FINAL REPORT

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Project: Evaluation of potential habitat for Peregrine Falcon reintroduction in southern Illinois.

Funding Source(s): Illinois Dept. of Natural Resources, State Wildlife Grant T-30-P

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SUMMARY (from attached thesis)

In Illinois, the peregrine falcon (*Falco peregrinus anatum*) has not yet recolonized natural cliff sites, remaining restricted to urban areas. I identified cliffs in southern Illinois using slope in a Digital Elevation Model, visited 38% of these cliffs, and assessed their suitability as nesting sites based on agreement with attributes reported in the literature for existing peregrine populations. Most (18 of 23) of the cliffs identified, possessed attributes consistent with good peregrine falcon nesting sites, suggesting that slope can be used as a simple habitat model. Using this model, southern Illinois should be capable of supporting approximately 5-16 territorial pairs on 69 km of cliffs, primarily along the Mississippi and Ohio Rivers and in the Shawnee National Forest. I identified 10 possible reintroduction sites that lacked great horned owls, a predator of young peregrines, with top sites in Monroe and Jackson Counties.

Using the slope model, I constructed a habitat map for southern Illinois and the surrounding region and linked it with a stage-structured population matrix to analyze peregrine population viability and reintroduction strategies. I derived habitat-specific demographic rates from peregrines in the central Mississippi River region during 1982-

2006. Mark-recapture analysis showed that juveniles fledged from cliffs had an annual survival rate of 20%, whereas juveniles from urban areas had an annual survival rate of 24%. Annual survival rates of subadults from cliff sites (84%) were similar to subadults from urban sites (85%) and to adults from both habitat types (85%). I also estimated average number of fledglings from cliff sites (1.8 ± 0.5) and urban sites (2.6 ± 0.1) during 2000-2005. Population viability analysis results indicated that the peregrine population in the study region is stable and slowly increasing. Cliff populations are stable, but not increasing. However, recolonization of cliff sites in southern Illinois will occur via dispersal from urban populations. A cost-benefit analysis indicates that the most cost-effective reintroduction strategy would cost approximately $280,000 and would result in only 2 additional breeding pairs from the no action scenario. Thus, funds would be more effectively used in other management efforts such as habitat preservation.

Job 1.1 Identification and survey of sites potentially suitable for peregrine falcons

a. Identify sites with habitat features required by peregrine falcons and verify their suitability with site visits.

A 10-m resolution digital elevation model (DEM) was used to identify cliffs and bluffs by querying for slopes ≥45 degrees. Based on the query, 69 km of cliffs, primarily located along the Mississippi and Ohio Rivers and in the Shawnee National Forest were identified. Cliffs in natural areas, nature preserves, state conservation areas, and parks were visited to assess site suitability. Eighteen of 23 sites visited were considered suitable for nesting based on cliff height, distance to water, elevation, and cliff dominance. Ten of these 18 sites were considered suitable for reintroduction based on the absence of great horned owls, a predator of young peregrine falcons. The most suitable reintroduction areas include 4 sites in Monroe County and the Little Grand Canyon in Jackson County.

Job 1.2 Population viability analysis

a. Determine viability of theoretical peregrine falcon population under different reintroduction scenarios, based on availability of suitable habitat in southern Illinois.

A spatially-explicit population viability analysis was constructed for southern Illinois and the surrounding region (416-km buffer) using program RAMAS/GIS. Multiple reintroduction scenarios were modeled with varying cohort sizes, supplementation schedules, and number of reintroduction sites. A base scenario with no reintroductions was also modeled. Under all scenarios, the peregrine falcon population did not go extinct in the region surrounding Illinois. Without reintroductions, cliffs in southern Illinois are expected to be recolonized in approximately 11 years and 2 breeding pairs are expected in the region after 50 years. The most cost-effective reintroduction strategy released 8 juveniles from each of 2 sites every 3 years during a 10 year period and cost approximately $300,000. Under this release strategy, cliffs in southern Illinois are expected to be recolonized in 3 years and should contain 4 breeding pairs in 50 years.

Job 1.3 Recommendations and report

a. Provide the IDNR with data necessary to evaluate the feasibility and likelihood of successful reintroductions of peregrine falcons to southern Illinois.

A final report including a Master's thesis was submitted to the IDNR discussing potential reintroduction sites, cost-effectiveness of release strategies, and viability of peregrine populations in southern Illinois. Two manuscripts have emerged from the thesis. Based on Jobs 1.1 and 1.2, southern Illinois and the surrounding region can support a healthy peregrine falcon population. However, at a cost of $300,000, reintroductions would increase the population in southern Illinois by only 2 breeding pairs after 50 years, compared to the no-action scenario. The current peregrine population in the lower Midwest appears to be healthy, and in time peregrines should naturally recolonize the cliffs in southern Illinois without human assistance. We recommend investing funds in other conservation efforts.

A HABITAT AND POPULATION VIABILITY ANALYSIS FOR POTENTIAL PEREGRINE FALCON REINTRODUCTIONS IN SOUTHERN ILLINOIS

by

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B.S., University of Maryland – College Park, 2004

A Thesis
Submitted in Partial Fulfillment of the Requirements for the
Masters of Science Degree

Department of Zoology
in the Graduate School
Southern Illinois University Carbondale
December 2008

THESIS APPROVAL

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in the field of Zoology

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December, 2008

AN ABSTRACT OF THE THESIS OF

SARAH M. WAKAMIYA, for the Master of Science degree in ZOOLOGY, presented on JULY 28, 2008, at Southern Illinois University Carbondale.

TITLE: A HABITAT AND POPULATION VIABILITY ANALYSIS FOR POTENTIAL PEREGRINE FALCON REINTRODUCTIONS IN SOUTHERN ILLINOIS

MAJOR PROFESSOR: Dr. Charlotte Roy

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Using the slope model, I constructed a habitat map for southern Illinois and the surrounding region and linked it with a stage-structured population matrix to analyze peregrine population viability and reintroduction strategies. I derived habitat-specific demographic rates from peregrines in the central Mississippi River region during 1982-2006. Mark-recapture analysis showed that juveniles fledged from cliffs had an annual survival rate of 20%, whereas juveniles from urban areas had an annual survival rate of 24%. Annual survival rates of subadults from cliff sites (84%) were similar to subadults from urban sites (85%) and to adults from both habitat types (85%). I also estimated average number of fledglings from cliff sites (1.8 ± 0.5) and urban sites (2.6 ± 0.1) during 2000-2005. Population viability analysis results indicated that the peregrine population in the study region is stable and slowly increasing. Cliff populations are stable, but not increasing. However, recolonization of cliff sites in southern Illinois will occur via dispersal from urban populations. A cost-benefit analysis indicates that the most cost-effective reintroduction strategy would cost approximately $280,000 and would result in only 2 additional breeding pairs from the no action scenario. Thus, funds would be more effectively used in other management efforts such as habitat preservation.

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CHAPTER 1

IDENTIFICATION OF PEREGRINE FALCON NESTING HABITAT AND POTENTIAL REINTRODUCTION SITES IN SOUTHERN ILLINOIS

INTRODUCTION

The North American peregrine falcon (*Falco peregrinus anatum*) historically ranged from the subarctic boreal forests of Alaska and Canada south to Mexico before widespread population declines resulted from the extensive use of DDT (Dichloro-diphenyl-trichloroethane) during 1950-1970 (USFWS 1999). In Illinois, peregrines historically occupied the bluffs of the Mississippi River (Bohlen, 1978). Prior to their extirpation from the state in 1957, nesting peregrine falcons were documented in Jackson, Jersey, and Union counties (Ridgeway 1889, Widmann 1907, Bohlen 1978, Enderson et al. 1995). Southern Illinois contains many cliffs and bluffs adjacent to open agricultural areas, rivers and lakes, yet these cliffs remain unoccupied >20 years after successful recovery efforts elsewhere in their former range.

Peregrines were removed from the federal list of endangered and threatened wildlife in 1999, yet in Illinois they remain state-threatened (USFWS 1999) and among the "species of greatest need of conservation concern" (IDNR 2005). Illinois began reintroducing peregrine falcons in 1986, with all efforts focused on buildings in the Chicago metropolitan area. Currently, 11 pairs hold territories in Illinois and all nest on buildings and bridges in Chicago (Redig et al. 2007). High site fidelity may partially explain why they have not returned to historic cliff sites in Illinois (Newton 1988, Tordoff et al. 1998). Therefore, reintroductions to cliffs may be necessary to promote the recolonization of these sites.

The return of peregrine falcons to cliff sites in southern Illinois would have many benefits. First, these efforts would initiate a population in a more natural setting, which has both ecological and aesthetic value, and would likely enhance viability of the current population. Maintaining solely urban populations is problematic because of the risks associated with urban environments (e.g., airplane collisions, building collisions and electrocutions), and because of the possibility of shifting public attitudes about their presence on buildings (Garrott et al. 1993, Septon et al. 1995, Sweeney et al. 1997). A stable cliff-nesting population in southern Illinois may link regional populations of urban falcons (i.e., Chicago, IL, St. Louis, MO, Indianapolis, IN) and reduce the risk of local and metapopulation extinction. Finally, restoring peregrine falcons to their historic range in the Midwest may become increasingly important as the U.S. Fish and Wildlife Service may soon implement the harvest of wild peregrines for falconry in the western United States and throughout Canada (USFWS 2006).

Reintroduction programs can facilitate recovery, but they are costly and time consuming. Therefore, the feasibility of reintroduction should first be evaluated by assessing habitat availability, identifying suitable reintroduction sites, and modeling populations with varying reintroduction strategies (IUCN 1998). A first step in this process is to identify important habitat variables and possible limiting factors.

Habitat requirements and potential limiting factors

Historical records of nesting peregrine falcons in southern Illinois are sparse as systematic surveys were rare prior to the species' decline (Enderson et al. 1995). However, published literature on restored populations identifies important habitat variables that may influence nest site selection. Guidance on other variables that might limit peregrine falcon populations, such as predator abundance, the availability of prey and competition with conspecifics also may be gleaned from the literature.

Prey availability is not expected to affect the distribution or number of peregrines (Hickey 1942). Peregrine falcons are well known for their catholic diet and typically prey upon common, medium-sized birds such as European starlings (*Sturnus vulgaris*), morning doves (*Zenaida macroura*), rock doves (*Columbia livia*) and Eastern Meadowlarks (*Sturnella magna*; Sherrod 1978, White et al. 2002, Carter 2003). Furthermore, peregrines will fly an average of 13 km from their nest to forage in croplands, pastures, waterways, swamps, or marshes (Enderson and Craig 1997). During the nonbreeding season, cliff-nesting peregrines in the northern portion of the continent typically migrate south. However, at latitudes similar to southern Illinois, where prey bases are more constant, migration patterns are unclear (Enderson 1965, Hickey and Anderson 1969, White et al. 2002).

Predators are unlikely to limit populations of peregrine falcons. The great horned owl (*Bubo virginianus*) is the most formidable predator of peregrine falcons in the Midwest, although the threat to adult birds is ambiguous due to reported instances of cohabitation and other reports of competitive exclusion (Hickey 1942, Herbert and Herbert 1965, White et al. 2002, Craig and Enderson 2004). However, the predation of hacked young and fledging birds by great horned owls was the primary reason for reintroduction failures in the Midwest and East (Barclay and Cade 1983, Redig and Tordoff 1988). Red foxes (*Vulpes vulpes*), gray foxes (*Urocyon cinereoargenteus*), raccoons (*Procyon lotor*) and rat snakes (*Elaphe spp.*) are also opportunistic predators of peregrine nestlings, but these predators are typically unable to negotiate their sheer nesting sites (Herbert and Herbert 1965).

Peregrines are highly territorial at nesting sites so competition with conspecifics may limit their population size. Peregrines also typically interact with other predatory birds during nest defense, but do not directly interact in competition for food or nesting sites (Hickey 1942, Bond 1946). Densities vary widely depending on the region, but historical data along the upper Mississippi River indicate approximately one pair every 30 km (Hickey and Anderson 1969, Berger and Mueller 1969). Therefore, suitable sites may remain unoccupied if a pair is already nesting nearby.

Peregrines are primarily restricted by the distribution of cliffs, which are used for nesting and provide protection to young from predators (Hickey 1942, Ratcliffe 1993). According to Hickey (1942), first-class peregrine cliffs are "usually overlooking water and generally dominating the surrounding countryside." Such cliffs are desirable because they provide a large riparian prey base, an expansive view of passing prey and a height advantage for the falcon's stoop (Newton 1988). Cliff orientation varies widely and appears to be more important in harsh climates than in the temperate Midwest (White et al. 2002, Christopher 1980). In the Illinois landscape, cliffs are fairly unique geographic features that are limited in number and distribution. Therefore, the availability of cliffs appears to be the primary factor in determining peregrine falcon nesting habitat in the Midwest.

This study constitutes an initial step in the development of a recovery plan for peregrine falcons in southern Illinois. I determined the amount of suitable nesting habitat and identified possible reintroduction sites. This information can then be used to decide if reintroductions are desirable in the region. Macdonald et al. (2000) and Dzialak et al. (2005) suggested using a combination of Geographic Information Systems (GIS) and field analysis to reduce the costs of assessing landscapes for wide-ranging species. I identified cliff sites using GIS, conducted on-site field assessments, and used literature-based expert knowledge and a multicriteria decision-making process to rank suitable reintroduction sites.

METHODS

Study area and cliff identification

The study area encompassed 35 counties in southern Illinois, bounded by the Mississippi River to the west and the Ohio River to the east (Figure 1). Raster-based Digital Elevation Models (DEM) of 10 m resolution and land cover data of 30 m resolution were downloaded from the National Elevation Dataset (NED) and National Land Cover Dataset (NLCD) provided by the Seamless Data Distribution System (USGS 2001, 2004). Cliffs were identified by querying for slopes ≥45° in the DEM using ArcGIS 9.2 (ESRI 2006).

Potential reintroduction sites

Reintroduction sites were selected from cliff sites based on site protection, microhabitat attributes, and presence of great horned owls. I first identified cliffs that were located in nature preserves, natural areas, state conservation areas, or parks. These areas were deemed "protected" because they were unlikely to become developed, although some were still being considered for protection. I defined 23 sites based on property boundaries, of which 12 were on protected public land, 7 on private land, and 4 on protected private land (e.g., Nature Conservancy).

I visited each cliff to measure site attributes. Peregrines are not currently nesting on cliffs in Illinois, so I compared measured attributes to values reported in the literature (Table 1) and historical nesting accounts to determine cliff suitability. Comparisons were made at the landscape and site levels. At the landscape level, I used GIS to determine elevation and the distance to water for each cliff. At the site level, I measured cliff height, slope, and horizontal extent using a rangefinder, clinometer and global positioning system unit during Jun.-Aug. 2007. I also calculated cliff dominance and the proportion of agriculture in a 13 km radius around each site in GIS. I defined cliff dominance as the average change in elevation from the top of the cliff to ground level at 1, 2, and 3 km from the cliff edge and made these measurements using the vertical terrain profile in GIS (Gainzarain et al. 2000). Cliffs that exceeded the limits of what peregrines have been documented using were deemed unsuitable for reintroduction.

I assessed great horned owl presence using broadcast surveys at each site not excluded by cliff measurements ($n = 18$). Twenty-four surveys at 5 sites were conducted in 2007 and 76 surveys at 17 sites in 2008. Four sites were surveyed in both years. All surveys were conducted between sunset and 2200 h during Jan.–mid-Mar. Recorded owl vocalizations (provided by the Cornell Lab of Ornithology) were broadcasted using a Sony CFS-B11 Cassette Recorder in 2007 and a FoxPro ZR2 wildlife caller in 2008. To maximize detection, surveys were conducted on clear nights with wind speeds <20 km/h and were replicated $\geq$2 times (McGarigal and Fraser 1984, Morrell et al. 1991, Takats et al. 2001). Broadcasts were at the approximate center of the site unless the cliff was >1.6

km long, in which case multiple surveys were conducted along the cliff at intervals of 1.6 km (Takats et al. 2001). Each survey consisted of a 2-min silent pre-broadcast listening period, a 7-min broadcast period and a 5-min silent post-broadcast listening period (McGarigal and Fraser 1984, Morrell et al. 1991). The number, distance, direction and behavior of owls were recorded. I used program PRESENCE 2.2 (Hines 2006) to estimate great horned owl occupancy and detection probability. Sites where great horned owls were present were deemed unsuitable for reintroduction.

Site Ranking

I used literature-based expert information and the analytical hierarchy process (AHP), a multi-criteria decision making process, to objectively rank reintroduction sites (Saaty 1980, Kovacs et al. 2004). The AHP uses available literature (Table 2) on site selection by peregrine falcons to assist in weighting variables (Clevenger et al. 2002). These weights are then applied to site attributes, providing a ranking of most to least suitable reintroduction sites. Empirical studies were primarily conducted on other continents (Gainzarain et al. 2000, Sergio et al. 2004, Wightman and Fuller 2005, Brambilla et al. 2006). Therefore, nonempirical studies from areas closer to the study region were also included (Hickey 1942, Bond 1946, Porter et al. 1973, Grebence and White 1989, Craig and Enderson 2004). Variables in the AHP analysis included cliff height, slope, dominance, length, proportion of agriculture, distance to water, and human disturbance.

RESULTS

Cliff identification and suitability

A total of 69 km of cliff line exist in southern Illinois, primarily along the Mississippi and Ohio Rivers and in the Shawnee National Forest (Figure 1). Of the 32 counties included in the study, >70% of cliffs were in Monroe (30.2%), Jersey (13.8%), Jackson (11.9%), Pope (9.2%) and Union (9.0%) counties. At the landscape level, all cliffs were within the range of values reported in the literature for proximity to water and very few (1.0 %) exceeded values reported for elevation (Table 1). Of the cliffs identified in GIS, 38% were on "protected" land and visited to assess suitability as a reintroduction site. Monroe County contained the tallest ($\bar{x} \pm$ SE, 74.5 ± 8.8 m, Appendix A) and most dominant cliffs (100.2 ± 6.1 m). Cliffs in Jersey and Madison counties were the longest (2500 ± 1450 m) and closest to water (40 ± 10 m). At the site level, all cliffs possessed characteristics consistent with those reported in the literature for height, but 6 of the 23 sites did not meet minimum values reported for dominance (i.e., 50 m). However, dominance at Tower Rock was close (48.7 m), and was therefore considered habitat. Thus, 78% of the sites visited were deemed suitable for nesting.

Of the 18 suitable nesting cliff sites, great horned owls were detected at 8, including La Rue Pine Hills, Fountain Bluff, Principia, Potato Hill, Oblate/Nature Conservancy, Demint Hill, Chautauqua and Miles Prairie. Great horned owl presence was the same at the 4 sites surveyed in both years. Pooling across years, detection probability and probability of occupancy corrected for detection were estimated to be 0.46 ± 0.12 and 0.44 ± 0.12, respectively.

Site Rank

A total of 10 cliffs were suitable for reintroduction based on site characteristics and owl absence. Literature-based expert opinion indicated that cliff dominance was the most important factor in peregrine falcon habitat selection (Table 3). Cliff dominance was 1.6 times more important than cliff height and 2.5 times more important than slope. Distance to water, disturbance, and percent agriculture each had similarly low ranked scores. Based on these weights, Fults emerged as the most suitable reintroduction site, followed by Monroe City, Saltpeter Cave Area, and Renault Herpetological Area (Table 4). These top 4 sites were located in Monroe County, whereas the remaining 6 sites were in Pope, Johnson, Hardin, and Jackson Counties.

DISCUSSION

This study identifies peregrine falcon nesting habitat in southern Illinois and ranks possible reintroduction sites. My approach is objective and uses the best available information to guide planning for a bird that declined before its habitat use in the region was well documented, yet for which habitat degradation was not the impetus for its decline. Few to no peregrines exist in their natural habitat in the Midwest, so I could not use nesting locations to create or test a habitat model. However, querying for slopes $\geq$ 45° identified suitable nesting sites based on consistency with descriptions in the literature for other regions and site visits. Of the cliffs I visited, 78% possessed characteristics necessary for nesting. Only 1% of all cliffs identified were considered unsuitable on the basis of distance to water and elevation. Thus, slope ($\geq$45°) can act as a simple habitat model for peregrines. Other predictive avian habitat models report accuracies of 60–99% (Gottschalk et al. 2005).

All sites deemed unsuitable during site visits had low dominance values. Although dominance can be measured on a per site basis using vertical terrain profiles, calculating dominance over large landscapes is very computationally intensive. Nevertheless, the majority of cliff attributes (e.g., slope, elevation, proximity to water) could be examined within GIS, and therefore, many cliffs could be eliminated as habitat without a site visit. Small cliffs that would probably be unsuitable for nesting were also excluded in GIS before site visits because the highest resolution DEM currently available is 10 m (Holmes et al. 2000).

My conclusion that slope (≥45°) can be used as a simple habitat model was based on site visits that were restricted to areas that were protected or being considered for protection. If protected sites were more likely to possess cliffs suitable for nesting, then the accuracy of slope as a predictor of nesting habitat may have been positively biased. For example, if cliffs at protected sites were taller or more dominant than unprotected sites, then slope may be less accurate at predicting good habitat than was indicated by site visits. However, my findings are consistent with a peregrine falcon habitat model developed for Maine that used slope and aspect to identify peregrine habitat and had 91% accuracy when compared to actual nesting locations (Banner and Schaller 2001). Because aspect has little influence on nesting sites in the temperate Midwest (White et al. 2002), slope would be expected to perform reasonably well as a habitat model in the Midwest, and it did.

Furthermore, limited historical records and nesting habitat identified in this study were in agreement, further promoting confidence in the use of slope as a simple habitat model for peregrines in the Midwest. Most of the nesting habitat I identified was in Jackson, Jersey and Union counties, which historically held nesting peregrines (Ridgeway 1889, Widmann 1907, Bohlen 1978). Although there is no historical documentation of peregrines nesting in Monroe County, this data gap is most likely due to a lack of intensive surveying and documentation prior to their extirpation (Enderson et al. 1995).

MANAGEMENT IMPLICATIONS

Peregrine falcons historically nested 30 km apart in the Upper Midwest (Berger and Mueller, 1969). Assuming this spatial distribution holds true in southern Illinois, approximately 5 nesting pairs can be supported, with Jersey, Madison, Jackson/Union, Johnson, and Pope Counties expected to hold one pair each. However, other studies have documented much higher densities of nesting peregrines. For example, cliff eyries were separated by an average of 11.3 km in New York (Hickey 1942). Using this spatial distribution, southern Illinois can support approximately 16 pairs. As Illinois currently holds 11 nesting peregrine falcon pairs, a successful reintroduction program may increase this population by 45–145%.

Eighteen sites on protected land possessed suitable nesting habitat. However, 8 of these sites were occupied by great horned owls, making them less than ideal for reintroduction. Detection probability of great horned owls in this study was <0.5, so I likely missed owls at some sites. Reintroductions at sites occupied by great horned owls can be successful (Powell et al. 2002); however, I do not recommend reintroducing peregrines to sites where owls are known to be present, if other suitable sites without owls are available. If a reintroduction effort is pursued by the state of Illinois, more focused owl surveys should be conducted at selected reintroduction sites. Great horned owls typically show high site fidelity unless prey densities are low (Rohner 1996). Therefore, owl occupancy at sites is not likely to change dramatically across years and surveys do not need to be conducted continuously.

Ten sites possessed suitable nesting habitat and lacked owl detections. The 4 highest-ranked sites (i.e., Fults, Saltpeter Cave area, Renault Herpetological area, Monroe City) were located in Monroe County bordering the Mississippi River floodplain. These cliffs were tall ($\geq$ 98 m), long and generally dominated over the landscape. Most of the floodplain in these areas has been converted to agriculture, but may still be beneficial to peregrines. Agricultural fields are important for many peregrine prey species (e.g., European starlings, mourning doves, meadowlarks) and unlike forested areas, do not provide escape cover to prey (Sherrod 1978, Carter et al. 2003, Dzialak 2005).

The number of hack sites and their spatial distribution should be considered to maximize the efficiency and minimize the costs of reintroduction attempts. Reintroducing peregrines in Monroe County may facilitate the transition from a solely urban population to one with both natural and urban nesting locations. St. Louis, Missouri has supported an urban peregrine population for over 10 years and is <50 km away. This proximity is advantageous because releases should be <170 km from other release or nest sites to increase chances for interaction and mating (Barclay and Cade 1983, Septon et al. 1995). Multiple hack sites in close proximity are also advantageous because personnel costs are reduced, as is the risk of young birds flying towards unfamiliar and potentially hazardous areas due to a lack of alternate landing sites (Redig and Tordoff 1988). In Monroe County, Fults, Saltpeter Cave area, and Renault Herpetological area form a near continuous line of cliffs. Sherrod et al. (1982) recommended at least two hack stations with 3-8 young hacked per year per station to ensure the return of breeding adults. Assuming peregrine territorial behavior will result in nest sites spaced 30 km apart (Berger and Mueller 1969), Monroe County can be expected to support 1-2 pairs.

Reintroduction efforts at a second location buffer against localized risks (e.g., disease or predators), provide linkage between sites and expedite the process of recolonization by exposing peregrines to more potential nesting sites. The next highest-ranked reintroduction sites had comparable weights and thus should be of very similar quality. Of these, Little Grand Canyon in Jackson County is the best candidate for an additional reintroduction area because it is managed by the Shawnee National Forest and is <80 km from the Monroe County sites (Figure 1). This site is more forested, but overlooks the Big Muddy River, which should provide an ample prey base of riparian birds and some open area for foraging. Additionally, the Little Grand Canyon area is <15 km from 2 potential nesting areas (e.g., Fountain Bluff and La Rue Pine Hills) that may have historically held a nesting pair. I recommend 2 hack stations each at the Monroe County area and Little Grand Canyon area, should the state decide to pursue a reintroduction program.

Peregrine falcons are naturally restricted to areas containing bluffs and cliffs for breeding. Identification of these breeding sites and core areas for reintroduction provides information necessary for the effective planning of peregrine falcon recovery in southern Illinois. By linking this landscape information with a population viability analyses, one can determine the most cost-effective and viable release strategies for southern Illinois, determine if the region can sustain a viable population, and determine if reintroductions will expedite the process of recolonization. Reintroduction programs are expensive; reintroductions in the Midwest during the 1980's cost $80,000 to hack 25 birds in one year (Sherrod et al. 1982, Redig and Tordoff 1988). Therefore, such programs should be carefully planned before reintroduction efforts and carefully executed to ensure success. This study illustrates the process involved in an initial query about the availability of habitat for peregrine falcons in southern Illinois, and some of the challenges that are presented by limited historical records and habitat information for a wide-ranging animal.

CHAPTER 2

A SPATIALLY-EXPLICIT POPULATION VIABILITY ANALYSIS FOR PEREGRINE FALCONS IN SOUTHERN ILLINOIS

INTRODUCTION

Over the past few decades, the number of reintroductions and translocations of threatened and endangered species has markedly grown (Seddon et al. 2007). Success rates of such programs, however, have been low and few have used an adaptive management strategy where results of post-release monitoring are used to improve future reintroduction success (Sarrazin and Barbault 1996, Fischer and Lindenmayer 2000). Monitoring programs following reintroductions provide years and sometimes decades of data on individuals and populations. Such data can be used to gather basic life history traits of rare species, to understand habitat selection, or to study colonization and founder effects that otherwise would have been unfeasible (Sarrazin and Barbault 1996). Most importantly, field data from monitoring programs can provide demographic information for population viability analyses (PVA) of future reintroduction programs.

With the advent of landscape analysis tools such as geographic information systems (GIS), spatially-explicit forms of population viability analyses are increasingly being used for reintroduction planning (South et al. 2000, Seddon et al. 2007). These spatially-explicit population models (SEPMs), which incorporate spatial structure, can be more accurate for species that are divided into subpopulations, such as reintroduced populations, and for species that have high dispersal rates (Southgate and Possingham 1995, Akçakaya 2000, South et al. 2000). Furthermore, SEPMs are often used for modeling reintroductions, translocations, or other management scenarios because of their ability to capture subpopulations (Southgate and Possingham 1995, Akçakaya 2000, South et al. 2000). Despite their usefulness, SEPMs are often criticized because they are extremely data hungry, requiring detailed landscape and dispersal information along with habitat-specific demographic rates, and because they usually are not validated (Dunning et al. 1995, Beissinger and Westphal 1998).

The American peregrine falcon (*Falco peregrinus anatum*) is an ideal candidate for using monitoring data to derive habitat-specific demographic rates for spatially-explicit models. Studies conducted in California found significant differences in peregrine falcon demographic and dispersal rates among coastal, inland, and urban habitat types (Wootton and Bell 1992, Kauffman et al. 2003, Kauffman et al. 2004). Applying these habitat-specific rates to population viability analyses provided considerable insight into improving species management. Unlike its western counterpart, the Midwestern peregrine population is almost entirely composed of urban-nesting pairs. Initially, midwestern reintroduction programs attempted to release peregrines from cliff sites but were terminated because of heavy loses due to predation (Redig and Tordoff 1988). Reintroductions resumed several years later, but were primarily concentrated in urban areas where tall buildings functioned as cliff sites.

Although peregrine falcon reintroduction programs were hailed as a success throughout the nation, the majority of the population in the Midwest still resides on man-made structures rather than their natural cliffs. In Illinois, peregrines historically nested on bluffs adjacent to the Mississippi River in the south (Ridgeway 1889, Widmann 1907, Bohlen 1978). Currently, 11 pairs hold territories in the state, but all are nesting on buildings and bridges in Chicago (Redig et al. 2007). High site fidelity may in part explain why they have not returned to their historic cliff sites (Newton and Mearns 1988, Tordoff et al. 1998). Therefore, reintroductions to cliff sites may be necessary to promote the recolonization of these natural sites. A stable cliff-nesting population in southern Illinois may also link regional populations of urban falcons (*i.e.*, Chicago, IL, St. Louis, MO, Indianapolis, IN) promoting local and metapopulation stability.

Reintroduction programs, though desirable are expensive, costing $80,000 to hack 25 falcons in the early Midwestern efforts (Redig and Tordoff 1988). Therefore, the feasibility and cost-effectiveness of reintroduction should first be evaluated by modeling and evaluating different reintroduction strategies (IUCN 1998). In this study, I derive habitat-specific survival and fecundity rates using monitoring data from the Midwestern peregrine falcon populations, use these rates to develop a spatially-explicit population viability analysis for a cost-benefit analysis of reintroduction scenarios, and use an independent dataset to verify model accuracy.

METHODS

Analysis of Monitoring Data

The Midwest Peregrine Society web database (midwestperegrine.org) has monitoring data for the entire Midwestern U.S., as well as Ontario and Quebec, Canada. Only peregrines from the central Midwestern region (<250 km from the Mississippi River), were used to estimate demographic rates, reserving birds from the upper Midwestern region for model verification.

Survival.—The Midwest Peregrine Society dataset provided banding, resighting, and recovery data for 924 falcons during 1982–2006 from which I could interpret individual life histories. Of these 924 falcons, 148 were from cliffs and 776 were from man-made structures (e.g. buildings, bridges, and smokestacks). Survival rates were estimated using the Barker model in program MARK (White and Burnham 1999). The Barker model allows the use of multiple data sources, including recaptures, live resightings between marking occasions, and dead recoveries, to produce more accurate parameter estimates (Barker and White 2001). The parameters in this model as defined by Barker and White (2001) are:

S_i = the probability an animal alive at time i is alive at time i+1;

p_i = the probability an animal alive and at risk of capture at time i is captured (i.e., banded or resighted);

r_i = the probability a marked animal that dies between i and i+1 is found and reported;

R_i = the probability that a marked animal alive at i+1 is resighted alive between i and i+1;

R'_i = the probability that a marked animal that dies between i and i+1 is resighted alive in this interval before it died;

F_i = the probability that an animal at risk of capture at time i is again at risk of capture at time i+1;

F'_i = the probability that an animal not at risk of capture at time i is at risk of capture at time i+1.

For this study, "recaptures" were actually captures or resightings occurring from 1 May – 1 Aug, as banding and observational effort is most intensive during this time period.

I used Akaike's Information Criterion adjusted for effective sample sizes and lack of fit in Program MARK to rank a set of candidate models. I expected survival to be affected by age class (i.e., juvenile, subadults, and adults) because juvenile mortality is likely to be high during dispersal from natal sites, subadult mortality is likely to be lower as they become familiar with their surroundings, but need to fight for territories, and adult mortality following territory establishment is the lowest (Ratcliffe 1993, Kauffman et al. 2003). I was also interested in determining if natal location (urban vs. cliff) or an interaction of age and natal location affected survival. Previous studies indicate juveniles raised in urban areas may have a higher survival rate because of the large, easily accessible prey base (e.g., rock doves [*Columba livia*]) and the lack of predators (Septon et al. 1995, Sweeney et al. 1997, Tordoff and Redig 1997, Kauffman et al. 2003). Therefore, I tested models of survival with age effects, natal site effects, additive age and natal site effects, and interactive age and natal site effects.

Recapture (*p*) and recovery (r) rates are likely influenced by age and natal sites because breeding adults in urban areas are more likely to be seen or reported. I did not expect any interaction between age and natal sites in recapture or recovery rates and therefore did not test for such effects. Resightings (R) take place during the non-breeding season and were therefore probably not influenced by age. Natal sites may have an effect on resighting because falcons are more conspicuous in urban areas. Resighting' (*R'*) was held constant because there were too few of these events to model covariates. Emigration (*F*) and immigration (*F'*) were unconstrained to simulate Markov emigration (risk of capture at time *i* depends on whether an animal was at risk of capture at time *i*-1; Barker and White 2001) because peregrines show high site fidelity to breeding sites. Because these recaptures were essentially resightings at breeding sites, individuals are more likely to be at the breeding site (e.g., at risk of capture) if they were present the previous year.

I used a reduced-parameter approach to sequentially find the most parsimonious model starting with resighting probability, then recapture, then recovery, and finally survival (Barker and White 2001, Brown et al. 2006). Rather than running all possible combinations of variables per parameter, this approach first accounted for nuisance parameters and then determined the best fit model. I examined 15 candidate models of survival using this approach (Table 5). The median ĉ approach, the only available goodness-of-fit test for Barker models, was used to test for overdispersion (Cooch and White 2008). Models within 4 AIC points were averaged for parameter estimation to account for model uncertainty (Burnham and Anderson 2002).

Productivity.— Average number of fledglings for cliff-nesting ($n = 11$) and urban-nesting birds ($n = 54$) were estimated between 2000-2006 because peregrines were not nesting on cliffs in the region prior to this time period. I used a mixed-model ANOVA to estimate temporal variance for urban-nesting peregrines, where within-year variance estimates sampling variance and between-year variance estimates environmental variance (White 2000). All statistical procedures were carried out in SAS® version 9.1.3 with $\alpha = 0.05$ (SAS Institute 2004). Homogeneity of variance and normality were tested by assessing residual and normal quantile plots.

Spatially-explicit population viability analysis

I used the software program RAMAS®/GIS software, version 5.0 (Akçakaya 2005) to develop a stage-structured, spatially-explicit population model of peregrine falcon populations in southern Illinois and the surrounding region. This model was developed based on a habitat map I created using slope as an indicator of suitable nesting sites (see Chapter 1) and on the demographic data derived from 25 years of monitoring data in the midwestern U.S.

Habitat map and patch structure.—The peregrine falcon is a habitat specialist, relying primarily on cliffs for nesting (Hickey 1942, Ratcliffe 1993). A cartographic map identifying nesting habitat was created by querying for slopes ≥45 degrees. Digital Elevation Models (DEMs) of 10 m resolution were downloaded from the U.S. Geological Survey National Map Seamless Server (www.seamless.usgs.gov) for a 950,581 km^2 area. This study region encompassed 35 counties in southern Illinois and a 416 km radius buffer from their edge (Figure 2). This radius was chosen because it is the farthest distance a peregrine of the more philopatric sex (male) dispersed in my dataset. Thus, some females may disperse outside the study area, but the majority would be expected to remain within the study boundary. The final map was converted to a 300 x 300 m grid and imported into RAMAS/GIS. The Spatial Program in RAMAS/GIS delineates landscape patchiness based on habitat suitability and a neighborhood distance where suitable cells separated by a distance less than the neighborhood distance are considered the same patch (Akçakaya 2005). I used a neighborhood distance of 30 km, the distance within which 80% of juveniles disperse (see *Dispersal* below). This distance also allowed southern Illinois patches to be distinct. However, some cliffs in Missouri and Kentucky were also incorporated in the "southern Illinois patches" thus inflating calculations of carrying capacity. RAMAS/GIS does not support individual-based modeling, so patches that could only support one breeding pair were grouped with the nearest adjacent patch producing a total of 9 cliff patches.

Most peregrines nest on buildings in the Midwest, so I also incorporated 24 urban centers in the study region that were occupied by ≥ 1 nesting pair in 2005 (Table 6). A point shapefile of these cities or towns was obtained through ESRI (2006) and converted to a 300 m resolution grid for import. Like cliff habitat, urban centers within a neighborhood distance of 30 km were grouped to form one patch. Urban centers that supported only one nesting pair in 2005 were grouped with the nearest adjacent urban patch producing a total of 4 urban patches. Because Kansas City, MO and Chattanooga, TN each contained only one pair and were >250 km from the next nearest occupied urban center, I assumed they had little influence on metapopulation dynamics and removed them from the analysis.

Stage structure.—The model included 3 stage classes: juveniles (0-1 yrs), subadults (1-2 yrs), and adults (2+ yrs). Only males were modeled because they are the philopatric sex and because RAMAS/Metapop allows the user to define only one dispersal-distance function. Although most species are modeled using females, many bird populations are modeled with males because they establish territories, and therefore, limit population growth (McCarthy et al. 2004, Shriver and Gibbs 2004, Alldredge et al. 2004).

Separate matrices were created for cliff-nesting and urban-nesting populations using the parameter estimates derived from the monitoring data (Table 7). Peregrines typically begin breeding the second year they return to a breeding site (Mearns and Newton 1984, Tordoff and Redig 1997). Therefore, both subadults (fecundity when almost, but not quite two-year olds) and adults were assumed to breed (Akçakaya 2005). Fecundity rates were calculated as the product of age-specific survival, average number of fledglings (f), and sex ratio (Akçakaya 2005). Assuming a 1:1 sex ratio (Restani and Mattox 2000), and a post breeding census (Mearns and Newton 1984, Tordoff and Redig 1997), I obtained the following stage matrix:

$$\begin{bmatrix} 0 & 0.5S_sf & 0.5S_af \\ S_j & 0 & 0 \\ 0 & S_s & S_a \end{bmatrix}$$

Initial abundance.— In the study region, peregrines are currently nesting only on man-made structures. I used the most recent available census (i.e., 2005) of breeding pairs in the study region to estimate initial abundance. Based on the number of breeding pairs, I estimated the number of fledglings and subadults at a stable age distribution for each urban patch.

Stochasticity.—Environmental stochasticity was incorporated into the model using the unconditional variance estimate for each model-averaged survival derived in Program MARK. This unconditional estimate removes variance due to sampling error and model uncertainty, leaving only process variance (Cooch and White 2008). Process variance in the average number of fledglings for urban-nesting peregrines was estimated using a one-way ANOVA (see *Productivity* above). Because sample sizes were too small to estimate process variance for cliff-nesting peregrines, I used a 50% increase in the standard deviation of average fledglings from urban-nesting birds. A 50% increase in standard deviation seemed reasonable because the standard deviations of survival rates differed 50% between urban and cliff-nesting populations. The variance of the product of two independent variables, *i* and *j*, is:

$$\mathrm{Var}_{ij} = \mathrm{var}_i(\mathrm{mean}_j)^2 + \mathrm{var}_j(\mathrm{mean}_i)^2 + \mathrm{var}_i\mathrm{var}_j$$

(Goodman 1960), which was used to estimate total process variance in fecundity. Demographic stochasticity was modeled by drawing the number of survivors and dispersers for each class from a binomial distribution and the number of young from a Poisson distribution (Akçakaya 2005).

Density dependence.—Peregrine falcon populations are believed to be regulated by the number of breeding sites in the region because of their highly territorial nature (Hickey 1942, Hunt 1988, Ratcliffe 1993, Kauffman et al. 2004). Therefore, I modeled density dependence using contest competition, or the Beverton-Holt function. Contest competition occurs in highly territorial populations where the number of territories does not change, but the number of individuals seeking territories may change a lot (Akçakaya 2005). This type of density dependence has a stabilizing effect because breeding individuals that die are quickly replaced. Peregrine falcons are known for their remarkably stable populations in spite of fluctuating breeding success, indicating contest competition is likely (Newton and Mearns 1988). Density dependence was assumed to affect both fecundity and survival because of limitations on breeding territories and intense territorial battles leading to lower survival after carrying capacity has been reached (Herbert and Herbert 1965, Hunt 1988, Ratcliffe 1993, Tordoff and Redig 1999, Kauffman et al. 2004).

In RAMAS, contest competition is implemented by altering rates in the stage matrix so that the growth rate at time t (R_t) is given by the following equation:

$$\frac{R_{\max} \times K}{(R_{\max} \times N_t) - N_t + K},$$

where R_{max} is the maximal growth rate in the absence of density dependence, N_t is adult abundance at time t, and K is the carrying capacity of territorial adults. Because demographic rates were estimated on peregrine falcon populations that have not yet reached carrying capacity, I assumed these populations should be growing at their maximum growth rate. Thus, the maximum growth rate for cliff (Rmax = 1.002) and urban birds (Rmax = 1.094) were based on eigenanalysis of their respective stage matrices.

Carrying capacity

Densities for peregrines are typically expressed based on length rather than area because of the linear nature of their breeding habitat. I used ArcGIS to estimate total patch length because RAMAS/GIS did not have this capability. As stated previously, RAMAS grouped several cliffs in Missouri and Kentucky with the southern Illinois patches so total carrying capacity for southern Illinois was slightly inflated. Peregrine densities vary widely depending on region. In New York, peregrines were estimated nesting approximately 11 km apart (Hickey and Anderson 1969), whereas Midwestern peregrines were estimated to nest every 30 km historically (Berger and Mueller 1969). Therefore I simulated a low, medium, and high case scenario of carrying capacity using densities of 1 territorial male every 10, 20, and 30 km. I assumed all adult males became territorial until carrying capacity was reached.

I assumed each urban patch was approaching carrying capacity and could be occupied by only 1 additional territorial male. This is a reasonable assumption because Midwestern urban centers have been occupied for >25 years, so occupancy of suitable man-made nesting structures are likely nearing the capacity dictated by peregrine territorial spacing behavior.

Dispersal.—Dispersal was modeled using a distance-dependent dispersal function. Only juvenile dispersal was modeled because adults typically show extreme site fidelity once territories are established (Mearns and Newton 1984). Natal and breeding site locations for falcons from the lower midwestern region, as described in the Midwestern peregrine falcon database, were converted to UTM coordinates using Topozone (www.topozone.com). Using the Animal Movements tool in Hawths Analysis tools for ArcGIS, I determined dispersal distances for 92 male peregrines that remained within habitat types (i.e., cliff-cliff or urban-urban movement). The number of dispersing juvenile males in each distance class was divided by the total sample size to determine the proportion dispersing in each class (Akçakaya and Atwood 1997). The proportion was plotted against the midpoint of each distance class and fitted to a negative exponential curve. With a maximum dispersal distance of 416 km, the probability of a juvenile male dispersing from its natal patch (i) to another (j) was modeled using the function $p_{i,j}$=1.08*e$^{-0.01d_{i,j}}$, where d_{ij} is the distance between the 2 patches (Figure 3).

Sample sizes were too small to estimate the distance-dispersal function between cliff and urban sites. Because peregrines have been nesting in cities in the study region for >10 years and have not yet recolonized their cliff habitat, it is likely that dispersal rates within habitat types (e.g. urban-urban and cliff-cliff) is different from dispersal rates between habitat types (e.g. urban-cliff and cliff-urban). In California, dispersal rates from cliff to urban habitats were 3.3 times higher than dispersal rates from urban to cliff habitats (Kauffman et al. 2004). Dispersal rates between habitat types were also approximately 80-90% lower than those within habitat types. Therefore, I reduced the distance dispersal function by 67% for cliff to urban patches and by 90% for urban to cliff patches. Density of the breeding peregrine population does not affect dispersal, so I did not model density-dependent dispersal (Restani and Mattox 2000).

Simulations and sensitivity analysis.— I simulated 19 metapopulation models (Table 8) representing different reintroduction strategies. These 19 models were run for the low, medium, and high cases of carrying capacity producing a total of 57 simulations (Appendix B). Models included a scenario with no reintroductions and several scenarios of reintroduction with varying cohort sizes (8-24), supplementation schedules (every 1, 3, or 5 yrs), and reintroduction sites (1-2 sites). Reintroductions were simulated in the Monroe/Jersey county area and the Little Grand Canyon/La Rue Pine Hills area (Figure 1). Only juveniles were reintroduced and all strategies were implemented over 10 years. These scenarios are representative of the reintroduction actions possible given the time and budget constraints of a wildlife agency. Simulations were projected for 50 years and replicated 1,000 times.

Cost-benefit analysis.—I conducted a cost-benefit analysis using the low carrying capacity scenario to determine the most cost-effective reintroduction strategy. Assuming each falcon and hack station costs $5,000 and $21,000 respectively (Redig and Tordoff 1988, adjusted for inflation), I calculated the total cost of each management scenario run over a 50 year time period. The best reintroduction strategy had the lowest cost to benefit ratio, where benefit was defined as the increase of the following metrics from the base scenario: minimum expected abundance, average number of adults in cliff patches, and average number of adults in southern Illinois cliff patches.

Sensitivity analysis.— Finally, I assessed model sensitivity to uncertainty in dispersal rates, carrying capacity, and to Rmax, by varying each parameter by ±10% and examining the relative influence on the minimum expected abundance, the number of patches occupied in 50 years, and the median time to ≥ 1 male adult in southern Illinois. Because reducing Rmax by 10% produced a value <1, I set the lower limit of Rmax at 1. Sensitivity analysis was conducted on the low case scenario with no reintroductions.

Model Verification

To evaluate model accuracy, I produced a spatially-explicit PVA for peregrine falcons nesting along the north shore of Lake Superior using the same parameters in the Illinois model (i.e., from the lower Mississippi River region; Table 7). Reintroduction scenarios followed those actually implemented in the region (Table 9). I visually compared model and observed population trajectories for the Lake Superior region and used the standard deviates test (McCarthy & Broome 2000) to compare the observed number of male adults over the predicted probability distribution of male adults.

The latter method expresses observed values in terms of the number of standard deviations from the predicted mean using the formula:

$$\frac{x_{obs} - \overline{x}_{pred}}{\sigma^2_{pred}},$$

where x_{obs} is the observed number of male adults, $\bar{x}_{pred}$ is the predicted mean number of male adults, and σ^2_{pred} is the predicted standard deviation. The standard deviates method was used because it considers model stochasticity and accounts for increasing model variation over time (McCarthy & Broome 2000). Predicted population sizes were based only on the abundance of the previous year to ensure independence of predicted values

(McCarthy & Broome 2000). If the standard deviates have a mean of 0 and a variance of 1 then the model predictions are accurate. A mean >0 indicates the model overestimated predicted population size, while a mean <0 indicates the model underestimated population size. Likewise, a variance of standard deviates >1 indicates model stochasticity is too small, while a variance <1 indicates model stochasticity is too large. Significant deviations from these values were determined using a one-sample *t*-test in SAS and chi-squared test for standard deviations in DataPlot (McCarthy & Broome 2000, Heckert 2001).

RESULTS

Monitoring data

Survival model selection.—The global model fit the data well with a median $\hat{c}$ of 1.01, indicating no overdispersion. Detection probability appeared to be influenced by age classes while recovery rates appeared to be influenced by age classes and natal sites (Table 5). Age and natal site interacted to influence survival. Natal site may have had some influence on resighting rates, however the evidence was weak. The top four models carried 99% support of the data.

Survival rate estimation.— The top 4 models (ΔAICc < 2.39) were averaged for survival rate estimation. Model averaged survival was 0.20 ± 0.06 for cliff-fledged juveniles and 0.24 ± 0.02 for urban-fledged juveniles. Process variance was 0.03 for cliff-fledged juveniles and 0.02 for urban-fledged juveniles. Survival of subadults and adults were similar between cliff (0.84 ± 0.14 and 0.85 ± 0.05) and urban (0.85 ± 0.07

and 0.85 ± 0.02) habitats. Process variance was 0.13 for cliff-fledged subadults, 0.07 for urban-fledged subadults, 0.04 for cliff-fledged adults, and 0.02 for urban-fledged adults.

Productivity estimation.— Average number of fledglings was 1.8 ± 0.5 for cliff-nesting peregrines and 2.6 ± 0.1 for urban-nesting peregrines. Process variance of average number of fledglings for urban-nesting peregrines was estimated at 0.13. Implementing the 50% increase of the urban estimate, process variance of average number of fledglings for cliff-nesting peregrines was estimated at 0.20.

Population viability analysis

The current peregrine population without reintroduction appears to be quite stable, even under different levels of carrying capacity. The model predicted a minimum abundance of 23 adult males, for the study area in all three scenarios, and an average abundance of 40 adult males in 50 years, indicating a 22.5% increase without reintroductions (Table 10). Changing carrying capacity did not appear to have an influence on population trends (Figure 4A), even when the southern Illinois carrying capacity was reached with reintroductions under the low case scenario, but not under the high case scenario (Figure 4B). Total carrying capacity for the study region was not reached in any of the models, with or without reintroductions, but the population appears to be slowly increasing with time (Figure 4).

Without reintroductions, the models predict that peregrines will recolonize cliffs in the study region in approximately 3 years, but that recolonization of cliffs in southern Illinois will take approximately 11 years (Table 10). Only about half (4.5 of 8) of the cliff patches are expected to be occupied in 50 years, by a total of 13 adult males. In all three scenarios of carrying capacity, only 1 of the 2 southern Illinois patches was occupied after 50 years, for a total of 2 adult males.

Reintroducing juveniles to cliffs in southern Illinois increased the expected minimum abundance, number of occupied patches after 50 years, and the average number of adults (Figure 5). As expected, all of these metrics increased with the number of individuals released, however the expected minimum abundance appeared to reach a plateau when more than 100 juveniles were released (Figure 5A). Reintroductions reduced the median time to at least one adult male in southern Illinois by 74.6 – 78.1% (Table 11). Hacking all birds at one site versus multiple sites did not largely influence average number of patches occupied in 50 years or the total average number of adult males on cliffs as indicated by scenarios 4 and 10, 5 and 11, and 6 and 12. Hacking at multiple sites did appear to slightly increase the average number of adult males on cliffs in southern Illinois as indicated by scenarios 4 and 10 and 5 and 11.

Cost-benefit analysis— Reintroducing 16 juveniles from one site every 3 years (scenario 5), had the lowest cost-benefit ratio for increasing the expected minimum abundance (i.e. decreasing extinction risk; Figure 6). Reintroducing 8 juveniles from 2 sites every 3 years (scenario 11) had a similar cost-benefit ratio for increasing the expected minimum abundance. To increase the number of adult males in southern Illinois, reintroducing 8 juveniles from 2 sites every 3 years (scenario 11) had the lowest cost-benefit ratio, followed by scenario 2 (reintroducing 8 juveniles from one site every 3 years; Figure 7A). Finally, to increase the number of cliff adult males in the study region, reintroducing 16 juveniles from 2 sites every 3 years, had the lowest cost-benefit ratio (Figure 7B). Scenarios 17 and 11, which required reintroducing 24 juveniles and 8 juveniles, respectively, from 2 sites every three years also produced low cost-benefit ratios for increasing the number of adult males on cliffs in the study region.

Sensitivity analysis—The model was most sensitive to changes in Rmax. Increasing Rmax by 10% produced an 18.7% increase in expected minimum abundance, a 34.1% increase in patch occupancy, and 13.2% decrease in median time to ≥ 1 adult in southern Illinois. Decreasing Rmax to 1 produced a 17.4% decrease in expected minimum abundance, a 14.6% decrease in patch occupancy, and a 1.7% increase in the median time to ≥ 1 adult (Table 12). In comparison, altering carrying capacity produced a maximum change of 7.8%. Changing the dispersal matrix, only affected the median time to ≥ 1 adult on cliffs in southern Illinois.

Model testing

The model population trajectory for the Lake Superior region increased steadily until reintroductions ended in 1995, after which time the number of adult males remained approximately constant (Figure 8A). In the actual population trajectory, the number of adult males continued to increase to the present. Therefore, during the first 8 years of reintroduction, the model appeared to follow the actual population trend well. Based on the population trajectories, the model overpredicted the number of adult males during 1991-2003 and underpredicted during 2003-2005. The standard deviates test indicated the model underpredicted population size and stochasticity, where the mean of standard deviates was significantly less than 0 ($n = 10$, $t = -2.12$, $p > 0.05$), and the variance was significantly higher than 1 ($n = 10$, $\chi^2 = 33.8$, df = 9, $p > 0.05$; Figure 8B).

DISCUSSION

Population growth and demographics

Results from this study show that the lower Midwestern population of peregrine falcons is stable. However, despite being well below carrying capacity, cliff-nesting peregrines showed essentially no growth. Instead, it appears that metapopulation growth relies primarily on the peregrines nesting in cities. These results are similar to those found in studies of peregrine falcons in California, where on average the rural population declined 1% per year, while urban populations grew an average of 29% per year during 1980-1998 (Kauffman et al. 2003). However, in California, the rural population acted as a sink, despite increasing growth rates with population size, which was likely due to declining levels of DDE and increasing eggshell thickness (Kauffman et. al. 2003, 2004). Such trends are unlikely for the Midwestern population because DDT has been banned from use in the U.S. for over 30 years and because Midwestern peregrines typically prey on resident species which have significantly lower levels of DDE residues than migratory species, which are often consumed by the California population (Banasch et al. 1992).

One possible explanation for the finding that cliff-nesting populations are not growing is that Rmax for cliff-nesting birds (1.002) may have been underestimated. However, I do not believe Rmax was underestimated because peregrines show strong density dependence in growth rates (Herbert and Herbert 1965, Hunt 1988, Ratcliffe 1993, Tordoff and Redig 1999, Kauffman et al. 2004) and the demographic rates I used to estimate Rmax were from a population that was well below its carrying capacity so the population should have been growing at its maximum rate. However, sensitivity analysis indicated that Rmax had the largest influence on population predictions. Rmax sets the strength of density dependence and was nearly 1 for cliff-nesting populations, so carrying capacity had little influence on model outcomes. Given that the Lake Superior population model showed growth only during years of supplementation and plateaued after reintroductions stopped, and the observed population trajectory continued to increase after reintroductions ceased, further attention to the demographic rates used in estimating Rmax was warranted. Comparing my survival rates used in the estimation of Rmax to literature rates indicate that juvenile survival (0.20-0.24) was on the low end of the range of literature values (0.16-0.65; Table 13), but were similar to those estimated by Mebs (1971), Tordoff and Redig (1997), and Kauffman et al. (2003). My results show that nesting sites may have an effect on juvenile survival, but that the difference in rates was not as strong as that in California (Kauffman et al. 2003). However, California estimates for urban fledglings may have been biased high because young from problematic urban areas were moved and released in other areas. Survival rates may also be lower in the Midwest because of higher levels of predation by great horned owls on cliff-nesting peregrines due to differences in reintroduction techniques. In the Western U.S., most peregrines were reintroduced using fostering methods, so young were protected by adults. In the East and Midwest, however, peregrines were reintroduced using hackboxes, where juveniles were unprotected and thus more vulnerable to predation (Barclay and Cade 1983, Septon et al. 1995). Despite the differences in survival rates of juvenile peregrines to other literature values, adult survival rates were quite comparable. Subadult and adult survival rates matched those in California and were well within the range of estimates found worldwide (Table 13).

While survival rates for juveniles in the Midwest appeared to be low compared to other regions, productivity of Midwestern peregrines appeared to be quite high (Table 14). The average number of fledglings per territorial pair in the Midwest was comparable to those in Alaska and Greenland, but higher than those throughout the rest of the U.S. Furthermore, the average number of fledglings of urban Midwestern peregrines was among the highest documented in North America. Because most of the literature values were taken from rural populations, it is likely that the high urban rate can be attributed to the large and easily accessible prey base found in cities and the lack of natural predators. Furthermore, cliff peregrine productivity rates in the Midwest were comparable to literature values of similar latitudes.

Thus, the parameters used in the population model appear to be accurate and reasonable. One of the reasons the model may have underpredicted population growth in the Lake Superior region is because productivity seems to be greater at more northern latitudes. Indeed, the estimates of peregrine productivity in Canada and Greenland (Mattox and Seegar 1988, Bird and Weaver 1988) were the highest among literature values. Higher fecundity would have produced a larger Rmax value, and thus increased population growth. However, increasing Rmax for the southern Illinois region seemed unreasonable because the demographic rates I derived appeared to be comparable to other populations at similar latitudes. An increase in Rmax would have likely overestimated population growth for cliff-nesting peregrines, producing overly optimistic trends.

The model results provide much explanatory power with regard to population growth in the lower Midwest. These results explain why cliff-nesting peregrine populations are recovering so slowly, despite healthy urban populations. Although cliff populations are stable, they are not increasing, and thus for growth to occur, birds from other areas must colonize cliffs. These findings explain why many cliffs remain without peregrine falcons, despite decades of recovery efforts following the elimination of the use of DDT.

Management implications for southern Illinois

Based on model results, peregrine falcon reintroduction to southern Illinois is unnecessary for maintaining regional population stability. The current growth rate of urban peregrines and the stability of cliff-nesting peregrines appear to be sufficient to maintain a healthy, albeit slow-growing population. Reintroducing juveniles to southern Illinois would decrease the time to recolonization and increase the number of cliff breeding pairs, but the benefits do not seem to outweigh the costs. Of the reintroduction scenarios, the most cost-effective strategies were those that released juveniles every 3 years. Of the top 5 scenarios, reintroduction strategy 11 appeared to be the best. This strategy released 8 juveniles from two sites every 3 years for a cost of approximately $282,000. However, if implemented, the state would only see an increase of 2 cliff-breeding pairs compared to the base scenario of no reintroduction. Spending nearly $300,000 to increase the population by 2 pairs hardly seems a worthy investment, especially if the population is not at risk of extinction. Furthermore, the model predicts that even without reintroductions, peregrine falcons will recolonize cliffs in southern Illinois in approximately 11 years. Although there is uncertainty in dispersal rates between urban and cliff populations, and hence predictions of colonization time, the model indicates that peregrines will recolonize their natural habitat in time.

Thus, I recommend that monetary funds would be better used elsewhere such as purchasing land containing peregrine habitat. Many of the bluffs in Monroe County, Illinois, for example are still being used for mining. Purchasing and preserving such areas would ensure isolation from human disturbance, which peregrines typically prefer for nesting (Hickey 1942). Additionally, many of these sites need to be actively managed as cliffs may become more overgrown with vegetation as a result of fire suppression (Septon 1993). Furthermore, many of the cliffs suitable for peregrine nesting in Illinois are located in the unique remnant hill prairie ecosystem. Protecting these areas would not only ensure the availability of habitat for peregrine recolonization, but would also likely help sustain other endemic species such as the Missouri coneflower, narrow-leaved green milkweed, plains scorpion, and dark-sided salamander (IDNR 2005).

This study shows the value of monitoring data for adaptively managing species of concern and of conducting population viability analyses prior to reintroduction efforts. In the past, reintroductions of peregrine falcons were necessary to reestablish an extirpated population in the East and Midwest and to sustain a sink population in California (Barclay and Cade 1983, Septon et al 1995, Kauffman et al. 2004). Without analyzing monitoring data and conducting a population viability analysis of the current population, it would have been unclear that the Midwestern population is stable and self-sustaining. If the state had initiated reintroductions without such an analysis, they would have spent thousands of dollars with relatively little return. Instead, managers can reserve their funds for areas that are more likely to benefit the species such as in habitat preservation. Furthermore, this study highlights the importance of incorporating spatial structure in population models. By modeling two distinct population types (i.e. cliff and urban), it is clear that urban peregrines play an important role in maintaining metapopulation growth for the lower Midwest.

Future research

While this study has brought new insight into the growth rates and dynamics of the Midwestern peregrine population, it also highlighted areas that are in need of further research. Sensitivity analysis indicated that Rmax plays a crucial role in determining population trends. Further research should be conducted to estimate annual population growth for the Midwestern population to assess density-dependent trends. Likewise, I recommend further analysis of dispersal between cliff and urban populations, which would be useful in obtaining more accurate estimates of time to recolonization. Because monitoring data via the Midwest Peregrine Society is so easily accessible, such research can be conducted with relatively little cost.

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Table 1. Cliff attribute values reported in the peregrine falcon literature, measured in GIS at the landscape level, and measured during site visits in southern Illinois.

Variable	Literature*	Landscape level	Site level
Distance from water (km)	0 – 11	0 - 11	0 - 8
Dominance (m)	50 – 475	NA	22 - 144
Elevation (m)	100 – 2057	81 – 247	110 - 247
Height (m)	8 – 400	NA	11 - 101

* Studies located primarily in Utah and Greenland (Porter and White, 1973; Grebence and White, 1989; White *et al.*, 2002; Wightman and Fuller, 2005.

Table 2. Literature values of occupied and unoccupied cliff site attributes for peregrine falcons

Variable	Occupied ($\bar{x} \pm$ SD)	Unoccupied ($\bar{x} \pm$ SD)	Occupied (Range)	Location
Distance from water (m)				
	2000 ± 1800	4100 ± 3400	-	Italy/Switzerland[1]
	500 ± 700	5700 ± 7800	0 - 3000	Utah[4]
	4000 ± 3200	-	200 - 10800	Utah[5]
	2700 ± 3200	-	300 - 6400	Utah[5]
	500 ± 500	600 ± 500	0 - 2800	Greenland[6]
Dominance (m)				
	361 ± 230	229 ± 290	-	Italy/Switzerland[1]
	250 ± 150	142 ± 132	-	Spain[2]
	456 ± 235	275 ± 224	-	Italy[3]
	205 ± 92	180 ± 89	50 - 475	Greenland[6]
Elevation (m)				
	649 ± 268	917 ± 471	-	Italy/Switzerland[1]
	1467 ± 189	1505 ± 218	1207 - 1865	Utah[4]
	1520	-	1024 - 2057	Utah[5]
	834 ± 306	1094 ± 380	-	Italy[3]
	288 ± 115	265 ± 124	100 - 550	Greenland[6]
Height (m)				
	152 ± 121	95 ± 71	-	Italy/Switzerland[1]
	63 ± 41	47 ± 31	-	Spain[2]
	141 ± 54	126 ± 65	79 - 305	Utah[4]
	54	-	12.2 - 121.9	Utah[5]
	187 ± 129	96 ± 48	-	Italy[3]
	99 ± 66	61 ± 33	14 - 365	Greenland[6]
	-		8 - 400	Worldwide[7]
Length				
	892 ± 756	181 ± 186	-	Italy/Switzerland[1]
	953 ± 947	392 ± 328	-	Italy[3]
% agriculture				
	10 ± 11	14 ± 14	-	Italy/Switzerland[1]
	21 ± 16	18 ± 15	-	Spain[2]
	28 ± 16	16 ± 16	-	Italy[3]

Sources: [1]Brambilla et al. 2006, [2]Gainzarain et al. 2000, [3]Sergio et al. 2004, [4]Grebence and White 1989, [5]Porter et al. 1973, [6]Wightman and Fuller 2005, [7]White et al. 2002

Table 3. Rank and weight of variables used to determine suitability of potential peregrine falcon reintroduction sites in southern Illinois using the analytical hierarchy process

Habitat Variable	Weight	Rank
Dominance	0.35	1
Height	0.24	2
Slope	0.14	3
Length	0.10	4
Distance to water	0.06	5.5
Disturbance	0.06	5.5
% Agriculture	0.04	7

Table 4. Top-ranked peregrine falcon reintroduction sites ranked from most to least suitable based on site attributes and weights from the analytical hierarchy process.

Site	Height (m)	Length (km)	Slope (°)	Elevation (m)	Dominance (km)	Distance to water (km)	% Agriculture	Disturbance type	Ownership	Weight
Fults	100.7	3.9	90	169.6	114.0	1.1	48.0	Road	Public	0.113
Monroe City	88.4	1.7	85	181.8	112.0	3.8	44.4	Road	Private	0.075
Saltpeter Cave	80.6	2.0	90	168.3	108.7	3.5	47.3	Road	Public	0.070
Renault Herp.	61.8	2.0	90	154.6	98.7	0.5	49.3	Road	Public	0.052
Pine Hollow	18.5	0.5	87	162.9	81.0	0.6	34.6	None	Private	0.038
Draper's Bluff	22.9	0.8	90	207.6	98.3	4.6	39.5	Climbing	Private*	0.038
Tower Rock	31.0	0.1	90	122.2	48.7	0.0	27.3	Hikers	Public	0.036
Little Grand	47.4	0.6	90	142.7	68.8	0.1	28.3	Hikers	Public	0.034
San Damiano	43.6	0.5	80	112.7	60.3	0.0	25.3	Residents	Private	0.030
Cedar Bluff	21.9	0.3	65	179.0	69.0	4.7	38.7	Hikers	Public	0.018

*Protected private land

Table 5. Markov emigration models of survival (s), detection (p), recoveries (r), resighting (R), resighting prior to death (R'), emigration (F), and immigration (F') for peregrine falcons in the lower midwestern United States.

Model	K	AICc	Δ AICc	Weights	Description
{s(a3)p(a3)r(a3+n)R(.)R'(.)F(.)F'(.)}	14	2942.26	0.00	0.39	Survival with age effects; detection with age effects; recovery with age and natal site effects
{s(a3*n)p(a3)r(a3+n)R(.)R'(.)F(.)F'(.)}	17	2942.66	0.40	0.32	Survival with age effects, natal site effects, and interaction; detection with age effects; recovery with age and natal site effects
{s(a3)p(a3)r(a3+n)R(n)R'(.)F(.)F'(.)}	15	2944.13	1.87	0.16	Survival with age effects; detection with age effects; recovery with age and natal site effects; resighting with natal site effects
{s(a3*n)p(a3)r(a3+n)R(n)R'(.)F(.)F'(.)}	18	2944.65	2.39	0.12	Survival with age effects, natal site effects, and interaction; detection with age effects; recovery with age and natal site effects; resighting with natal site effects
{s(a3*n)p(a3+n)r(a3+n)R(n)R'(.)F(.)F'(.)} (GLOBAL model)	15	2994.56	52.30	0.00	Global model: Survival with age effects, natal site effects, and interaction; detection with age and natal site effects; recovery with age and natal site effects; resighting with natal site effects
{s(a3+n)p(a3)r(a3+n)R(.)R'(.)F(.)F'(.)}	15	3018.37	76.11	0.00	Survival with age and natal site effects; detection with age effects; recovery with age and natal site effects
{s(.)p(a3)r(a3+n)R(.)R'(.)F(.)F'(.)}	12	3083.56	141.30	0.00	Survival unconstrained; detection age effects; recovery age and natal site effects

Table 5. Continued

Model	K	AICc	Δ AICc	Weights	Description
{s(.)p(a3+n)r(a3+n)R(.)R'(.)F(.)F'(.)}	13	3170.05	227.79	0.00	Survival unconstrained; age and natal site effects; recovery with age effects
{s(n)p(a3)r(a3+n)R(.)R'(.)F(.)F'(.)}	13	3219.96	277.70	0.00	Survival with natal site effects; detection with age effects; recovery with age and natal site effects
{s(.)p(n)r(a3+n)R(.)R'(.)F(.)F'(.)}	11	3711.97	769.72	0.00	Survival unconstrained; detection with natal site effects; recovery with age and natal site effects
{s(.)p(.)r(a3+n)R(.)R'(.)F(.)F'(.)}	10	3753.82	811.56	0.00	Survival unconstrained; recovery with age and natal site effects
{s(.)p(.)r(n)R(.)R'(.)F(.)F'(.)}	8	3764.43	822.18	0.00	Survival unconstrained; recovery with natal site effects
{s(.)p(.)r(a3)R(.)R'(.)F(.)F'(.)}	9	3766.67	824.42	0.00	Survival unconstrained; recovery with age effects
{s(.)p(.)r(.)R(.)R'(.)F(.)F'(.)}	6	3775.34	833.08	0.00	No constraints
{s(.)p(.)r(.)R(n)R'(.)F(.)F'(.)}	7	3776.02	833.76	0.00	Survival unconstrained; resighting with natal site effects

Table 6. Current number of breeding pairs, from 24 urban areas incorporated in a population viability analysis of peregrine falcons in southern Illinois and the surrounding region.

Urban center	Current no. breeding pairs
Oak Creek, WI	1
Racine, WI	1
Kenosha, WI	1
Pleasant Prairie, WI	1
Waukegan, IL	1
Cedar Rapids, IA	1
Evanston, IL	1
Chicago, IL	8
Des Moines, IA	1
Michigan City, IN	1
South Bend, IN	1
East Chicago, IN	3
Davenport, IA	1
Gary, IN	1
Fort Wayne, IN	1
Columbus, OH	1
Dayton, OH	1
Indianapolis, IN	2
Kansas City, MO*	1
Cincinnati, OH	1
Clayton, MO	1
St. Louis, MO	1
Louisville, KY	1
Chattanooga, TN*	1

*Urban centers excluded from model due to isolation

Table 7. Parameters used for a population viability analysis of peregrine falcons in southern Illinois and the surrounding region. Mean values and process variance are provided.

	Cliff	Urban
Initial abundance	0	31
Juvenile survival	0.20 ± 0.03	0.24 ± 0.02
Subadult survival	0.84 ± 0.13	0.85 ± 0.07
Adult survival	0.85 ± 0.04	0.85 ± 0.02
Sex ratio	1:1	1:1
Subadult fecundity	0.76 ± 0.15[†]	1.11 ± 0.11
Adult fecundity	0.77 ± 0.09[†]	1.11 ± 0.06

*Source: Wootton and Bell 1992

[†]Based on a 50% increase of standard deviation of urban population fertility rates

Table 8. Number of release sites, cohort size, and supplementation schedules over a 10 year period for alternative reintroduction strategies simulated in a 50-year population viability analysis of peregrine falcons in southern Illinois.

Scenario	No. release sites	Cohort size per site	Supplementation schedule	Total juveniles released
Base	0	0	0	0
1	1	8	1	80
2	1	8	3	24
3	1	8	5	16
4	1	16	1	160
5	1	16	3	48
6	1	16	5	32
7	1	24	1	240
8	1	24	3	72
9	1	24	5	48
10	2	8	1	160
11	2	8	3	48
12	2	8	5	32
13	2	16	1	320
14	2	16	3	96
15	2	16	5	64
16	2	24	1	480
17	2	24	3	144
18	2	24	5	96

Table 9. Peregrine falcon reintroductions in the Lake Superior region used to validate a population model of peregrine falcons in southern Illinois and the surrounding region.

Year	Total released	Location (no. released)
1984	4	Mt. Leaveaux, MN
1985	5	Mt. Leaveaux, MN
1986	14	Mt. Leaveaux, MN
1987	6	Isle Royale, MI (3), Rouchleau Pit Virginia, MN (3)
1988	17	Bergland MI (4), Isle Royale, MI (3), Rouchleau Pit Virginia, MN (4), Wolf Ridge ELC Finland, MN (6)
1989	35	Bergland, MI (7), Isle Royale, MI (3), Pictured Rocks MI (5), Rochleau Pit Virginia, MN (9), Sturgeon Bay, ON (10), Wolf Ridge ELF Finland, MN (1);
1990	12	Isle Royale, MI (8), Sleeping Giant Provincial Park, ON (4)
1991	22	Isle Royale, MI (9), Pictured Rocks, MI (9), Ruby Lake, ON (4)
1992	14	Grand Island, MI (5), Ruby Lake, ON (9)
1994	14	15 mi E. Ely, MN (2), Sault Ste. Marie, ON (7), Sleeping Giant, ON (5)
1995	11	Sault Ste. Marie, ON (6), Sleeping Giant, ON (5)

Table 10. Population viability of peregrine falcons in southern Illinois and surrounding regions under a best, medium, and worst case scenario of carrying capacity, and without reintroductions.

	High	Medium	Low
Total carrying capacity	496	264	171
Southern Illinois carrying capacity	47	24	16
WHOLE POPULATION			
Expected minimum abundance	23.2	23.0	23.0
No. occupied patches in 50 years	8.2 ± 1.8	8.2 ± 1.8	8.2 ± 1.7
Ave. no. breeding males in 50 years	40.1 ± 12.0	39.2 ± 11.7	39.3 ± 12.1
ALL CLIFFS			
Median time to ≥ 1 breeding male	3.3	3.4	3.4
Ave. no. patches occupied in 50 years	4.5 ± 1.5	4.4 ± 1.6	4.5 ± 1.5
Ave. no. breeding males in 50 years	12.8 ± 7.4	12.6 ± 7.4	12.8 ± 7.4
SOUTHERN ILLINOIS CLIFFS			
Median time to ≥ 1 breeding male	10.9	11.4	11.4
Ave. no. patches occupied in 50 years	1.0 ± 0.8	1.0 ± 0.8	1.0 ± 0.8
Ave. no. breeding males in 50 years	2.2 ± 2.5	2.1 ± 2.4	2.1 ± 2.4

Table 11. Population viability of peregrine falcons at cliff sites in southern Illinois under various reintroduction strategies using the low case scenario of carrying capacity.

	All cliff patches			Southern Illinois cliff patches			
Scenario	Median time to ≥ 1 adult male	Ave. patches occupied in 50 years	Ave. adult males in 50 years	Median time to ≥ 1 adult male	Ave. patches occupied in 50 years	Ave. adult males in 50 years	Total cost ($)
Base	3.4	4.5 ± 1.5	12.8 ± 7.4	11.4	1.0 ± 0.8	2.1 ± 2.4	0
1	2.6	5.5 ± 1.5	18.0 ± 9.4	2.9	1.5 ± 0.7	4.6 ± 4.0	421000
2	2.6	5.0 ± 1.5	14.7 ± 8.3	2.9	1.3 ± 0.7	3.2 ± 3.2	141000
3	2.6	4.8 ± 1.5	13.7 ± 7.8	2.9	1.2 ± 0.8	2.6 ± 2.8	101000
4	2.5	6.1 ± 1.4	23.0 ± 11.3	2.6	1.7 ± 0.6	6.1 ± 4.6	821000
5	2.5	5.3 ± 1.5	16.4 ± 8.7	2.6	1.4 ± 0.7	3.9 ± 3.5	261000
6	2.5	4.9 ± 1.6	14.5 ± 8.1	2.6	1.3 ± 0.7	3.1 ± 3.0	181000
7	2.5	6.5 ± 1.2	26.7 ± 12.5	2.6	1.8 ± 0.5	7.7 ± 3.6	1221000
8	2.5	5.6 ± 1.5	18.7 ± 9.7	2.6	1.5 ± 0.7	4.6 ± 4.1	381000
9	2.5	5.2 ± 1.6	16.2 ± 8.8	2.6	1.3 ± 0.8	3.5 ± 3.3	261000
10	2.5	6.2 ± 1.3	23.4 ± 10.7	2.6	1.8 ± 0.5	6.7 ± 4.9	842000
11	2.5	5.4 ± 1.5	17.5 ± 9.4	2.6	1.5 ± 0.7	4.4 ± 3.8	282000
12	2.5	5.0 ± 1.6	15.0 ± 8.3	2.6	1.3 ± 0.7	3.1 ± 2.9	202000
13	2.5	6.8 ± 1.1	33.2 ± 14.0	2.5	1.9 ± 0.3	10.7 ± 6.6	1642000
14	2.5	6.0 ± 1.4	22.1 ± 11.0	2.5	1.7 ± 0.5	6.0 ± 4.8	522000
15	2.5	5.3 ± 1.6	16.7 ± 9.2	2.5	1.4 ± 0.8	4.0 ± 3.8	362000
16	2.5	7.2 ± 0.9	41.4 ± 16.1	2.5	2.0 ± 0.2	14.0 ± 7.9	2442000
17	2.5	6.3 ± 1.3	25.8 ± 12.2	2.5	1.8 ± 0.5	7.7 ± 5.3	762000
18	2.5	5.6 ± 1.5	19.1 ± 10.0	2.5	1.4 ± 0.7	3.8 ± 3.6	522000

Table 12. Results of sensitivity analysis for peregrine falcons in southern Illinois and the surrounding region using a low case scenario of carrying capacity with reintroductions to cliffs.

		Expected minimum abundance	No. patches occupied in 50 years	Median time to ≥ 1 male adult in southern IL
Rmax				
	+10%	+18.7%	+34.1%	-13.2%
	-10%	-17.4%	-14.6%	+1.7%
Dispersal matrix				
	+10%	0	0	-19.3%
	-10%	-1.3%	-3.7%	+10.5%
Carrying capacity				
	+10%	+3.9%	0	-0.9%
	-10%	-7.8%	-3.7%	+4.4%

Table 13. Comparison of peregrine falcon survival rates from this study compared with other studies conducted worldwide.

Location	Juvenile	Subadult	Adult	Source
Lower Midwest	0.24 ± 0.02	0.85 ± 0.07	0.85 ± 0.02	This study
Cliff	0.20 ± 0.06	0.84 ± 0.14	0.85 ± 0.05	
Urban	0.24 ± 0.02	0.85 ± 0.07	0.85 ± 0.02	
California	0.38 ± 0.08	0.86 ± 0.05	0.85 ± 0.02	Kauffman et al. 2003
Cliff	0.28 ± 0.04	-	-	
Urban	0.65 ± 0.15	-	-	
Midwest	0.16 - 0.23†	-	0.86	Tordoff and Redig 1997
Colorado	0.54 ± 0.08	0.67 ± 0.1	0.80 ± 0.05	Craig et al. 2004
British Columbia	-	-	0.68	Nelson 1988
Finland	0.29	-	0.81	Mebs 1971
Germany	0.44	-	0.72	Mebs 1971
Scotland	0.44	-	0.91	Newton and Mearns 1988

†Range of estimate

Table 14. Comparison of peregrine falcon productivity values from this study compared with other studies conducted in North America.

Location	Fledglings/territorial pair	Study
Lower Midwest	2.49 ± 0.13	This study
Cliff	1.81 ± 0.52	
Urban	2.61 ± 0.13	
Midwest	1.78 ± 0.19	Moen and Tordoff 1993
Alaska	1.25	Ambrose et al. 1988
Alaska North Slope	1.80	Mattox and Seegar 1988
Arizona	1.66	Ellis 1988
Greenland Southeast	1.60	Mattox and Seegar 1988
Greenland West	2.40	Mattox and Seegar 1988
New Jersey	1.38	Steidl et al. 1991
Quebec	2.85	Bird and Weaver 1988
Washington	1.30	Hayes and Buchanan 2002

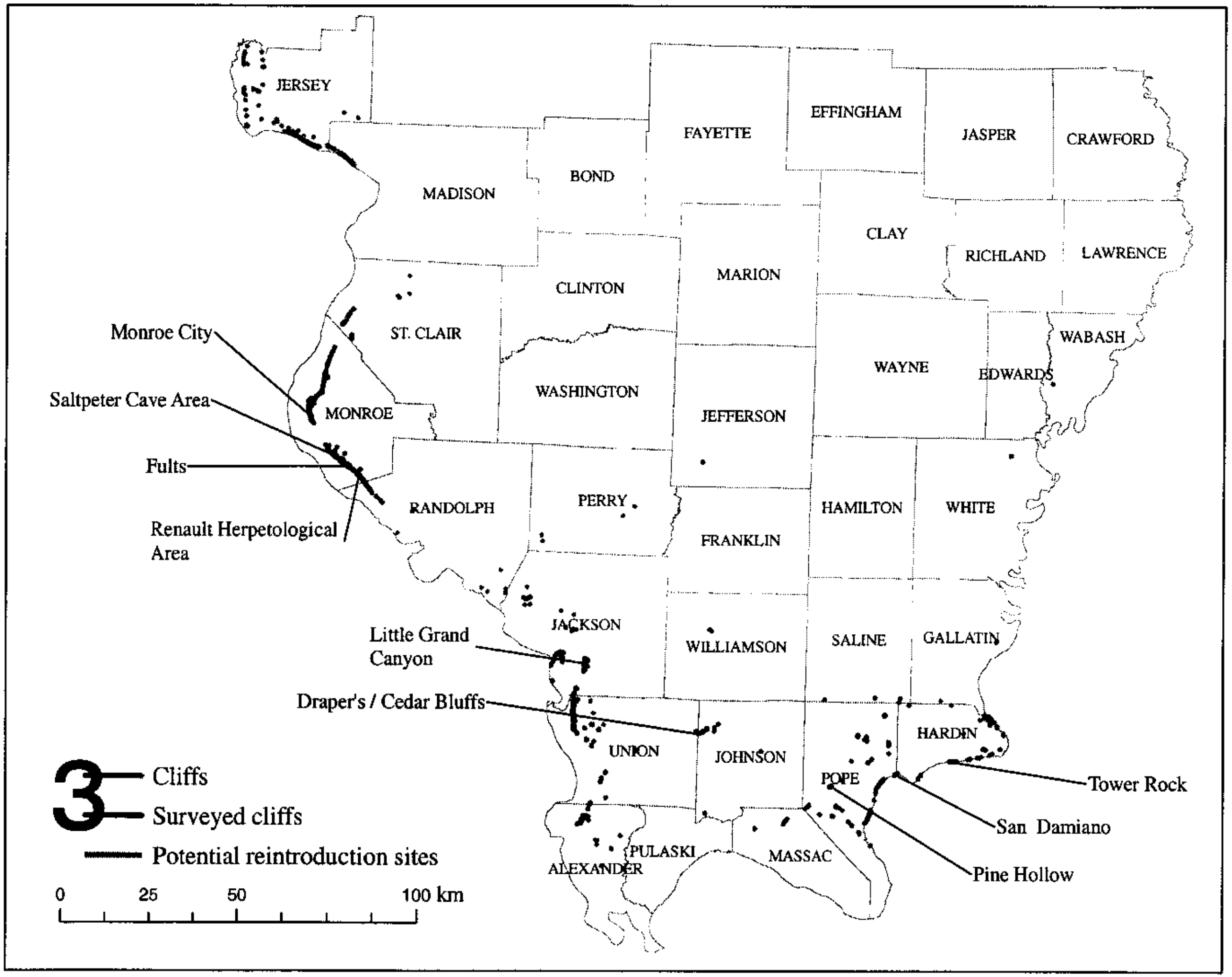

Figure 1. The 35 counties in southern Illinois included in the habitat analysis, suitable peregrine falcon breeding habitat identified by slope, sites surveyed, and potential reintroduction sites.

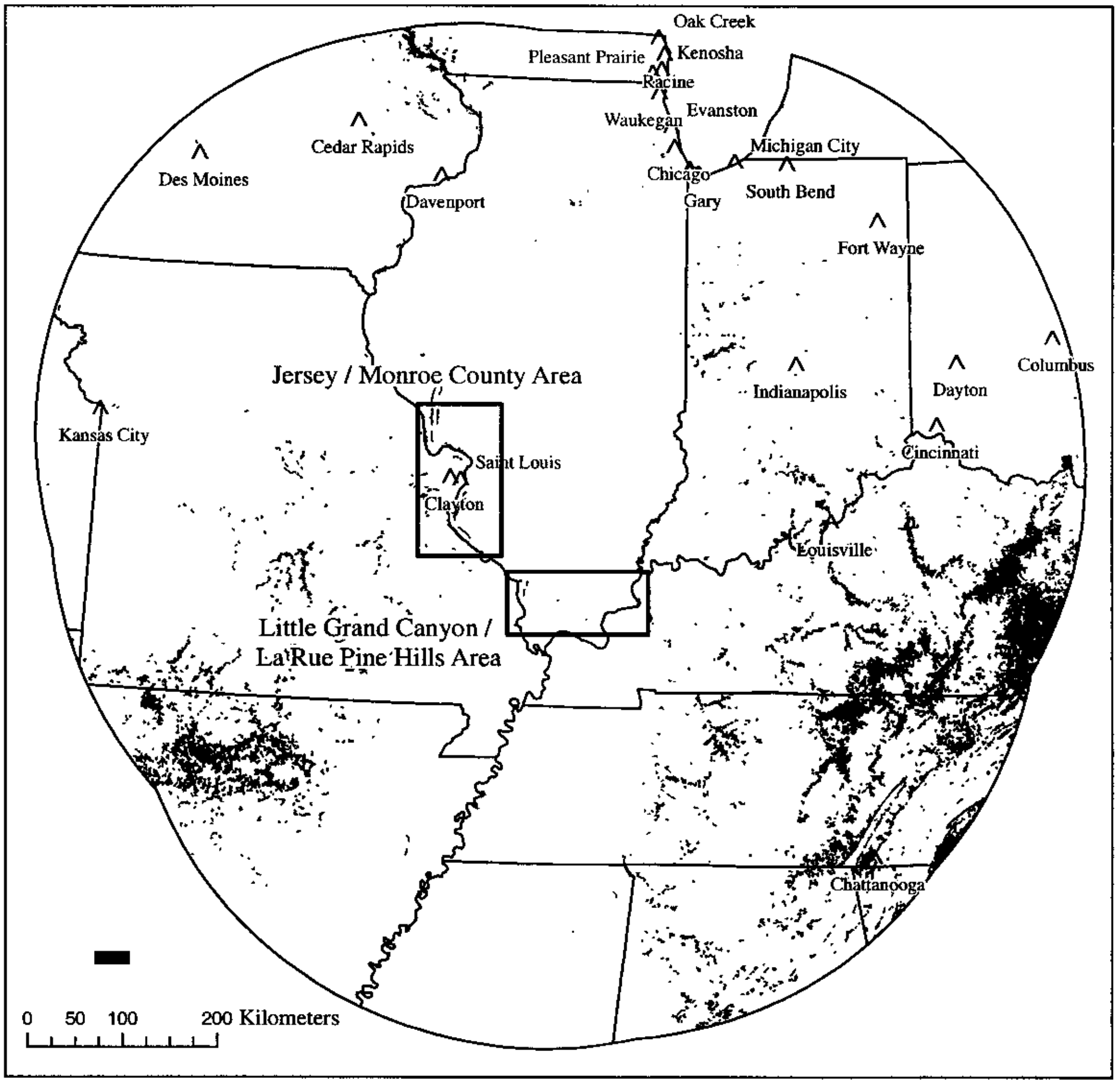

Figure 2. Peregrine falcon habitat map used in population viability analysis with the two major southern Illinois patches. Stars represent urban centers occupied by peregrines in 2005.

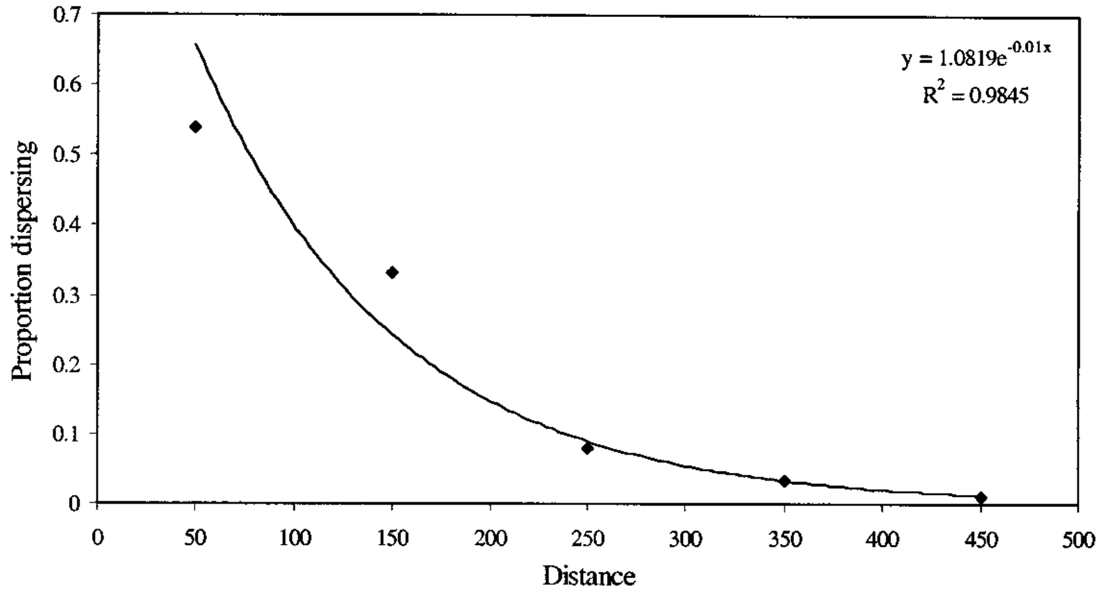

Figure 3. Distance dispersal function for male peregrine falcons in the lower Midwestern United States.

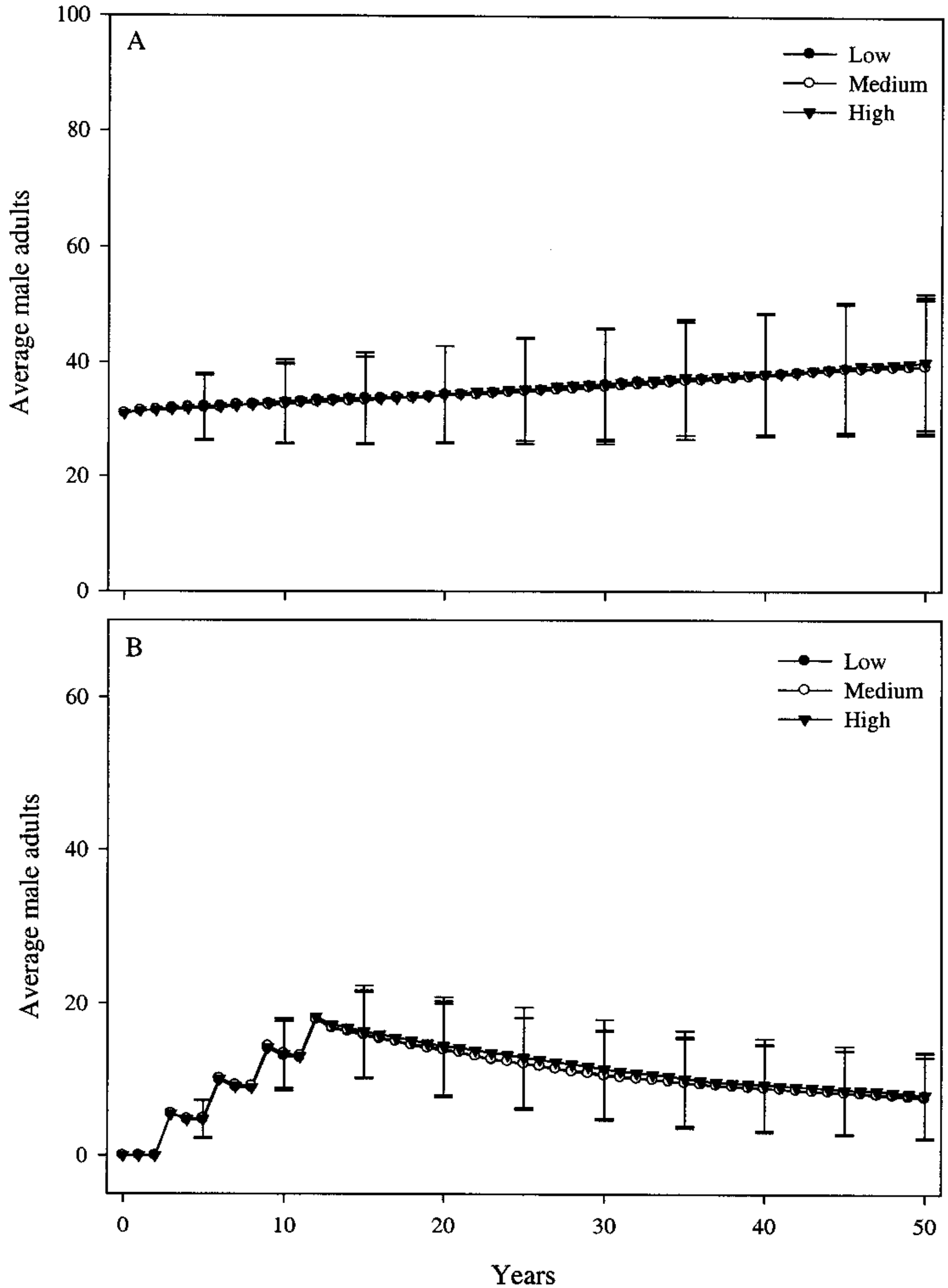

Figure 4. Average number of adult male peregrine falcons predicted by the model under low, medium, and high levels of carrying capacity for (A) the whole study region without reintroductions and (B) for southern Illinois with reintroduction of 24 juveniles at 2 sites every 3 years.

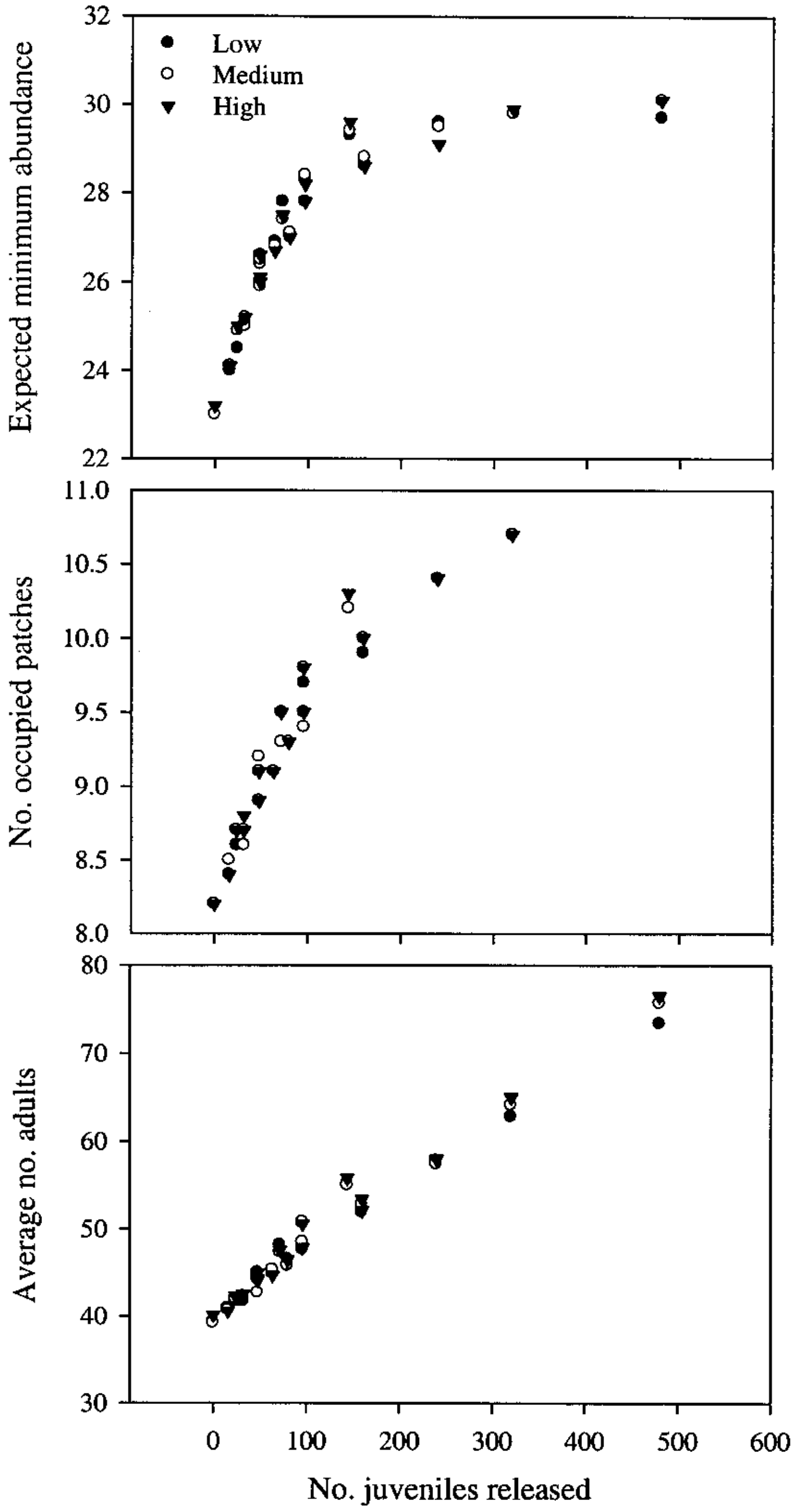

Figure 5. Effects of reintroducing peregrine falcons to southern Illinois on population viability for low, medium, and high carrying capacities.

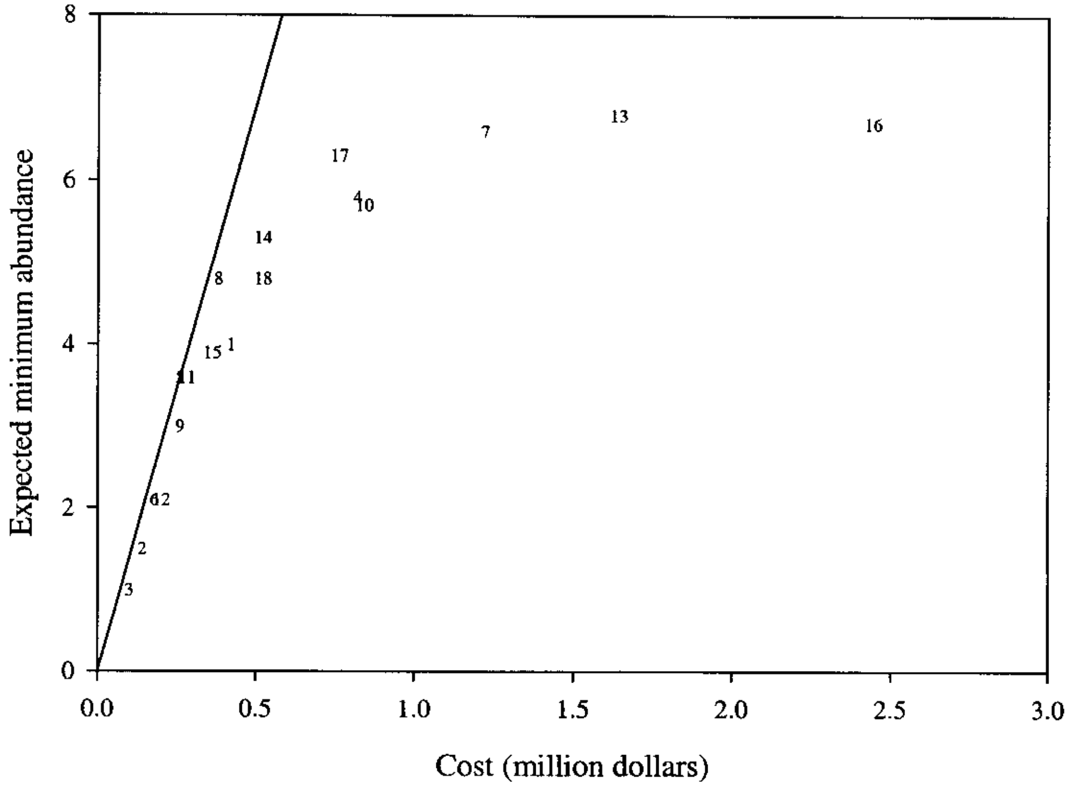

Figure 6. Cost-benefit analysis of peregrine falcon reintroductions using the increase in expected minimum abundance from the no-action scenario. Numbers represent reintroduction scenarios and the diagonal line indicates the lowest cost-benefit ratio with all points below having higher ratios.

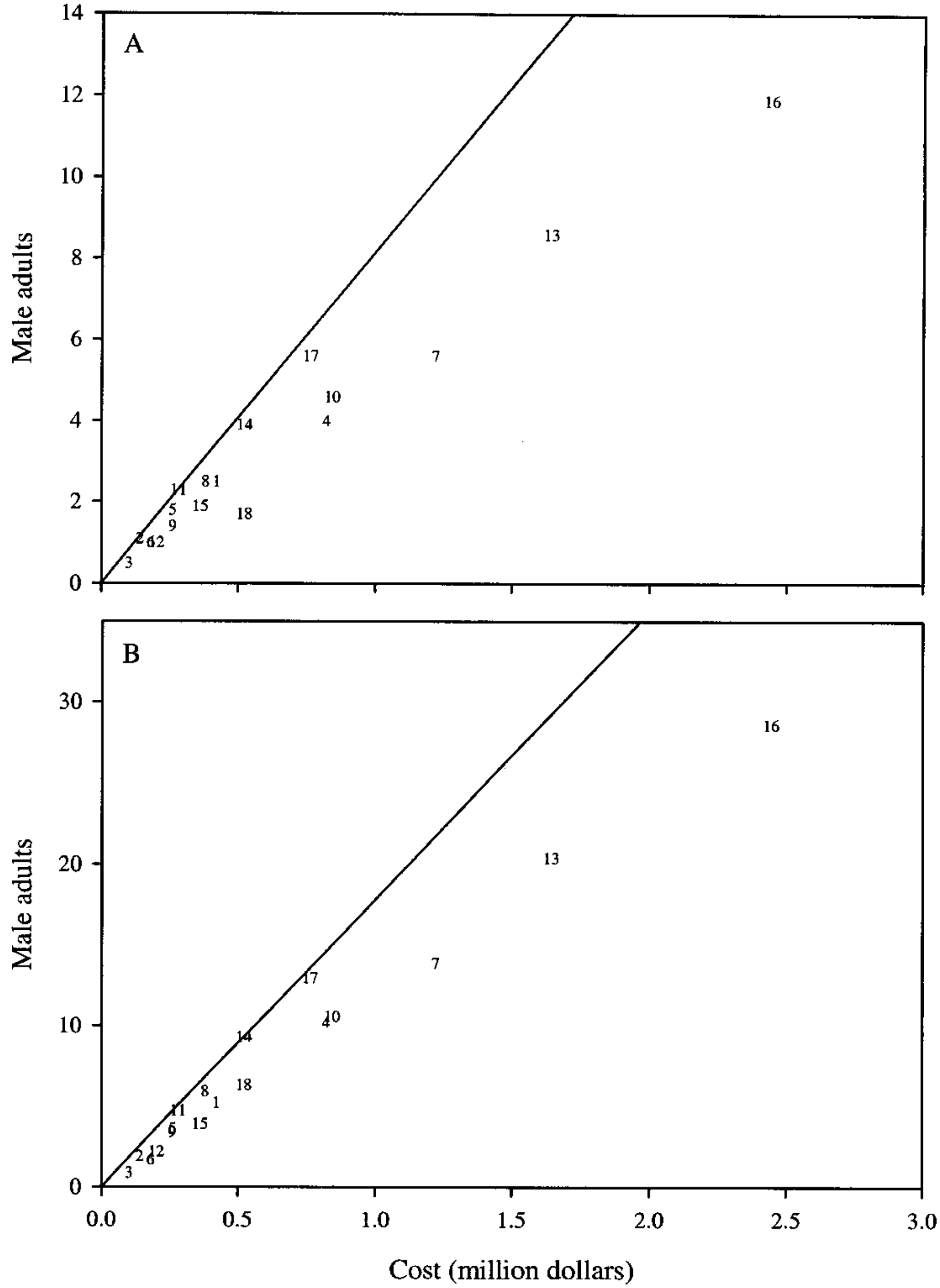

Figure 7. Cost-benefit analysis of peregrine falcon reintroductions using the increase in average number of male adults at (A) southern Illinois cliff sites and at (B) all cliff sites. Numbers represent reintroduction scenarios and the diagonal line indicates the lowest cost-benefit ratio with all points below having higher ratios.

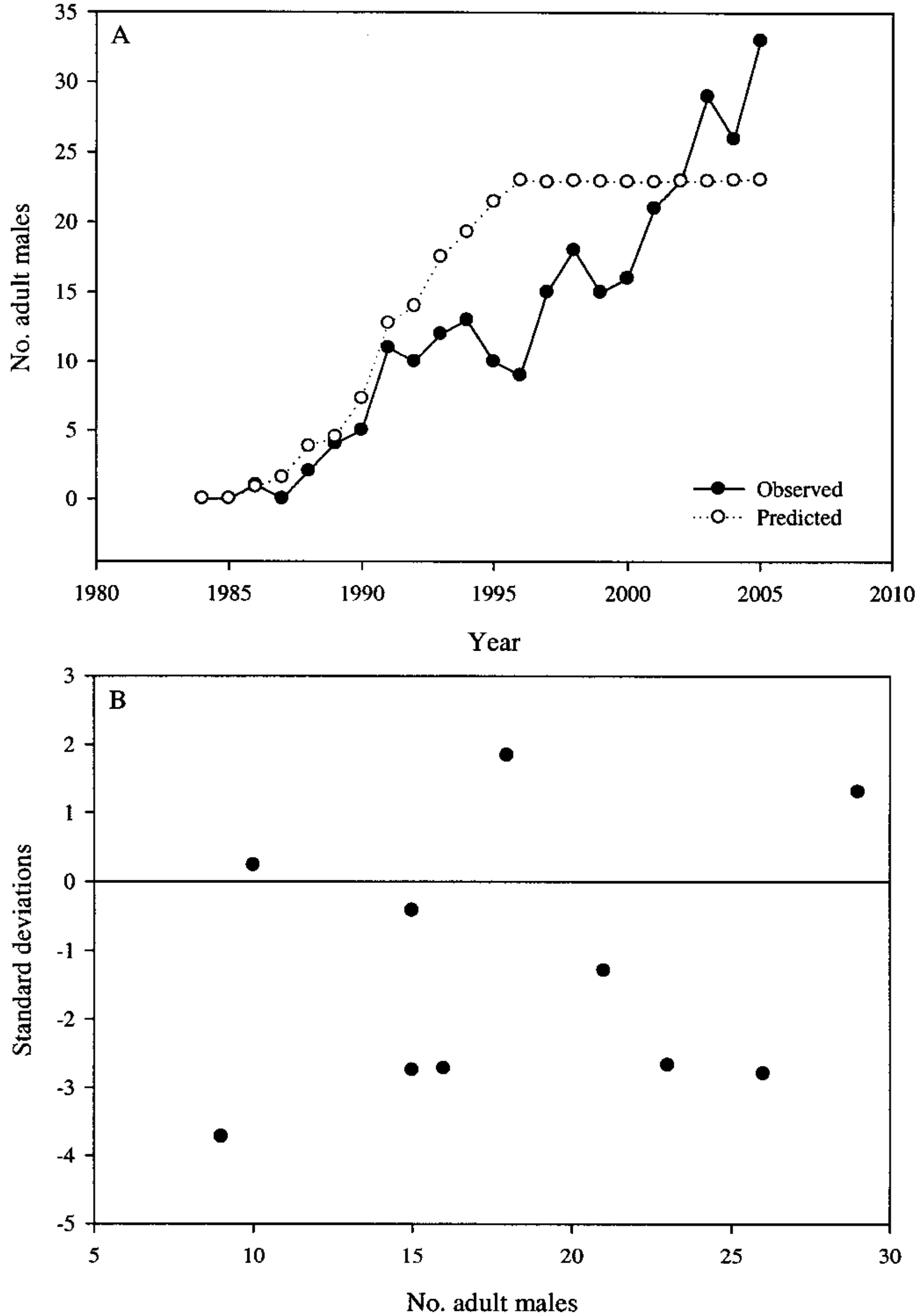

Figure 8. Plot of (A) trajectories for observed (closed circles) and predicted (open circles) number of adult males and (B) standard deviations of the observed number of adult males from the predicted mean over different initial abundances, where good model fit is indicated by a mean of 0 and a standard deviation of 1, for the Lake Superior region during 1984-2005.

APPENDICES

Appendix A. Measurements of potential peregrine falcon cliff sites in southern Illinois grouped by counties.

Site	Height (m)	Length (km)	Slope (°)	Elevation (m)	Dominance (m)	Dist. to water (km)	% Agriculture	Disturbance type	Ownership	Owls
HARDIN										
Cave in rock*	25.1	0.4	90	116.8	28.7	0.0	28.2	Hikers	Public	N
Panther hollow*	12.7	0.2	87	143.0	46.0	1.6	38.4	Hikers	Public	-
San damiano	43.6	0.5	80	112.7	60.3	0.0	25.3	Residents	Private†	N
Tower rock	31.0	0.1	90	122.2	48.7	0.0	27.3	Hikers	Public	N
JACKSON / UNION										
Fountain bluff	64.8	2.5	85	145.6	115.0	1.3	37.6	Road	Public	Y
La rue pine hills	67.4	2.2	90	151.9	144.3	1.2	25.3	Hikers	Public	Y
Little grand canyon	47.4	0.6	90	142.7	68.8	0.1	28.3	Hikers	Public	N
Reed creek canyon*	27.7	0.5	85	162.6	27.3	2.9	47.7	None	Private†	-
JERSEY / MADISON										
Chautauqua	43.5	1.4	90	172.8	88.0	0.0	44.9	Road	Private	Y
Oblate / nature	42.5	0.7	86	157.0	67.3	0.1	32.6	Road	Private	Y
Principia	56.5	5.4	90	164.0	102.7	0.1	43.0	Road	Private	Y
JOHNSON										
Cedar bluff	21.9	0.3	65	179.0	69.0	4.7	38.7	Hikers	Public	N
Draper's bluff	22.9	0.8	90	207.6	98.3	4.6	39.5	Climbing	Private	N
Little black slough*	11.3	0.0	87	124.9	22.3	1.7	53.7	None	Public	-
Miles prairie	33.8	1.4	90	163.7	66.0	5.4	46.1	Road	Private†	Y

*Sites deemed unsuitable for nesting based on cliff dominance, elevation, and distance to water
†Unprotected sites

Appendix A. Continued

Site	Height (m)	Length (km)	Slope (°)	Elevation (m)	Dominance (m)	Dist. to water (km)	% Agriculture	Disturbance type	Ownership	Owls
MONROE										
Demint hill	62.9	4.0	90	157.4	100.0	2.4	51.3	Mining	Private†	Y
Fults	100.7	3.9	90	169.6	114.0	1.1	48.0	Road	Public	N
Monroe city	88.4	1.7	85	181.8	112.0	3.8	44.4	Road	Private†	N
Potato hill	93.0	0.8	90	171.7	102.3	1.9	43.7	Road	Private†	Y
Renault herp. area	61.8	2.0	90	154.6	98.7	0.5	49.3	Road	Public	N
Saltpeter cave area	80.6	2.0	90	168.3	108.7	3.5	47.3	Road	Public	N
POPE										
Lusk creek*	27.2	0.1	90	146.2	28.7	7.8	15.1	Hikers	Public	-
Pine hollow	18.5	0.5	87	162.9	81.0	0.6	34.6	None	Private†	-

*Sites deemed unsuitable for nesting based on cliff dominance, elevation, and distance to water
†Unprotected sites

Appendix B. Population viability analysis results for peregrine falcons at cliffs in the study region and southern Illinois under low, medium, and high levels of carrying capacity.

					All cliff patches			Southern Illinois cliff patches		
K	Scenario	Expected min abun.	Total occupied patches	No. adult males in 50 years	Median time to ≥ 1 adult male	No. patches occupied in 50 years	No. adult males in 50 years	Median time to ≥ 1 adult male	No. patches occupied in 50 years	No. adult males in 50 years
Low	Base	23.0	8.2 ± 1.7	39.3 ± 12.1	3.4	4.5 ± 1.5	12.8 ± 7.4	11.4	1.0 ± 0.8	2.1 ± 2.4
	1	27.0	9.3 ± 1.6	46.6 ± 13.9	2.6	5.5 ± 1.5	18.0 ± 9.4	2.9	1.5 ± 0.7	4.6 ± 4.0
	2	24.5	8.6 ± 1.7	41.7 ± 13.1	2.6	5.0 ± 1.5	14.7 ± 8.3	2.9	1.3 ± 0.7	3.2 ± 3.2
	3	24.0	8.4 ± 1.7	41.0 ± 12.7	2.6	4.8 ± 1.5	13.7 ± 7.8	2.9	1.2 ± 0.8	2.6 ± 2.8
	4	28.8	9.9 ± 1.4	51.9 ± 14.4	2.5	6.1 ± 1.4	23.0 ± 11.3	2.6	1.7 ± 0.6	6.1 ± 4.6
	5	26.6	9.1 ± 1.6	44.3 ± 12.8	2.5	5.3 ± 1.5	16.4 ± 8.7	2.6	1.4 ± 0.7	3.9 ± 3.5
	6	25.1	8.6 ± 1.7	41.7 ± 12.0	2.5	4.9 ± 1.6	14.5 ± 8.1	2.6	1.3 ± 0.7	3.1 ± 3.0
	7	29.6	10.4 ± 1.3	57.8 ± 15.4	2.5	6.5 ± 1.2	26.7 ± 12.5	2.6	1.8 ± 0.5	7.7 ± 3.6
	8	27.8	9.5 ± 1.6	48.2 ± 14.4	2.5	5.6 ± 1.5	18.7 ± 9.7	2.6	1.5 ± 0.7	4.6 ± 4.1
	9	26.0	8.9 ± 1.7	42.8 ± 12.9	2.5	5.2 ± 1.6	16.2 ± 8.8	2.6	1.3 ± 0.8	3.5 ± 3.3
	10	28.7	10.0 ± 1.4	52.5 ± 15.2	2.5	6.2 ± 1.3	23.4 ± 10.7	2.6	1.8 ± 0.5	4.9 ± 6.7
	11	26.6	9.1 ± 1.6	45.1 ± 13.3	2.5	5.4 ± 1.5	17.5 ± 9.4	2.6	1.5 ± 0.7	4.4 ± 3.8
	12	25.1	8.7 ± 1.8	42.2 ± 13.0	2.5	5.0 ± 1.6	15.0 ± 8.3	2.6	1.3 ± 0.7	3.1 ± 2.9
	13	29.8	10.7 ± 1.2	62.8 ± 17.3	2.5	6.8 ± 1.1	33.2 ± 14.0	2.5	1.9 ± 0.3	10.7 ± 6.6
	14	28.3	9.7 ± 1.6	50.7 ± 15.6	2.5	6.0 ± 1.4	22.1 ± 11.0	2.5	1.7 ± 0.5	6.0 ± 4.8
	15	26.9	9.1 ± 1.6	45.0 ± 13.4	2.5	5.3 ± 1.6	16.7 ± 9.2	2.5	1.4 ± 0.8	4.0 ± 3.8
	16	29.7	11.1 ± 1.0	73.4 ± 19.1	2.5	7.2 ± 0.9	41.4 ± 16.1	2.5	2.0 ± 0.2	14.0 ± 7.9
	17	29.3	10.2 ± 1.3	55.1 ± 15.3	2.5	6.3 ± 1.3	25.8 ± 12.2	2.5	1.8 ± 0.5	7.7 ± 5.3
	18	27.8	9.5 ± 1.5	47.7 ± 13.7	2.5	5.6 ± 1.5	19.1 ± 10.0	2.5	1.4 ± 0.7	3.8 ± 3.6

Appendix B. Continued

K	Scenario	Expected min abun.	Total occupied patches	No. adult males in 50 years	All cliff patches: Median time to ≥ 1 adult male	All cliff patches: No. patches occupied in 50 years	All cliff patches: No. adult males in 50 years	Southern Illinois cliff patches: Median time to ≥ 1 adult male	Southern Illinois cliff patches: No. patches occupied in 50 years	Southern Illinois cliff patches: No. adult males in 50 years
Med	Base	23.0	8.2 ± 1.8	39.2 ± 11.7	3.4	4.4 ± 1.6	12.6 ± 7.4	11.4	1.0 ± 0.8	2.1 ± 2.4
	1	27.1	9.3 ± 1.6	45.9 ± 13.3	2.6	5.6 ± 1.4	18.4 ± 9.6	2.9	1.5 ± 0.7	4.4 ± 3.8
	2	24.9	8.7 ± 1.7	42.1 ± 12.7	2.6	5.0 ± 1.5	15.1 ± 8.4	2.9	1.3 ± 0.7	3.2 ± 3.0
	3	24.1	8.5 ± 1.7	40.7 ± 12.1	2.6	4.8 ± 1.5	14.1 ± 7.6	2.9	1.1 ± 0.8	2.7 ± 3.1
	4	28.6	10.0 ± 1.4	52.3 ± 14.6	2.5	6.1 ± 1.3	23.3 ± 10.9	2.7	1.7 ± 0.5	6.2 ± 4.7
	5	26.4	9.2 ± 1.5	44.9 ± 12.9	2.5	5.3 ± 1.5	17.4 ± 9.7	2.6	1.4 ± 0.7	4.0 ± 3.6
	6	25.0	8.6 ± 1.7	42.2 ± 12.4	2.5	4.9 ± 1.6	14.8 ± 8.3	2.6	1.2 ± 0.8	3.2 ± 3.4
	7	29.5	10.4 ± 1.2	57.4 ± 16.1	2.5	6.4 ± 1.2	27.9 ± 12.5	2.6	1.8 ± 0.5	8.2 ± 5.8
	8	27.4	9.3 ± 1.6	47.4 ± 14.1	2.5	5.6 ± 2.0	18.9 ± 9.7	2.6	1.6 ± 0.6	4.9 ± 4.2
	9	25.9	8.9 ± 1.6	42.8 ± 12.5	2.5	5.2 ± 1.5	16.3 ± 8.9	2.6	1.4 ± 0.7	3.6 ± 3.5
	10	28.8	10.0 ± 1.4	52.8 ± 15.4	2.5	6.2 ± 1.2	23.4 ± 10.7	2.6	1.8 ± 0.5	7.1 ± 5.3
	11	26.5	9.1 ± 1.7	44.5 ± 13.4	2.5	5.4 ± 1.5	17.3 ± 9.5	2.6	1.4 ± 0.8	4.4 ± 4.1
	12	25.2	8.7 ± 1.8	42.4 ± 12.9	2.5	5.0 ± 1.6	15.2 ± 8.8	2.6	1.3 ± 0.7	3.1 ± 3.1
	13	29.8	10.7 ± 1.2	64.1 ± 17.7	2.5	6.9 ± 1.1	33.5 ± 14.3	2.5	1.9 ± 0.3	11.0 ± 6.5
	14	28.4	9.8 ± 1.5	50.8 ± 15.1	2.5	6.0 ± 1.3	21.7 ± 11.0	2.5	1.7 ± 0.6	6.3 ± 4.8
	15	26.8	9.1 ± 1.8	45.4 ± 14.0	2.5	5.3 ± 1.5	17.1 ± 8.9	2.5	1.4 ± 0.8	4.2 ± 3.9
	16	30.1	11.1 ± 1.1	75.7 ± 20.3	2.5	7.2 ± 1.0	42.3 ± 16.6	2.5	2.0 ± 0.2	14.6 ± 8.1
	17	29.4	10.2 ± 1.3	55.0 ± 15.0	2.5	6.3 ± 1.3	25.5 ± 12.3	2.5	1.8 ± 0.5	13.5 ± 5.6
	18	27.8	9.4 ± 1.6	48.5 ± 14.4	2.5	5.8 ± 1.4	19.9 ± 9.6	2.5	1.6 ± 0.6	5.1 ± 4.4

Appendix B. Continued

K	Scenario	Expected min abun.	Total occupied patches	No. adult males in 50 years	All cliff patches: Median time to ≥ 1 adult male	All cliff patches: No. patches occupied in 50 years	All cliff patches: No. adult males in 50 years	Southern Illinois cliff patches: Median time to ≥ 1 adult male	Southern Illinois cliff patches: No. patches occupied in 50 years	Southern Illinois cliff patches: No. adult males in 50 years
High	Base	23.2	8.2 ± 1.8	40.1 ± 12.0	3.3	4.5 ± 1.5	12.8 ± 7.4	10.9	1.0 ± 0.8	2.2 ± 2.5
	1	27.0	9.3 ± 1.6	46.5 ± 13.5	2.6	5.6 ± 1.4	18.6 ± 10.0	2.8	1.5 ± 0.7	4.3 ± 3.9
	2	25.0	8.7 ± 1.7	42.4 ± 12.6	2.6	4.9 ± 1.6	14.7 ± 8.3	2.9	1.2 ± 0.8	3.1 ± 3.14
	3	24.1	8.4 ± 1.8	40.6 ± 12.0	2.6	4.8 ± 1.6	13.9 ± 7.7	2.9	1.1 ± 0.8	2.5 ± 2.6
	4	28.6	10.0 ± 1.4	52.1 ± 15.2	2.5	6.1 ± 1.4	23.0 ± 11.3	2.6	1.7 ± 0.6	6.7 ± 5.3
	5	26.1	9.1 ± 1.6	44.4 ± 13.5	2.5	5.3 ± 1.6	17.2 ± 9.6	2.6	1.5 ± 0.7	4.3 ± 3.9
	6	25.2	8.7 ± 1.7	42.7 ± 13.0	2.5	5.0 ± 1.6	15.1 ± 8.5	2.6	1.3 ± 0.7	3.1 ± 3.2
	7	29.1	10.4 ± 1.4	58.0 ± 17.1	2.5	6.5 ± 1.2	28.0 ± 12.4	2.6	1.8 ± 0.5	8.3 ± 5.8
	8	27.5	9.5 ± 1.5	47.6 ± 13.8	2.5	5.6 ± 1.5	19.3 ± 10.3	2.6	1.5 ± 0.7	4.8 ± 4.1
	9	26.0	8.9 ± 1.7	44.2 ± 12.9	2.5	5.2 ± 1.5	15.9 ± 8.9	2.6	1.3 ± 0.8	3.6 ± 3.5
	10	28.6	10.0 ± 1.5	53.4 ± 15.8	2.5	6.2 ± 1.2	23.8 ± 11.6	2.6	1.8 ± 0.5	7.2 ± 3.4
	11	26.6	9.1 ± 1.7	45.1 ± 13.3	2.5	5.3 ± 1.5	17.3 ± 9.0	2.6	1.5 ± 0.7	4.6 ± 4.3
	12	25.2	8.8 ± 1.7	42.6 ± 12.8	2.5	5.0 ± 1.6	14.9 ± 8.2	2.6	1.3 ± 0.8	3.4 ± 3.3
	13	29.9	10.7 ± 1.2	65.0 ± 17.8	2.5	6.9 ± 1.0	34.9 ± 14.4	2.5	1.9 ± 0.3	11.5 ± 7.0
	14	28.2	9.8 ± 1.5	50.6 ± 15.0	2.5	6.0 ± 1.3	22.3 ± 10.7	2.5	1.7 ± 0.5	6.3 ± 4.9
	15	26.7	9.1 ± 1.7	44.8 ± 13.5	2.5	5.4 ± 1.5	17.5 ± 9.6	2.5	1.4 ± 0.7	4.4 ± 4.0
	16	30.1	11.1 ± 1.1	76.5 ± 21.6	2.5	7.3 ± 0.9	44.9 ± 17.9	2.5	2.0 ± 0.2	15.2 ± 8.2
	17	29.6	10.3 ± 1.3	55.8 ± 16.0	2.5	6.4 ± 1.3	26.6 ± 12.5	2.5	1.8 ± 0.5	8.1 ± 5.6
	18	27.8	9.5 ± 1.6	47.9 ± 14.1	2.5	5.6 ± 1.5	19.6 ± 10.2	2.5	1.6 ± 0.6	5.1 ± 4.2

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Thesis Title:
A habitat and population viability analysis for potential peregrine falcon reintroductions in southern Illinois

Major Professor: Dr. Charlotte Roy

Project: Evaluation of potential habitat for Peregrine Falcon reintroduction in southern Illinois.

Objectives:

1. Identify and survey areas potentially suitable for peregrine falcons.
2. Model the viability of populations under various reintroduction scenarios.
3. Provide specific recommendations for the recovery of peregrine falcon populations in southern Illinois.

Funding Source(s): Illinois Department of Natural Resources, State Wildlife Grant T-30-P

Principal investigator(s): Charlotte Roy Nielsen and Eric Hellgren

Graduate Research Assistant/Staff: Sarah Wakamiya (Graduate Research Assistant)

Introduction

Declines of peregrine falcon populations in the mid 1900s were associated with the use of DDT pesticide and the consequent eggshell thinning among birds of prey. In response, peregrines were listed as federally endangered in 1970. Widespread bans of DDT in the U.S. in 1972, in combination with successful reintroduction programs coordinated by federal, state, and independent biologists, culminated in the federal delisting of peregrine falcons in 1999. However, in Illinois peregrines remain state threatened (Endangered and Threatened Species Protection Board, IDNR 2004).

Peregrine falcons historically inhabited southern Illinois in Jackson, Jersey, and Union counties, suggesting that suitable breeding habitat exists in the region (Ridgeway 1889, Widmann 1907, Bohlen 1978). Although peregrine falcon reintroductions in Illinois began in 1985, all hack sites have been in the Chicago metropolitan area, including the University of Illinois-Chicago, Fort Sheridan, Illinois Beach State Park, and the College of DuPage (Field

Museum 2006). Due to high site fidelity, peregrine falcons have not returned to their historic cliff sites in Illinois. Maintaining solely urban populations is problematic not only because of the increased risk of collisions with man-made objects, but also because of the possibility of shifting public attitudes. Furthermore, a stable, cliff-nesting population in southern Illinois may provide linkage to regional populations of urban falcons (i.e., Springfield, Illinois; St. Louis, Missouri; and Evansville, Indiana), and may reduce the risk of local and metapopulation extinction. Reintroductions have proven to be successful in several other Midwestern states and protocols to maximize the probability of success have been established.

Although reintroduction programs are desirable, they are also costly and time consuming. Therefore, the feasibility of reintroduction should first be evaluated by assessing habitat availability, identifying suitable reintroduction sites, and modeling populations under various reintroduction strategies (IUCN 1998). Population viability analysis (PVA) is a tool for modeling populations and is often used for endangered species management. A PVA can help determine 1) the most effective release strategies for southern Illinois, 2) if the region can sustain a peregrine falcon population, and 3) if reintroductions will expedite the process of re-colonization or stabilize regional populations. Wildlife managers can use PVA to decide whether a reintroduction is a worthwhile venture and to plan a reintroduction scenario that is most likely to succeed (Akçakaya et al. 2004).

Methods

Cliff presence is an essential component of peregrine falcon habitat (Hickey 1942, Christopher 1980); peregrines rely on cliff ledges for protection from predators during egg-laying and chick-rearing phases. Minimum dimensions for suitable peregrine falcon cliffs are 30-m wide and 12-m high, and cliffs should possess at least a 45° slope (Bollengier 1979, Ratcliffe 1980). Sherrod et al. (1982) suggested that cliffs at least 70-100 m high are most desirable for hacking, and empirical studies indicated cliffs with a slope of $\geq 70^{\circ}$ are preferred (Sergio et al. 2004). These characteristics were therefore used to identify optimal habitat for reintroduction.

We used Geographic Information Systems and field surveys to identify both suitable habitat and potential reintroduction sites. Raster-based Digital Elevation Models (DEM) of 10-m resolution and land cover data of 30-m resolution were downloaded from the National Elevation Dataset and National Land Cover Dataset provided by the Seamless Data Distribution System (USGS 2001, 2004). Suitable habitat was identified by querying for slopes $\geq 45^{\circ}$ and analyzing vertical terrain profiles in the DEM that fit the above criteria. Sites that were not federally or state protected were excluded as potential reintroduction sites.

Following identification of potential cliff nesting sites, the quality of these sites were preliminarily assessed via ground truthing and with presence-absence surveys for great horned owls. Great horned owls are an important predator of peregrine falcons and have significantly contributed to mortality of reintroduced peregrines (Redig and Tordoff 1988). Broadcast surveys were conducted at 5 sites between January and mid-March 2007 including Fountain Bluff, La Rue Pine Hills, Principia College, San Damiano, and Tower Rock. Surveys occurred between 1700 and 2200 hrs on clear nights with wind speeds < 20 km/hr (McGarigal and Fraser 1984, Morrell et al. 1991, Takats et al. 2001). Surveys were replicated a minimum of 3 times for sites where no owls were heard to increase the probability of detecting owls if they were present. Recorded owl vocalizations (provided by the Cornell Lab of Ornithology) were broadcasted using a Sony CFS-B11 Cassette Recorder at the approximate center of the site, unless the cliff was >1.6 km long, in which case multiple surveys were conducted along the cliff at intervals of 1.6 km (Takats et al. 2001).

Results

Habitat

Four major regions in southern Illinois contain potential peregrine falcon habitat. These include the Alton area, Jackson/Union County area, Monroe County area, and the Ohio River region (Fig. 1). Within these 4 regions, 32 cliff sites were located in federally or state protected lands. These sites will be surveyed for the presence of owls and the suitability for reintroduction.

Owl Surveys

Two of the 5 sites surveyed, consistently showed no owl presence (Table 1). Both of these sites were along the Ohio River and included San Damiano and Tower Rock. Great horned owls were heard most frequently at Principia College, with an average of 4.5 owls detected per survey night. La Rue Pine Hills and Fountain Bluff both averaged 1 owl per survey night; however at each of these sites, 2 owls were heard on the same night rather than 1 owl per night.

Discussion

Currently, we are collecting several additional measurements and attributes at cliff sites. We are characterizing cliff physiography by measuring cliff height, slope, and horizontal extent with a rangefinder and clinometer. We are also recording cliff orientation, number of potential ledges, observed competitors and prey, human disturbance factors, and cliff dominance, which is a measure of how prominent the cliff is over the landscape. Cliff dominance will be assessed in GIS by averaging the change in elevation from the top of the cliff to ground level at 1, 2, and 3 km (Fig. 2, Gainzarain et al. 2000). Dominant cliffs allow peregrines to spot and stoop down on prey.

We are also currently examining surrounding land use, which influences prey availability and abundance (Hickey 1942, Newton 1988). Peregrine falcons are usually found near large bodies of water, which provide a commanding view and abundant prey base (Hickey 1942, Ratcliffe 1980). They have an indiscriminate diet of bird species and will forage in croplands, pastures, waterways, swamps, or marshes. Average foraging distance for peregrine falcons is approximately 13 km (Enderson and Craig 1997). Therefore, land-use classifications within a 13-km radius of cliff sites are being identified from the land-cover dataset.

Following field data collection, we will select the best reintroduction sites with literature-based expert information and a multicriteria decision-making process. Nine reintroduction site variables (i.e., cliff height, slope, dominance, orientation, adjacent habitat, nearest water type, distance to water, owl presence, and human disturbance) will be compared. We will then use a spatially explicit PVA model to 1) evaluate the time to establish a breeding population of peregrine falcons in southern Illinois and 2) to determine the most viable reintroduction program with the lowest cost to benefit ratio.

The results of this population viability analysis will provide state wildlife managers with information to determine whether reintroductions are necessary for the Illinois peregrine falcon population, if a reintroduction would likely be successful, and how beneficial a reintroduction would be to the current regional population. Data continue to be collected and analysis and interpretation will be included in the project Final Report.

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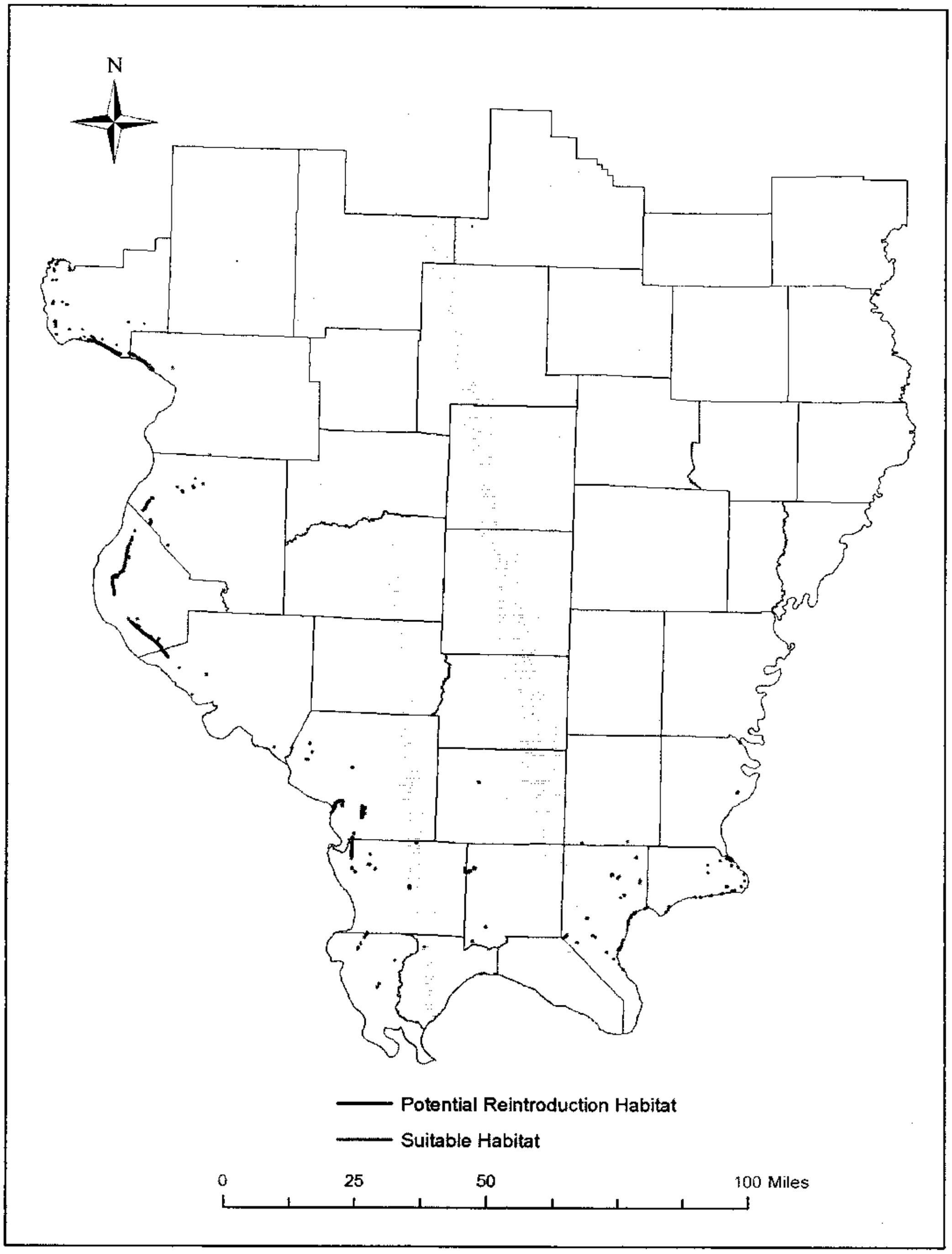
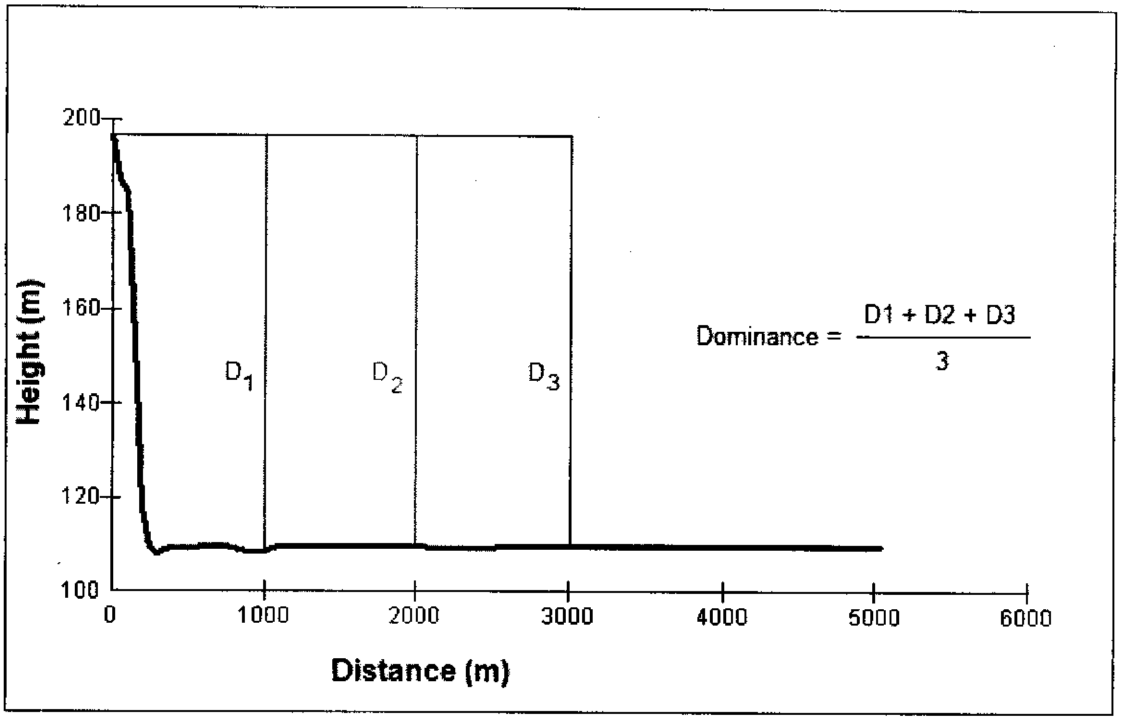
Table 1. Great horned owl presence-absence survey results for January – March 2007 in Southern Illinois.

Site Name	No. surveys	No. owls detected / survey night
Fountain Bluff	2	1
La Rue Pine Hills	2	1
Principia College	2	4.5
San Damiano	3	0
Tower Rock	3	0

Figure 1. Potential peregrine falcon habitat and reintroduction sites in southern Illinois.

Figure 2. Assessing cliff dominance of peregrine falcon reintroduction sites where D_i is the difference in elevation.


N
Potential Reintroduction Habitat
Suitable Habitat
0
25
50
100 Miles


200
180
160
140
120
100
Height (m)
0
1000
2000
3000
4000
5000
6000
Distance (m)
D_1
D_2
D_3
Dominance = $\frac{D1 + D2 + D3}{3}$