitness should be a focus of investigation (Keane 1999, Squires and Kennedy 2006, Keane et al. 2006). Comparable studies focused on ecological relationships, both direct and indirect, between primary production (i.e. Douglas-fir cone production) and goshawk-prey relationships, in areas such as the Westside Sierra Nevada mixed conifer forests between 750 to 1500 m (2046.6 to 4921.3 ft.) elevation and eastside pine forests which have been highly affected by human management activities, are especially needed (Keane et al. 2006, Squires and Kennedy 2006). A combination of ecosystem and landscape level perspectives and investigations will further our understanding of the relationships between population patterns and system wide structure (i.e. mature vs. young forest) and function (i.e. nesting vs. foraging sites) (Keane et al 2006).

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ECOLOGY OF THE MERLIN IN CALIFORNIA

By Madeleine Wieland

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The merlin (*Falco columbarius*) is a small, agile falcon native to Europe, Asia, and North America. Three subspecies of merlin occur in North America: Richardson's or prairie merlin (*F. c. richardsonii*), black merlin (*F. c. suckleyi*), and taiga merlin, also known as the boreal or northern merlin (*F. c. columbarius*). The prairie merlin is the lightest in coloring of the three subspecies while the black merlin is markedly darker than the other two. All three are known to occur in California.

In 1978, the Merlin was labeled as "highest priority" on the California Bird Species of Special Concern list (Remsen 1978). Shuford and Gardali (2008) removed the merlin from the list after numbers of migrant and wintering merlins increased in California. The International Union for the Conservation of Nature (IUCN) described the species as "least concern" on the 2004, 2008 and 2009 Red Lists citing a large range and high population numbers.

The merlin can be found in every province and state in North America at some point in time throughout the year (Bent 1938). During the breeding season, these birds can be found throughout Canada and Alaska except in the extreme north above the treeline (Bent 1938, Friedmann 1950). This bird is also a rare breeder in the northern United States. Wintering merlins can be found throughout the western U.S. from the Rocky Mountains west to California, Oregon and Washington (Peeters 2005). Merlins have also been known to winter along the Gulf of Mexico and many of the southern states along the East Coast (Bent 1938, Clark 1985). Central and eastern states where merlins neither breed nor winter may see migrant birds as they travel from one range to the other (Friedmann 1950).

The taiga merlin has the largest breeding range stretching throughout the entire boreal forest from northwestern Alaska to the east coast of Canada as well as parts of parts of Montana, Idaho, Washington and Oregon (Trimble 1975, Friedmann 1950, Warkentin et al. 2005). Prairie merlins breed on the northern Great Plains spanning from the southern edge of the Canadian boreal forests in Alberta, Saskatchewan, and Manitoba down into Montana and the Dakotas (Bent 1938, Trimble 1975, Friedmann 1950). Black merlins breed primarily in the coastal forests of British Columbia, but sometimes nest in the southernmost tip of Alaska and in coastal regions of northern Washington (Bent 1938, Oliphant 1985, Laing 1935).

One of the smallest falcons in North America, the merlin has a wingspan of 53 to 68 cm and a length of 24 to 30 cm. Like many raptors, they display reversed sexual size dimorphism where the females are generally larger than the males (Temple 1972b). The average male weighs only 160 to 170g while the female weighs 220 to 240g (Warkentin et al 2005). The species has a dark gray-blue beak and a bright yellow cere. The "mustache" facial mark that is typical to other falcons is usually present but much fainter in the merlin and may be absent in the lightest individuals. The crown is light brown to almost black depending on the subspecies; a white or cream superciliary line extends over each dark brown eye giving the appearance of a pale eyebrow. The breast and underside of wings are cream colored with brown streaks. Frequently, the sides of the breast have a rusty wash. The tail is long, rounded at the end, with a white band at the tip. Varying numbers of lighter bands extend the length of the tail depending on the subspecies. In males, the dorsal feathers range from a pale, slate blue (in the prairie merlin) to a dark gray approaching black (in the black merlin). In females, the dorsal plumage is typically a muted brown. Males can also be distinguished by rufous feathers on their legs. In both sexes, the legs and feet are yellow, and in breeding males the feet and legs are orange. Male and female juvenile merlins resemble the adult female in coloration (Trimble 1975, Palmer 1988, Warkentin et al 2005).

The merlin gained popularity among falconers as a “lady’s hawk” in medieval Europe and was a favorite of both Catherine the Great of Russia and Mary, Queen of Scots, and is still a popular species in falconry today (Warkentin et al. 2005). During 2006 through 2010 in California, 44 wild merlins were harvested for falconry; of these, 40 were immature and 4 were of unknown age. Since merlins are found in California in the non-breeding season, no nestlings were harvested. Harvest was dispersed throughout the state, with the highest numbers taken from Solano, Santa Barbara, and Riverside counties (7, 7, and 6 respectively), accounting for nearly half of all harvested individuals. While in captivity, 6 merlins died. Reported causes of death included the following: proventriculitis, aspergillus, heat exhaustion, and predation by a red-tailed hawk. All six were immature birds. Twenty-three merlins were released back into the wild between 2006 and 2010, and 6 were lost (CDFG 2012).

Habitat

Merlins prefer to nest in partially wooded locations that provide both nest protection and open areas for hunting. Merlins rarely build their own nests. Instead, they will use old raptor and corvid nests, particularly those of common crows and black-billed magpies (Warkentin and James 1988). Nest sites tend to be close to watercourses or lakes (Craighead and Craighead 1940, Lawrence 1949, Oliphant and Thompson 1978).

Foraging habitats can vary widely. During the breeding season, male merlins have been found hunting as far as 9 km away from the nests in multiple habitats types (Becker and Sieg 1987). The preferred foraging habitats of birds in one Montana study include ponderosa pine habitat, riparian areas and mixed sagebrush and grass habitat (Becker and Sieg 1987). A study in Alaska found that merlins foraged in dense spruce forest as well as sparse alpine tundra (Laing 1985). Merlins wintering on the coasts of California and Washington have been reported foraging in estuaries where shorebirds are abundant (Page and Whitacre 1975, Buchanan et al. 1988). Gaines (1988) reported that merlins in Yosemite will hunt anywhere except dense forests as long as there are small birds to be eaten. Merlins have even been known to hunt within towns and residential areas (Rosenberg et al. 1991).

Across their range, wintering merlins will roost in trees near open areas where small flocking birds are abundant such as grasslands, marshes, dune systems and shorelines (Palmer 1988). Favorite perches include fence posts and trees (Fox 1964). Merlins wintering within the city of Saskatoon, Saskatchewan often perch on utility poles and buildings as well (Warkentin and Oliphant 1990).

Wintering and migrant merlins in California use a broad range of habitats. Small (1994) stated that wintering habitat consists of coastal areas, tidal estuaries, woodlands, grasslands, edge habitat associated with grasslands and deserts, and farms and ranches. Patches of trees were important habitat for roosting. Gaines (1988) also reported seeing merlins in woodlands as well as willow thickets, mudflats, meadows and sagebrush scrub. Grinnell and Miller (1944) reported that the wintering habitat of the taiga merlin was “chiefly open type of woodland; foothills and valleys, often where scatteringly grown to blue oaks, black oaks, live oaks or valley oaks." Peeters (2005) adds farmlands, dairies, savannahs and the edge of deserts to the list of habitats merlins use in California.

Migration

Each subspecies of merlin has its own distinct pattern of migration. Black merlins are the least migratory. Most individuals of this subspecies are resident within range. Generally, only those birds at the northernmost edge of their range migrate south for the winter (Warkentin et al. 2005). Those individuals that do migrate south tend to stay close to the Pacific coast. The majority of migrating black merlins winter in Washington, Oregon and northern California. However, black merlins have been reported as far south as New Mexico and Southern California and as far east as Wisconsin (Friedmann 1950, Palmer 1988).

Nearly all prairie merlins migrate south to overwinter in Wyoming and Colorado and are sometimes seen as far south as Texas and northwestern Mexico (Bent 1938, Warkentin et al. 1990, James et al 1987). This subspecies is rare in California (Friedmann 1950). The presence of nonmigratory merlins in cities of the Great Plains is a fairly recent phenomenon and may be connected to the high number of conifers now available due to human planting (Warkentin and James 1990).

Taiga merlins, whose breeding grounds have the harshest winters, are almost completely migratory. When migrating, they pass over the breeding and wintering ranges of the prairie merlin to reach warmer regions of the southern United States, Central America, the Caribbean, and even the most northerly parts of South America (AOU 1957, 1998, Clark 1985). Taiga merlins traveling south from Alaska and western Canada winter along the coast of British Columbia and south through the western United States (Friedmann 1950, Trimble 1975).

In general, female merlins migrate south for the winter before the males, possibly because females require larger prey and are following the migration of the birds they hunt (Clark 1985). In the spring, males return to the breeding range first. Aft Cape May Point, New Jersey, Clark (1985) theorized that merlins use migratory flight strategies similar to those of peregrine falcons. Like peregrines, they may use powered flight in the early hours of the morning and switch to gliding flight using afternoon thermal updrafts. Later in the day as thermals weaken, they probably fly at lower altitudes to facilitate hunting. Merlins flew later into the day than other raptors and also continued migration on overcast days when thermals were weak or nonexistent.

Raptor counts conducted by the Golden Gate Raptor Observatory in Marin County, California, showed that merlins begin migrating south along the coast of California in September (Figure 1). The number of merlins passing by the observatory peaked in late October and declined through November (Elliot and Fish 2010). The Golden Gate Raptor observatory does not have a spring migration profile for the merlin but Lehman (1994) states that most merlins have left San Diego county by the beginning of April and one record shows a merlin lingering until April 30. Elsewhere, data have shown that spring migration may occur over a shorter time period than fall migration. Bent (1938) stated that spring migration of Merlins in New England occurs in the last two weeks of April and the first two weeks of May. This is corroborated by Clark (1985) who found that most merlins migrating through New Jersey had passed through raptor banding stations between April 20 and May 5.

Figure 1. Migration profile for the merlin based on 25 years of data from the Golden Gate Raptor Observatory in the Marin headlands, California. This graph shows the average number of merlins sighted per hour throughout the fall migration season (Elliot and Fish 2010).

Food Habits

Preferred prey species of the merlin are birds that are locally abundant and that forage in the open. Small birds weighing less than 50 g make up the largest part of the merlin's diet (Becker 1985, Laing 1985, Warkentin et al. 2005). Within the city of Saskatoon, Saskatchewan, the house sparrows (*Passer domesticus*) was the most common prey species; on the Great Plainsof Saskatchwan and Montana the horned lark (*Eremophila alpestris*) and the lark bunting (*Calamospiza melanacorys*) were the primary prey species (Fox 1964, Becker 1985, Sodhi and Oliphant 1993). Merlins wintering along the coasts of California and Washington hunt mostly sandpipers (Page and Whitacre 1975, Buchanan et al. 1988). Insects comprised a much smaller percentage of the merlin's diet but may be a seasonally important source of food for recent fledglings learning how to hunt (Lawrence 1945, Fox 1964). Though much less common, merlins also prey on small mammals, reptiles and amphibians (Bent 1938).

Peeters (2005) reports that merlins in California will perch at the tops of snags and trees in the morning and search for prey. They may also try to flush prey by skimming over mudflats, grasslands and salt marshes (Peeters 2005).Page and Whitacre (1975) documented several hunting attempts made by a lone, female merlin wintering on California's central coast. After locating potential prey from a perch, this female surprised her prey by approaching in a low glide, seizing shorebirds either directly from the ground or as they tried to rise in flight. In contrast, Buchanan et al. (1988) reported that merlins in Washington most commonly attacked flocks of birds that were already in flight, diving into a group of shorebirds from a great height, splitting the flock into two or more smaller flocks, thus making it easier to isolate individual birds. Although attacking birds on the wing was the most common tactic, Buchanan et al. (1988) also reported a strategy similar to the one recorded by Page and Whitacre (1975). Both studies suggested that surprise attacks are an integral part of the merlin's hunting strategy. Merlins also take young birds directly out of nests. Lawrence (1949) documented the capture of an eastern kingbird nestling by a merlin; Kermott (1981) witnessed merlin predation on two robin nestlings.

After catching a bird, merlins kill their prey by severing the spinal cord (McClure 1957, Warkentin et al. 2005). Male merlins hunting for their mates may remove the head, legs and feathers of prey near the site of capture, reducing the amount of weight carried back to the nest (Sodhi 1992). Merlins may cache their food. Uneaten birds and mammals may be placed on a tree branch or in a squirrel's nest to be consumed at a later time (Pitcher et al. 1979, Warkentin and Oliphant 1985). During the breeding season, a male may cache a bird if he brings it to his mate and she refuses it (Palmer 1988, Sodhi in Warkentin et al. 2005).

Breeding

According to Oliphant (1985), there are no historical records of merlin nests in California. Fisher (1893) mentioned that merlins living in California build their nests in April but cited no specific records of nesting. Dixon and Dixon (1938) reported a merlin seen multiple times in Mono County in June that appeared to be nesting, though they were never able to find a nest. Although there may be unconfirmed accounts of breeding in California, there is no substantiated evidence that merlins have ever nested in California. Merlins inhabiting areas of California are found during the wintering and migration months only.

Merlins are typically monogamous throughout a breeding season but do not mate for life (Warkentin et al. 1991). This may be a product of differing levels of site fidelity between the sexes. Males often return to the same breeding grounds in successive years while females are much less likely to do so (Wiklund 1996). Merlins may switch mates within a single breeding season if the eggs are unsuccessful or their mate dies. Sodhi (1989) documented a single case of polygyny in which a male merlin simultaneously fathered broods with two different females during one breeding season.

Migrating males return to the breeding grounds in late winter or early spring and seek out a potential breeding territory. Female merlins follow about a month later (Fox 1964, Becker and Sieg 1985, Clark 1985). Males attempt to attract prospective mates to nest sites by performing aerial courtship displays described in detail by Feldsine and Oliphant (1985).

After a breeding pair is established, the male and female will copulate for many weeks before laying eggs. Eggs are usually laid between April and May. The female will lay between 4 and 6 eggs at an interval of one egg about every 2 days (Palmer 1988). Becker and Sieg (1985) noted breeding merlins in Montana had an average clutch size of 4.3 eggs. During incubation, the male merlin provides all of the food for the female who rarely leaves the nest except to consume prey supplied by the male. While the female is briefly away from the nest during feeding, the male will incubate the eggs (Oliphant 1974, Laing 1985). Usually, only one set of eggs is laid per season, however, a second clutch may be laid if the first set fails or is removed from the nest (Bent 1938).

The eggs are incubated until they hatch 31 to 32 days after the final egg is laid (Ruttledge 1985). The female broods the nestlings consistently for the first week after they hatch. During the nestling period, the male continues to provide all of the food for the female and the young. Though the female spends increasingly more time away from the nest, she does not hunt until after the nestlings have fledged (Oliphant 1974, Laing 1985, Warkentin et al. 1991).

Oliphant and Tessaro (1985) reported that hand-raised nestlings fledged about a month (29-34 days) after hatching. These young began hunting first insects and then birds somewhere between 40 and 50 days old and had become completely independent by 60 days old. It is suspected that wild merlin nestlings probably fledge and disperse a few days sooner than the hand raised merlins (Oliphant and Tessaro1985).

Home Range

There are no data on wintering range sizes of merlins in California. Urban merlins wintering in Saskatoon, Saskatchewan had highly variable home range sizes with a mean of 19.63 km^2 (196.3 ha). The smallest range was a mere 9.40 km^2 (94.0 ha) while the largest was 39.84 km^2 (398.4 ha) (Warkentin and Oliphant 1990). Winter home range sizes of rural merlins have not been studied.

The size of the merlin's home range is related to the abundance of prey in their habitat. Areas rich in prey result in smaller hunting ranges (Sodhi 1993), likely due to the shorter flight distance to acquire an adequate amount of food. Another possible determinant of home range size is familiarity with the area. Sodhi and Oliphant (1992) reported that resident merlins living within the city of Saskatoon, Saskatchewan year-round had smaller hunting ranges than immigrant birds that only lived in the city during the breeding season. This study reported a mean hunting size for resident males of 6.3 ± 1.3 km^2 and 33.7 ± 12.1 km^2 for immigrant males. All of the immigrant males spent at least some of their time hunting outside of the city while most of the resident males stayed within the city.

While merlins have been known to vigorously defend their nest area from potential predators such as other birds and humans, most do not defend their entire home range. Overlaps in home ranges have been shown in both urban and rural populations of merlins (Sodhi and Oliphant 1992, Becker and Sieg 1987). Since small birds are the main prey of merlins, hunting them may involve giving chase over an unpredictable distance and area, making it difficult to defend a set territory.

Historical and Current Distribution

Merlins do not nest in California but occur as migrants and regular winter visitors. Historically, merlins could be found in almost every region of California between the months of September and May. According to Grinnell and Miller (1944) wintering and migrant black merlins concentrated along the coast and in the northern regions while prairie merlins were more likely to be found in southern California. Taiga merlins had been found throughout the length of the state but were more common west of the Sierra Nevada mountain range (Grinnell and Miller 1944).

Currently, the range for wintering and migrant merlins in California is the same. They can be found across the western half of the state and are only absent from elevations above 3,900 ft, dense forests and open desert (Garrett and Dunn 1981, CWHR 1999). In Southern California, merlins can be found along the coast and in the high valleys (Garrett and Dunn 1981). In Northwestern California, merlins congregate in the lowlands along the coast but there are some records of these birds farther inland (Harris 1991).

Historical and Current Abundance, Population Trends

In California, Christmas Bird Count (CBC) data showed merlin detections increased significantly between 1960 to 2010 (Figure 2; NAS 2012). In San Diego County, wintering merlins increased during the 1990's (Unitt 2004). Daily counts from the Golden Gate Raptor Observatory, Marin County (Figure 3) also corroborate an increasing population of merlins. Migration counts of merlins increased every year through 2006, with a slight decrease since 2007. According to data submitted to eBird (http://www.avianknowledge.net/content/datasets/projects/ebird, retrieved February 17, 2012), the taiga merlin is by far the most common subspecies of merlin found in California during the winter months. Wintering and migrant prairie merlins and black merlins have also been reported throughout the state, though less frequently (Munson et al. 2011).

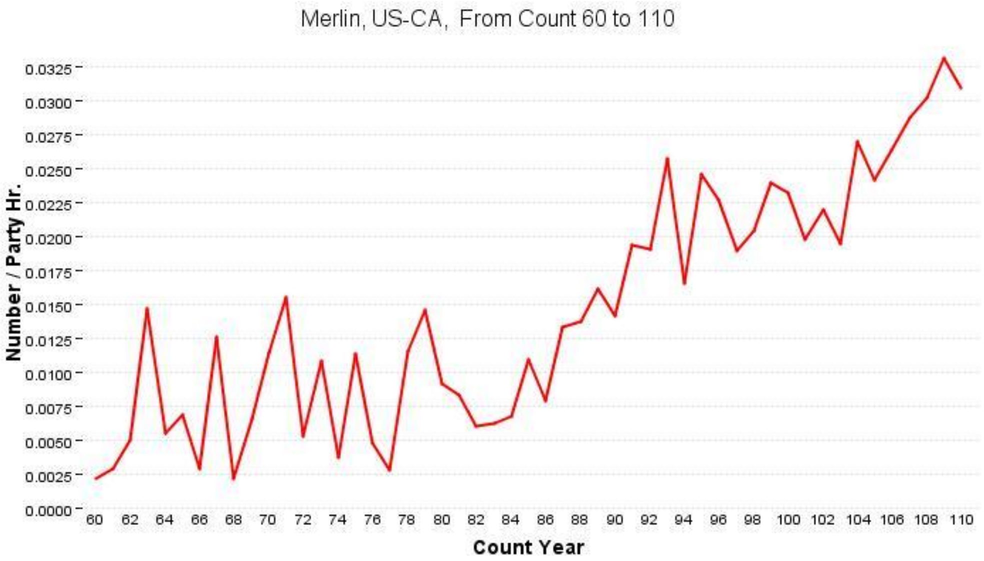

Figure 2. The number of merlins counted per party hour on Christmas Bird Counts in California from 1960 to 2010. Yearly changes in the number of circles or surveyors in a party are accounted for by dividing the number of birds seen by each party by the number of hours the party spent surveying (NAS 2012).

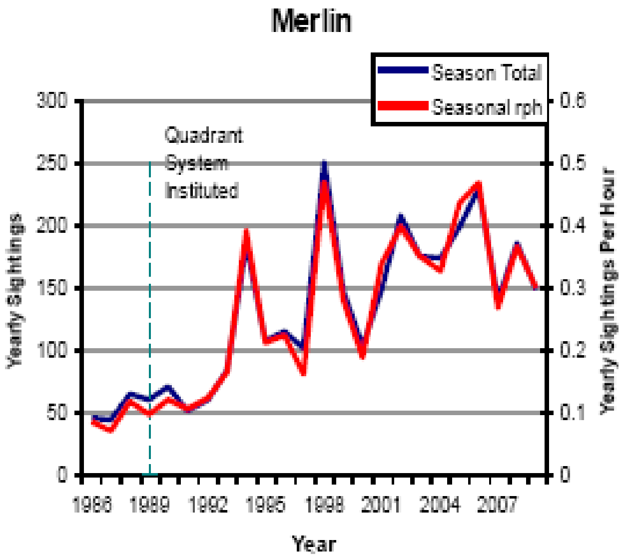

Figure 3. Daily hawk counts at the Golden Gate Raptor Observatory in the Marin headlands, California from 1986-2009. (Elliot and Fish 2010).

In the 1950's and 1960's, merlins in North America experienced a significant population decline, particularly taiga merlins living in eastern Canada and prairie merlins across their range (source). Egg shell thinning due to pesticide (chlorinated hydrocarbons e.g., DDT) contamination and loss of habitat were likely factors for this decline (Hickey and Anderson 1968, Fox 1971, Temple 1972a, Trimble 1975).

Currently, merlin numbers in North America are increasing. According to the Breeding Bird Survey (BBS; Figure 4), the North American trend index for this species between 1966 and 2010 is 2.9 (n = 520, CI = 1.2, 4.2) showing a steady increase in merlin numbers across the continent (Sauer et al. 2011). In recent years, the increase has been more dramatic. Between 2000 and 2010 the trend index was 4.6 (CI = 1.3, 7.2) (Sauer et al. 2011). Analyses of CBC and BBS data from the 1960s through the 1980s showed that most taiga and prairie merlin populations had stabilized or were increasing throughout this period (Kirk and Hyslop 1988). Data on black merlin numbers before and after 1950 is lacking. Hawk migration monitoring sites reported increases in merlin sightings between 1972 and 1987 (Titus and Fuller 1990). Farmer et al. (2008) performed a comparative analysis of BBS, CBC and data from monitoring sites and concluded that merlin numbers have “increased dramatically” since the 1970’s and 1980’s. Currently, it is estimated that there are approximately 600,000 merlins in North America (Rich et al. 2004). There are no data on the population size of wintering and migrant merlins in California.

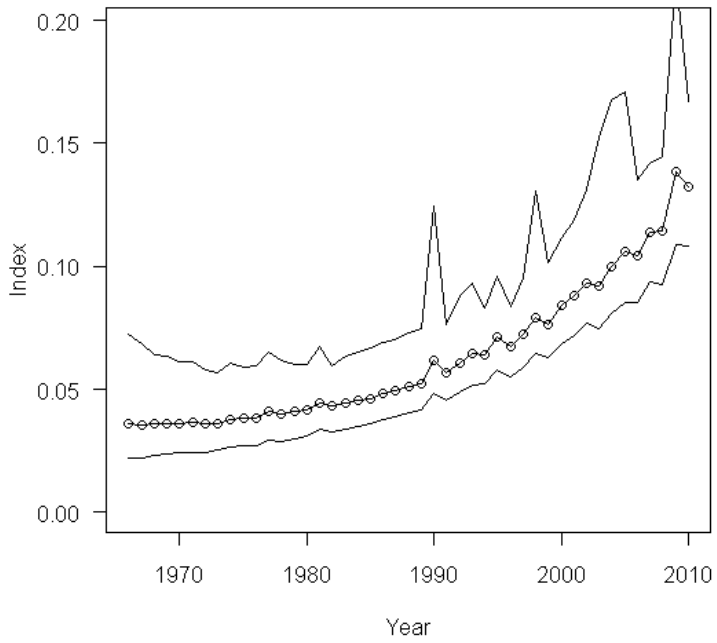

Figure 4. Breeding bird survey trend estimate of merlins in North America from 1966-2010 (Sauer et al. 2011)

Threats

Merlin populations are generally increasing throughout North America but they are still vulnerable to some threats. Oliphant (1985) suggested that the greatest threat to prairie merlins is habitat loss. The removal of trees and sagebrush and grassland habitats for agricultural and urban development reduces the amount of available nest and roost sites for both merlins and their prey species (Fox 1971, Trimble 1975, Becker and Sieg 1987). However, since the 1960’s, increasing numbers of prairie merlins have been roosting in cities on the Great Plains such as

Saskatoon, Saskatchewan; Bozeman and Missoula, Montana; Boulder, Colorado; Bismarck, North Dakota and other cities (Servheen 1985, James et al. 1986) . This adaptation to urbanization may counteract some the effects of rural habitat loss. Habitat loss or alteration could be a potential threat to taiga merlins. As peat is harvested for an alternate energy source, foraging habitat for merlins is destroyed (Evans 1982).

Some diseases and parasites are more common among merlins than others raptors; however, there are currently no known pathogens that pose a significant threat to the species (Evans 1982).

In Sweden, Wiklund (1995) reported losses of nestlings and eggs to corvids (Family Corvidae), rough-legged hawks (*Buteo lagopus*) and weasels (*Mustela sp.*) with a higher occurrence of predation of eggs than nestlings. Young merlins may also be lost due to human-related impacts. Oliphant and Thompson (1978) reported three instances in which a nest tree was climbed during egg-laying or incubation that eventually resulted in the failure of the nests to produce young.

According to analyses performed by the Point Reyes Bird Observatory in conjunction with the California Department of Fish and Game, the merlin has a low vulnerability to climate change. The merlin was judged to have minimal sensitivity to climate change in six out of seven categories. It was ranked high sensitivity in the migratory status category because migratory species can be more sensitive to changes in climate changes than non-migratory species (http://data.prbo.org/apps/bssc/index.php?page=climate-change-vulnerability, accessed June 8, 2012).

Data Gaps and Monitoring Needs

Almost all of our knowledge of merlin nesting habitats, home range and breeding behavior is the result of studies of prairie merlin populations, with the greater proportion of those being urban populations outside of California. These studies may not portray accurately the ecology of the species in California.

Many taiga merlins travel through California to winter in South America where the pesticide DDT is still widely used to kill the mosquitoes that transmit malaria (NPIC 1999). It is unknown if merlins wintering in areas that continue to use DDT are being impacted. Since DDT is stored in fatty tissues, studies of the fat stores of migrant merlins passing through the state could indicate whether DDT is present in large enough quantities to be deleterious to the health of merlins or their reproductive productivity.

Taiga merlins living in the boreal forests of Canada and Alaska make up a significant proportion of North American merlins and are also the most migratory of the three subspecies. Since this is the subspecies most commonly found in California it would be beneficial to have a better understanding of habitat requirements on wintering grounds, and potential threats in wintering areas. While data on the taiga merlin are scant, data on the black merlin entirely lacking. There have been little or no studies on abundance, population trends or basic ecology (e.g. breeding cycle, migration patterns and timing, food and hunting habits) for this subspecies. In addition, it is not known whether their partial migration strategy (i.e. some are resident, while others migrate) is related to climate, prey species, genetics or learned behavior.

Monitoring the effects of climate change on this species may be necessary to determine if alterations in habitat or diet of the merlin severely impact other sensitive species or ecosystems.

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ECOLOGY OF PRAIRIE FALCON IN CALIFORNIA

By Justin D. Garcia
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The prairie falcon (*Falco mexicanus*) is endemic to the dry ecosystems of western North America, distributed from southern Canada to central Mexico and from the Great Plains to the Pacific Ocean (Fisher 1893, Peeters and Peeters 2005). Evolutionary forces have allowed this species to exploit medium to small-sized prey that inhabit hilly grasslands, arid plains, open prairies, sparse woodlands, and desert sagebrush communities (Steenhof 1998). Predatory behavior has inexorably linked the abundance of this falcon to the seasonal abundance of prey throughout its range (Haak 1995). Nest-site selection also plays an important role in prairie falcon distribution; typical breeding territories contain escarpments, shelves, bluffs, and outcroppings interspersed with fertile hunting grounds (Steenhof 1998). Prey selection, foraging techniques, and changes in foraging habitat in response to changing weather and prey availability are additional factors defining the prairie falcon niche.

This falcon is similar in morphology and behavior to its closest related relative, the well-known peregrine falcon (*Falco peregrinus*) (Nittinger et al. 2005). Both males (554 g; 19.5 oz) and females (863 g; 30.4 oz) are similar in plumage, but like many Falconiformes they are sexually dimorphic in size (Enderson 1964, Steenhof 1998). Plumage is sandy brown above, with streaks of light brown on its white underparts (Dunn and Alderfer 2005). Facial markings include a brown crown with a dark malar stripe and a dark ear-patch extending from the supercilium, with a distinctive white area in between (Cade 1982, Johnsgard 1990, Wheeler 2003). In flight, the tell-tail field mark of a prairie falcon are the dark axillaries (arm-pit patches) extending into the secondary wing coverts, which are variable in each individual (Wheeler 2003, Unitt 2004). Their wings are long, narrow, and unslotted, creating a wing load that allows for high speed stooping and precise aerobatic maneuvers (Cade 1982). Proportionally larger eyeballs than any other falcon, strong talons, and a razor sharp hooked bill, complete an arsenal of predatory adaptations (Cade 1982).

The prairie falcon has been described as a noble, unpredictable, and fierce creature (Bent 1937, Tyler 1923, Pierce 1935). This aggressive, high-strung bird is genetically adapted for hunting in open areas of low vegetation (Cade 1982). In western North America, the only bird that can compare with the prairie falcon’s aerobatic hunting skills is the peregrine falcon (Cade 1982, Anderson and Squires 1997). While most successful meals come from perch-and-wait low level attacks, when soaring from high in the sky the prairie falcon can stoop at high speeds comparable to its cousin, until it glides dangerously low to the ground with a stealthy element of surprise (Dawson 1924, Fowler 1935, Haak 1982, Kaiser 1986, Haak 1995). The peregrine falcon may stoop to catch birds in flight, but it does not maneuver through ground-dwelling prey quarries (Cade 1982).

An early publication concerning the economic relationship of prairie falcon to agriculture found that, based on a diet of rodents and game birds, there was not enough evidence to warrant either protection or harassment (Fisher 1893). The possible benefit to farmland by suppressing ground squirrels and other rodents provided impetus to include the prairie falcon in the California fish and wildlife “protective laws” of 1901 (Mallette and Schlorff 1978). Peregrine falcon, Cooper’s hawk (*Accipiter cooperii*), and sharp-shinned hawk (*Accipiter striatus*) were not afforded protection in California, due to their perceived depredation on game birds and domestic fowl (Mallette and Schlorff 1978). By the late 1960s, widespread reproductive failure caused by the thinning of eggshells from organochlorines, DDE in particular, prompted further research and immediate action to protect peregrine falcon, bald eagle (*Haliaeetus leucocephalus*), and California condor (*Gymnogyps californianus*) (Ratcliffe 1967, Hickey and Anderson 1968, Mallette and Schlorff 1978). The decline of peregrine falcon and the uncertain status of California prairie falcon led to a prohibition of take (for both species) from the wild for falconry in 1968 (Mallette 1975). It was believed prairie falcon may have suffered the same fate as its cousin, the peregrine; but the difference in diets allowed prairie falcon populations to persist at higher levels of stability.

Beginning in 1969, a preliminary nesting survey was done to locate historical eyrie sites statewide, followed by observations of occupancy and productivity for 218 different historical territories in a period of four years (Garrett and Mitchell 1973). This study indicated noticeable reductions of eyrie occupancy surrounding the western rim of the Central Valley, as well as high frequencies of nonproductive pairs or lack of pairs from traditional nest sites in the deserts of southern California (Garrett and Mitchell 1973). A later study focused on manipulating prairie falcon nests for the recovery of peregrine falcons and concluded that occupancy and productivity within the study area (coast ranges from Santa Barbara to south of the San Francisco Bay Area) was at historic levels (Walton 1977). Prairie falcon was then placed on the Bird Species of Special Concern (3rd priority) list because of concerns over potential impacts on productivity, including: persistent pesticides, shooting, egg robbing, and recreational activities, especially in the southern deserts (Remsen 1978). The species account noted: “The total population within California is very small and vulnerable” (Remsen 1978). In the same year, the Fish and Game Commission lifted the prohibition of wild prairie falcon harvest for falconry, but only after stable reproduction was reported in California (Schlorff 1981). The report noted: “In 1980, a large sample of eyries were sampled and normal reproduction was assumed; thus, a harvest was authorized” (Schlorff 1981).

The prairie falcon remained on the Bird Species of Special Concern list of 1992, but failed to make the most recent list based on a study in the 1970s and a statistically significant positive Christmas Bird Count trend from 1959-1988 (Boyce et al. 1986, CDFG 1992, Sauer et al. 1996, Shuford and Gardali 2008). The research by Boyce et al. (1986) was extensive and included observations at 520 nesting territories over 10 years; reporting there was still a high abundance in the “metropolis of range” defined by Grinnell and Miller (1944). Although, there was still a concern for lowered productivity observed in the Mojave Desert based on a master’s thesis by Boyce (1982). Since then, the prairie falcon has been placed on the federal Bird of Conservation Concern (BCC) list for the Mojave/Sonoran Bird Conservation Region (BCR) (also, the southern Rockies/Colorado plateau, badlands and prairies, shortgrass prairies BCRs, and region 6) (USFWS 2008).

Prior to the large decline of peregrine falcon and the passage of the Endangered Species Act, the prairie falcon was not a popular bird for falconry (Haak 1992). However, its aggressive nature, small size, speed, and its ability to capture large prey with multiple attack styles have made it the best alternative to the de-listed, California fully protected peregrine falcon (Haak 1992).

Reported harvest in California from 2006-2010 is 46 prairie falcons, averaging 9.2 per year; 2 adults, 34 immature, 8 nestlings, and 2 of unknown age (CDFG 2012). Four died while in captivity, two from collision with automobile, one from old age, and one from unknown causes (CDFG 2012). Most take is centralized in the “metropolis” of range in: Kern (5), Los Angeles (7), San Bernardino (4), Riverside (5), and San Luis Obispo (5) Counties (CDFG 2012).   Seventy-four percent of the prairie falcons harvested were female (34/46), most of which were immatures

(27). Females may be selected more often by falconers due to their ability to capture larger prey, such as upland game birds and waterfowl (Haak 1992). For the 5 year period of analysis (2006-2010), 19 prairie falcons were released and 3 escaped in California (CDFG 2012). It is unknown if these birds survive and successfully reproduce.

Habitat

The prairie falcon is particular with regard to habitat selection, requiring arid open rangeland and a safe place to raise their young (Dawson 1913). The diversity of habitats utilized throughout its range includes prairies, valleys, foothills, sparse woodlands, plains, pastures, grasslands, buttes, canyons, river bluffs, high mountain meadows, desert shrub-steppe, marshland, cropland, and ocean shores (Bent 1938, Grinnell and Miller 1944, Haak 1982, Palmer 1988, Steenhof 1998, Peeters and Peeters 2005, Buranek 2006). Where abundant food is found, these critical habitats may provide year-round occupancy throughout the various geographic regions of California (California Floristic Province, southern Desert Province, and eastern Great Basin Province) (Bent 1938, Boyce et al. 1986, Buranek 2006, Baldwin et al. 2012). While prairie falcons can nest in woodland and chaparral habitats, grasslands that provide food and cover for ground squirrel colonies are usually within 5-20 km of an eyrie (Marzluff et al. 1997, Buranek 2006). Non-native pasture and cropland can also supply abundant alternate prey such as voles and passerines, which are important during years of poor ground squirrel production (Haak 1982, Marzluff et al. 1997).

The inner coast ranges and parts of the eastern foothills of the Central Valley contain habitat of importance for nesting and wintering prairie falcons (Boyce et al.1986, Peeters and Peeters 2005). This central California population forages primarily in the expansive grasslands and valleys within the coast ranges and the grasslands and blue-oak/pine woodlands of the eastern foothills of the Central Valley (Bent 1938, Garrett and Mitchell 1973, Verner et al. 1980, Boyce et al. 1986, Buranek 2006, Peeters and Peeters 2005, Bousman 2007). In the southern coast ranges, breeding territories are found within coastal slope woodlands, chaparral, sage scrub, and desert grassland (Unitt 2004).

Summer and fall use areas expand above the coniferous treeline to include high Sierra alpine meadows and mountain valley grasslands (Grinnell and Miller 1944, Verner et al. 1980, Gaines 1988). Wintering individuals are strongly associated with grasslands in the Central Valley, with no significant difference between grazed and ungrazed grasslands (Pandolfino et al. 2011). The coastal prairies of California are also visited during winter, from the river bottoms of northern California to the coastal slope in southern California (Garrett and Dunn 1981, Roberson 1985, Harris 1991, Unitt 2004, Peeters and Peeters 2005, Elliot and Fish 2010).

Prairie falcons also nest and winter in the southeastern desert ecosystems of California (Fisher 1893, Bent 1938, Grinnell and Miller 1944, Boyce et al. 1986, Peeters and Peeters 2005). High reptile species richness supplements a diverse diet of desert dwelling mammals and birds (Boyce 1985, Kaiser 1986, Hill 2003). The Mojave Desert is characterized by arid mountain ranges of pinyon/juniper woodland and valleys filled with creosote bush scrub (*Larrea tridentata*), Joshua tree woodlands (*Yucca brevifolia*), Mojave yucca (*Yucca schidigera*), white-bursage (*Ambrosia dumosa*), saltscrub, and alkali sinks (Baldwin et al. 2012). Most of the Mojave Desert is above 600 m (1,970 ft), ranging from -83 m (-272 ft) to above 1,500 m (4,920 ft) (Boyce 1982). The lower, warmer Sonoran desert is similar, but also includes blue palo verde (*Parkinsonia florida*), ocotillo (*Fouquieria splendens*), chuparosa (*Justicia californica*), ironwood (*Olneya tesota*), and some agriculture (Baldwin et al. 2012). Granite, limestone, and conglomerate cliffs are extensive in the desert region, whereas sandstone cliffs are infrequent (Runde and Anderson 1986, Boyce 1987). Selection of a stable microclimate to raise juveniles is important in an otherwise severe desert climate (Boyce 1987).

The Great Basin supports another area of suitable habitat, which is comprised of the high plains in the Modoc Plateau and the region east of the Sierra Nevada (Fisher et al. 1893, Ryser 1985, Boyce et al. 1986, Gaines 1988, Peeters and Peeters 2005). The northeastern California high plateau averages about 1550 m (5,085 ft) in elevation, while the region south of Lake Tahoe ranges from 1100 m (3,609 ft) in Owens Valley to 4300 m (14,107 ft) in the White Mountains (Baldwin et al. 2012). The Great Basin region experiences low rainfall, temperature extremes similar to the desert, and is dominated by pinyon/juniper woodlands, sagebrush steppe, meadows, and scattered pine forest (Gaines 1988, Huenneke 1989, Young et al. 2007, Baldwin et al. 2012). The prey base of this region has historically supported populations of many species of raptors, including the prairie falcon (Dixon and Bond 1930, Garrett and Mitchell 1973).

Prairie falcons are also limited by available eyrie sites during the breeding season in spring and summer (Boyce et al. 1986, Peeters and Peeters 2005). Normally, eyrie sites are situated on cliff ledges and rock outcroppings in woodland foothills, with foraging habitat patchily distributed within the nesting home range (Bent 1938, Grinnell and Miller 1944, Haak 1982, Marzluff et al. 1997). The average cliff size selected in the Mojave Desert, California is 29.3 m (96 ft) high and 19.3 m (63 ft) wide, with an average cliff area of 1414 m2 (15,000 ft2) (Boyce 1987). This species does not build its own nest, and must be flexible when choosing the exact location to lay its eggs (Brown and Amadon 1968, Boyce 1987). Suitable nests are scrapes found in potholes, crevices, recessed ledges, subterranean caves, platforms inside chimneys, vertical fissures, cliff top shelves, canyon walls, and artificial platforms (Bent 1938, Haak and Denton 1979, Boyce et al. 1980, Runde and Anderson 1986, Haak 1995, Steenhof 1998). Throughout its range, prairie falcons most frequently use potholes found in sandstone cliffs (Runde and Anderson 1986). Old stick nests built by common ravens (*Corvus corax*), red-tailed hawks (*Buteo jamaicensis*), and golden eagles (*Aquila chrysaetos*) are used when alternate nest sites in a territory do not exist (Decker and Bowles 1930, Boyce 1987, Peeters and Peeters 2005). In the Mojave, prairie falcons use stick nests built by other species (49% of the time) in cliff sites near hunting grounds that would otherwise be unusable (Boyce 1987). Prairie falcons occasionally utilize stick nests built by other species in trees and on transmission towers (MacLaren et al. 1984, Roppe et al. 1989). This added flexibility is advantageous when competing with other cliff nesting species (e.g., Great horned owl – *Bubo virginianus*), because the availability of cliff eyries is limited on the landscape (Runde and Anderson 1964, Boyce 1987).

The various broad connected habitats the prairie falcon utilizes all have a common theme, low sparse vegetation with seasonally changing prey populations and suitable rock formations for nesting (Haak 1982, Boyce et al. 1986). Their hunting style is tailored to the open prairies, grasslands, and shrub-steppe where a diverse faunal community of mammals, birds, reptiles, and insects exists (Steenhof 1998, Steenhof et al. 2005). The prairie falcon occupies a specialized niche that may require vast movements across their range, because many habitats (e.g., Snakes River NCA) do not sustain their nutritional needs throughout the entire year (Steenhof et al. 2005). In California, prairie falcons can remain year-round in select geographic regions of the state, changing their foraging habitat to local hotspots of prey density (Bent 1938, Grinnell and Miller 1944, Buranek 2006, D. Bell East Bay Regional Parks personal communication 2012).

Migration and Winter Movements

Due to seasonal fluctuations in prey populations, distribution of this species can vary greatly between the nesting and non-nesting seasons, based on studies conducted in Idaho (Steenhof 1998, Steenhof et al. 2005). While some falcons remain near the nesting site all year, others abandon breeding areas in order to exploit nutritional resources from distant rangelands (Ryser 1985, Steenhof et al. 2005, Buranek 2006). Until recently, their wandering nature and degree of genetic interchange among regionally different populations has been poorly understood. Where the largest abundance of nesting prairie falcons occurs (as many as ~200 pairs), in the Snake River Birds of Prey National Conservation Area (NCA), satellite telemetry confirmed that three different seasonal areas are used, and with a high level of site fidelity (nesting, summering, wintering) (Steenhof et al. 1999, Steenhof et al. 2005). It was determined the grassland habitat of the Great Plains serves as a vital non-breeding area for a large number of the NCA prairie falcons (DeLong and Steenhof 2004, Steenhof et al. 2005).

Arrival at nesting locations in most of the prairie falcon breeding range occurs in late winter (Steenhof 1998). In Idaho, nesting territories are occupied through the spring nesting season, until fledging of juveniles in early summer (Steenhof et al. 1984). The estivation of ground squirrels (Belding's, Piute, etc.) at nesting territories is usually what signals a migration in late summer (Haak 1982, Steenhof 1998). In the NCA, females depart to summer use areas first, probably requiring males to provide food and protection to young fledglings (Steenhof et al. 2005). Migrant prairie falcons then move north and east across the Continental Divide, exploiting prey populations at the higher elevations of Canada and the northern Great Plains (Steenhof et al. 1984). Distances of 822 +/- 291 km (510 +/- 181 miles) are traveled from southwestern Idaho to reach initial summer use areas, at an average of 21-299 km/day (13-186 miles/day) (Steenhof et al. 2005). Summer distribution in the Great Plains at high mountain ridges, plateaus, and pastures has been shown to closely resemble the distribution of Richardson's ground squirrels, a species that does not estivate until September (MacLaren et al. 1988, Hartson 1999, Steenhof et al. 2005).

A change in weather and declining prey availability then prompts a southeast movement to wintering grounds in the southern Great Plains, an integral migration route in the species' life cycle (Enderson 1964, DeLong and Steenhof 2004, Steenhof et al. 2005). Alternatively, some individuals travel back across the Continental Divide, moving west of the Rocky Mountains by using the Intermountain Flyway to reach wintering grounds (Schmutz et al. 1991, Steenhof et al. 2005). Some fly up to 1948 km (1,200 miles) to reach winter use areas, while others stay within 200 km from the natal eyrie, taking an average of 2 weeks to reach their destination (Palmer 1988, Steenhof et al. 2005). Large concentrations of horned larks are located and pursued in winter rangeland and cropland, until the flocks disperse back to their nesting grounds (Enderson 1964, Schmutz et al. 1991, Beauvais et al. 1992). The prairie falcon then takes a direct route from winter areas back to the breeding grounds. Total distance traveled from the NCA during the non-nesting season ranges from 281 to 4636 km(174-2880 miles), revealing the energetic cost of long distance flight is low, when compared to the benefit of winter survival and exploitation of abundant populations far from the nesting territory (Steenhof et al. 2005).

In California, Walton (1977) found that adults and their fledglings from the central Coast Range disperse to flat open areas (females) and woodland foothills (males), with the young leaving the study area. Other accounts found prairie falcons also move to the valleys of the Coast Ranges, the Central Valley, and up to high Sierra alpine meadows (Grinnell and Miller 1944, Verner et al. 1980, Bousman 2007, Pandolfino et al. 2011). Some select foraging areas up to 4000 m

(13,123 ft) in the White Mountains, Mount Langley, and Whitney Meadows (Fisher et al. 1893, Grinnell and Miller 1944, Small 1994). Winter snow drives bird prey populations and the prairie falcon downhill to the wetland and grassland habitat of the Central Valley and to the sagebrush steppe/woodland mosaic east of the Sierras (Fisher et al. 1893, Verner et al. 1980, Gaines 1988, Small 1994, Haak 1995).

Prairie falcons move into the state, traveling to northern California during the fall and winter, thus outnumbering spring resident birds 4 to 1 (Small 1994, Steenhof et al. 2005). A smaller fall migration travels south down the coast line going as far as San Diego and Baja California (Figure 1) (Bent 1938, Small 1994, Howell and Webb 1995, Unitt 2004). Anecdotal accounts have documented wintering prairie falcons throughout the southern deserts and Great Basin, but definite migratory patterns are still uncertain in these regions (Fisher et al. 1893, Bent 1938, Gaines 1988). Overall, the wintering prairie falcon is found in areas beyond its normal breeding range in California, largely because it is not restricted to an eyrie at this time (Garrett and Dunn 1981, Gaines 1988, Harris 1991, Peeters and Peeters 2005). More recent research regarding migratory patterns found that prairie falcons in the east Bay Area and Pinnacles National Monument (NM) in central California are year-round residents (Buranek 2006, D. Bell personal communication 2012).

Enderson (1964) found that 4 of 5 prairie falcon nestlings banded in California were recovered east of the Continental Divide, suggesting not all birds are year-round residents of the state. More recently, 2 nestlings banded in the east Bay Area have been recovered moving east of the continental divide (D. Bell, personal communication 2012).

However, another factor to consider regarding winter movements of prairie falcon in California, is that Belding’s ground squirrel does not hibernate completely until September and California ground squirrel does not hibernate until later in the year, if at all (Fitch 1948, Johnson 1999). Ground squirrel colonies also attract a high abundance of grassland birds which may serve as alternate prey in winter, similar to the horned lark flocks of the Great Plains (Enderson 1964, Beauvais et al. 1992, Lenihan 2007). This year-round abundant food supply allows many prairie falcons to remain resident in California’s extensive rangelands, as their life cycle is intertwined with populations of ground squirrels and passerines (Bent 1938, Palmer 1988, Steenhof and Kochert 1988, Buranek 2006, D. Bell personal communication 2012).

One caveat to the anecdotal evidence of winter movements in California is that only two studies (Pinnacles NM and east Bay Area) to date have tracked year-round movements by use of radio telemetry, and one of those is still in press (Buranek 2006, D. Bell, personal communication) . There have been some other telemetry studies tracking breeding home range and some banding recoveries of California prairie falcons east of the Continental Divide (Enderson 1964, Harmata et al. 1978, Haak 1982, D. Bell, personal communication). But only one adult has been tracked via satellite telemetry from Snake River NCA to a winter use area in northern California; the bird died in California (Steenhof et al. 2005). Furthermore, only 992 (average ~19/year) prairie falcons have been banded in California since 1960, and only 33 banded prairie falcons (regardless of original banding location) have been recovered in the state (U.S.G.S Bird Banding Lab 2012). The results reflect the difficulty of recovering enough banding data to reach a statistically significant conclusion regarding the seasonal movements of prairie falcons in California.

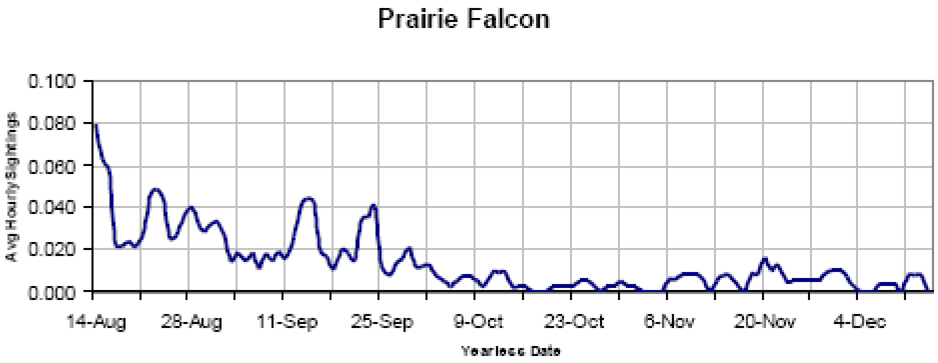


Figure 1. Migration profile at the Marin Headlands, California, from 1986 to 2009 (Elliot and Fish 2010).

Food Habits

This agile predator consumes a wide variety of prey, including: small to large mammals, birds, reptiles, and even insects (Palmer 1988, Peeters and Peeters 2005). The most important food type is a seasonally high supply of medium sized rodents; namely ground squirrels of the genus *Spermophilus* (Bailey 1917, Steenhof 1998). Recently, this genus was divided into 8 different genera (Helgen et al. 2009). Various species of ground squirrels preyed upon throughout its range are: Townsend's (*Urocitellus townsendii*), Piute (*Urocitellus mollis*), Belding's (*Urocitellus beldingi*), Wyoming (*Urocitellus elegans*), thirteen-lined (*Ictidomys tridecemlineatus*), spotted (*Xerospermophilus spilosoma*), Richardson's (*Urocitellus richardsonii*), golden-mantled (*Callospermophilus lateralis*), California ground squirrel (*Otospermophilus beecheyi*), etc. (Steenhof 1998). The specialized preference towards ground squirrels causes this raptor to move in response to the ephemeral fluctuations in the life cycles of numerous ground squirrel species (Steenhof and Kotchert 1988, Steenhof 1998).

Most of these ground squirrel species weigh between 100-300 grams (3.5-10.6 oz), making capture easy for both male and female prairie falcon adults (Kaiser 1986, Hartson 1999). In the Mojave, (Boyce 1985) found 84% of prey captured by males weighed less than 150g (5.3 oz), averaging 107g (3.7 oz). This average prey weight is 20% of the average male weight, which was highly correlated with 4 other species in the genus Falco (Boyce 1985). Reverse sexual dimorphism may facilitate resource partitioning between males and females (Sellander 1966, Buranek 2006). Considering females are larger and use open grasslands more often than males near Pinnacles NM, the smaller male probably cannot consistently capture large prey such as an adult California ground squirrel, 565 gram (20 oz) average (Boyce 1985, Buranek 2006, D. Bell personal communication 2012).

The California ground squirrel inhabits semi-arid open rangeland throughout the state, except parts of the Great Basin and the southeastern deserts (Grinnell and Dixon 1918, Polite and Ahlborn 1999). Where it is common, this rodent comprises a majority of the biomass of prairie falcon nesting diets (Bond 1924, Fowler 1931, Bond 1936, Bent 1938, Garrett and Mitchell 1973, D. Bell personal communication 2012). Furthermore, California ground squirrels disturb grasslands during foraging and burrowing, thus creating a montage of low vegetation habitat that attracts a diverse assemblage of ground-feeding passerines and invertebrates (Lenihan 2007). Important alternate prey for prairie falcon includes horned larks, western meadowlarks, and other ground nesting passerines (Tyler 1923, Fowler 1931, Enderson 1964, Steenhof

1998). In Contra Costa and Alameda counties, these alternate prey items are found in higher abundances, especially during the fall and winter, at ground squirrel colony sites (Lenihan 2007). Burrowing owl (*Athene cunicularia*), a historical prey item of prairie falcon, is also found nesting and roosting in California ground squirrel burrows (Fowler 1931, Brown and Amadon 1968, Lenihan 2007). Combining these various biotic factors, California ground squirrel colonies serve as attractive foraging microhabitat to prairie falcon year round.

In the California Mojave Desert, Boyce (1985) found a wide prey base consisting of 9 different species of mammals (72% of pellets), twenty-five species of birds (24% of pellets), five species of reptiles (4% of pellets), and rarely, the armored stink beetle (*Eleodes armata*). Prey items most frequently found at nest sites are desert woodrat (*Neotoma lepida*), valley pocket gopher (*Thomomys bottae*), mourning dove (*Zenaida macroura*), and horned lark (Boyce 1985). These small captures are offset by larger mammals such as whitetail antelope squirrel (*Ammospermophilus leucurus*), Mojave ground squirrel (*Xerospermophilus mohavensis*), California ground squirrel, black-tailed jackrabbit (*Lepus californicus*), and desert cottontail (*Sylvilagus audubonii*) (Boyce 1985). Chuckwalla (*Sauromalus obesus*), desert horned lizard (*Phrynosoma platyrhinos*), and other reptiles are utilized during nesting as well (Fowler 1935, Boyce 1985). Out of forty species, not one was determined to be the primary prey in the Mojave, but mammals (~179 g) still accounted for a majority of prey biomass (Boyce 1985). In contrast, Kaiser (1986) found whitetail antelope squirrel (>50% frequency of occurrence) to be the most important prey item in the west Mojave. The latter two studies detected low predation on the Mohave ground squirrel, which is a historical prey species in this region (Bond 1936).

In Lava Beds National Monument of northern California, reported prey items were mainly Belding’s ground squirrel (35% frequency of occurrence), voles, horned lark, and other passerines. In the San Joaquin Desert, the San Joaquin antelope squirrel (*Ammospermophilus nelsoni*) has been historically documented at falcon eyries in San Luis Obispo County, but this threatened species has suffered declines in the last century (Fowler 1931, Bond 1936a, Germano et al. 2001, Germano et al. 2012). Other ground squirrel species may play a role in the diet of prairie falcons in California, but have not been documented. Overall, ground squirrels are preferred when abundant because: (a) most juveniles and adults are easily transported to the nest by males or females; (b) ground squirrels are common in open rangeland where suitable eyries exist; (c) colonies foraging in low vegetation are vulnerable to stealthy hunting tactics; (d) the period of above ground abundance is concurrent with the falcon nesting season (Kaiser 1986).

The ability to locate and capture prey is essential for daily survival and successful reproduction, and requires a large home range (Marzluff et al. 1997). Prairie falcons will use different techniques depending on the topography and prey habits, but more important is the location of local prey abundance (Kaiser 1986, Buranek 2006). Once a particular species is found to be common in a home range, the prairie falcon will restrict its flight path to that specific locale, feeding almost exclusively until exhaustion, migration, or estivation by the prey (Bond 1936b, Haak 1982, Ryser 1985). When squirrels are available, there is a strong correlated preference toward them, even when other species increase in density (Steenhof and Kotchert 1988). Even though prairie falcons have an overall generalist diet comprised of over 60+ different species that vary regionally, in a long-term Snake River NCA study, ground squirrels were the only prey that accounted for over 5% frequency of occurrence (Steenhof and Kochert 1988). Preference for alternate prey has been shown to be inversely correlated with ground squirrel abundance, as opposed to increased alternate prey abundance (Steenhof and Kochert 1988).

Using radio-telemetry in northern California, Haak (1982) found during courtship and incubation of eggs that males spent more time hunting (for voles and passerines) than females. Juvenile squirrels begin emerging from winter dens and reach carrying capacity as young prairie falcons hatch in the spring; these squirrels remain above ground through the falcon nesting period, providing a stable food source for both parents to hunt (Haak 1982). After fledging, young falcons develop hunting instincts by capturing ground squirrels, until the heat forces these prey populations to estivate below ground in late summer (Haak 1982).

The prairie falcon exhibits different aerial hunting strategies in order to surprise and flush prey (Steenhof 1998). Soaring above 100 m (328 ft) or perching at high cliffs, this bird stoops from a steep angle at high speeds, and drops out of the sky to capture prey at ground level (Dawson 1924, Dekker 1982, Haak 1982). Most successful hunting attacks result when scanning from perches, where the falcon can swoop down in a “low course” above vegetation, concealing its location (Kaiser 1986, Peeters and Peeters 2005). Potential perches include trees, poles, cliffs, rocks, hay bales, or anything with a good vantage point (Fisher et al. 1893, Haak 1982, Beauvis et al. 1992, Haak 1995). Direct attacks occur while flying 6-37m (20-120 ft) off the ground, resulting in a low stoop towards a quarry (Kaiser 1986). “Undulating flight” is another low course stoop used by males to hunt voles, and by females when hunting ground squirrels (Haak 1982). The “incline and stoop” method is used by females that fly upwind of a perch, only to gain momentum and maximum speed before stooping downwind towards a squirrel colony (Haak 1982). When pursuing birds that flush, a close tail chase occurs until the falcon reaches out and captures the prey with its talons (Palmer 1988, Fesnock 2002). Piracy is also a common hunting behavior, when prairie falcons stoop down and attack larger raptors such as the northern harrier (*Circus cyaneus*), golden eagle, and red-tailed hawk, forcing them to drop a recently caught prey item (Ryser 1985, Wheeler 2003). All of these aerial attack styles require low vegetation (<30 cm) (<1 ft) found in rangelands and the element of surprise (Haak 1982, Steenhof 1998). In the end, capture of prey either comes by the gripping talons (mice, voles, passerines) or a sharp beak to the skull/neck (passerines, medium-large sized mammals) (Bond 1936b, Haak 1982). The prairie falcon plucks bird feathers or skins mammals and reptiles before feeding to young eyasses (Fowler 1931, Bond 1936b).

Breeding

Prairie falcons abandon their winter foraging areas between January and March, and return to their traditional eyrie locations (Walton 1977). Breeding usually occurs at 2 years of age, and occasionally first year females will breed successfully (Platt 1981, Runde 1987). Location of nest sites/eyrie may vary annually, but overall these birds have a high fidelity to a traditional breeding territory (Bent 1938, Runde 1987, Steenhof et al. 2005). Because the male and female falcons do not migrate together, pair bonds are formed at the nesting site. Courtship begins in late February/March and includes a variety of different aerial maneuvers, cliff perching, ledge displays, courtship feeding, and copulation (Cade 1982, Steenhof et al. 2005). Elaborate courtship behaviors occur at various potential cliff ledges, until the pair selects a particular nest scrape (Cade 1982). Competition over the same nesting localities frequently brings the common raven and red-tailed hawk into aggressive interactions with prairie falcons (Holthuijzen and Oosterhuis 2004).

Males strengthen the pair bond by bringing small birds and mice to the fasting female (Haak 1982, Kaiser 1986). Because female prairie falcons spend a higher proportion of time incubating the clutch, it is imperative that the male establish his hunting prowess early on in courtship (Haak 1982, Kaiser 1986). During courtship and pre-incubation, females spend considerably more daylight hours at the eyrie (82%) than males (17%), and always roost at the nest site or adjacent cliff (Kaiser 1986). “Egg-laying lethargy” prevents the female from hunting, and probably facilitates the uncooperative hunting behavior observed in prairie falcon (Haak 1982, Kaiser 1986).

Average clutch completion is variable across the landscape, depending on latitude and local climate (Williams 1985). In California, most clutches are completed by April 14th and vary between mid-March and early May (Walton 1977, Boyce 1982, Haak 1982, Kaiser 1986). Historical (1900-1960) average clutch-laying date was reported as April 4th (Walton 1977). Clutch size is usually 4-5 eggs, unless nest failure occurs (Kaiser 1986, Steenhof 1998, Peeters and Peeters 2005). Incubation lasts about a month and hatching dates vary from early April to late May (Walton 1977, Boyce 1982, Haak 1982, Williams 1985, Kaiser 1986).

After hatching occurs, female time spent at the nest decreases and hunting gradually increases as the nestlings grow older and require larger prey items to meet metabolic needs (Kaiser 1986). However, females still account for 94% of daytime brooding and weather shading between both parents during the nestling phase (Kaiser 1986). Furthermore, the timing of prairie falcon hatching occurs at peak juvenile ground squirrel abundance, providing a preferred nutritional food source through nesting (Grinnell and Dixon 1918, Howell 1938, Fitch 1948, Haak 1982, Kaiser 1986, Steenhof 1998, Lenihan 2007). The average time young eyasses spend in the nest is about 5 weeks, and ranges from early April to late June in California (Walton 1977, Boyce 1982, Haak 1982, Kaiser 1986, Peeters and Peeters 2005).

Throughout the breeding season prairie falcons must protect their nesting territory from intruding conspecifics and interspecific predators with sympatric home ranges (Kaiser 1986). Aggressive behavior (vocalization, chasing, striking) is directed towards intruders within 400 m (1,312 ft) away, from and 100 m (328 ft) above the eyrie, while the remainder of the home range and foraging habitats are undefended (Haak 1982). Common ravens are responded to most frequently (49%), because both species breed in close proximity of each other and have similar nesting preferences (Haak 1982, Kaiser 1986, Holthuijzen and Oosterhuis 2004). The red-tailed hawk, golden eagle, and northern harrier also trigger a high amount of aggressive behavior from prairie falcons (Kaiser 1986, Holthuijzen and Oosterhuis 2004). The only significant territoriality difference between the sexes is that males tend to chase the small, aerobatic American kestrel, while the female chases away larger enemies such as bobcats (Holthuijzen and Oosterhuis 2004). This interspecific aggression is rare during winter, when home ranges become more flexible (Beauvais et al. 1992). In areas such as the Lava Beds National Monument and the Snake River NCA, a large prey base and abundance of nesting sites allows many of these different raptors to coexist with minimal competition (Dixon and Bond 1937, Ogden and Hornocker 1977).

Brooding by the female is terminated once nestlings become full grown, just before fledging. Because they become aggressive at this stage, food is dropped off at the eyrie ledge (Kaiser 1986). Juveniles fledge between mid-May and June 20th (Walton 1977, Boyce 1982, Haak 1982). Food is still brought to the fledglings for a few weeks, but they must learn how to fly and hunt before dispersing from the nesting territory (Haak 1982, McFadzen and Marzluff 1996). Prior to dispersal to summer foraging grounds, a mortality rate of 31% has been reported for dependent fledglings (McFadzen and Marzluff 1996). While hematophagous ectoparasites can have a profound effect on nestlings, they are very susceptible to predation by great horned owls and golden eagles (Runde 1987, McFadzen and Marzluff 1996). Out of the safe confines of the nest, bobcats and coyotes add to the list of juvenile prairie falcon predators (Holthuijzen and Oosterhuis 2004).

Overall, spring and early summer is a labor intensive time focused on reproduction. Many stochastic events factor into nestlings successfully becoming the next generation of breeding individuals, as indicated by a low juvenile survival rate of 15-35% (Denton 1975, Runde 1987). Adult survivorship ranges from 65% to 81% (Denton 1975, Runde 1987). The estimated average life span calculated from banding data is 2.4 years old to 4.9 years, while calculated longevity is 15.6 years (Shor 1975, Platt 1981, Runde 1987). The latter calculation likely holds true when looking at banding data, as the oldest known prairie falcon was a male recovered at 17.25 years of age (Lutnmerding and Love 2011). When combining all of these factors, there is an optimal rate of rate of productivity that must be achieved in order to maintain stable populations and perpetual existence of the prairie falcon.

The optimal rate of productivity has been calculated from numerous studies to average 2.46 (2.2-2.7) fledglings/occupied nest/year (Runde 1987). One can further look at the productivity (mean 3.49) of total successful nests for all studies examined (61-79%), by excluding the number of unsuccessful nesting pairs and the number of unoccupied historical nesting eyries (Runde 1987). To compare with Runde's (1987) calculations, averages over a 24 year study at Snake River Birds of Prey NCA were: 2.46 ± 0.61 fledglings/occupied nest/year; 2.76 ± 0.74 fledglings/successful nest/year; 3.9 ± 0.29 brood size at fledging; 63 ± 14% successful pairs; and 70 ± 17% pairs laying eggs (Steenhof et al. 1999). The only other long-term study comparable in California is a 26-year legacy data set from Pinnacles NM, combining core and non-core nesting areas (Emmons 2011). This area has supported relatively stable numbers of prairie falcons, each year averaging: 11.9 territorial pairs; 9.9 nesting pairs; 7.8 successful pairs; 2.24 fledglings/territorial pair; and 3.42 fledglings/successful nest (Emmons 2011). Complete failure of every nest within the study area in 1998 skews these averages (Emmons 2011).

An average of 3.2 fledglings per productive pair and 1.59 fledglings for all observed pairs was reported in California, later averaged to 2.18 fledglings per pair (Garrett and Mitchell 1973, Boyce et al. 1986). The low production rate of 1.59 was attributed to lower than historical levels of breeding pairs in the Central Valley region and in the southwestern deserts, with an increased amount of non-breeding pairs (Garrett and Mitchell 1973). Subsequent studies in the "central coast" populations showed historic occupancy/production levels, while Mojave Desert pair studies were only successful in fledging young 21-54% of the time (Boyce 1982, Boyce et al. 1986). The number of fledglings per productive pair was 3.26, while the number of fledglings per pair studied was 1.97 (Boyce 1982). Using the population model developed by Platt (1981), which averages 2.43 (2.12-2.73) young/occupied territory, it was determined that declines in occupancy and fledgling rates were synergistically creating an overall population reduction of 25 % in the Mojave Desert during the 1970s (Boyce 1982). Every nest in this region had some degree of influence by human disturbance and urban sprawl (and possible prey base reduction and pesticide contamination), undoubtedly altering the breeding success of prairie falcons (Boyce and Garrett 1977, Boyce 1982).

Home Range

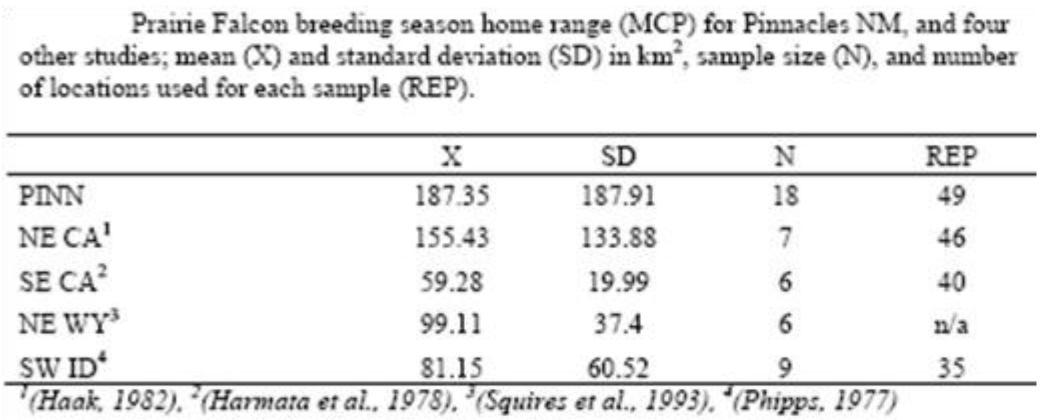
The prairie falcon utilizes large home ranges that contain nesting, soaring, and foraging habitats (Haak and Jarvis 1980). Home range size (minimum convex polygon, MCP) during the breeding season varies regionally and among different studies, with averages from 59 to 315 km2 (23-122 miles2) (Phillips 1977, Harmata et al. 1978, Haak 1982, Squires et al. 1993, Marzluff et al.

1997, Buranek 2006). A thesis by Buranek (2006) compared MCP home range data from Pinnacles NM with four other home range studies, and found no significant difference (P=0.079) (Table 1); a Snake River NCA study with a large sample size (n=95) was not included in the analysis (Marlzluff et al. 1997). Marzluff et al. (1997) found the largest reported average home range (MCP) of 315 km2 (122 miles2) for all falcons radio-tagged. The large average home range size may have been higher due to a drought during the study period, which greatly effected ground squirrel abundance and prairie falcon foraging behavior (Marluff et al. 1997).

Home range size between nesting male and female (average 187 km2) (72 miles2) prairie falcons in Pinnacles NM was not significantly different (P=0.12); males averaged 251 km2 (97 miles2) and females averaged 87 km2 (33 miles2) (Buranek 2006). Other studies found males had larger home ranges as well (Harmata et al 1978, Haak 1982, Marluff et al. 1997). Sizes between home range vary greatly, and are unique to each individual (Buranek 2006). In northern California, Haak (1982) found a range from 34 to 389 km2 (13-150 miles2). At Pinnacles NM, home ranges varied from 20 to 720 km2 (8-278 miles2) (Buranek 2006). Home ranges may overlap in areas of high density nesting pairs, such as Pinnacles NM and Snake River NCA, with core use areas being utilized by multiple individuals (Marzluff et al. 1997, Buranek 2006). Home range overlap and multiple use of core foraging areas likely applies to other local populations in California (e.g., east Bay Area, D. Bell personal communication 2012).

Most flights are 3 to 21 km (2-13 miles) from the nest to foraging areas, averaging about 7 km (4 miles) (Haak 1982, Marzluff et al. 1997). Longer distances from the nest have been recorded (Marlzluff et al. 1997). The longer flights are possible because little energy is spent when soaring to the hunting grounds (Haak 1995). Although, adult falcons at Snake River NCA that increased foraging larger than 300 km2 (116 miles2) showed poor reproduction (Marzluff et al. 1997). In contrast, eyries close to optimal ground squirrel habitat require much less travel time, potentially benefiting productivity (Marzluff et al. 1997). This likely applies in other regions, considering prairie falcon is a central-place forager restricted to a nest during the breeding season (Rosenberg and McKelvey 1999).

Select habitats patchily distributed within the home range are used as hunting grounds, and these core use areas may be repeatedly used each year, with a high degree of fidelity (Marzluff et al. 1997, Buranek 2006). High site fidelity for seasonal core use areas during the non-breeding season has been confirmed as well (Steenhof et al. 2005). Prairie falcons can travel great distances post-nesting, but once they arrive at summer, fall, and winter use areas, home range size shrinks to an average of 30 km2 (12 miles2) (Beauvais et al. 1992, Steenhof et al. 2005). At Pinnacles NM, where prairie falcons are non-migratory, winter home range was smaller than the breeding season as well (126 km2 vs. 97 km2) (49 miles2 vs. 37 miles2), but sample size was low (n=4) and the difference was not significant (Buranek 2006). Overall, a smaller home range suggests these raptors hone in on prey populations (e.g., horned lark flocks) when not restricted to a nesting cliff (Enderson 1964).



Prairie Falcon breeding season home range (MCP) for Pinnacles NM, and four other studies; mean (X) and standard deviation (SD) in km², sample size (N), and number of locations used for each sample (REP).

	X	SD	N	REP
PINN	187.35	187.91	18	49
NE CA[1]	155.43	133.88	7	46
SE CA[2]	59.28	19.99	6	40
NE WY[3]	99.11	37.4	6	n/a
SW ID[4]	81.15	60.52	9	35

[1]*(Haak, 1982)*, [2]*(Harmata et al., 1978)*, [3]*(Squires et al., 1993)*, [4]*(Phipps, 1977)*

Table 1. Statistical analysis examining different home range studies of prairie falcon (from Buranek 2006, page 62).

Historical and Current Distribution

The most comprehensive analysis of the historical distribution of prairie falcon in California is from Grinnell and Miller (1944). Original accounts from Grinnell (1915) were reworked and manuscripts anecdotal accounts from other ornithologists added. Information on prairie falcon occurrence included the capture of nestlings, removal of eggs, or the shooting of adults/juveniles. Various “life-zones” previously described by Merriam (1898) were also used by Grinnell and Miller (1944) to describe where the concentration of breeding prairie falcons resided in the state. The general distribution of prairie falcon was considered “…extensive, throughout state save the northwest humid coast belt. Metropolis appears to lie in southeastern deserts and thence northwest along inner coast ranges…Breeding life-zones, (commonly) Lower Sonoran and (less commonly) Upper Sonoran ” (Grinnell and Miller 1944). Merriam’s Lower Sonoran zone is characterized as the arid southern deserts and the seasonally hot Sacramento-San Joaquin Valley; while the Upper Sonoran includes the valleys, foothills, and coast ranges that surround the Central Valley, the southern California mountain ranges, the coast line south of Santa Barbara, and portions of the eastern Great Basin (Merriam 1898, Pickwell 1944).

To further describe the geographic range within the state, Grinnell and Miller (1944) turned to older accounts of prairie falcon. While records from the extreme northeastern Modoc Plateau are scant, prairie falcon was reported near Camp Bidwell, near Tule Lake at the Lava Beds National Monument, the Warner Mountains, and at Eagle Lake (Henshaw 1879, Dawson 1916, Dixon and Bond 1930, Grinnell et al. 1930, Bond 1939). Moving southwest to Shasta Valley in Siskiyou County, Merriam (1899) saw year-round resident prairie falcons nesting in cliffs close by (Bent 1938). Shasta Valley is one of the major islands of sagebrush steppe that extends from the high plateau into the California Cascades, just north of the Sacramento Valley (Baldwin et al. 2012). Breeding territories were also documented extending south from the top of the Sacramento Valley (Townsend 1887). Wintering individuals have also been found in the vicinity of Marysville and Arbuckle (Bent 1938, Belding 1879).

In the San Francisco Bay Area region, the farthest record north into the humid coastal belt is Marin County and Petaluma (winter) in Sonoma County (Grinnell and Wyeth 1927, Stephens and Pringle 1933, Bent 1938). Grinnell and Wyeth (1927) documented prairie falcon as expanding its distribution throughout the Bay Area in the winter, but nested “in the drier, interior parts of the region, mostly south and east of the Bay". While Cohen (1903) was already reporting prairie falcon retreating from the encroachment of civilization in the Bay Area, he did find active eyries in the foothills of Mount Diablo, Contra Costa County. Other historical accounts confirmed breeding pairs in Berkeley and Altamont of Alameda County; Mount Diablo and near Redwood Canyon, Contra Costa County; the canyons east of Mount Hamilton, Stanislaus County; and Sargent, Santa Clara County (Keeler 1891, Grinnell and Wyeth 1927, Fowler 1931, Fowler 1935, Bent 1938). Wintering individuals were also found in Martinez, Contra Costa County; Presidio, San Francisco County; Redwood City, Menlo Park, and Pescadero, San Mateo County; Berryessa, Santa Clara County; and just east of Stockton (Belding 1879, Grinnell and Wyeth 1927). Prairie falcons have even been reported to venture out to the Farallon Islands off the coast of San Francisco in the winter (Bryant 1888).

In the coast range mountains, Bent (1938) found there were "scores" of prairie falcons throughout the "rough, broken foothills," and Grinnell (1915) noted the Upper Sonoran life zone "metropolis" was centered in "…the arid inner coast ranges west of the San Joaquin Valley." Tyler (1923) found numerous eyries in the "arid hills along the western rim" of the San Joaquin Valley (notably Fresno County). In Monterey County, prairie falcons were seen by Pemberton and Carriger (1915), flying through the scattered oak woodlands during the breeding season. They even witnessed one hunting individual "sail into a flock of seagulls flying near the beach and strike one of them into the sand…after performing a trick the bird flew away…" (Pemberton and Carriger 1915). A concentration of breeding territories also existed in the "…low range of hills which run north to south in eastern San Luis Obispo County" (Dawson 1913, Bond 1936). Each fall as the wind storms blew through the San Joaquin Valley, prairie falcons were seen soaring "for the sheer pleasure of it" (Tyler 1923).

There are records of prairie falcon distribution throughout the southern California coast, deserts, and mountain ranges. An account from Willet (1912) was initially intended to resurvey Grinnell (1898) in Los Angeles County, but it was expanded to include the Pacific coast line from Santa Barbara to Mexico. Prairie falcon was determined by Willet (1912) to be a "…resident east of the mountains…Occurs on the Pacific slope in fall, winter, and spring." A few years later, Willett (1933) confirmed a few breeding pairs on the Pacific slope, and this falcon was "frequently noted in winter throughout southern California." Dawson (1916) found a pair nesting in the "seaward slope of the Santa Ynez Range"…hunting "over the coastal lowlands" (Santa Barbara County). Non-breeding prairie falcons have been found to travel up to Mount Pinos (2,400 m) (7,874 feet) in Ventura County, Cajon Pass (nesting) and Bear Lake (2,000 m) (6,562 feet) in the San Bernardino Mountains (Grinnell 1905, Van Rossem and Pierce 1915, Pierce 1916, Bent 1938). Multiple breeding pairs were also found in Piru Canyon (close to Los Padres National Forest) and non-breeders near to the coast at Hueneme in the fall, both in Ventura County (Willet 1933). There are old records of prairie falcon observations in Los Angeles County, including: breeding pairs in Cerritos, and wintering birds in the San Fernando Valley, Eaton Canyon, and Arroyo Seco near Pasadena (Willett 1912, Bent 1938, Willet 1933). Chino, Claremont, Colton, Santa Ana Canyon, Oceanside, and a few other coastal areas in Orange and San Diego counties round-out the breeding/wintering territories in the coastal slope of southern California (Bent 1938, Pierce 1932, Willett 1933). Nesting pairs were also known to occur somewhat inland to Hemet and the Cuyamaca Mountains of San Diego (Willett 1912, Grinnell and Swarth 1913, Pierce 1932). While some falcons have been observed on Santa Barbara Island and San Miguel Island off the coast, offshore island visits are rare (Streator 1888, Garrett and Dunn 1981).

In what was called the Death Valley Expedition, a team of biologists concluded that “prairie falcons were seen in a number of localities throughout the desert regions as well as among the mountain ranges of southern California and Nevada” (Fisher et al. 1893). Extending from the eastern border, the expedition observed prairie falcons in these locales: Amargosa Desert, Death Valley, Panamint Range, Panamint Valley, Inyo Mountains, Owens Valley, Sierra Nevada, the San Joaquin Valley, and the Tehachapi Mountains (Fisher et al. 1893). About fifty years later, Grinnell and Miller (1944) concurred with the Death Valley expedition findings, stating that prairie falcon “altitudinally, occurs from below sea level, as in Death Valley and around the Salton Sea, up, in summer wanderings, into the high Sierras; for example, above the summit of Mount Langley, 14,000+ feet, Inyo County…”. Furthermore, Bent (1938) mentions banding of three prairie falcons in the Lucerne Valley, Mojave Desert, that were recovered south of the Salton Sea (one) and near the site of banding. Although much exploration was done to the east of the Sierras and in the southern coastal range, prairie falcon distribution in the interior of the southeastern deserts was not explained in detail by Grinnell and Miller (1944). However, they did consider the entire southeastern desert area as the “metropolis” of the range.

In the early 1970’s, there was an observed decline in occupancy of traditional nest sites about 30 km from the Central Valley and this curtailment in distribution was greatest along the eastern side of the coast range, in the northern and central regions (Garrett and Mitchell 1973). Further research (observation of 520 different nests from 1970 to 1979) indicated the prairie falcon exhibits regional variation in nesting density, but distribution of historical territories remained the same and nests can be found in each floristic province of California (Boyce et al. 1986). More recent accounts indicating statewide distribution include: the Great Basin, the Central Valley and its foothills, northern California, Coast Ranges from the Bay Area and south to San Diego County, the Mojave Desert, and Pinnacles NM (Walton 1977, Boyce 1982, Haak 1982, Eakle 1984, Kaiser 1986, Small 1994, Berner et al. 2003, Unitt 2004, Peeters and Peeters 2005, Bousman 2007, Emmons 2011). Nesting pairs have been found from sea level to elevations greater than 3,000 meters (9,842 ft) by Garrett and Mitchell (1973). Aside from observations of some wintering individuals, the lack of nesting territories in the northwestern corner of the state is thought to be due to a lack of suitable habitat (Grinnell and Miller 1944, Boyce et al. 1986, Harris 1991, Sullivan et al. 2009). Although there are an abundance of cliffs, much of the habitat north of Marin County is forested with unconnected open foraging areas (Boyce et al. 1986).

In the Modoc Plateau, where there had been comparatively little urban development at the time, Garrett and Mitchell (1973) found there was a stable population and no change in distribution. In southern California, areas of vast urban development have been noted (Hoshovsky and Kauffman 2003), and such areas are not beneficial to the species (Berry et al. 1998). The prairie falcon was initially considered “tolerably common in fall and winter in the foothill and mesa regions” of Los Angeles County (Grinnell 1898). From a resurvey of Willett (1933) in the coastal slope of southern California, a curtailment in distribution is evident (Garrett and Dunn 1981). The only nesting territories found by Garrett and Dunn (1981) were in the western Santa Monica Mountains, while most historical territories in this region have become vacant. Although, the prairie falcon has always been rare in coastal areas such as San Diego, the closest nest found recently is 37km (23 miles) from the San Diego coast (Unitt 2004). The arid interior on the eastern side of the coastal slope and into the desert, still supports a population of breeding pairs, though wintering birds on the southern California coast are still rare (Tijuana River Valley, coastal Orange County, and coastal Santa Barbara) (Garrett and Dunn 1981, Unitt 2004). A similar loss of rangeland to urban development and related eyrie vacancy has occurred in the east Bay Area (D. Bell, personal communication 2012).

A more current representation of prairie falcon distribution can be gained from Breeding Bird Survey (BBS), Christmas Bird Count (CBC), and Ebird citizen count maps. Breeding season distribution has changed slightly between 1982-1986, 1994-2003, and 2006-2010 (Figures 2, 3, and 4). The central Coast Ranges have experienced a range expansion from Santa Barbara County north to the Pinnacles NM area (Monterey/San Benito County) (Figure 4). Figure 4 also represents a recent disconnect in range, showing a separation (south San Joaquin Valley/Tehachapi Range) between the central Coast Ranges and the Inyo/Mono County Area (compared with Figures 2 and 3). In fact, the large prairie falcon occurrence on the eastern side of the Sierras has largely disappeared from the Lake Tahoe region and south to the desert region; however, the southern deserts appear to have experienced a large increase in distribution. Aside from the central Coast Range, the only constant distribution depicted on the BBS maps is in the northeastern Modoc Plateau. Winter distribution (Figure 5) covers the entire state and ecological provinces seem unimportant in defining winter distribution of the prairie falcon in California. Ebird distribution in the last 10 years uses more discrete point occurrences, confirming the description provided by Peeters and Peeters (2005) is relatively unchanged (Sullivan et al. 2008). However, no recent publications exist, that are comparable to the extensive work done in the 1970s, documenting prairie falcon distribution at the statewide level (Garrett and Mitchell 1973, Boyce et al. 1986).

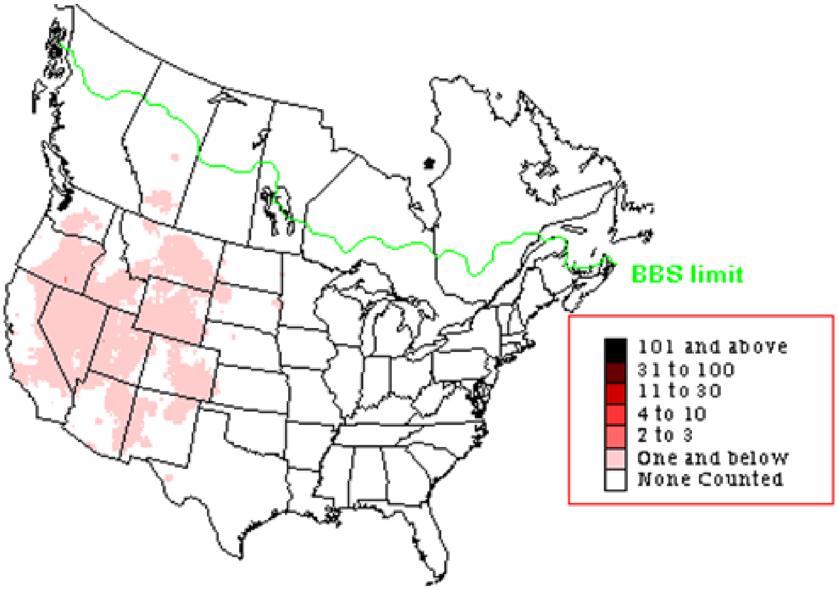


Figure 2. Prairie falcon summer distribution based on Breeding Bird Survey data from 1982 to 1996 (Sauer et al. 1997).

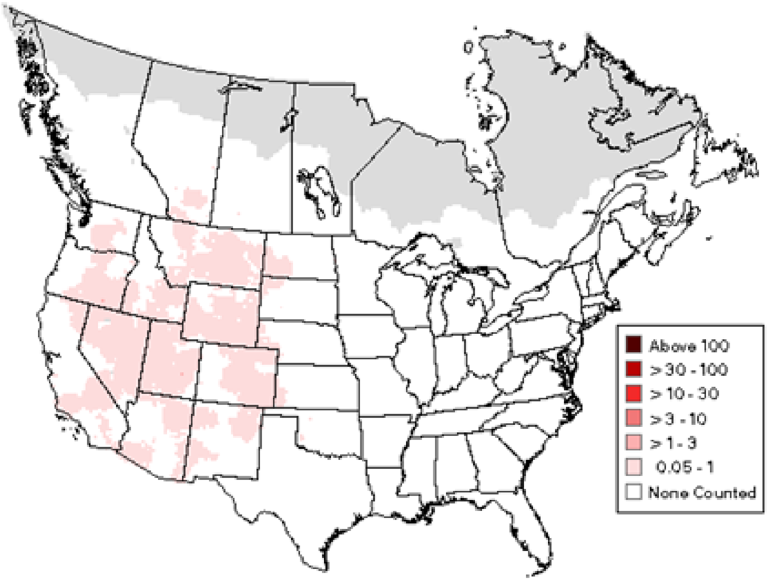


Figure 3. Prairie falcon summer distribution based on Breeding Bird Survey data from 1994 to 2003 (Sauer et al. 2008).

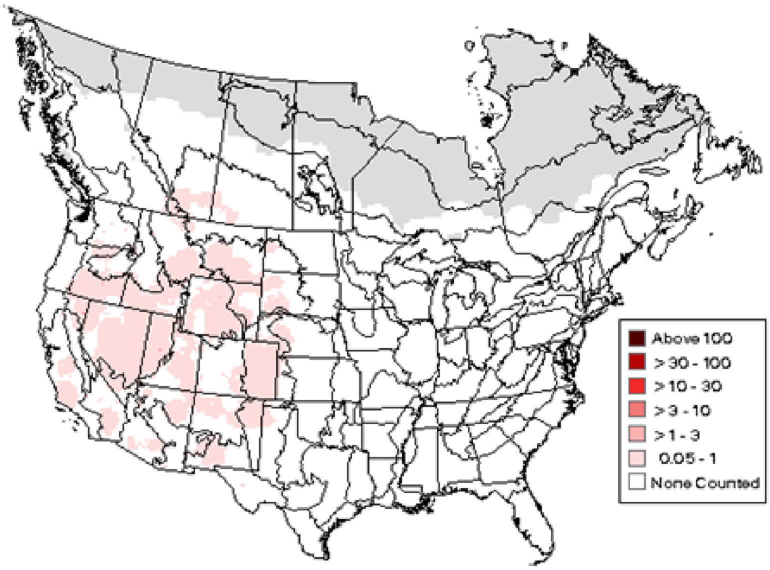


Figure 4. Prairie falcon summer distribution based on Breeding Bird Survey data from 2006 to 2010 (Sauer et al. 2011).

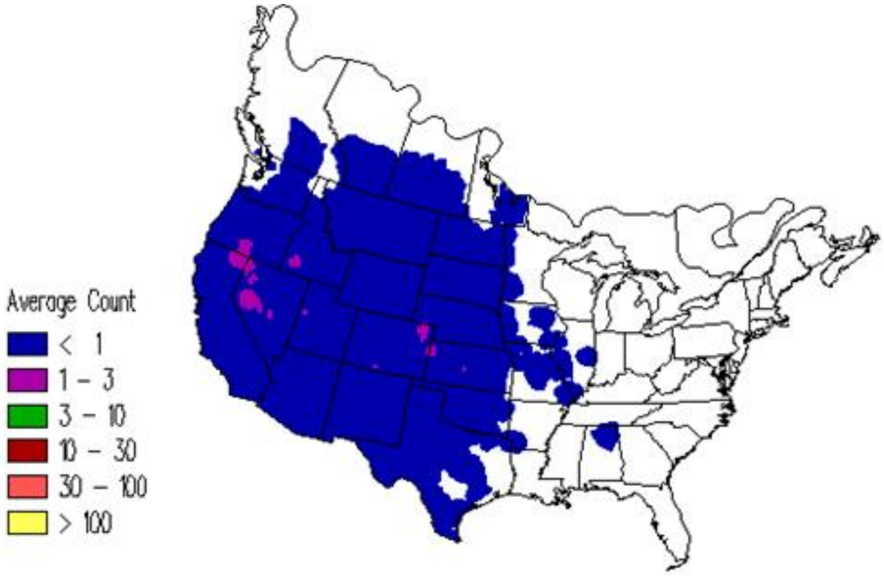


Figure 5. Prairie falcon winter distribution/relative abundance based on Christmas Bird Count data from 1959 to1988 (Sauer et al. 1996).

Historical and Current Abundance

The processes of natural selection have allowed the prairie falcon to fill a particular niche; the exploitation of patchily distributed, high density faunal diversity across broad connected landscapes (Steenhof et al. 2005). Prairie falcon abundance is closely correlated with the abundance of prey in areas of low vegetation, with interspersed cliffs during the nesting season (Boyce et al. 1986, Steenhof et al. 1999). The first analysis of statewide abundance indicated the prairie falcon is a year-round resident, “common in metropolis of range”…which “appears to lie in southeastern deserts and thence northwest along inner Coast Ranges” (Grinnell and Miller 1944).

The Mojave Desert (91 territories) and Coast Ranges (136 territories) of California were confirmed to still provide the greatest abundance of nesting territories (Boyce et al. 1986). The Modoc Plateau has harbored an abundant area as well, 63 territories (Boyce et al. 1986). The other regions with fewer prairie falcon territories are as follows: Southern Coast/Tehachapi Mountain Range, 44 territories; Inyo/Mono region, 39 territories; Cascade Mountain Range (north of Lake Tahoe), 30 territories; the Sonoran Desert, 25 territories; the North Coast forest/Klamath Basin, 9 territories; and the Sierra Nevada Mountain Range (south of Lake Tahoe), 5 territories (Boyce et al. 1986). The old “biotic provinces” of California were used in Boyce’s work, which placed 78 nesting territories in the Central Valley (Dice 1943, Boyce et al. 1986). The Central Valley is large, and it is uncertain as to what part of the valley these particular territories should be placed. Future statewide abundance estimates would benefit from methods more similar to Garrett and Mitchell (1973), where exact locations of eyries were documented. Also, using the updated state geographical/floristic province map from Baldwin et al. (2012) would be more descriptive. A caveat of the Boyce et al. (1986) survey is that the work took place over ten years, thus, the 520 nesting territories found are not necessarily 100% occupied every year. It is known there is regional variability in occupancy and productivity (Garrett and Dunn 1981, Boyce 1982, Boyce et al. 1986).

Early range-wide estimates of population size by Enderson (1964) indicated a nationwide population of 2,000-3,000 prairie falcons in North America. A compilation of 18 population studies across the entire range was completed by Steenhof (1998), with results totaling at least 4,273 pairs. A more recent population size extrapolated from Breeding Bird Survey results in the 1990s, estimated 17,280 individuals (Rich et al. 2004). The estimate by Millsap and Allen (2006), using the same BBS data from the 1990s, was more conservative at 8,640 prairie falcons, of which half are juveniles.

California statewide abundance has been estimated to be 300 to 500 breeding pairs and 650 to 1,100 fledglings per year, based on observations at nesting territories over a ten year period, based on work conducted in the 1970s (Boyce et al. 1986). The large variation in abundance likely either reflects the ephemeral nature of the prairie falcon or the uncertainty behind the estimate. Partner's In Flight population size, using 1990s BBS data, estimated 2,900 individuals (Rich et al. 2004). Using Bird Conservation Regions (BCR) within California, the population estimate by Partners in Flight indicates, with moderate BBS coverage, that 500 individual prairie falcons exist in the Coastal California BCR, 1,200 individuals in the Great Basin BCR, and 1,100 individuals in the Mojave/Sonoran BCR (Rich et al. 2004). Populations for the Sierra BCR and north coast BCR both are estimated at 20 individuals based off adjacent BCR's (Rich et al. 2004). This latter estimate seems unlikely based on the amount of potential nesting territories found by Boyce et al. (1986). It should be noted that the surveys done by Boyce et al. (1986) were not a full population census, and should be only cited as a rough estimate. Lastly, abundance of prairie falcon observations changes seasonally, with more individuals sighted in winter months (Figure 6). This could be due to range expansion in winter (Garrett and Dunn 1981, Peeters and Peeters 2005), or by the presence of wintering prairie falcons from outside of California (Small 1994).

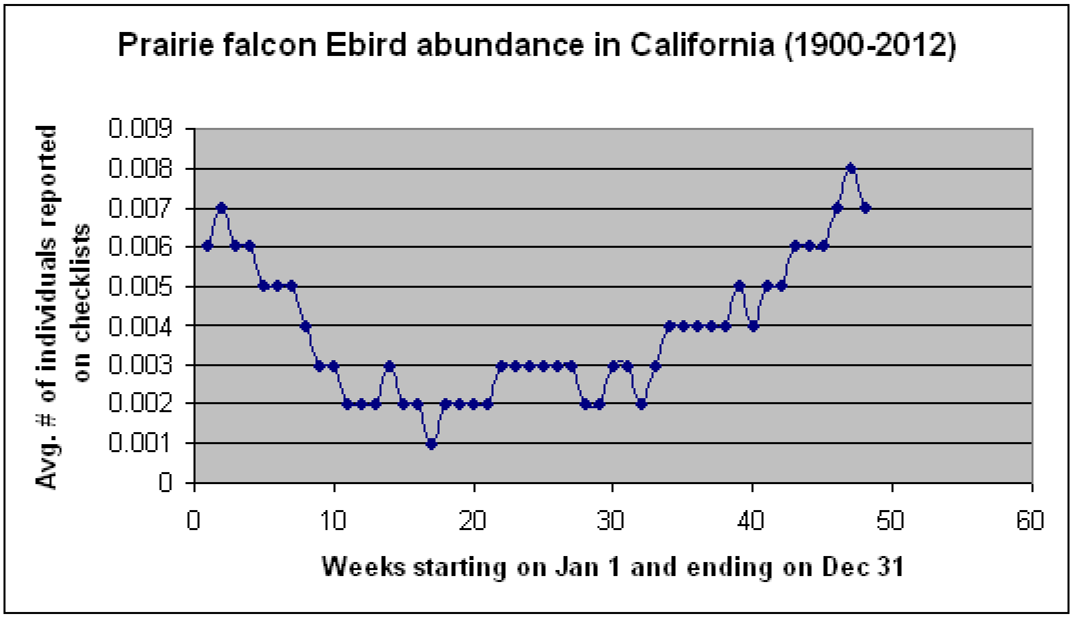


Figure 6. Prairie falcon abundance from the citizen science reporting database, Ebird, 2002 to 2012 (Sullivan et al. 2009).

Population Trend

Because there have been no follow-up studies assessing prairie falcon population size since Boyce et al. (1986), which was based on fieldwork conducted in the 1970s, available data is limited to Christmas Bird Count (CBC) and Breeding Bird Survey (BBS) datasets. Other than these two datasets, there are no recent, focused, statewide studies on population trend of prairie falcon in California, for either the breeding or non-breeding season. For prairie falcon in California, population estimates made from CBC and BBS are extrapolations from small sample sizes.

Christmas bird count data from Sauer et al. (1996) showed a statistically significant positive trend for wintering prairie falcons from 1959 to 1988, however this trend may have been misleading. In California, an average of 30.8 prairie falcons per year was observed in the first 20 years, and the 30 year average was 77.6 prairie falcons per year (Sauer et al. 1996, National Audubon Society 2010). This difference is apparent in Figure 7. When interpreting CBC trends in California, it should be noted that from 1951 to 1987 there was a 5-fold increase in count circles, from 20 to 100, and many of the old count circles were in urbanized areas, while newer CBCs were often located in areas with fewer people and more open habitats (Garrison 1990). As noted in the Garrison (1990) study on ferruginous hawks and CBC data from California, one would not expect an open-habitat forager like the ferruginous hawk to be associated with urbanized areas. The same conclusion is likely for prairie falcons wintering in California, and is supported by work conducted in Colorado where researchers found reduced abundance of wintering prairie falcons on plots that included significant amounts (5-7%) of urban habitat (Berry et al. 1998). For the period of 2001 to 2010, CBC data for California revealed an average of 183.5 (0.0278 = #/party hour) prairie falcon sightings per year, and the population appears stable since the early 1980s (Figure 8) (National Audubon Society 2010).

BBS data collected for prairie falcon on California routes from 2006 to 2010 includes sample sizes of 10, 10, 5, 8, and 9 individuals sighted. BBS data for California show a statistically insignificant positive trend (Figure 9). However, as Thogmartin (2010) cautioned, all Partner’s in Flight extrapolations of BBS data should only be a “crude first approximation.” The latest population trend map from the BBS is shown in Figure 10, indicating both declining and increasing trends in various parts California.

The only closely-monitored raptor migration route in California, the Marin Headlands (GGRO), has trend data from 1986 to 2009 (Figure 11). While sample sizes are low, sharp variations from year-to-year are evident, with a range of 1 to 16 (~0.0025 to ~0.03 yearly sightings per hour) individual sightings/year by GGRO (Figure 11). From 2000 to 2009, an average of 8 (0.02 yearly sightings per hour) prairie falcons per year were seen in the Marin Headlands, and only 2 in 2010 (Fish and Hull 2010). Due to the more inland winter distribution of prairie falcon in California, an inland raptor migration count site is needed to more accurately assess prairie falcon numbers and trend.

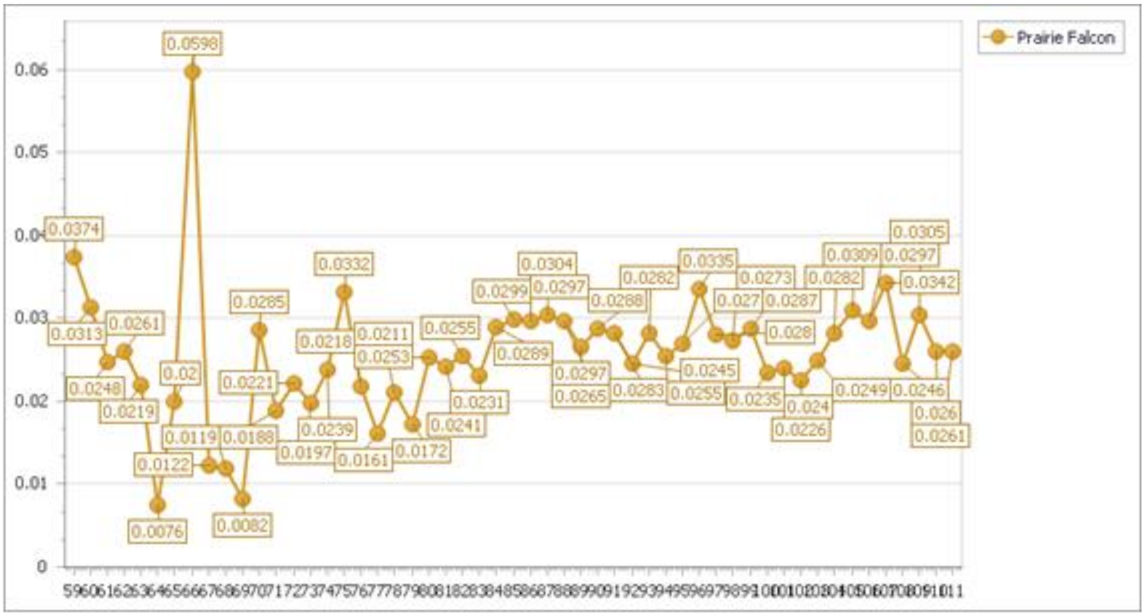


Figure 7. Prairie falcon observations (#/party hour) during California Christmas Bird Counts, from 1958-2010 (National Audubon Society 2010).

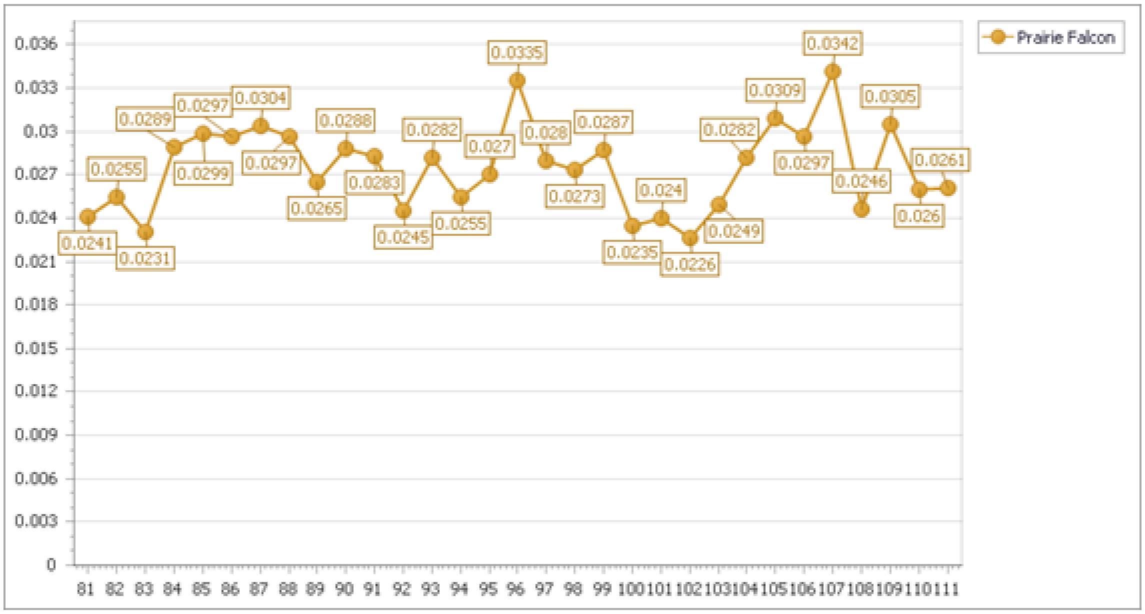


Figure 8. Prairie falcon observations (#/party hour) during California Christmas Bird Counts, from 1980-2010 (National Audubon Society 2010).

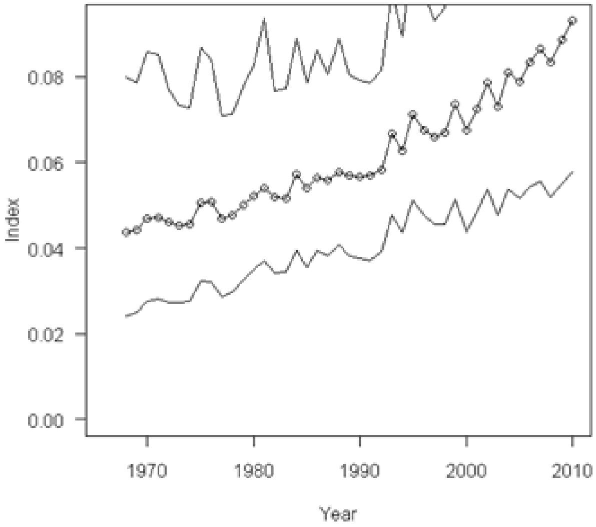


Figure 9. California Breeding Bird Survey trend estimate from 1966 to 2010. (Sauer et al. 2011).

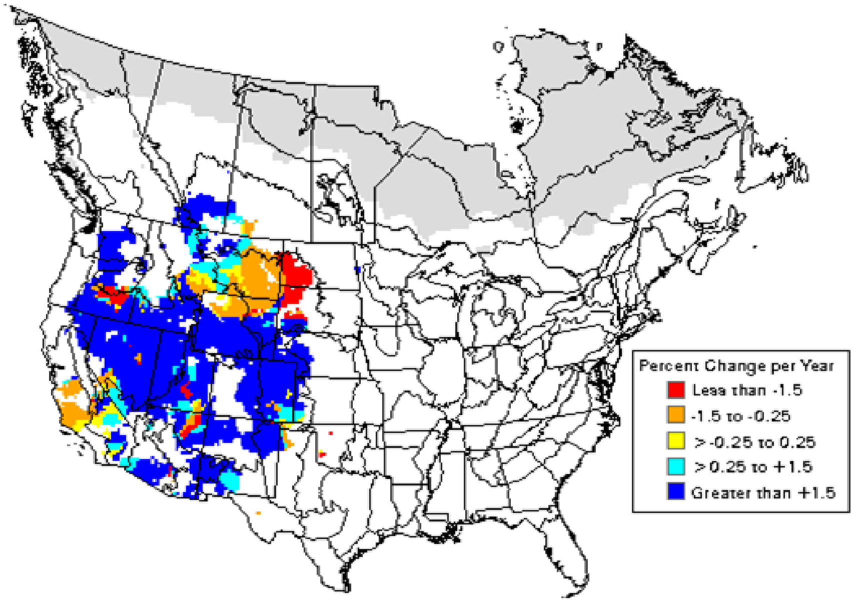


Figure 10. Breeding Bird Survey trend map for prairie falcons, from 1966 to 2010 (Sauer et al. 2011).

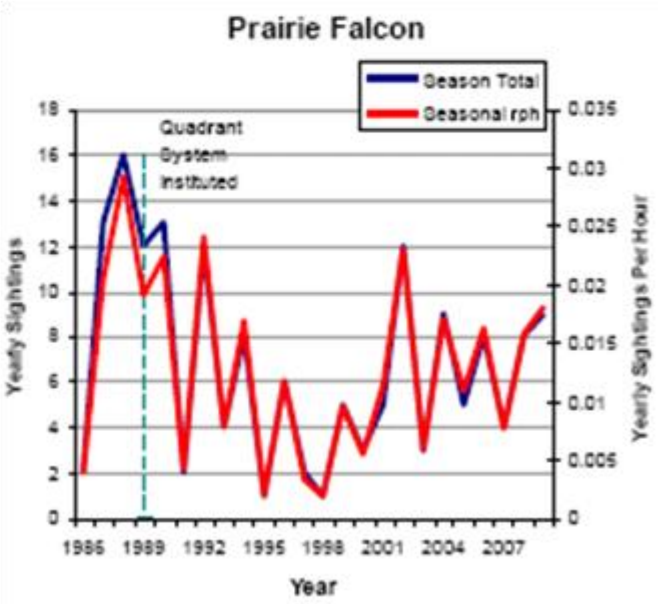


Figure 11. Yearly trend in prairie falcon observations at the Marin Headlands, from 1986 to 2009 (Elliot and Fish 2010).

Threats

One of the most significant threats affecting prairie falcon nest site occupancy and productivity is human related disturbance; the disturbance factors and other threats include habitat loss along with fragmentation and alteration, illegal shooting, pesticide accumulation in prey, military training, road construction, hiking, camping, rock climbing, collisions with man-made objects, illegal falconry take, mining, and off-road vehicle use (Boyce and Garrett, 1977, Boyce 1982, Holthuijzen 1989, Jarman et al. 1996, Steenhof 1998). The significance of these threats depends on the degree, timing, and distance to nesting sites and the surrounding foraging grounds (Harmata et al. 1978). In the central coast range, where most land is privately owned or protected, occupancy rates were near historic levels at the time of study in the 1970s, but the majority of the region has not been resurveyed since (Walton 1977, Boyce et al. 1986). In Pinnacles NM, a more recent study found territory occupancy rates to be comparable to occupancy rates since 1984-2011 (Emmons 2011). However, much of California is experiencing increasing threats to the large tracts of land needed for protection of historical prairie falcon nest sites, and long-term population sustainability.

Of the threats listed above, habitat alteration, loss, fragmentation, and degradation may have the largest effect on the prairie falcon and prey populations it depends on. The first European influence on California rangeland was large-scale livestock grazing, which reduced vast amounts of native grasses and allowed invasive species to disrupt the ecological balance of the landscape (Thompson 1961, Germano et al. 2001, Barry et al. 2006). The drainage of Buena Vista Lake in Kern County removed vital habitat for many avian winter migrants that served as prey to prairie falcon and other predators (Garrett and Mitchell 1973). The pre-European landscape of California consisted of about 60,000 to 80,000 km2 (20,000 to 30,000 miles2) of grasslands (~20% in the Central Valley), and a more recent estimate found less than 10% of the Central Valley grasslands remain (Thompson 1961, Huenneke 1989, CDF 2002). Furthermore, agriculture and other human-related activities have converted much of the San Joaquin Valley from a desert landscape where desert flora such as the endangered Bakersfield cactus (*Opuntia* *basilaris var. treleasei*), annual wildflowers, and salt scrub (*Atriplex spinifera* and *Atriplex polycarpa*) were dominate (Germano et al. 2001, Germano et al. 2011). As a result, desert-adapted natives such as the Tipton kangaroo rat (*Dipodomys nitratoides nitratoides),* giant kangaroo rat (*Dipodomys ingens*), Fresno kangaroo rat (*Dipodomys nitratoides exilis*), San Joaquin antelope squirrel (*Ammospermopholis nelsoni*), and blunt-nosed leopard lizard (*Gambelia sila*) have all become endangered or threatened with extinction (Germano et al. 2001, Germano et al. 2012). The state threatened San Joaquin antelope squirrel has been historically documented as prairie falcon prey (Fowler 1931, Bond 1936a). Their disappearance may impact the prairie falcon population in the central Coast Ranges and the San Joaquin Desert region.

In the Mojave Desert, where California ground squirrels are rarely present, prairie falcons have the most diverse, generalist diet (39 vertebrate species) when compared to every other region in its range (Boyce 1985, Steenhof 1998). The habitat fragmentation and degradation caused by urban development in the southeastern deserts (situated between Los Angeles and Las Vegas) has impacted prey species such as the Mohave ground squirrel and western burrowing owl (Leitner 2008, Wilkerson and Siegel 2011). Both of these species are prey items of the prairie falcon (Bond 1936a, Fowler 1935, Boyce 1985, Kaiser 1986). Declines in prairie falcon occupancy and productivity observed by Boyce (1982) in the Mojave region may be linked to the decline of a western Mojave Desert endemic, the state threatened Mohave ground squirrel.

While certain types of agriculture are not conductive to the prairie falcon, some types of agriculture (e.g., alfalfa, irrigated pasture) provide suitable foraging habitat (Pandolfino et al. 2011). Over 4,000 km2 (1,544 miles2) of agricultural land have been converted to urban land use around the perimeter of cities located in California’s agricultural valleys and coastal areas since 1984 (California Department of Conservation 2011). In addition, intensive agriculture such as vineyards and orchards has been increasing rapidly (Volpe et al. 2010). The South Coast has experienced losses in coastal prairies and sage scrub habitat, and historical wintering habitat and nesting territories due to heavy urban development (Garrett and Mitchell 1981, Hoshovsky and Kauffman 2003, Rundel 2007, U.S. Fish and Wildlife Service 2010). Similar losses in rangeland have occurred in the east Bay Area region, leading to the loss of foraging habitat and vacancy of historical nest sites (Hoshovsky and Kauffman 2003, D. Bell personal communication 2012).

Both urban (> 5% development) and intense agriculture are shunned by prairie falcons, based on a recent study in the Central Valley, showing only a strong positive association with grassland habitat (Berry et al. 1998, Pandolfino et al. 2011). As human populations continue to increase, more rural areas with a historically and current high abundance of nesting pairs (e.g., Pinnacles NM) may face similar encroachment by urban development, ranchettes, and vineyard mansions (Buranek 2006). While there are relatively lower human population levels in the southeastern deserts, compared to the South Coast and Bay Area, a new form of habitat loss/fragmentation is on the horizon, “Big Solar” (Hoshovsky and Kauffman 2003, Leitner 2009). The Mojave Desert of California receives some of the most intense solar radiation in the country, contributing to a large-scale solar energy land rush on BLM land; twelve projects have been authorized since 2009 (Leitner 2002, Leitner 2009, BLM 2012).

Vital components of prairie falcon diet (i.e., California and Belding’s ground squirrels, and other rodents) have become significant pests on croplands and are actively controlled with poisons (Bent 1938, Bond 1936, Haak 1982, Garrett and Mitchell 1973, Marsh 1998). These vertebrate control programs use strychnine and anticoagulant rodenticides, which reduce the prey base near the agricultural landscapes found throughout California (Unitt 2004, Buranek 2006, Emmons 2011, Proulx 2011). Strychnine is not as commonly used today as it was previously, because first and second generation rodenticides have proven more effective (Garrett and Mitchell 1973, Lima and Salmon 2010). Each year, close to 500 tons of chlorophacinone and diphacinone (1st generation for “agricultural” use) are sold by the California Agricultural Commissioners in order to kill ground squirrels, voles, and other rodents, and an even larger quantity of second generation anticoagulants are sold to “homeowners” and “urban” businesses (Lima and Salmon 2010). Diphacinone has recently been shown to be lethal in the closely related American kestrel (*Falco sparverius*) (Rattner et al. 2011). The second generation rodenticides, such as bromadiolone, can have a higher potential for toxicity than first generation chemicals, but the impacts on raptors are inconclusive (Lima and Salmon 2010). There are studies showing the lethal effects of bromadiolone on great horned owls, so one may surmise that the primary and secondary poisoning of prairie falcons is likely, despite zero cases reported to date (Mendenhall and Pank 1980). The difficulty in testing for rodenticides in prairie falcon and other animals lies in the sampling method; a piece of liver is needed to properly detect poisoning, and lethal levels have not been estimated.

While prairie falcon distribution and abundance did not plummet due to organochlorine (DDE) poisoning like its sister peregrine falcon (15-20,000 µg/kg), the prairie falcon is actually far more sensitive, and the impacts of eggshell thinning occurs at lower critical concentration levels (~2,000 µg/kg) (Fyfe et al. 1976, Fyfe et al. 1988). Even though DDT (DDE is a metabolite of DDT) was banned in 1972, it has continued to persist in the environment. In the Mojave Desert, Boyce (1982) found that that eggshells were 14.5x thinner than pre-DDT levels, and eggshells found in unsuccessful nests averaged 17% thinner. Boyce (1982) used historical specimens from the Western Foundation of Vertebrate Zoology, but did not formally test for the presence of DDE and its effects on productivity. A long term study monitoring productivity at Pinnacles NM detected similar reproductive failures characteristic of DDE poisoning (Rechtin 1992). Further study was warranted, and Jarman et al. (1996) set out to test for pesticides at Pinnacles NM and other territories in central and northern California. There were prairie falcon eyries in Pinnacles NM that had DDE levels 4 to 5 (8,200-9,700 µg/kg) times higher than the critical concentration level, and eggshells outside of Pinnacles NM had levels of 1,000-4,200 µg/kg (Jarman et al. 1996). Another harmful chemical detected in prairie falcon eggs is hexachlorobenzene (HCB), a fungicide that was globally banned in 2001. Studies have shown the lethal effects of HCB on other species, and it is likely the combined effects with DDE adversely impact prairie falcon productivity (Gilbertson and Reynolds 1972, Boersma et al. 1986, Jarman et al. 1996). The agricultural Central Valley and Salinas Valley are likely sources for HCB and other chemical values in Pinnacles NM prairie falcons; it has been recommended that lethal biocides on rangeland pests within core foraging areas of prairie falcons should be prohibited (Jarman et al. 1996, Buranek 2006).

Direct human disturbances at nest sites during courtship and incubation times can adversely affect prairie falcon productivity (Steenhof 1998). Uniformed hikers and rock climbers can provoke nest territory defense that can lead to improper temperature regulation of clutches, especially in extreme environments like the Mojave Desert (Boyce 1982, Emmons 2011). Even conducting trapping of adults (females, in particular) during the nesting season can cause complete reproductive failure (Harmata et al. 1978). Off-road vehicles also increase the accessibility to otherwise remote nesting locations, and can degrade foraging habitat for prairie falcons and their prey (Boyce 1982). Boyce (1982) found that nests located closer to roads had significantly higher levels of human disturbance and lower rates of productivity when compared to more isolated nest sites (Boyce 1982).

Intensive military training can also disrupt foraging activities, which could be adverse during the nesting season (Steenhof et al. 1999). Public lands in areas such as the Mojave Desert are subject to a great deal of human disturbance, resulting in high rates of nest failure and unoccupied territories (Boyce and Garrett 1977, Boyce 1982). Nests located higher in canyons, mountain slopes, or on buttes have a better chance of successful reproduction (Boyce 1982). Nesting territories that are currently secluded may receive increasing pressure in the future, as human population levels continue to rise.

The effects of human activities can be detrimental in the long-term, especially when land use changes are involved. Habitat alteration in California's rangelands and deserts sets in motion a trophic cascade that may alter ecosystem function, starting at the level of primary production to prey species the prairie falcon depends on. Additionally, we do not fully understand what impact climate change and a trend towards a drier climate will have on prairie falcon prey populations (Unitt 2004).

Another threat to survival of the prairie falcon is the increase in wind energy projects throughout California, especially in areas where collisions with wind turbines may impact local breeding populations. An estimate of annual prairie falcon deaths in the Altamont Pass Wind Resources Area (APWRA) was initially 1.1 (0.2-2.0) (Smallwood and Thelander 2008). Later estimates by Smallwood (2010) used two different scavenger removal rates to estimate annual wind turbine-caused mortality, with the latter estimate accounting for scavenger swamping (Smallwood 2007, Smallwood et al. 2010). Searcher detection and scavenger removal rates based on national averages estimated 1.3 (0.3-2.2) APWRA fatalities to prairie falcons per year (Smallwood 2007, Smallwood 2010). A more recent rate using a camera-trap study at Vasco Caves Regional Preserve yielded a higher estimate of 2.2 (0.5-3.8) APWRA fatalities per year (Smallwood 2010, Smallwood et al. 2010). While these three different estimates appear to be low, extensive telemetry studies (in press) in the east Bay Area indicate about 5 to 7 pairs nest in the Altamont Pass area (D. Bell, personal communication 2012). Losing a couple breeding adults per year could have significant impacts on productivity for this local breeding population (D. Bell, personal communication). If these mortality rates apply to other current (e.g., Tehachapi Pass, San Gorgonio Pass) and future large-scale wind energy farms, then the impacts to the California prairie falcon population would be cumulative, and may be significant, especially when coupled with other threats.

Data Gaps and Monitoring Needs

Future scientific study on the prairie falcon in California should be modeled after the extensive body of research conducted by Karen Steenhof and her associates at Snake River NCA (Steenhof et al. 1984, Steenhof and Kochert 1988, Vakasy et al. 1996, Steenhof et al. 1999, Lehman et al. 2000, DeLong and Steenhof 2004, Steenhof et al. 2005, Steenhof et al. 2006). While there have been statewide studies assessing the distribution and productivity of select breeding populations in California, only one study in Pinnacles NM has assessed the importance of multiple core use areas and resource selection (Garrett and Mitchell 1973, Boyce et al. 1986, Buranek 2006). Findings from Buranek's (2006) thesis work emphasize the lack of baseline data on population dynamics for the California prairie falcon.

To fill statewide data gaps on the prairie falcon, regional population assessments using year-round radio/satellite telemetry and nestling/adult banding in the northern California foothills, portions of the Great Basin in California, central California, and the southern California deserts are all warranted. Studies similar to work done by Steenhof et al. (2005) and Buranek (2006) would provide valuable information on regional differences between home range, habitat use, key foraging areas, migration, mortality, site fidelity, and territoriality. This baseline data, for both the breeding and non-breeding seasons, would be beneficial for guiding land acquisition and the conservation of important habitat elements. Because we know prairie falcons center their foraging activities on available, abundant prey (mainly ground squirrels) across a vast landscape, conducting a long-term analysis of prey population dynamics in conjunction with core use area research would be helpful in making future land use decisions favorable to prairie falcons and their prey.

A follow-up study to the research done in the 1970s is important to estimate long-term changes in population size, historical eyrie occupancy, productivity per breeding pair, and overall status of the prairie falcon in California (Garrett and Mitchell 1973, Boyce et al. 1986). A prairie falcon monitoring protocol has been developed for Pinnacles NM, and would be useful elsewhere (Emmons et al. 2011). Conducting genetic research via blood samples from regional nesting populations would be beneficial in determining the degree of genetic interchange throughout the state (D. Bell, personal communication 2012). The results of the latter study could profoundly affect monitoring needs. If regional populations are genetically unique, it would be important to maintain those pockets of genetic diversity for the long-term viability of the species.

Determining levels and types of human disturbance near nesting areas would be valuable in assessing changes in population status when compared with past disturbance levels. Effects from consuming poisoned prey and the threat of secondary poisoning warrants further research. At Pinnacles NM, where much long-term monitoring on raptors has been done, prairie falcons are thought to be vulnerable to rodenticides and other pest control chemicals (Jarman et al. 1996, Buranek 2006, Emmons 2011). Concretely determining this is difficult, as fatalities to prairie falcons are likely to be scavenged before being found and starvation may be the more important indirect effect of prey poisoning (Emmons 2011). This highlights the importance of avoiding use of these biocides in core areas determined important to prairie falcons through telemetry studies (Buranek 2006, Emmons 2011). Habitat monitoring for this species should consider each seasonal use area, at all trophic levels (Steenhof et al. 1999). Prohibiting rodenticide and other biocide use near small or declining prairie falcon populations would have high conservation value.

With recent emphasis on “green energy”, assessing core resource areas for the prairie falcon will be vital to assist in suitable placement of large-scale wind turbine farms and solar facilities. The impacts from these developments are largely unknown, and may result in direct losses to breeding populations and prey habitat. Close monitoring is needed to maintain the “metropolis” of range for the California prairie falcon, especially in the desert public lands, where disturbance levels are known to be high.

Large-scale agricultural and urban development of suitable habitat results in habitat loss and fragmentation, leading to degradation of natural communities that have evolved over millennia (Huenneke 1989, Buranek 2006, Pandolfino et al. 2011). In order to conserve well-distributed nesting populations of prairie falcons in California in perpetuity, it is important we understand the combined effects of habitat loss and fragmentation, and the cumulative impacts of all threats.

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ECOLOGY OF THE SHARP-SHINNED HAWK

By Kevin P. Cahill

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The sharp-shinned hawk (*Accipiter striatus*) is named for its long thin laterally-compressed tarsometetarsi and tibiae that are only feathered on the proximal anterior quarter causing the appearance of a “sharp shin” (Bildstein and Meyer 2000, Wheeler 2003). An especially long middle toe and large eyes are adapted for capturing highly mobile prey (Bildstein and Meyer 2000). Short rounded wings and a long tail enable dexterous maneuverability through their woodland habitat features in pursuit of prey (Palmer 1988). Other names include sharpy, sharpshin, and little blue darter due to their fast bursts of speed in pursuit of prey and tendency to dart from concealment in surprise attack (Johnsgard 1990). Three of ten recognized subspecies are found in the United States, *A. s. perobscurus*, *A. s. velox*, and *A. s. suttoni*.

These hawks are the smallest of the three North American accipiters and have the greatest sexual size-dimorphism of any North American bird (Palmer 1988). Females are the larger of the two sexes and have little to no size overlap with males. Females weigh 120 to 210 grams (g) with a 51 to 68.5 centimeter (cm) wingspan. Males weigh 82 to125 g with wingspans from 25.7 to 34 cm (Palmer 1988, Wheeler 2003).

Adults have fine rufous horizontal barring that covers white under parts, and heavily barred flight feathers. The cheeks are rufous and accentuated next to the dark crown and grayish-blue nape and back. Females have more brownish-olive tinges on the crown, nape and back, with softer muted coloring than males (Clark and Wheeler 1987, Palmer 1988). The tail has thick, alternating dark and light barring with a white terminal band, and is typically square at the tip and can be notched in the center (Palmer 1988, Clark and Wheeler 1987). Eye color changes with age, being grey at hatching and transitioning to pale yellow, orange, and then red. Male eyes tend to change earlier than females and the darker reds indicate older birds (Palmer 1988, Wheeler 2003). Juveniles have a dark brown head with a pale superciliary line, and brown back and upperwing coverts. Below, juveniles are white with thick, reddish-brown, vertical streaking and flight feathers similar to adults (Clark and Wheeler 1987). Juveniles usually reach adult plumage at 2 years of age but males can acquire adult plumage within 1 year (Wheeler 2003).

The larger female sharp-shinned hawk is often confused with the male Cooper’s hawk (*Accipiter cooperii*). However, the Cooper’s hawk is always longer in length and typically wider in wingspan. In flight, the head of the sharp-shinned hawk appears much smaller and barely extends beyond the protruding wrists, while Cooper’s hawk appears to have a large head protruding far beyond the straight leading edge of the wings (Wheeler, 2003). When perched, Cooper’s hawk is more robust with thicker, more heavily feathered legs and eyes set farther forward. It can raise its hackle creating a squarish head and has a darker cap. The Cooper’s hawk tail may also appear more rounded with a wider terminal band (Clark and Wheeler 1987).

During 2006 through 2010 in California, seventeen wild sharp-shinned hawks were harvested for their use on falconry; six immature, nine nestlings, and two of unknown age. While in captivity, one nestling suffered predation, and one immature male died from the respiratory disease (Aspergillosis). Seven sharp-shinned hawks were released back into the wild, and one was lost (CDFG 2012).

Habitat

Sharp-shinned hawks are a forest species occurring in young to mid-aged conifers and mixed deciduous-coniferous stands from sea level to near alpine altitudes throughout the year (Palmer 1988). Forest stands with dense, shallow canopies and an abundance of dead limbs beneath the crowns are preferred in the vicinity of clearings and brushy areas where prey (i.e. small birds) is plentiful (Reynolds et al. 1982, Palmer 1988).

In California, nesting occurs from near sea level to over 2133 meters (m) (7000 feet (ft)) (Grinnell and Miller 1944, Small 1994). Hemlock (*Tsuga* spp.), pine (*Pinus* spp.), and Douglas-fir (*Pseudotsuga menziestii*) are the most important nest trees throughout North America but other fir (*Abies* spp.), spruce (*Picea* spp.), oak (*Quecus* spp.), maple (*Acer* spp.), poplar (*Populus* spp.), and willow (*Salix* spp.) are also used. Nest sites are frequently in moist shaded forested ravines (Apfelbaum and Seelbach 1983, Reynolds et al. 1982). Of the three accipiter species found in North America (the others being Cooper's hawk and northern goshawk (*Accipiter gentilis*)), sharp-shined hawks use stands with the smallest diameter trees and shallowest crowns (Reynolds et al. 1982, Siders and Kennedy 1996). Diameter at breast height (dbh) ranges from15.2 to 35.7 cm in the west but can reach up to 60 cm (Reynolds et al. 1982, Siders and Kennedy 1996, Carlson 2006). Nest sites also have the greatest mean percent canopy cover and tree density when compared to Cooper's hawks and northern goshawks (Reynolds et al. 1982, Siders and Kennedy 1996, Moore and Henny 1983). Canopy closure and tree diversity in the Southern Sierra was higher in stands used by sharp-shinned hawks than stands used by Cooper's hawks (Carlson 2006).

Across North America nests heights average 9 m (30 ft.). Nests tend to be placed higher in the Pacific Northwest and lower in the Western Canadian provinces (Apfelbaum and Seelbach 1983). Nests are usually located a few meters from the tops of trees against the trunk in conifers or at branch points slightly below the tops in deciduous trees (Apfelbaum and Seelbach 1983, Reynolds et al. 1982). In Oregon, nesting habitat consisted of relatively young (25 to 60 years), dense, even-aged conifer stands, typically fir, ranging in height from 9.2 to 16.7 m, on gentle slopes with northwest to northeast exposures (Moore and Henny 1993, Reynolds et al. 1982). Nest trees were generally within 91 to 180 m of water, and nests were placed 12 to 23 m high in the denser portion of the canopy against the trunk or in a crotch of a split trunk (Reynolds et al. 1982). In California's Southern Sierra Nevada mountains, nest trees were located on a mean 35% grade and included second growth giant sequoia and white fir that averaged 30.5 m tall (Carlson 2006). Within these trees, nest were placed an average of 21.9 m high (Carlson 2006). This same study found the presence of snags important for hunting, observation, and roosting, as well as providing habitat for potential prey. In New Mexico, tree heights averaged 12 to 19 m (Siders and Kennedy 1996). In Utah, nests were located in dense mixed conifer and deciduous stands below the top of a well-developed canopy (Platt 1976). Moore and Henny (1983) suggest that concealment of nests and shading from sunlight may determine nest placement.

Coleman et al. (2002) found urban sharp-shinned hawks in Montreal, Canada, nested near edges that were in close proximity to open space. These open spaces often had considerable recreational activity, including a heavily used all-terrain vehicle trail, walking trails and a golf course. While they used stands with significant deciduous cover, every nest was located in a conifer (Coleman et al. 2002). Coleman et al. (2002) suggest that the location of nests near edges may be due to the lack of available large tracts of forested areas in the study area.

Migration

Sharp-shinned hawks routinely follow north-south migration pathways along leading lines such as mountain ranges and coastlines, and exhibit age and sexually differential migration (Mueller and Berger 1967, Bildstein and Meyer 2000, Hoffman et al. 2002). Migration generally takes place from early- to mid-August through early-December, with the bulk of it occurring from early-September to late-October. Peak migration periods are variable and coincide with weather factors such as cold fronts, wind direction and velocity, and sunshine (Bent 1961, Mueller and Berger 1967). Increasing temperatures, days preceding cold fronts and north headwinds were correlated with higher counts (Hall et al. 1992). Peak captures at the Palomarin Field Station in coastal Marin County, California, occurred from late-September to mid-November (Culliney and Gardali 2011). Likewise, counts at the Golden Gate Raptor Observatory (GGRO) in California's Marin Headlands increase rapidly in early-September peaking in late-September (Figure1, Elliot and Fish 2010).

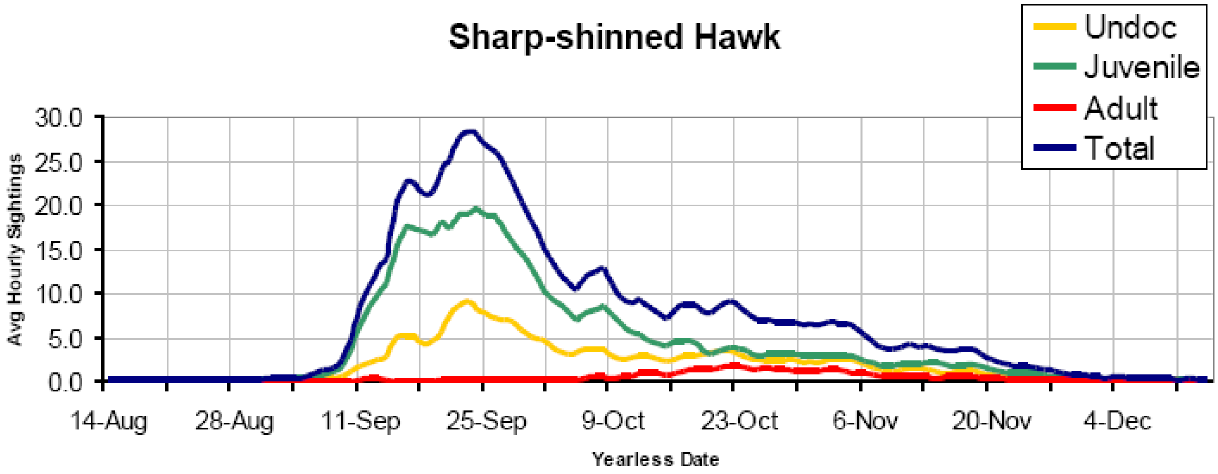

Figure 1. Migration profile for sharp-shinned hawk at the GGRO, Marin County California, from 1986 to 2009

These hawks are high altitude migrants that typically use soaring flight, but also engage in active flap and sail flight (Bildstein and Meyer, 2000). Mueller and Berger (1967) also found that sharp-shined hawks will take advantage of thermal updrafts, fly higher in tail winds and lower in headwinds, with less of a tendency to follow leading lines in tail winds.

Juvenile hawks precede adults principally during the first month of fall migration while adults dominate the last one and a half months, and within age groups females precede males (DeLong and Hoffman 1999, Mueller et al. 2004, Rosenfield et al. 2011). Hoffman et al. (2002) found that adults tended to winter farther north than younger birds, however juveniles and some adult females will travel as far south as Middle America, Panama, the West Indies and Costa Rica (Wheeler, 2003). According to DeLong and Hoffman (1999), age-specific timing may reflect the ability of experienced adults to capitalize on a diminishing food supply longer than inexperienced juveniles. Sex-specific timing may be due to an earlier completion of molt by females or may reflect faster female flight with fewer foraging stops then males (DeLong and Hoffman 1999)

Migration patterns in the west may reflect the "coastal effect" proposed by Ralph (1981); this theory suggests a higher proportion of juvenile birds use coastal migration pathways, whereas adult birds use safer inland migration pathways. For California, this theory has not been confirmed, however about 80% of the accipiters identified to age at GGRO were juvenile birds, and 88% of the hawks captured at the Palomarin Field Station were juveniles (Figure 1, Elliot and Fish 2010, Culliney and Gardali 2011). Inland hawk monitoring stations reported a more balanced juvenile representation at 41.9% to 70.3% (DeLong and Hoffman 2004, Mika and Hawks 2011a, Mika and Hawks 2011b, Smith 2010).

Shifts in migration timing as well as declines in number of hawks observed have been documented at Eastern North American hawk watch sites (HMS 2012, CMBO 2012, Rosenfield et al. 2011). Fall migration from 1974 to 2009 shifted an average of four days later at Hawk Ridge Bird Observatory, Duluth, Minnesota and was consistently eight to nine days later during late September through October from 1994 to 2009 compared to 1974 (Rosenfield et al. 2011). This pattern may reflect extended food availability in breeding grounds due to delayed songbird migration resulting from longer summers. Migratory "short-stopping" (pausing before full migration is complete) may also be occurring in relation to abundant songbird populations concentrated at increasingly common backyard bird feeders along migratory pathways (DeLong and Hoffman 1999, Rosenfield et al. 2011).

Food Habits

Sharp-shinned hawks primarily prey on small birds but will also prey on mammals, reptiles, and insects (Bent 1961, Bildstein and Meyer 2000, Palmer 1988, Wheeler 2003). The size of prey taken varies between males and females. Breeding males in New Mexico and Arizona almost exclusively took prey smaller than 25 g, while females took prey larger than 25 g in addition to smaller prey (Snyder and Wiley 1976). In one Alaska study, an adult female killed and began plucking a marbled murrelet (*Brachyramphus marmoratus*) weighing 200 g (Marks and Naslund 1994). Females have also been known to attack black-crowned night herons (*Nycticorax nycticorax*), weighing from 727 to 1014 g, as well as pileated woodpeckers (*Dryocopus pileatus*), weighing from 228 to 355 g (Bull and Jackson 2011, Gross 1923, Palmer 1988). Hawks are more successful when solitary prey is attacked, especially if prey is actively feeding (Roth et al. 2006).

In the Southern Sierra Nevada Mountains, 100% of analyzed prey remains were passerines; of this, 60% were 30 g or less and 40% were larger than 30g (Carlson 2006). Reynolds and Meslow (1984) found that 308 of the combined 315 prey items noted in eastern Oregon were bird species. Yellow-rumped warblers (*Setophaga coronata*), American robins (*Turdus migratorius*), white-crowned sparrows (*Zonotrichia leucophrys*), and dark eyed juncos (*Junco hyemalis*) were the most common prey species in Colorado, where 91.1% of prey consumed at successful nests was birds, and 60% of the mammals taken were voles (Joy et al. 1994). Joy et al. (1994) also found that 67% more prey was collected during the nestling period than during other life stages, with mammal consumption increasing from incubation to fledging and avian prey mass decreasing between incubation and nestling stages. Hawks in Indiana preyed almost exclusively on birds and had a distinct preference (in order of importance) for house sparrows (*Paser domesticus*), northern cardinals (*Cardinalis cardinalis*), European starlings (*Sturnus vulgaris*), American robin, and dark-eyed juncos (Roth et al. 2006).

Prey is hunted in the upper canopy of tall trees, the lower canopy of small trees, and from concealed perches with a clear view of the ground (Palmer 1988, Wheeler 2003). Hunting also occurs in open areas, along forest edges, hedgerows, and at backyard bird feeders. Attacks are made from fence posts, rocks, and concealed perches as well as in aerial pursuit (Bent 1961, Bildstein and Meyer 2000, Palmer 1988, Wheeler 2003). Hawks will glide low to the ground making abrupt turns or drops as well as dashing madly through dense foliage and branches (Bent 1961). Natural and man-made obstructions such as bushes, fences, or buildings are utilized as concealment to surprise prey (Bent 1961, Palmer 1988, Wheeler 2003). Sharp-shinned hawks will also chase their prey across the ground attacking with a series of wing aided jumps (Bent 1961). Prey is almost completely consumed in dense cover on the ground or on a perch less than one meter above the ground. Hawks are also known to cache larger kills for consumption the following day (Roth et al. 2006). These small hawks will steal fresh kills from human hunters and other avian and mammalian species (LaFave 1955, Bent 1961). In one instance LaFave (1955) witnessed repeated attacks on a bobcat (*Lynx rufus*) apparently in competition for a ruffed grouse (*Bonasa umbellus*).

Backyard bird feeders are a common source of prey. In a continent-wide winter survey, Dunn and Tessaglia (1994) found that sharp-shinned hawks were the top-ranked predator at feeders and were responsible for 35% of all predation deaths in which the predator was identified. Sites more likely to report avian predators were located near woodlots, in well foliaged yards, in yards with a variety of bird feed, and in yards with many feeders (Dunn and Tessaglia 1994). Hawks wintering in Indiana had a 20.5% attack success rate at feeders (Roth and Lima 2007a). Historically sharp-shinned hawks took advantage of human fed birds as well, especially poultry. Bent (1961) describes this hawk's crafty approach to the poultry yard and the seizure of small chickens before detection. This behavior prompted the name “chicken hawk”, and facilitated stigmatizing labels such as “the terror of all small birds and the audacious murderer of young chickens” (Bent 1961).

Sharp-shinned hawks may the victims of intraguild predation from owls and other hawks. Forty-seven percent of mortality suffered by wintering hawks in Indiana was caused by predation by owls and Cooper's hawk (Roth et al. 2005). Because of this, these hawks show aggressive behavior when presented with a potential predator. Just over 73% of migrating hawks that approached a great horned owl model in an east coast study went out of their way to reorient their course, allowing them to make aggressive passes from behind, which in some cases were accompanied by vocalizations (Kerlinger and Lehrer 1982). Forage timing may also reflect the high risk of owl predation experienced by these hawks. Indiana hawks left the roost at sunrise or later and returned to roost well before sunset, a timing that coincides with higher predation risk from owls around sunrise and sunset (Roth and Lima 2007b).

Breeding

Sharp-shined hawks generally arrive at breeding sties between April to mid-May, at which point males set up territories and pairs start refurbishing or building nests (Henny et al. 1985, Bent 1961, Palmer 1988). In California, these hawks mainly nest in northern portions, and inhabit semi-open lowlands and arid regions of the state from September through April (Hunter et al. 2005). These hawks do not appear to show site fidelity, however nest groves are often re-used with new nests being constructed (Platt 1976, Palmer 1988, Johnsgard 1990). In Oregon new nests were usually built within 100 m (328 ft) of old ones (Reynolds and Wight 1978). Occasionally old nests are refurbished and used, and nests of other species, such as American crow (*Corvus brachyrhynchos*), great-horned owl (*Bubo virginianus*), and squirrels (*Sciuridae*), may be used (Wheeler 2003). Both the male and female provide nesting material, but females do most of the nest construction (Bildstein and Meyer 2000).

Males attract females with territorial calling and flight displays such as high-circling over the nest grove and sky-dancing, which includes shallow undulations and high velocity dives as well as tail-fanning in flight and from a perch (Palmer 1988, Wheeler 2003). The pair may also engage in pursuing one another in flight (Bent 1961). Once a pair-bond is formed, copulation occurs repeatedly within the defended territory but away from the nest (Palmer 1988). After copulation, the pair will perch together for extended periods, mating multiple times (Bent 1961). During breeding, both sexes will vigorously defend the territory against intruders (Bent 1961, Bildstein and Meyer 2000).

Egg laying starts from mid-March to early-June, soon after completion of the nest (Johnsgard 1990). A pair in Oregon laid their first egg 6 days after first being noted on the nest (Reynolds and Wight 1978). Eggs are laid on alternating days and incubation starts after the final egg is laid (Johnsgard 1990). In California, completion of a clutch generally occurs between May 22 and June 11 (Palmer 1988). Average clutch size is 3.9 eggs in North America (Apfelbaum and Seelbach 1983, Bildstein and Meyer 2000).

Incubation is done exclusively by the female for an average of 30 to 35 days (Henny et al. 1985, Palmer 1988). During incubation males roost nearby and provide food to the female (Snyder and Wiley 1976, Delannoy and Cruz 1988, Bildstein and Meyer 2000). Males leave the nest vicinity once eggs hatch but continue to deliver food to the females and young (Bildstein and Meyer 2000). Females begin foraging when nestlings are 10 to 15 days old (Bildstein and Meyer 2000, Johnsgard 1990, Snyder and Wiley 1976).

Young attain full feathers after two weeks of age, and fledging occurs between 21 and 30 days after hatching (Platt 1976, Reynolds and Wight 1978, Henny et al. 1985, Bildstein and Meyer 2000). First flights occur below the canopy (Bildstein and Meyer 2000). Both adults continue food delivery after the young fledge by dropping prey into the nest or by mid-air transfer (Delannoy and Cruz 1988, Bildstein and Meyer 2000). Young attempt hunting on their own by 40 to 47 days of age (Mueller et al 1981, Delannoy and Cruz 1988). Food delivery rapidly decreased when young are 42 to 47 days old and by day 49 young start dispersing from their natal territory at distances up to 2 kilometers (km) (Mueller et al 1981). In Oregon, Wyoming, and Utah, nests fledged between 2.7 and 3.5 young with Oregon having an 81.2% fledgling successful rate (Reynolds and Wight 1978). Parents do not appear to distinguish between their own young and young dispersing from other nesting territories, but resident young will aggressively defend against intruders (Bildstein and Meyer 2000).

Juveniles reach breeding age by 2 years, however younger individuals are known to breed successfully (Bent 1961, Henny et al. 1985, Palmer 1988, Bildstein and Meyer 2000, Coleman 2002). Montreal's urban population for example had a large proportion of breeders in immature plumage at 33.5% and 38.5%, male and female respectively (Coleman 2002).

Home Range

Reynolds (1989) cites several studies which estimate home ranges and density of sharp-shinned hawks. The home range size for Alaska hawks was approximately 380 hectares (ha) (939 acres (ac)) with 23.8 nests per 10,000 ha (24710.5 ac), Canada hawks ranged 706 ha (1,744.6 ac), Oregon hawks 460 ha (1136.7 ac) and 3.6 to 5.3 nests per 10,000 ha (24710.5 ac), range was 100 to 1,248 ha (247.1 to 3083.9 ac) in Wyoming, and 4.0 nests per 10,000 ha (24710.5 ac) in Utah (Reynolds 1989). Mean distance between neighboring nests was 4.1 to 5.5 km in Oregon, where actual densities were likely higher than reported but difficulty in locating nest sites caused a sampling bias. Nesting areas in Oregon were approximately 4 ha and often contained several nests, indicating variable nest site use over multiple years (Reynolds and Wight 1978). Some nesting areas in Utah contained up to five old nests (Platt 1976). In both the Oregon and Utah studies, only one pair was observed using the previous year's nest (Platt 1976, Reynolds and Wight 1978). In the southern Sierra Nevada, Carlson (2006) confirmed a density of between 0.12 and 0.29 sharp-shinned hawks per 100 ha (247.1 ac), with one nest every 340 to 849 ha (840.2 to 2097.9 ac).

Males are known to actively defend an area of about 75 m in diameter, and fly up to 1.7 km from their defended area to forage, resulting in a home range of approximately 120 to 270 ha (296.5 to 667.2 ac) (Palmer 1988, Bildstein and Meyer 2000). Females have a more circular range approximately 90 to 140 ha (222.4 to 346 ac) centered on their nest (Bildstein and Meyer 2000).

Historical and Current Distribution

Historically, sharp-shinned hawks were common in fall, winter, and early-spring in the low country of the Southern California Pacific slope and on the Santa Barbara Islands (Willit 1912). Nesting was confirmed above 7000 ft. in the San Bernardino Mountains (Willit 1912). They were also considered a common resident throughout Placer County (Adams, 1909). Grinnell and Miller (1944) described the sharp-shinned hawk's California distribution as wide and abundant in the winter from September to April below heavy winter snows and in Southern California. Nests were found in Shasta County near Redding; Mendocino County at Mount Sanhedrin; Lake County near Kelseyville; Berkeley, Oakland, and San Lorenzo in Alameda County; Stanford in Santa Clara County; Monterey County near Carmel in the Carmel River bottom; and Bear Valley in the San Bernardino Mountains (Grinnell and Miller 1944). Grinnell and Miller (1944) also confirmed the dark subspecies (*A. s. perobscurus*) on the coast of British Columbia but were uncertain of its presence in California due to the lack of morphological data on California variants at the time. Bent (1961) notes breeding records in the Inyo Mountains, Star Lake, Mount Sanhedrin, Mineral, and Redding.

Today, sharp-shinned hawks have a generally northern distribution in summer but extend southward in the west and east wherever forest cover is sufficient (Wheeler 2003). During the winter northern birds migrate south as far as Central America, the West Indies, and Costa Rica (Bildstein and Meyer, 2000, Wheeler 2003). Two of its ten subspecies, *A. s. perobscurus* and *A. s. velox*, occur in the United States and Canada. A third, *A. s. suttoni*, rarely extends its normal range in Mexico into the extreme southeast Arizona, southwest New Mexico and southeast Texas (Bildstein and Meyer 2000, Wheeler 2003). *A. s. velox*, the most common subspecies, breeds throughout Canada, southeastern and coastal Alaska, and in forested regions of the eastern and western United States (Bildstein and Meyer 2000, Hunter et al. 2005). *A. s. perobscurus* breeds on Queen Charlotte Island, British Columbia and possibly the coastal mainland of Southeast Alaska and British Columbia, and winters along the Pacific Coast from Northern Oregon to as far south as Los Angeles and Riverside Counties in California (Wheeler 2003). Some birds are residents year-round or considered partially migratory moving only short distances to their wintering grounds, while others travel as far south as Central America, Panama, the West Indies, and Costa Rica (Palmer 1988, Bildstein and Meyer 2000, Wheeler 2003).

Distribution in California varies by county. While not considered numerous, according to the Humboldt County Breeding Bird Atlas, sharp-shinned hawks can be found nearly anywhere in Humboldt County during the breeding season especially in the region of the Hoopa Valley Indian Reservation. However, only three confirmed nests were reported (Hunter et al. 2005). These hawks are usually encountered in oak woodlands, coniferous forests and riparian zones from fall through winter in Shasta County but are considered rare in the summer (WAS 2005). In Plumas County these hawks are considered a year-round resident (PAS 2007). Data is lacking for the Klamath-Trinity Mountains and much of the Central Sierra Nevada Mountain Range.

From 1998-2002 sharp-shinned hawks nested and were observed throughout Contra Costa County (CCBBA 2002). Sharp-shinned hawks can be found almost anywhere in Napa County during the winter but becomes less common during the breeding season (Berner et al. 2003). Santa Clara surveyors documented nesting in the Diablo Range as well as the Santa Cruz Mountains although breeding was considered rare and largely absent from the valley floor (Bousman 2007). In Santa Cruz County, this hawk is noted as fairly common during the fall, winter, and spring, but is uncommon during the summer (Suddjian 2009). Today, habitat in the Carmel River bottom of Monterey County is likely too fragmented to support nesting however most of the remaining Monterey County nesting habitat is protected within Los Padres National Forest (Roberson 1993). In Santa Barbara County, this hawk is an uncommon to fairly common transient and winter visitor, and a fairly common migrant and winter visitant in Orange County (Hamilton and Willick 1996, Lehman 1994). The winter population in San Diego is considered widespread but uncommon, and although nesting has been confirmed, the last known breeding pair was recorded in 1978 (Unitt 2004).

In Yosemite National Park, the sharp-shinned hawk is a rare winter resident below 1,219 m (4,000 ft), rare summer and possibly winter resident from 1,219 to 2,133.6 m (4000 to 7000 ft), uncommon to rare below 2,438.4 m (8000 ft) east of the Pacific Crest, and an uncommon transient from the foothills to the tree line on the western slope of the Sierra Nevada Mountains (Gaines 1988). Only one breeding record from 1930 is documented in the Yosemite Valley and breeding is suspected but unconfirmed on the west slope at Crane Flat and on the east slope near Walker Lake and the Glass Mountain region (Gaines 1988). In Sequoia National Forest, breeding was documented near Dinkey Creek (Gaines 1988). Carlson (2006) estimated nine pairs in Tulare County at a managed 3,398 ha (8396.6 ac) study area in the Southern Sierra Nevada Mountains.

Historical and Current Abundance

During the early part of the 20th century, sharp-shinned hawks were widely distributed throughout North America and considered very common in practically all of the United States and Canada during at least one season of the year, and especially abundant along migration routes (Bent 1961). The hawk was known to breed rarely throughout its range but known breeding was concentrated in the Eastern Provinces of Canada (Bent 1961). In California, Grinnell and Miller (1944) described sharp-shinned hawks as abundant and widely distributed during winter below the level of heavy snows. Grinnell and Miller (1944) noted that summer abundance was low with breeders chiefly located in the northern half of California and south along mountain ranges, with only a few summer occurrences in the southern mountain ranges.

Due to declines in breeding numbers in Contra Costa, Alameda, and Santa Cruz Counties, and a decline in winter numbers in the 1950’s through early 1960’s, the sharp-shinned hawk was placed as a third priority species on California’s Bird Species of Special Concern in 1978, and again on the unpublished list in 1992 (Remsen 1978, California Department of Fish & Game 1992 unpub. list). It was also included on the National Audubon Society’s “Blue List” from 1972 to 1986 due to its rare breeding status in the United States (Tate 1986). The sharp-shinned was not included as a Species of Special Concern in 2008 due to a lack of evidence pointing to declines, but this species was included on the list of taxa to watch (Shuford and Gardali 2008).

In California, sharp-shinned hawks become widespread during fall and winter months as birds from higher elevations join migrants in the lowlands and southward (Fish and Hull 2010, Small 1994). According to Wheeler (2003) *A. s. velox*, is one of the most common raptors in Western North America. Sharp-shinned hawks are considered the most difficult raptors in North America to census during summer months because they are rarely seen during the breeding season (Bildstein and Meyer 2000). The Breeding Bird Survey (BBS) data can be used to asses abundance and trend, however, results should be interpreted cautiously as a certain level of bias is associated with BBS roadside survey because secretive birds that inhabit forested areas, like the sharp-shinned hawk, are not always detected during surveys (Blancher et al. 2007). According to BBS criteria, sharp-shinned hawk data contains important deficiencies which make data imprecise (Sauer et al. 2011).

Partners in Flight (PIF), using BBS data, estimated a population size of 600,000 sharp-shinned hawks in North America, representing 56% of the global population of 1,100,000 (Rich at al. 2004, PIF 2012). Based on the evaluation of 200 California BBS routes, PIF estimated 12,000 sharp-shinned hawks in California (Rich at al. 2004, PIF 2012). A slight to moderate decline in the future suitability of breeding conditions in the Sierra Nevada, Central, and Coastal Regions are predicted, with conditions in the northwestern portion of the state, including the coastal area north of San Francisco, expected to remain stable (Rich at al. 2004, PIF 2012).

Population Trends

Much of what is known regarding sharp-shinned hawk historic and current population trends are from migration count data. Early records were not consistently kept as this hawk was considered a pest species. Records that were kept were those where shootings of sharp-shinned occurred. One account from 1907 claimed that more hawks were migrating through Ontario, Canada, than existed in the area, and at one point 281 hawks were observed in a 20 minute span, all of which were “within gunshot” (Bent 1961).

During the early 1900s, sharp-shinned hawks were considered very common in North America, and especially abundant along migration routes (Bent 1961). In the mid-1900s, sharp-shinned hawk population declines were noted in the east, and were largely attributed to the use of dichlorodiphenyltrichloroethane (DDT) and other organochlorines, however numbers began to rapidly recover in the mid-1960’s (Bednarz et al. 1990). The affects of organochlorines on western populations was not thoroughly investigated.

Overall counts at Western North American watch sites increased from 1985 to 2005 however slight declines in the late 1990’s were possibly linked to widespread drought (Farmer et al. 2008). Recent shifts in timing and declines in count data have been documented in the east and are thought to be attributed to “short-stopping” at northeastern bird feeder stations during migration, as well as climate change affects on trophic networks such as the documented delayed songbird migration in conjunction with longer summers (Bildstein 2001, Chandler et al. 2004, Rosenfield et al. 2011). In California, from 2001 to 2009 sharp-shined hawks were consistently the third most numerous species observed behind red-tailed hawks and turkey vultures during GGRO fall migration counts; however numbers banded decreases from there; 767 individuals in 2006 to 362 in 2010, a phenomenon attributed to poor weather conditions (Fish and Hull 2010). Decade high GGRO migration counts occurred in 2002 and 2006 at 5,844 and 5,250 respectively, and lows occurred in 2001, 2009 and 2011 at 3,373, 3,332 and 2,609 respectively (Fish and Hull 2010, GGRO 2012). Low numbers in 2011 were possibly related to wet weather conditions (A. Fish 2012 pers. comm.).

Fifty years of National Audubon Society Christmas Bird Counts (CBC) for California from 1960 to 2010 show an increasing trend up through the early-1980’s followed by a stabilizing trend up until the last 10 years, where a slight decline can be detected (Figure 2, NAB 2012). Whereas the BBS data shows an increasing population trend in California (Figure 3, Sauer et al. 2011). Furthermore, BBS trend maps show detections increased more than 1.5 percent from 1966 to 2010 on routes along the Central Coast of California, in Mendocino National Forest in the southern portion of the Northern Coastal Ranges, and Klamath National Forest to the north in the Southern Cascade Range and Eastern Klamath Mountains (Figure 4, Sauer et al. 2011).

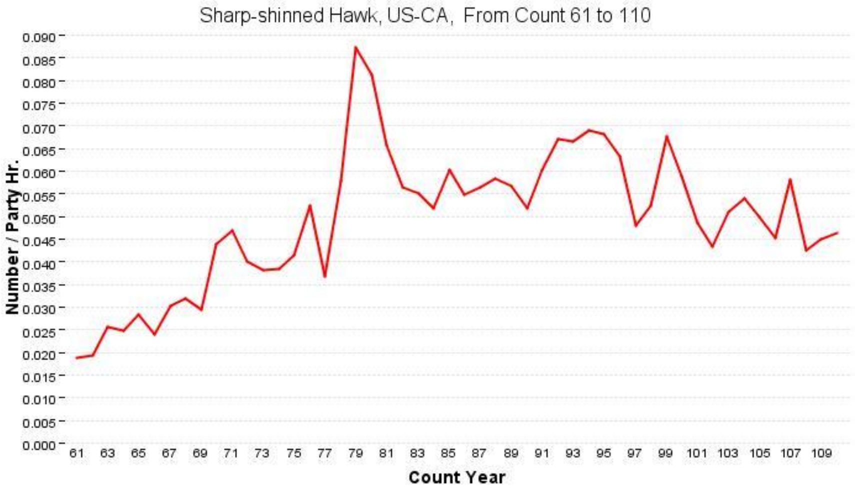

Figure 2. Number of sharp-shinned hawks detected per party hour from 1960-2010 in CA. (NAB 2012)

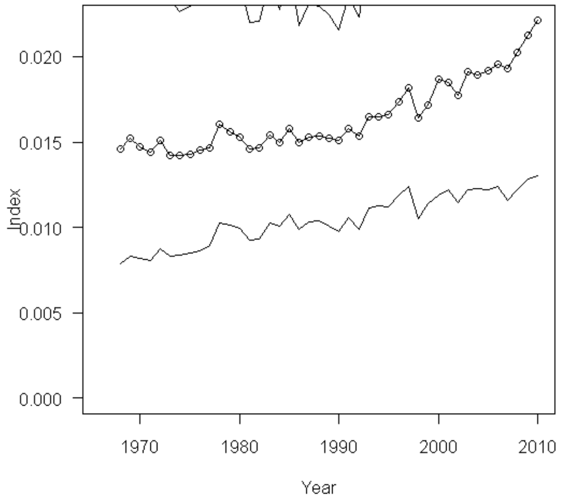

Figure 3. North American Breeding Bird Survey sharp-shinned hawk trend results 1966-2010 in CA (Sauer et al.2011).

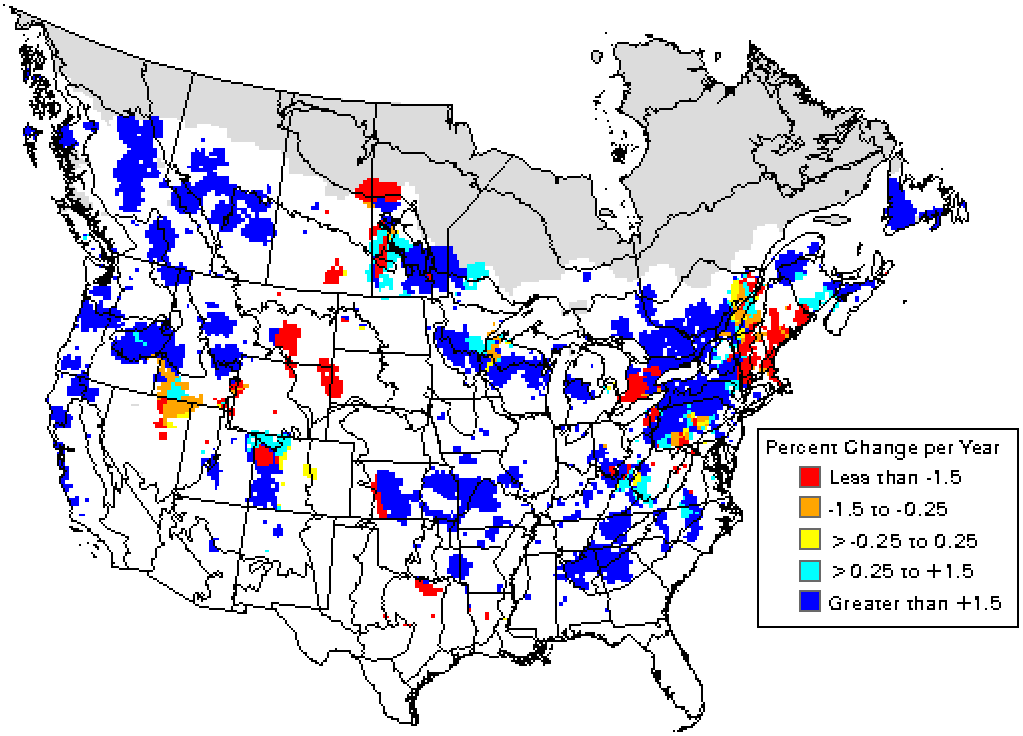

Figure 4. Geographic pattern in population trend map for sharp-shinned hawk in North America based on North American Breeding Bird Survey route detection data (Sauer et al. 2011).

Threats

Threats for sharp-shinned hawks are varied and include predation, collisions, contaminants, and loss of habitat. Owls and other predators may impact sharp-shinned hawk behavior causing them to adjust their own foraging times to avoid being preyed upon (Roth and Lima 2007b). A study in Illinois and Indiana attributed 12 of 23 known mortalities to intraguild predation (owls and Cooper's hawk) (Roth and Lima 2007b). Sharp-shinned hawks also exhibited aggressive behavior when presented with a threatening predator model (Kerlinger and Lehrer 1982)

Collisions with windows, automobiles, and other manmade structures are causes of mortality for sharp-shinned hawks (Bent 1961, Roth et al. 2005, Hagar 2008). Roth et al. (2005) reported several fatal collisions with windows and Bent (1961) relays an account of a sharp-shinned hawk crashing through double-layered greenhouse glass before being stopped by the third layer while in pursuit of prey. Hagar et al. (2008) reviewed 86 published sources and found that sharp-shinned hawks were the most frequently reported victim of window collisions among all raptors and that it was their leading source of mortality. Roth et al. (2005) observed several non-lethal collisions as well where hawks appeared to attack their own reflection likely believing it to be a competitor. Hoffman et al. (2002) found that the majority of band returns were due to collisions with human structures.

During the late 19th and early 20th centuries, these hawks were referred to as a "bushwhacker" and bounties were offered in order to eradicate this particular species of "vermin" as many raptors and all accipiters were labeled due to their predation on game and domestic stock (Bildstein 2001). Many of the historical observational accounts in Bent (1961) end with the subject being shot. Thousands were shot annually for sport along migration routes in the Eastern United States including at Hawk Mountain in Pennsylvania (Bildstein and Meyer 2000). Although data on shooting impacts is sparse, banding recoveries for the similar species, Cooper's hawk, suggest shooting accounted for 28 to 47% of first year mortality from 1929 to 1940 and 12 to 21% from 1946 to 1957 (Henny and Wight 1972). In 1934, however, a movement to protect the hawks began with the leasing of 565 acres north of Reading, Pennsylvania including Hawk Mountain, which became the first raptor sanctuary. However, the hawks were not fully protected under the law until the Migratory Bird Treaty Act was amended in 1972 to include raptors (Bildstein 2001). Due to federal and state protections, shooting deaths have substantially decreased and no longer have substantial impacts to sharp-shinned hawk populations in the United States (Bildstein and Meyer 2000). However, gunshot wounds still poses a risk for wintering hawks in Mexico with 96% of all shootings reported by Hoffman et al. (2002) occurring there.

Several contaminants are known to impact sharp-shinned hawks. Increased lead concentration in the blood of house sparrows and other common prey species may pose a significant threat, but few studies have been conducted to determine the level of threat (Chandler et al. 2004). Lead concentration levels in urban house sparrows in Vermont were more than 450% greater than sparrows found in agricultural settings (Chandler et al. 2004). Increasing urban sharp-shinned hawk numbers there correlated with increasing house sparrow abundance, which may lead to lethal levels of lead in the hawks (Chandler et al. 2004). The affects of organochlorines on western populations has not been thoroughly investigated, however, Snyder et al. (1973) found that while their sample size was small, sharp-shinned hawk eggshells had the highest concentration of dichlorodiphenyldichloroethylene (DDE) when compared to Cooper's hawk and northern goshawk. Sharp-shinned hawk numbers plummeted in the mid-1940s, but began to recover in the mid-1960's as the use of DDT declined (Bednarz et al. 1990). At Hawk Mountain, Pennsylvania, detection rates rebounded from approximately 6.27 per hour during the DDT era to approximately 14.2 per hour post-DDT era (1973 to 1995) (Bildstein and Meyer 2000). This suggests a correlation with the presence of DDT and hawk numbers. Accumulation in prey species of organochlorine pesticides used in Latin America still present a threat as many of the hawk's Neotropical prey items overwinter there and could pass sufficient contaminant levels up the food chain to effect eggshell thickness (Bildstein and Meyer 2000)

Timber harvest and loss of forest habitat pose a threat to nesting sharp-shinned hawks as their specific habitat requirements make them vulnerable to changes in forest stand structure (Reynolds 1983). Hawks may be able to relocate if enough suitable nesting sites remain available, however population reductions may occur if breeding habitat is limited (Moore and Henny 1983). Logging may also reduce prey availability and increase abundance of potential predators (e.g. great horned owl and red-tailed hawk (*Buteo jamaicensis*)) and competition for resources (Moore and Henny 1983). The effect of timber harvest has not been thoroughly studied and is hard to quantify due to the variability in re-occupancy and the elusive nature of this species (Carlson, 2006).

Captive sharp-shinned hawks are at risk of contracting the fungal respiratory disease Aspergillosis which is caused by the inhalation of spores produced from items such as moldy hay and is aggravated by stress (Abundis-Santamaria 2003). Poor sanitation or ventilation conditions, shipping, heat, capture, changes in management, and lowered immune response increase risk of contraction. The acute form which is seen most often in wild birds rapidly leads to death in less than one week. Prolonged treatment for the chronic form is generally ineffective.

Prevention includes good ventilation in housing facility, daily bedding changes, and general good management of birds. Vaccines exist but little research has been conducted as to their effects (Abundis-Santamaria 2003).

Data Gaps and Monitoring Needs

While some work has provided invaluable information on nest site selection and breeding biology in managed forests, there are no studies from California focused on this species (Carlson 2006). It is apparent that much is to be learned about population size, population dynamics, and habitat use of this species, as well as the implications of management decisions at the population level (Carlson 2006).

Much information could be gained from long-term studies over a broad area, designed to increase statistical rigor, especially in managed forests statewide (Carlson 2006). Determination of functional roosting habitat, screening trees, perch trees, and replacement trees within potential nesting and breeding territories will help forest managers when making decisions about which stands to maintain as suitable sharp-shinned hawk habitat (CalFire 2012). To ensure future nest site availability, forest managers should identify stands with appropriate vegetation characteristics and topography (CalFire 2012). Thinning of these stands will reduce tree densities, roost and nest materials, and create deeper crowns which are less than ideal for sharp-shinned hawks (Carlson 2006, Reynolds et al. 1982, Reynolds 1983). An understanding of the effects of forest use on breeding ecology and success, including the impacts of forest-patch size, age structure and species composition as well as how management effects interactions with other *Accipiters* is critical (Bildstein and Meyer 2000).

Focus should also be placed on the timing of breeding, fledging rates and fledgling development, as well as post-natal dispersal to better understand critical periods and landscape use during early development (Bildstein and Meyer 2000, Johnsgard 1990). The use of radio telemetry tracking has been successful for other secretive species such as northern goshawk, marbled murrelet, and Cooper’s hawk, and should be utilized for sharp-shinned hawks as well (Beier and Drennan 1997, Golightly et al. 2003, Mannan et al. 2004, Murphy et al. 1988).

Research into the effects of birdfeeders on winter distribution, abundance and survival, as well as sex differences in habitat use remain to be explored (Bildstein and Meyer 2000). Further studies are also needed to establish prey species preferences in California and to examine the effects that insect control methods, rodenticides, and avian poisons have on prey species and sharp-shinned hawks. The continued monitoring of biomagnification and contaminant levels is also warranted (Bildstein and Meyer 2000).

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