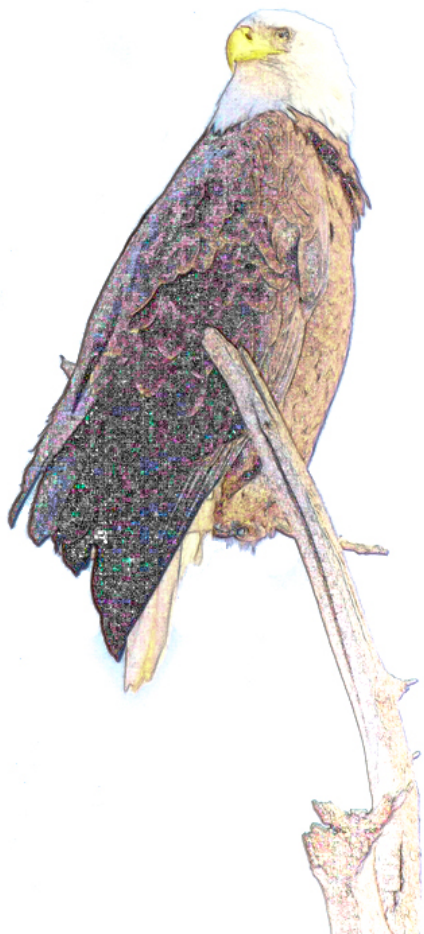


Bald and Golden Eagles

Population demographics and estimation of sustainable take in the United States, 2016 update



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April 26, 2016

Authors

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U.S. Fish and Wildlife Service
Division of Migratory Bird Management

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The information in this report is intended to aid in the development of regulations and inform eagle management decisions by the U.S. Fish and Wildlife Service. The recommendations and findings in the report do not constitute U.S. Fish and Wildlife Service policy, but they will be considered by the U.S. Fish and Wildlife Service as it sets eagle management policies. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Suggested Citation:

U.S. Fish and Wildlife Service. 2016. Bald and Golden Eagles: Population demographics and estimation of sustainable take in the United States, 2016 update. Division of Migratory Bird Management, Washington D.C., USA.



Acknowledgments

We gratefully acknowledge contributions of data from tagged golden eagles by Bryan Bedrosian, Pete Bloom, James Cain, Ross Crandall, Robert Domenech, Daniel Driscoll, Jamey Driscoll, Mark Fuller, Rick Gerhardt, Al Harmata, Todd Katzner, Robert Knight, Craig Koppie, Brian Latta, Mike Lockhart, Mark Martell, Carol McIntyre, Libby Mojica, Robert Murphy, Gary Roemer, Steve Slater, Jeff Smith, Dale Stahlecker, Brian Washburn, and Jim Watson. Andrew Dennhardt provided additional information on eastern U.S. golden eagle population estimates. Bruce Peterjohn and Danny Bystrak with the U.S. Geological Survey Bird Banding Lab helpfully provided banding data. The analyses and report greatly benefited from reviews and assistance provided by members of the Service's Eagle Technical Assessment Team, and we are especially grateful for a thorough review by Todd Katzner. Analyses of the bald eagle post-delisting survey data were improved by input from John Sauer and William Link, and population models and harvest analyses were improved by suggestions and input from Michael Runge, John Sauer, and Leslie New. Gabriela Chavarria (U.S. Fish and Wildlife Service, Science Applications) was instrumental in funding a significant portion of the on-going PTT-tagging study for golden eagles. We appreciated comments on an earlier draft by Leon Kolankiewicz, and Eliza Savage. Any errors or omissions are the responsibility of the senior author.



Executive Summary

In June 2014 the U.S. Fish and Wildlife Service (Service) announced its intent to consider several revisions to regulations at 50 CFR, part 22 that pertain to permits to take bald (*Haliaeetus leucocephalus*) and golden (*Aquila chrysaetos*) eagles. The Service is preparing a Programmatic Environmental Impact Statement (PEIS) to evaluate the potential effects of the revised regulations on eagle population status. The PEIS will analyze alternatives that include both conservative and liberal take rates for both species, consistent with the overall management objective of maintaining stable or increasing populations relative to estimated population levels in 2009. The liberal alternatives will use take rates estimated from the median values for relevant parameters (e.g., population size, growth rates), and the conservative alternatives will use the 20th quantile values of parameter estimates. The alternatives will also consider different configurations of eagle management units (EMUs): (1) the current EMUs, which are Bird Conservation Regions (BCRs) for golden eagles, and approximately Service regional boundaries for bald eagles based on nest densities; and (2) the four administrative migratory bird flyways (i.e., Atlantic, Mississippi, Central, and Pacific). To inform the evaluation of the PEIS alternatives, a subgroup of the Service's Eagle Technical Assessment Team compiled recent information on population size and trend of both species of eagle, generated estimates of recent survival and fecundity rates, and used these data in models to predict future population trends and the ability of each species to withstand additional mortality in the form of permitted take. This document summarizes the findings from those analyses.

The team (hereafter we) estimated population size for the bald eagle in the coterminous United States (U.S.) using a population model in conjunction with estimates of the number of occupied nesting territories (representing the number of breeding pairs) in 2009 from a comprehensive dual-frame aerial survey. That population size estimate combined with a previous estimate of population size for Alaska was 143,000 (20th quantile = 126,000) bald eagles for the entire U.S. in 2009. This represents an increase in population size since 2007 in the coterminous U.S. (the year the final rule for delisting under the Endangered Species Act was published). We attribute the difference to improved survey and estimation efforts, as well as increases in bald eagle numbers. Consistent with the population model, independent Breeding Bird Survey (BBS) indices indicated bald eagles are continuing to increase over much of the U.S. We used a potential biological removal model to estimate sustainable take rates and limits with the goal of maintaining at least the 2009 population level, and concluded that under the liberal alternative bald eagles over most of the country can support an annual take rate of 8% (20th quantile = 6% under the conservative alternative). The exceptions are the Southwestern U.S., where population growth potential is lower, and Alaska, where limited survey information led managers to select a lower management objective factor; there, the sustainable take rates are 4.5% (20th quantile = 3.8%) and less than 1%, respectively. Nationally, the annual bald eagle take limit with these rates would be approximately 6,300 eagles under the liberal alternative and 4,200 eagles under the conservative alternative.

We estimated population size for the golden eagle by first estimating a population size for the western coterminous U.S. using a composite model that integrated multi-year information from a late summer aerial transect survey over the interior western U.S. with information from the BBS. Population size for Alaska could not be estimated directly. Instead, we used results from mid-winter aerial transect surveys in 2014 and 2015 over the same area as the interior western U.S. summer transect survey to estimate the increase in population size between late summer and winter. The increase was used as a coarse estimate of the size of the overwintering migrant population. We allocated 24% of the winter increase to Alaska as a conservative population estimate, assuming migrants originated proportionately across western Canada and

Alaska. A population size estimate for eastern North America was available from the literature. Combining the western coterminous U.S., Alaska, and eastern U.S. estimates, total population size for the golden eagle in the U.S. (including Alaska) was approximately 39,000 (20th quantile = 34,000) in 2009 and 40,000 (20th quantile = 34,000) in 2014. The population trend estimate from the composite model was stable, but an estimate from a population model similar to that used for the bald eagle suggested the population in the coterminous western U.S. might be declining towards a lower equilibrium size. Thus, taking into account the uncertainty, the available data for golden eagles are somewhat equivocal, with count data suggesting a stable population but with demographic data forecasting a slight decline.

We used banding data obtained from the United States Geological Survey Bird Banding Lab from 1968–2014 to estimate contemporary age-specific survival rates. We also used a data set of unbiased cause-of-mortality information for a sample of 386 satellite-tagged golden eagles from 1997–2013 to estimate the effect of current levels of anthropogenic mortality on those survival rates. Anthropogenic factors were responsible for about 56% of satellite-tagged golden eagle mortality, but rates of anthropogenic mortality varied among age-classes, ranging from 34% for first-year eagles to 63% for adults. We estimated the maximum rate of population growth for the golden eagle in the U.S. in the absence of existing anthropogenic mortality was 10.9% (20th quantile = 9.7%). Sustainable take under these conditions is close to 2,000 individuals (20th quantile = 1,600). However, available information suggests ongoing levels of human-caused mortality likely exceed this value, perhaps considerably. Thus, the data from satellite tags lends further support to the suggestion from the demographic models that current survival rates may be leading to a decline in population size.

The Service also has a need to apply take values to nest disturbance and loss. We updated metrics for converting take via nest disturbance and nesting territory loss to debits from the EMU take limits for bald and golden eagles. The current policy is that for each instance of authorized take through disturbance in each year the nest is disturbed, the Service reduces EMU take limits by the median number of young that would have been expected to fledge from the disturbed territory. The updated median productivity values are 1.12 for the bald eagle (0.73 in the Southwest region only), and 0.54 for the golden eagle. By carrying forward the above debits from the EMU take limits for a period of years equal to the species or population-specific generation time (10 years for the bald eagle—12 years in the Southwest, and 11 years for the golden eagle), we also calculated a take value for nesting territory loss (i.e., the territory becomes permanently vacant).

In addition to setting EMU take limits, the Service has established local-area population (LAP) thresholds for permitted take when authorized take in a local area might have long-term negative consequences at that scale. The primary objective of LAP take limits is to minimize chances of extirpation of local breeding or wintering populations of eagles. The LAP take thresholds are cumulative, such that all ongoing Service-permitted take and any new take under consideration for a permit is taken into account. This take is in addition to any existing ongoing unpermitted take that is occurring in the LAP. As such, the LAP take analysis is a form of cumulative effects analysis for each eagle take permit. Unlike EMUs, the LAP area is unique to each prospective permit and is defined as the area of the permitted activity bounded by the 90th quantile of the natal dispersal distance for golden eagles (109 mi), and the median natal dispersal distance of females for bald eagles (86 mi). The Service has identified LAP take-rates of $\geq 1\%$ as being of concern, and rates of 5% being the maximum of what should be considered. We analyzed the effects of the 5% take threshold on LAPs for each species of eagle and showed that for bald eagles the additional take could result in a reduction of the equilibrium population size in the LAP area of 38%. For golden eagles, which currently appear to be at quasi-equilibrium, the 5% threshold could result in a decline of 80% to a new lower equilibrium. In both cases, extirpation of the local area population appeared unlikely under this policy.

When authorized take exceeds EMU take limits, Service policy is that take must be effectively offset by compensatory mitigation such that there is no net increase in mortality. Currently, the only offsetting mitigation measure the Service has enough information to confidently apply in this manner is retrofitting of power lines to reduce eagle electrocutions (although the Service does consider other offsetting mitigation

options on an experimental basis). Offsetting mitigation is mostly an issue affecting take authorization for golden eagles, as EMU take limits are set at zero requiring all authorized take to be offset. Based on the cause-specific mortality rates analyzed, we estimated that 500 (20th quantile = 280) golden eagles are electrocuted in the U.S. annually. This estimate provides an indication of the number of golden eagle deaths the Service can expect to offset through electrocution abatement until proven methods to reduce other forms of mortality are available.

The Service currently implements the eagle take permit program within the context of an adaptive management framework that requires regular updates of population size estimates. We offer a possible regime for conducting these surveys that balances information needs with costs and logistics, and that would allow updating of population size estimates every six years. Bald eagle surveys would be conducted in years three and six, and paired summer-winter golden eagle surveys in the first/second, and fourth/fifth years of each six-year period. Data collected should be used to re-assess population status, revise and update population size estimates by EMU, and to update—and if necessary modify—EMU or LAP take restrictions.

Table of Contents

Acknowledgments	iv
Executive Summary	v
Introduction	1
Eagle Management Objectives	1
Bald Eagle	3
Demographic Rates and Characteristics	3
Survival	3
Productivity	3
Matrix Population Model	3
Population Size	6
Number of Occupied Nesting Territories	6
Total Population Size	8
Population Trajectory	8
Management Unit Comparison	9
Golden Eagle	12
Demographic Rates and Characteristics	12
Survival	12
Causes of Mortality	13
Productivity	13
Matrix Population Model	15
Population Size	15
Western U.S.	15
Alaska	17
Eastern U.S.	18
Total U.S.	18
Population Trajectory	18
Management Unit Comparison	18
Effects of Take	22
Take Limits at the Scale of Eagle Management Units	22
Approach for Estimating Sustainable Take Limits	22
Sustainable Take for Bald Eagles	23
Sustainable Take for Golden Eagles	24
Metrics for Take as a Result of Nest Disturbance	25
Metrics for Take as a Result of Territory Loss	26
Take Limits at the Scale of the Local Eagle Population	27
The Role of Compensatory Mitigation	29

Population Monitoring	31
Literature Cited	32
Appendices	36
A1 Review of Eagle Productivity Data	36
Literature Review and Data Compilation	36
Bibliography: Bald Eagle Productivity Data	37
Bibliography: Golden Eagle Productivity Data	38
A2 Bald and Golden Eagle Productivity Meta-Analysis	40
Abstract	40
Introduction	40
Methods	41
Data	41
Model	41
Results	42
Discussion	43
A3 The 2009 National Bald Eagle Post-Delisting Survey and Estimation Results	58
Abstract	58
Survey Overview	58
Dual-frame Survey Design	58
Stratification for the 2009 Survey and Analysis	59
Methods	59
Sample Unit Selection and Sample Size	59
Surveys	60
Estimating Occupied Nests	61
List Estimation	61
Area Estimation: Correcting for Detection Probability With Multiple Observers	62
Area Estimation: New Occupied Nest Estimation	64
Area Estimation: Area-Only Estimation	64
Dual-Frame Estimation	65
List Coverage	65
Results and Discussion	65
List-Frame Estimates	65
Area-Frame Estimates	67
Dual-Frame Estimates	68
The Low-Density Strata	68
Area-Only Estimates	68
List Coverage	71
A4 Updated Golden Eagle Population Size Estimate in the Western U.S.	73
Summary	73

A5 USFWS Policy Update to the Use of Eagle Natal Dispersal Distances in Permitting Decisions	97
Background	97
Decision Point	98
Technical Issue	98
Inconsistent Terms Issue	100
Recommendations	100
Alternative 1a (ETAT-recommended alternative)	100
Alternative 1b	100
Alternative 2a	100
Alternative 2b	101
Decision	101

List of Tables

1	Candidate models for bald eagle survival rates and band recovery probabilities, 1995–2014.	4
2	Annual survival rate estimates for bald eagles, 1995–2014.	5
3	Estimated total bald eagle population size in 2009, sustainable harvest rates, and harvest limits by potential EMU.	8
4	Summary of bald eagle banding and recovery locations summarized by 2009 EMUs.	11
5	Summary of bald eagle banding and recovery locations summarized by Flyway.	11
6	Candidate models for golden eagle survival rates and band recovery probabilities, 1968–2014.	12
7	Annual survival rate estimates for golden eagles, 1968–2014.	13
8	Estimated total annual golden eagle mortality attributable to different causes.	14
9	Golden eagle annual survival rate estimates with and without anthropogenic mortality.	15
10	Estimated total golden eagle population size in 2014, sustainable harvest rates, and harvest limits by potential EMU.	19
11	Golden eagle banding locations and subsequent mortality recoveries by 2009 EMU.	20
12	Golden eagle banding locations and subsequent mortality recoveries summarized by Flyway.	21
13	Take associated with disturbance to nesting bald eagles and the loss of occupied nesting territories.	27
14	Take associated with disturbance to nesting golden eagles and the loss of occupied nesting territories.	28
A2-1	AIC values for models to explain variation in bald and golden eagle productivity.	44
A2-2	Regional prediction estimates of bald and golden eagle productivity.	45
A2-3	Productivity model random effect standard errors with 95% credible intervals.	45
A2-4	Bald eagle productivity model median random effects apart from region.	48
A2-5	Golden eagle productivity model median random effects apart from region.	49
A2-6	Bald eagle studies, simple productivity ratios, and final random effect model median estimates for each area and year combination.	52
A2-7	Golden eagle studies, simple productivity ratios, and final random effect model median estimates for each area and year combination.	56
A3-1	Geographic sampling areas for the 2009 post-delisting survey.	61
A3-2	Survey capture history summaries and estimated detection probabilities.	63
A3-3	List-frame estimates of total occupied bald eagle nests by strata.	66
A3-4	Estimates of new, occupied nests by area-frame strata.	67
A3-5	List-frame, area-frame, and dual-frame estimates of occupied nests by stratum.	68
A3-6	Number of recorded occupied nests for areas of low nest density.	69
A3-7	Estimates of occupied nests by strata based on area search plot data only.	70
A3-8	Total list-only, area-only, and dual-frame estimates of occupied nests for the high-density strata.	70
A3-9	Proportion of the total estimated nests included on nest lists by strata.	71
A4-1	Proportion of BCRs used to post-stratify BCR-specific golden eagle population size estimates to a Flyway scale.	74
A4-2	Population size estimates for golden eagles in the Central and Pacific Flyways, and the total western U.S., 1967–2014.	79
A4-3	Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014.	83

A5-1 Examples of bald and golden eagle natal dispersal distance criteria and implications for take benchmarks. 99

List of Figures

1	Eagle management unit configurations under consideration by the Service.	2
2	Diagram of the Lefkovich (stage-structured) population model for the bald eagle, and corresponding population projection matrix.	5
3	Example of the linear relationship used to incorporate density-dependent effects on fecundity into eagle population projection models.	7
4	Estimated number of occupied bald eagle nesting territories by 2009 bald eagle management units and the apparent change from 2007 to 2009.	7
5	Projected bald eagle population in the U.S. and the Southwest from 2009–2109.	10
6	Diagram of the stage-structured population model for the golden eagle, and corresponding population projection matrix.	16
7	Comparison of time series for golden eagles in the western U.S.	16
8	Population projection matrix trajectory for the golden eagle allowing for density-dependent effects on fecundity.	17
9	Harvest yield curve for the bald eagle in the U.S., excluding the Southwest region, under liberal and conservative alternatives.	24
10	Histogram of the difference between estimated current human-caused mortality of golden eagles and the estimated sustainable take rate at maximum sustainable yield.	25
11	Simulated effect of added take on golden eagle populations.	26
12	Effect on bald eagle local area populations (LAP) of a 5% additive harvest rate.	29
13	Effect on golden eagle local area populations (LAP) of a 5% additive harvest rate.	30
A2-1	Bald eagle productivity for the Southwest U.S. and the U.S. excluding the Southwest.	44
A2-2	Bald eagle productivity model random effects due to year given study.	46
A2-3	Bald eagle productivity model random effects due to study apart from regional and area differences.	47
A2-4	Golden eagle productivity for the U.S.	49
A2-5	Golden eagle productivity model random effects due to year given study.	50
A2-6	Golden eagle productivity model random effects due to the study random effects apart from regional differences.	51
A3-1	Survey strata for the 2009 bald eagle post-delisting survey.	60
A4-1	North American Bird Conservation Initiative bird conservation regions (BCRs).	75
A4-2	Updated golden eagle BCR-specific trends compared to those estimated by <i>Millsap et al. (2013)</i>	76
A4-3	Comparison of golden eagle trends in the Pacific and Central Flyways and the total trend for the entire western U.S.	77
A4-4	Comparison of Flyway-specific time series of golden eagle population size estimates from 1967–2014.	77
A4-5	Comparison of time series for golden eagles in the western U.S. based on data from 1967–2010 (<i>Millsap et al. 2013</i>) and updated data (1967–2014).	78
A4-6	Comparison between <i>Millsap et al. (2013)</i> and updated scaling factors used to adjust BBS indices to a population estimate for golden eagles.	78

Introduction

In June 2014 the U.S. Fish and Wildlife Service (Service) announced its intent to consider several revisions to regulations at 50 CFR, part 22 that pertain to permits to take bald (*Haliaeetus leucocephalus*) and golden (*Aquila chrysaetos*) eagles. As part of this process, the Service is preparing a Programmatic Environmental Impact Statement (PEIS) to assess potential effects of the revised regulations on bald and golden eagle populations. The PEIS will analyze alternatives that include both conservative and liberal take rates for both eagle species, consistent with the overall management objective of maintaining the potential for stable or increasing populations relative to 2009 estimates. The Service's Eagle Management Team (EMT) decided that for the liberal alternatives take rates would be estimated using the median values for relevant parameters (e.g., population size, growth rates), and that conservative alternatives would use the 20th quantiles of parameter estimates. The PEIS alternatives will also consider two different configurations of eagle management units (EMUs): 1) the current EMUs, which are Bird Conservation Regions (BCRs) for golden eagles, and management units based on nest densities for bald eagles (U.S. Fish and Wildlife Service 2009b); and 2) the four administrative migratory bird flyways (i.e., Atlantic, Mississippi, Central, and Pacific; U.S. Fish and Wildlife Service 2014, 2015).

The PEIS must consider the most current relevant information for both eagle species. A subteam of the Service's Eagle Technical Assessment Team (ETAT) began work in February 2015 assembling relevant scientific data and conducting analyses to support the PEIS. Much of this work has focused on gathering data to estimate sustainable take rates and take limits for both eagle species. The subteam (hereafter team, or we) compiled recent information on population size and trend of both species of eagle, generated estimates of recent survival and fecundity rates, and used these data in models to predict future population trends and the ability of each species to withstand additional mortality in the form of permitted take.

Herein, we describe approaches used in conducting technical analyses to inform the PEIS, then summarize the results with consideration of the Service's proposed management objectives. The trend and population status information are presented together under relevant subheadings for each species of eagle. Because the data available for bald and golden eagles differs, the subheadings and approaches used for the two species differ in some cases. The analysis of resilience to additional permitted take (harvest) by both species is covered in the final section of this document. Throughout, we present means or medians and 20th quantiles for parameter estimates that are used directly in the calculation of values under liberal and conservative PEIS alternatives, otherwise we present 95% confidence limits (or credible intervals when Bayesian methods are used) for estimates.

Eagle Management Objectives

In 2009, the Service established management objectives for bald and golden eagles as part of the Final Environmental Assessment on the Nonpurposeful Eagle Take Regulations (U.S. Fish and Wildlife Service 2009a,b). The management objectives at that time were to maintain stable or increasing populations of both species of eagle within a set of described EMUs, with 2009 as the baseline, for 100 years into the future. In 2009, EMUs for bald eagles were based on nest densities and approximated Service regions, and EMUs for golden eagles were BCRs in the western U.S. (Figure 1). The use of different management units for the two eagle species reflected biological differences and differences in data available.

The Service is proposing to retain the 2009 management objectives, but is considering revising the EMUs to follow the four administrative migratory bird flyways with some modifications (available data presented

later in this report suggest that breaking the Pacific Flyway into three separate EMUs may be warranted for the bald eagle, and combining the Mississippi and Atlantic Flyways may be warranted for the golden eagle). In addition, the Service is proposing to address uncertainty in estimates of population size and resiliency to increased permitted take by considering distribution medians for relevant parameters for liberal alternatives, and 20th quantiles for conservative alternatives. We use N_{2009} to refer to Service population objective(s) in formulas and in the text.

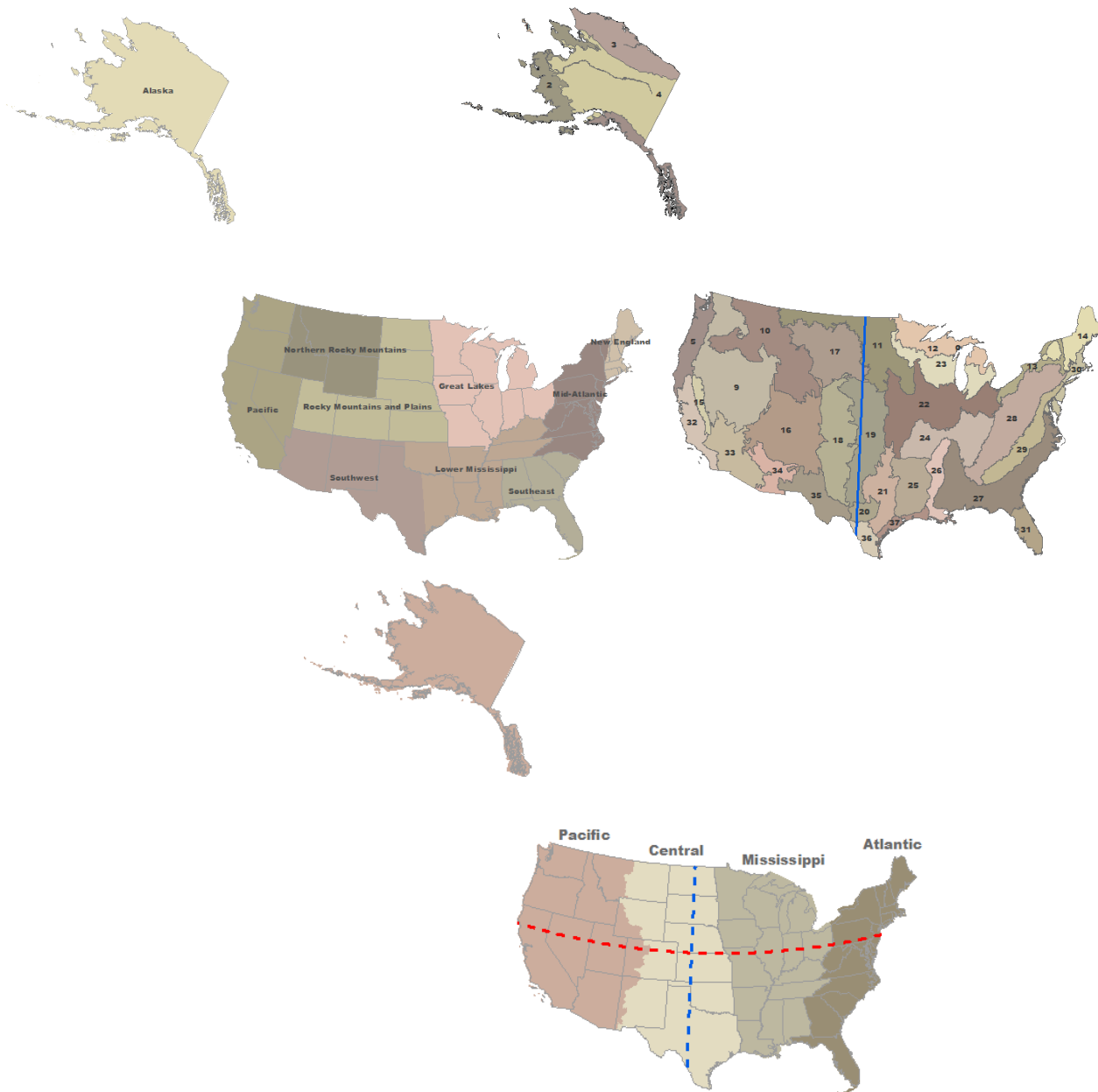


Figure 1. The 2009 eagle management units (EMUs) for bald eagles (top left) and golden eagles (top right, [U.S. Fish and Wildlife Service 2009b](#)), and the administrative flyways (bottom, [U.S. Fish and Wildlife Service 2014, 2015](#)), which are under consideration as alternative EMU configurations by the Service. The blue lines indicate 100°W longitude, and the red line indicates 40°N latitude.

Demographic Rates and Characteristics

Survival

We estimated bald eagle survival rates using banding data provided by the U.S. Geological Survey Bird Banding Laboratory (BBL). Given evidence of continued population growth (see [Population Trajectory](#)), we limited the survival analysis to 1995–2014. This data set included 14,805 banding records and 296 dead recoveries. We estimated annual survival rates using a dead-recovery model with the Seber parameterization in Program MARK ([Cooch and White 2014](#)). We evaluated a set of 10 candidate models that included age, year, geographic covariates on survival, and geographic variation in recovery probability (the probability that dead, marked individuals are reported). We then used the best-supported model (based on model-selection using an overdispersed and small-sample adjustment [QAIC_c] of the Akaike information criterion [[Burnham and Anderson 2002](#)]) from this set to evaluate a second set of six models that included linear and quadratic time trends ([Table 1](#)). The best-supported models from this analysis included two age-classes, first-year (hatching year [HY]) and older (after-hatching-year [AHY]), and two geographic areas—the Southwest region west of the 100th meridian and south of 40°N latitude (hereafter Southwest), and the rest of the U.S. We used the most parsimonious model within two QAIC_c units of the top model to estimate annual survival rates. For the bald eagle, the best supported model included a geographic effect on recovery probability, but a model with constant recovery probability that required estimates of fewer parameters had an only slightly greater QAIC value, so we used the latter model. We used the Markov chain Monte Carlo (MCMC) method in Program MARK so our final estimates were in the form of samples from the Bayesian posterior distributions of each parameter. Our estimates of annual survival were 66% for HY and 93% for AHY bald eagles in the Southwest, and 86% for HY and 91% for AHY bald eagles over the rest of the U.S. ([Table 2](#)).

Productivity

We conducted a thorough literature review and obtained estimates of bald eagle productivity from 17 study areas in the U.S. from 1995–2014 ([Appendix A1](#)). Productivity ranged from 0.52–2.29 young fledged per occupied nesting territory ([Table A2-6](#)). We then used a random-effects meta-analysis model and estimated predictive distributions for bald eagle productivity ([Appendix A2](#)). Productivity differed by region with lower productivity in the Southwest (median = 0.73, 95% credible interval = 0.40–1.36) than the rest of the U.S. (median = 1.12, 95% credible interval = 0.73–1.72).

Matrix Population Model

We used the demographic rates and variances described above to parameterize a post-breeding Lefkovich (stage-structured) matrix to model potential population growth and the stable-age distribution for the bald eagle ([Caswell 2001](#)). For population projections, our model included age categories 0–1, 1–2, 2–3, and > 3 years ([Figure 2](#)). The model allowed bald eagles > 3 years old to breed, but limited the number of total breeding opportunities to the estimated number of occupied nesting territories in 2009. This limitation reflects that availability of suitable nesting sites is a proximate limiting factor in many raptor populations ([Hunt 1998](#), [Millsap and Allen 2006](#)), and that younger individuals are less competitive than older individuals for breeding slots ([Turrin 2014](#), [Lien et al. 2015](#)). By using the estimated number of occupied bald eagle nesting territories in 2009, we conservatively capped projected population growth to levels consistent with known numbers of

Table 1. Candidate models evaluated to explain variation in bald eagle survival rates and band recovery probabilities, based on dead band recoveries from 1995–2014 analyzed using a Seber parameterization in Program MARK.

Models ^b	QAIC _c ^a	Δ QAIC _c	AIC _c Weights	Model Likelihood	Number of Parameters
S(age2_hy&ahy+zone2_SW) r(zone2_E&W)	3,555.41	0.00	0.32	1.00	6
S(age2_hy&ahy+zone2_SW) r(.)	3,556.64	1.23	0.17	0.54	5
S(age2_hy&ahy) r(.)	3,557.16	1.75	0.13	0.42	3
S(age3_hy&SA&ahy+zone2_SW) r(.)	3,557.58	2.17	0.11	0.34	4
S(age2_hy&ahy+zone2_SW) r(zone2_SW)	3,557.77	2.36	0.10	0.31	6
S(age4) r(.)	3,558.73	3.32	0.06	0.19	5
S(age3_hy&SA&ahy+zone2_SW) r(zone2_SW)	3,559.53	4.12	0.04	0.13	5
S(age2_hy&ahy+zone2_S&N) r(.)	3,559.83	4.42	0.04	0.11	5
S(age2_hy&ahy+zone2_E&W) r(.)	3,560.26	4.86	0.03	0.09	5
S(age+time+zone+age*time*zone) r(age+time+zone+age*time*zone)	5,979.11	2,423.70	0.00	0.00	1,248
Models with Time Trends					
S(zone2_SW) r(zone2_E&W)	3,556.64	0.00	1.00	1.00	5
S(zone2_SW+quadratic) r(.)	3,576.46	19.82	0.00	0.00	6
S(zone2_SW+quadratic) r(zone2_E&W)	3,576.93	20.29	0.00	0.00	7
S(zone2_SW+linear) r(.)	3,578.38	21.74	0.00	0.00	5
S(age_quadraticAHY + zone2_SW) r(zone2_E&W)	3,579.76	23.12	0.00	0.00	7
S(age+time+zone+age*time*zone) r(age+time+zone+age*time*zone)	4,051.09	494.45	0.00	0.00	312

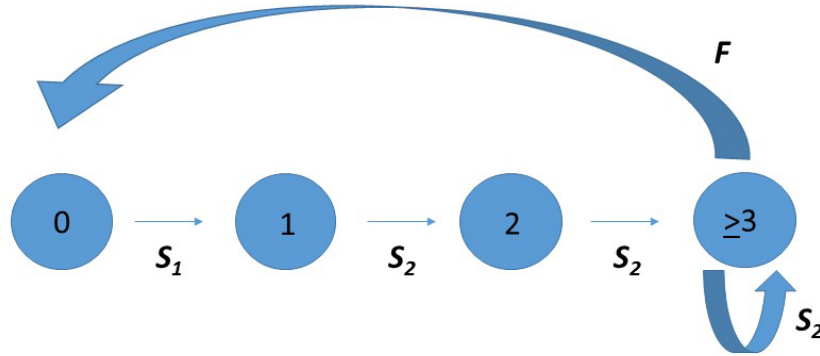
^a QAIC_c is an adjusted AIC_c value to account for overdispersion in the data, as measured by the value \hat{c} . $\hat{c} = 1.00$ in the absence of overdispersion. $\hat{c} = 1.23$ for the global model in this case.

^b Abbreviations are as follows: S= survival; r= recovery probability; age= age covariate, with the number of age-classes in the model specified by the numeral and the age-classes identified as hy= hatching-year, ah= after-hatching-year, SA= subadult (ages 2–4); zone= geographic covariate, with the numeral indicating the number of geographic zones in the model and the geographic zones identified as E=east of the 100th meridian, W= west of the 100th meridian, S=south of 40°N latitude, N= north of 40°N latitude, SW= south of 40°N latitude and west of the 100th meridian. Symbols are as follows: . = constant, +=additive effect, *= interactive effect. Linear refers to a model with a linear time effect over the full time-series, and quadratic refers to a model with a quadratic time effect over the full time-series.

Table 2. Annual survival rate estimates for bald eagles, 1995–2014, based on a dead-bird band recovery model with the Seber parameterization in Program MARK. Estimates are from the best-supported model in Table 1.

	Estimate	Lower 95% Credible Interval	Upper 95% Credible Interval
Annual Survival^a			
HY, U.S. excluding Southwest	0.86	0.80	0.90
AHY, U.S. excluding Southwest	0.91	0.86	0.94
HY, Southwest	0.66	0.31	0.87
AHY, Southwest	0.93	0.73	0.99
Recovery Probability	0.03	0.03	0.04

^a Abbreviations are: HY = hatching-year; AHY = after-hatching-year; Southwest = U.S. west of the 100th meridian and south of 40°N latitude.



$$\begin{pmatrix} N_{0,t+1} \\ N_{1,t+1} \\ N_{2,t+1} \\ N_{\geq 3,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0 & 0 & b(218, 23) \times n(0.59, 0.13) \\ b(158, 26) & 0 & 0 & 0 \\ 0 & b(218, 23) & 0 & 0 \\ 0 & 0 & b(218, 23) & b(218, 23) \end{pmatrix} \times \begin{pmatrix} N_{0,t} \\ N_{1,t} \\ N_{2,t} \\ N_{\geq 3,t} \end{pmatrix}$$

Figure 2. Diagram of the Lefkovich (stage-structured) population model for the bald eagle, and corresponding population projection matrix. On the diagram, stages correspond to year age-classes 0–1, 1–2, and 2–3. Stage > 3 includes all subsequent age classes. S denotes survival rates, where S_1 is the rate for first-year survival and S_2 is the rate for survival in all subsequent years (see Table 1 for justification for the 2-age class model and Table 2 for rates). F denotes per-individual fecundity, with reproduction contingent on breeding slots being available. The matrix shown is parameterized with values for bald eagles outside the Southwest U.S.; the model for the Southwest used demographic values specific to that region. Survival rates (rows 2–4) were sampled from beta distributions and fecundity (row 1) is shown as sampled from a normal distribution, but in the actual models we used the specific random-effects predictive distribution. To estimate the stable-age distribution, we used a similar model but with a non-reproductive 3–4 year-old stage, and a reproductive > 4 year-old stage (see text for details).

nesting territories (including the number of “floating” adults not associated with nesting territories) rather than assuming continued increases in breeding opportunities.

We used the stable-age distribution in combination with the direct estimates of numbers of occupied nesting territories in 2009 to estimate total bald eagle population size (see below). In using the stable-age distribution we made the implicit assumption that bald eagle populations are at equilibrium (Caswell 2001). We acknowledge this is unlikely given evidence of continued population growth, but believe this is a conservative assumption that likely leads us to underestimate total population size because in growing populations a larger percentage of the population is in the younger age classes (Lande et al. 2003). For this model we were not able to estimate the number of non-breeding individuals in the adult stage using the minimum number of known nesting territories. Rather, because bald eagles < 4 years old breed infrequently (Bowman et al. 1995, Turrin 2014), we revised our projection model to include a 3–4 year non-reproductive age class, and we added a fifth age class that included all adults ≥ 4 years of age. We assumed all individuals in the adults ≥ 4 years stage were associated with a nesting territory.

We ran 10,000 iterations of 100-year simulations for each population projection. Although we projected forward 100 years, we note that future predictions are only valid and relevant to the degree that environmental and biological conditions remain as they were over the time period when vital rates were measured. This critical assumption is less likely to be met the further into the future the projections go and should be kept in mind when evaluating this information. For each simulation, we sampled survival rates from beta distributions with shape parameters derived from the pertinent survival rates, and we randomly sampled the appropriate random-effects predictive distributions for fecundity values.

We incorporated a density-related response in fecundity by allowing mean productivity to increase as the proportion of adult floaters decreased. We implemented this as a linear increase in per-individual fecundity, ranging from the median of the estimated predictive distribution at current (2015) population levels assuming all territories are occupied, to the maximum annual observed rate (see ‘year:study area productivity’ combinations cited in Appendix A2 [Table A2-6]) when territory occupancy rates approached zero (Figure 3). There is strong evidence for density-related dampening of fecundity rates in increasing raptor populations (Kauffman et al. 2004, Bretagnolle et al. 2008, Fasce et al. 2011), as well as evidence for increasing per-individual fecundity rates in some decreasing populations (Whitfield et al. 2004b, 2007, Baldwin et al. 2012). This type of response in fecundity may be the result of interference competition (Kauffman et al. 2004, Bretagnolle et al. 2008) or increasing nesting habitat heterogeneity as lower-quality nesting territories are occupied at higher densities (Ferrer and Donazar 1996, Carrete et al. 2006).

In addition to using this basic model to estimate the population growth potential and stable age distribution of bald eagles, we used it to estimate demographic carrying capacity. Our estimates of carrying capacity assume nest site availability rather than food or other resources will be the proximate factor limiting growth, consistent with Moffat’s equilibrium theory (Hunt 1998).

Population Size

Number of Occupied Nesting Territories

We obtained estimates of the number of occupied bald eagle nesting territories in the coterminous U.S. from a dual-frame survey coordinated by the Service in 2009 (Tables A3-5, A3-8). We used the stratified survey estimates to calculate the number of occupied nesting territories for each EMU under consideration by redistributing the strata estimates to the EMUs according to the proportion of the total strata area within each EMU (Figure 4). The estimated number occupied nesting territories across EMUs in the coterminous U.S. was slightly lower than the dual-frame survey estimate of occupied nesting territories due to rounding and imperfect alignment of the survey strata and EMUs. We combined the EMU estimates for the coterminous U.S. with an existing estimate for Alaska from 2009 (U.S. Fish and Wildlife Service 2009b) and calculated

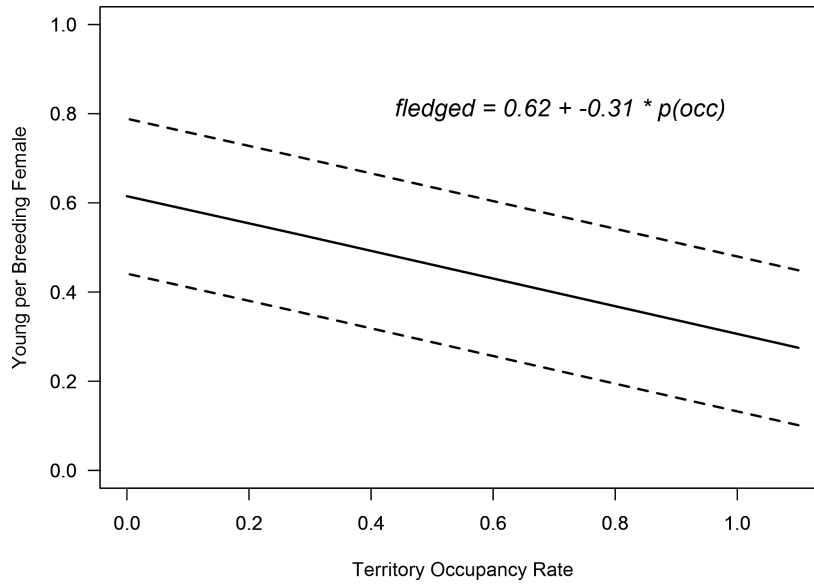
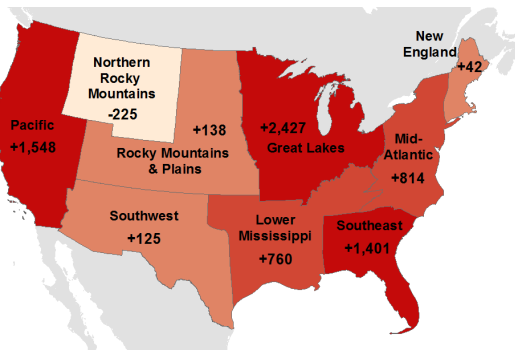


Figure 3. Example of the type of linear relationship used to incorporate density-dependent effects on fecundity into eagle population projection models. This example is for the golden eagle, where per-individual fecundity ranges from 0.275 young per year with 100% of nesting territories occupied (the mean of the predictive distribution derived from a literature review of contemporary fecundity rates), to 0.62 as the occupancy rate approaches zero (the maximum fecundity rate observed in any one year in the studies included in the literature review). Thus, fecundity in the model increases linearly according to the described linear model as populations fall and nesting territories go unoccupied.



EMU	2007	2009	2009 CI
Alaska	15,000	15,000	(12,471–17,529)
Great Lakes	3,452	5,879	(4,769–6,989)
Lower Mississippi	447	1,207	(753–1,661)
Mid-Atlantic	952	1,766	(1,373–2,159)
New England	603	645	(577–713)
Northern Rocky Mtns.	564	339	(0–751)
Pacific	1,039	2,587	(2,073–3,101)
Rocky Mtns. & Plains	200	338	(281–3950)
Southeast	1,210	2,611	(2,180–3,042)
Southwest	51	176	(119–233)
Total	23,518	30,548	(24,524–36,572)

Figure 4. A map (left) showing the apparent change in estimated occupied bald eagle nesting territories by bald eagle management unit (U.S. Fish and Wildlife Service 2009b) between the time of delisting in 2007 (data used were from 2007 or earlier) and 2009 (except Alaska, estimated from the post-delisting survey flown in 2009). The table (right) shows the number of estimated occupied nesting territories for both time periods and the 95% credible intervals for the 2009 estimates (the delisting numbers did not include explicit quantification of uncertainty). The different methods used to estimate nesting population size between the two intervals likely contribute to the differences shown here. Estimates for Alaska are based on limited local survey information (U.S. Fish and Wildlife Service 2009b).

nearly 30,600 (95% confidence interval = 24,500–36,600) occupied bald eagle nesting territories in the U.S. in 2009.

Total Population Size

The stable-age distribution from the five-stage matrix population model estimated that 40% of bald eagles in the Southwest and 43% elsewhere were ≥ 4 years old. Under the assumption that all of these individuals occupied nesting territories, we estimated the total population size for each region in the coterminous U.S. using the formula:

$$N_{Total} = \frac{N_{Occ.Terr} * 2}{p(Age \geq 4)},$$

where $N_{Occ.Terr}$ is the estimated number of occupied nesting territories (approximately 200 for the Southwest and 30,400 elsewhere in the U.S.), and $p(Age \geq 4)$ is the proportion of the population ≥ 4 years old. We estimated a median bald eagle population size of approximately 143,000 nationally (20th quantile= 126,000); estimates for each prospective bald eagle EMU are provided in Table 3.

Population Trajectory

The BBS index trend estimate for the bald eagle over the entire BBS coverage area for the period 1966–2012 is 5.3% (95% confidence interval = 4.1–6.6%), though trends for the area that includes Alaska

Table 3. Estimated total bald eagle population size in 2009 at the median (N) and 20th quantile (N_{20}) by potential eagle management unit (EMU). Estimated sustainable harvest rates (h) and harvest limits (H) are also presented with the median and 20th quantile for each EMU. Harvest rates and limits are constrained by a management objective factor (F_0) such that take is consistent with the objective of maintaining the potential for an equilibrium population size $\geq N$.

Management Unit	N	N_{20}	h	h_{20}	H	H_{20}	Source
Alaska ^a	70,544	62,935	0.007	0.008	494	494	USFWS (2009b)
Great Lakes	27,440	24,065	0.080	0.060	2,195	1,444	Post-Delisting Survey
Lower Mississippi	5,640	4,622	0.080	0.060	451	277	Post-Delisting Survey
Mid-Atlantic	8,244	7,201	0.080	0.060	660	432	Post-Delisting Survey
New England	3,017	2,729	0.080	0.060	241	164	Post-Delisting Survey
Northern Rocky Mountains	1,569	720	0.080	0.060	126	43	Post-Delisting Survey
Pacific	12,102	10,504	0.080	0.060	968	630	Post-Delisting Survey
Rocky Mountains and Plains	1,583	1,411	0.080	0.060	127	85	Post-Delisting Survey
Southeast	12,190	10,788	0.080	0.060	975	647	Post-Delisting Survey
Southwest	648	533	0.045	0.038	29	20	Post-Delisting Survey
Alaska-FW ^a	70,544	62,935	0.007	0.008	494	494	USFWS (2009b)
Atlantic Flyway	22,279	20,387	0.080	0.060	1,782	1,223	Post-Delisting Survey
Central Flyway	3,209	1,163	0.080	0.060	257	70	Post-Delisting Survey
Mississippi Flyway	31,706	27,334	0.080	0.060	2,537	1,640	Post-Delisting Survey
Pacific Flyway, South	447	391	0.045	0.038	20	15	Post-Delisting Survey
Pacific Flyway, North	14,792	13,296	0.080	0.060	1,183	798	Post-Delisting Survey
Total US	142,977	125,508			6,273	4,240	
Total US (excluding AK)	72,434	62,572			5,772	3,742	

^a Population size estimates for Alaska are approximations based on limited survey information. Because of this added uncertainty, the Service proposes to use a lower management objective factor for Alaska that results in a take limit comparable with that estimated in 2009 (U.S. Fish and Wildlife Service 2009b). The median value of h is used in every case.

have been closer to stable at 0.08% (95% confidence interval = -8.41–5.44%; Sauer et al. 2014). We observed increases in the number of occupied nesting territories and inferred population size between pre-2007 (the time of delisting under the Endangered Species Act; U.S. Fish and Wildlife Service 2009b) and 2009 in all current bald eagle EMUs except the Northern Rockies (Figure 4). Data used to support delisting were nest counts provided by the states, whereas the 2009 estimates are based on the dual-frame survey which corrected for detection and dealt with issues of sampling effort. The percent list coverage (the proportion of the total estimated nests, regardless of occupancy status, represented on state nest lists) for the dual-frame survey ranged from 48–100%, meaning some state nests lists were missing as many as 52% of the total estimated nest structures (Appendix A3). Thus, it is likely that some part of the difference in population size between the two time periods is a result of differences in survey and analysis methodologies. In particular, the decline indicated for the Northern Rockies EMU is not reflected in the BBS data, which shows a population change of 8.7% (95% confidence interval = 5.1–13.1%) from 2003–2013 (Sauer et al. 2014).

We estimated future bald eagle populations using the previously described population projection matrix model and the conservative assumption that the number of suitable bald eagle nesting territories will not increase above the 2009 estimate. Given limitations of the Alaska data and evidence from BBS indices that bald eagle populations are growing more slowly there, we did not model projections for Alaska and assumed that Alaska's bald eagle population will remain stable (though demographic rates suggested continued growth is possible). With these constraints, the model forecasts that the number of bald eagles in the U.S. outside the Southwest will continue to increase until populations reach equilibrium at about 228,000 (20th quantile = 197,000; Figure 5). The model predicts that bald eagles in the Southwest will continue to increase until reaching equilibrium at about 1,800 (20th quantile = 1,400). Again, these projections assume underlying demographic rates and other environmental factors remain unchanged through time and assume food and other factors will not become limiting. Additionally, these projections do not take into account forecasted changes in climate nor how such changes may affect bald eagle population vital rates, population size, food availability, or other factors.

Management Unit Comparison

To assess whether the EMU configurations under consideration (2009 EMUs and Flyway EMUs) differed in terms of capturing bald eagle movements across seasons and life stages, we used 1,021 band recovery records from the BBL data set from 1931–2014. We compared the frequency with which banded bald eagles were recovered within the same EMU as they were originally banded. Eagles were not banded systematically or randomly with respect to EMUs, however, if eagles frequently move distances or with directionality that is incongruent with the shape and size of a particular management unit configuration, we would expect a difference even in basic summary metrics. We found that 84% (range = 43–100%) of bald eagles were banded and later recovered in the same 2009 EMU (Table 4) compared to 94% (range = 67–96%) recovered in the same Flyway EMU (Table 5). In part, the difference may be a reflection of the larger geographic size of the Flyway EMUs. This is supported by the increase in the percentage of recoveries (98%) when adjacent 2009 EMUs are also considered. However, the consistently higher percentage of recoveries in the same Flyway EMU may also suggest a general association between bald eagle movements and Flyway EMUs.

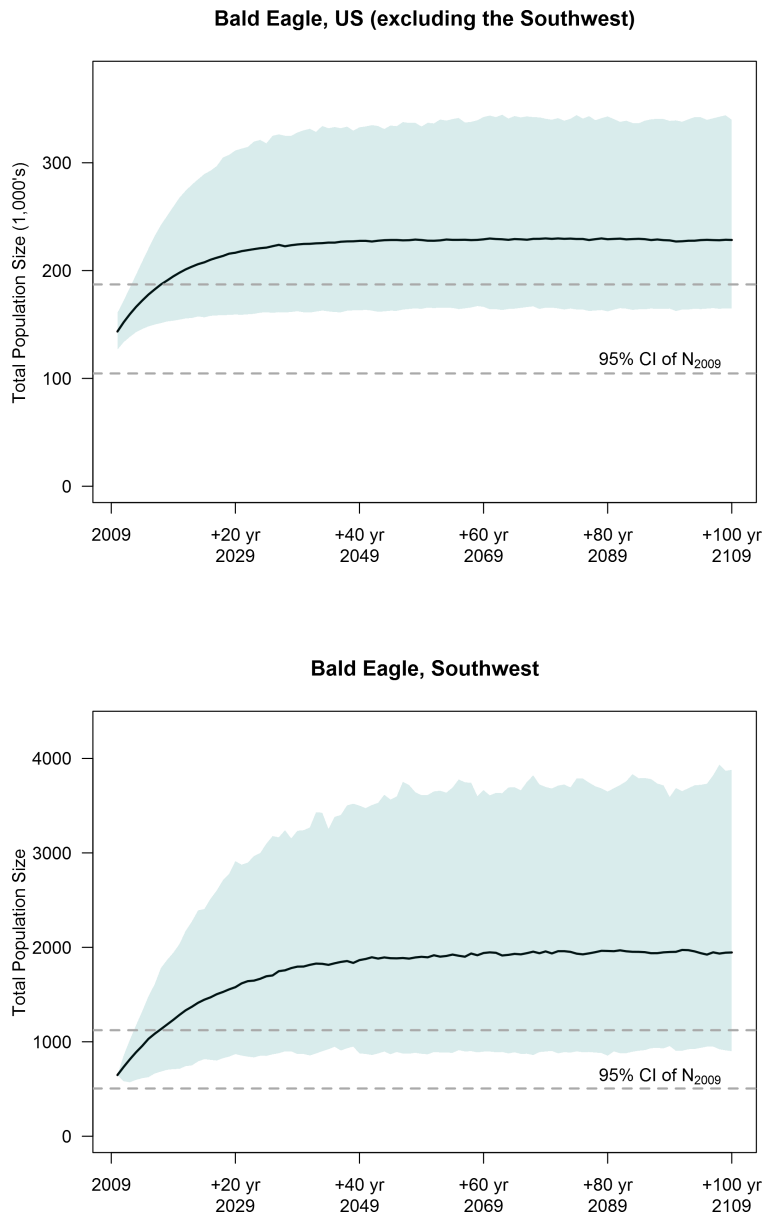


Figure 5. Projected bald eagle population in the United States excluding the Southwest (top) and in the Southwest (bottom) from 2009–2109 using a stage-structured population projection matrix and demographic rates derived from data over the period from 1995–2015. The blue shading indicates the 95% confidence intervals around the estimates (dark blue lines). The gray, dashed lines are the 95% confidence intervals for population size in 2009, which is the minimum bald eagle population size objective. Model projections assume demographic rates remain as estimated, and that the number of suitable nesting territories does not increase above 2009 levels, which accounts for the plateau in population size.

Table 4. Bald eagle banding locations by 2009 eagle management unit (EMU; [Figure 1](#)) and subsequent location of band recoveries (total recovered). Band recovery is classified as the percentage of the total recovered bands that were recovered in the same EMU, the same EMU or an adjacent EMU, or a non-adjacent EMU (other) relative to the banding location.

EMU Banded	Total Recovered	Recovered (%):		
		Same EMU	Same EMU or Adjacent EMU	Other EMU
Alaska	16	100	100	0
Great Lakes	510	90	99	1
Lower Mississippi	51	43	100	0
Mid-Atlantic	133	85	100	0
New England	140	86	99	1
Northern Rocky Mountains	46	70	96	4
Pacific	20	100	100	0
Rocky Mountains and Plains	36	58	100	0
Southeast	39	74	85	15
Southwest	30	73	87	13
Total	1,021	84	98	2

Table 5. Bald eagle banding locations by administrative Flyway ([Figure 1](#)) and subsequent location of band recoveries (total recovered). Band recovery is classified as the percentage of the total recovered bands that were recovered in the same Flyway, the same Flyway or an adjacent Flyway, or a non-adjacent Flyway (other) relative to the banding location.

EMU Banded	Total Recovered	Recovered (%)		
		Same EMU	Same or Adjacent EMU	Other EMU
Atlantic	319	94	100	0
Central	46	67	100	0
Mississippi	540	96	100	0
Pacific	116	93	98	2
Total	1,021	94	99.7	0.3

Demographic Rates and Characteristics

Survival

We estimated golden eagle survival rates using banding data from 1968–2014 provided by the BBL. A longer time series was necessary for reliable estimates than for the bald eagle, however, evidence suggests that golden eagle populations across the western U.S. have been largely stable over that longer period of time (Millsap et al. 2013), thus we assume survival rates have also been relatively stable. The data set included 10,627 banding records and 565 dead recoveries. As with bald eagles, we estimated annual survival rates using a dead-recovery model with the Seber parameterization in Program MARK (Cooch and White 2014). We evaluated a set of 10 candidate models that included age, time, linear and quadratic time trends in survival, and—for all but the global model—a constant recovery probability (Table 6). The best-supported model included four age-classes: <1 year (HY), second-year (SY), third-year (TY), and after-third-year (ATY). We used this model and the approach described previously for bald eagles to estimate annual survival of 70%, 77%, 84%, 87% for the respective golden eagle age classes (Table 7).

Table 6. Candidate models evaluated to explain variation in golden eagle survival rates and band recovery probabilities, based on dead band recoveries from 1968–2014 analyzed using a Seber parameterization in Program MARK.

Models ^b	QAIC _c ^a	Δ QAIC _c	AIC _c Weights	Model Likelihood	Number of Parameters
S(age4) r(.)	7,285.61	0.00	0.81	1.00	5
S(age3_hy&sa&aty) r(.)	7,288.53	2.93	0.19	0.23	4
S(age2 + hy&ahy) r(.)	7,299.89	14.29	0.00	0.00	3
S(age4 + quadratic) r(.)	7,303.94	18.33	0.00	0.00	7
S(age4 + linear_age2_hy) r(.)	7,308.11	22.50	0.00	0.00	6
S(age4 + linear) r(.)	7,312.46	26.85	0.00	0.00	6
S(age4 + quadratic_age2_aty) r(.)	7,328.14	42.53	0.00	0.00	7
S(age4 + linear_age2_ahy) r(.)	7,328.59	42.98	0.00	0.00	6
S(.) r(.)	7,345.03	59.42	0.00	0.00	2
S(age4 + time + age4*time) r(age4 + time + age4*time)	8,432.31	1, 146.70	0.00	0.00	728

^a QAIC_c is an adjusted AIC_c value to account for overdispersion in the data, as measured by the value \hat{c} . $\hat{c} = 1.00$ in the absence of overdispersion. $\hat{c} = 1.02$ for the global model in this case.

^b Abbreviations are as follows: S = survival; r = recovery probability; age = age covariate, with the number of age-classes in the model specified by the numeral and the age-classes identified as hy = hatching-year, ahy = after-hatching-year, SA = subadult (ages two through four). Symbols are as follows: . = constant, + = additive effect, * = interactive effect. Linear refers to a model with a linear time effect over the full time-series, and quadratic refers to a model with a quadratic time effect over the full time-series.

Table 7. Annual survival rate estimates for golden eagles, 1968–2014, based on a dead-bird band recovery model with the Seber parameterization in Program MARK. Estimates are from the best-supported model in [Table 6](#).

	Estimate	Lower 95% Credible Interval	Upper 95% Credible Interval
Annual Survival ^a			
HY	0.70	0.66	0.74
SY	0.77	0.73	0.81
TY	0.84	0.79	0.88
ATY	0.87	0.84	0.89
Recovery Probability	0.06	0.06	0.07

^a Abbreviations are: HY = hatching-year; SY = second year; TY = third-year, ATY = after-third-year.

Causes of Mortality

For golden eagles, the Service had access to a database of information (see [Acknowledgments](#)) on 386 satellite-tagged golden eagles over the period 1997–2013. As of 2013, 139 of those birds had died and were recovered; cause-of-death was known for 97 eagles. Radio- and satellite-tagged raptors are an important source of unbiased information on causes of death compared to bands, for which recovery probability varies by the type of death (e.g., raptors struck by vehicles are more likely to be encountered than raptors that die of starvation [[Kenward et al. 1993](#)]). The Service was particularly interested in the relative extent of human-caused (anthropogenic) mortality, in that this mortality is generally considered additive to natural mortality rates (but see [Chevallier et al. 2015](#)). Using the 139 eagles that had died and were recovered, we computed estimates of the overall proportion of annual mortality that was attributable to different factors using a binomial generalized linear model (GLM) in a Bayesian framework; this approach allowed us to include and account for the 42 eagles that died from unknown causes. Then, we used posterior distributions of the rates of cause-specific mortality, age-specific survival rates, and population size (partitioned by age class according to the stable-age distribution) to derive an estimate of the number of golden eagle mortalities annually by cause of death for each golden eagle age class for the total population ([Table 8](#)).

The relative importance of anthropogenic mortality increased with age, with 63% of adult golden eagle mortality caused by humans compared to 34% for HY golden eagles ([Table 9](#)). Moreover, annual golden eagle survival rates would be approximately 10% higher without human-caused mortality, assuming the reduction in anthropogenic mortality is additive, which may not be entirely the case, particularly for juvenile golden eagles ([Chevallier et al. 2015](#)).

Productivity

As with bald eagles, we reviewed the literature and obtained estimates of golden eagle productivity from 1995–2014. We included data from 12 study areas in the U.S. ([Appendix A1](#)) and used the same meta-analysis framework as for bald eagles to characterize the distributions of golden eagle productivity ([Appendix A2](#)). The best model for predicting productivity was a random-effects model with overdispersion ([Table A2-1](#)) that estimated median productivity for the continental U.S. ([Table A2-2](#)) as 0.54 (95% credible intervals = 0.40–0.75) young fledged per occupied nesting territory. Model selection did not support use of region-specific productivity values, though this may in part be to the limited productivity data available.

Table 8. Causes of death for satellite-tagged golden eagles in North America, 1997–2013, extrapolated to estimate total annual golden eagle mortality attributable to different causes. Extrapolations are based on estimated cause-specific proportions from a Bayesian binomial generalized linear model, and also account for uncertainty in annual mortality rates and in age-class population sizes. The extrapolation assumes that this sample of satellite-tagged deaths are representative of deaths overall, and was computed using the stable age distribution and age-specific survival rates described in the text. Age classes are age <1 year (HY), 1–3 years (Subadult), and >3 years (ATY).

Factor	Type	Observed Deaths			Total Deaths Projected per Year (95% Credible Interval)			
		HY	Subadult	ATY	HY	Subadult	ATY	Total
Shot	Anthropogenic	6		5	141 (55–292)	7 (0–153)	777 (280–1,600)	926 (336–2,046)
Electrocution	Anthropogenic	8	2	1	193 (88–364)	180 (27–526)	131 (9–604)	504 (124–1,494)
Poisoning	Anthropogenic	1	4	4	21 (2–100)	393 (126–806)	611 (188–1,360)	1,025 (316–2,266)
Collision	Anthropogenic	4		3	93 (27–218)	7 (0–154)	444 (106–1,137)	545 (133–1,509)
Trap	Anthropogenic	1	1	1	21 (1–98)	78 (5–362)	132 (9–611)	231 (15–1,071)
Lead Toxicosis	Anthropogenic	1		1	21 (2–97)	7 (0–160)	131 (9–609)	160 (10–867)
Starvation/disease	Natural	35	2	2	876 (605–1,205)	177 (28–538)	282 (48–883)	1,334 (681–2,626)
Injury	Natural	3	1	2	68 (17–180)	79 (5–370)	280 (44–888)	427 (66–1,437)
Fighting	Natural		1	4	3 (0–47)	78 (5–363)	609 (188–1,377)	690 (193–1,787)
Predation	Natural	2			44 (7–142)	8 (0–152)	17 (0–277)	69 (8–571)
Drowning	Natural	1	1		21 (2–97)	80 (5–369)	18 (0–281)	119 (6–747)
Total		62	12	23	1,503 (806–2,840)	1,093 (200–3,954)	3,432 (882–9,628)	6,029 (1,888–16,422)

Table 9. Golden eagle annual survival rate estimates with and without anthropogenic mortality. The proportion of mortality caused by humans was estimated from a sample of satellite-tagged golden eagles that died (see Table 8), and results presented here account for uncertainty in the proportions of cause-specific mortality, survival rates, and population size. Base survival rates were estimated from dead band recoveries using a Seber parameterization in Program MARK.

	Age Class		
	First-year	Subadult	Adult
Cause-of Death			
Anthropogenic	0.34 (0.23–0.46)	0.57 (0.32–0.81)	0.63 (0.44–0.80)
Natural	0.66 (0.54–0.77)	0.43 (0.19–0.68)	0.37 (0.20–0.56)
Survival Rate			
Only natural mortality	0.80 (0.76–0.85)	0.92 (0.86–0.96)	0.93 (0.89–0.96)
All mortality	0.70 (0.66–0.74)	0.80 (0.77–0.83)	0.87 (0.84–0.89)

Matrix Population Model

We used the above demographic rates and variances to parameterize a post-breeding Lefkovich matrix model for the golden eagle (Figure 6). We followed the same approach as for the bald eagle population projection model. As discussed above (see Bald Eagle: Matrix Population Model), we assumed no growth in the number of suitable nesting territories above 2009 levels. As for the bald eagle, we incorporated density-dependent effects on fecundity (Figure 3) such that per-individual fecundity ranged from approximately 0.28 young per year (the mean of the predictive distribution for productivity from the meta-analysis) with 100% of nesting territories occupied and an estimated floater to breeder ratio of 1.13:1 (the estimated condition in 2015), to 0.62 (the maximum fecundity rate observed in any one year for the studies included) as the territory occupancy rate approaches zero. Thus, fecundity in the model increases linearly as populations fall and nesting territories go unoccupied.

We used this model to estimate the stable age distribution of golden eagles and to project future population trajectory, assuming environmental conditions remain as they were over the time these data were collected, and that nest site availability and survival rates and not food will be the proximate factors limiting growth (Figure 7). For golden eagles we also have direct estimates of current and historical population size (e.g., Millsap et al. 2013), and we were able to project future population trajectory based on empirical trend estimates (Figure 8; Appendix A4).

Population Size

Western U.S.

Since 2006, the Service has funded an annual late-summer aerial transect survey to estimate golden eagle population size over four BCRs in the interior western U.S. that account for about 80% of the western U.S. golden eagle population (Nielson et al. 2014). Recently, Millsap et al. (2013) combined these data with BBS indices in a hierarchical model to produce a composite estimate of golden eagle population size and trend for the entire coterminous U.S. west of the 100th meridian over the years 1967–2010. We updated the Millsap et al. (2013) composite model estimates of golden eagle population size and trend through 2014 for this analysis (Appendix A4). The updated summer population size estimates do not differ substantially from those reported by Millsap et al. (2013), and indicate a late summer population averaging 31,000 (20th quantile = 29,000) individuals over the most recent decade (Figure 7). The updated composite model estimated the

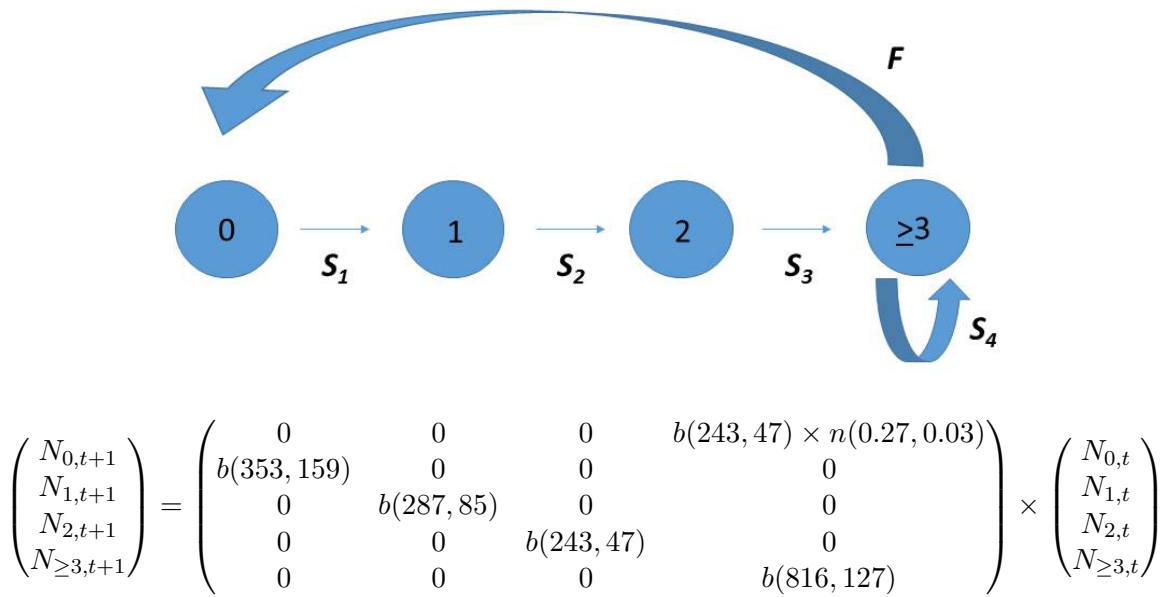


Figure 6. Diagram of the stage-structured population model for the golden eagle, and corresponding population projection matrix. See Figure 2 and text for a description of the model framework and vital rates. Fecundity, F (row 1) is shown as sampled from a normal distribution, but in the actual models we used the specific random-effects predictive distribution.

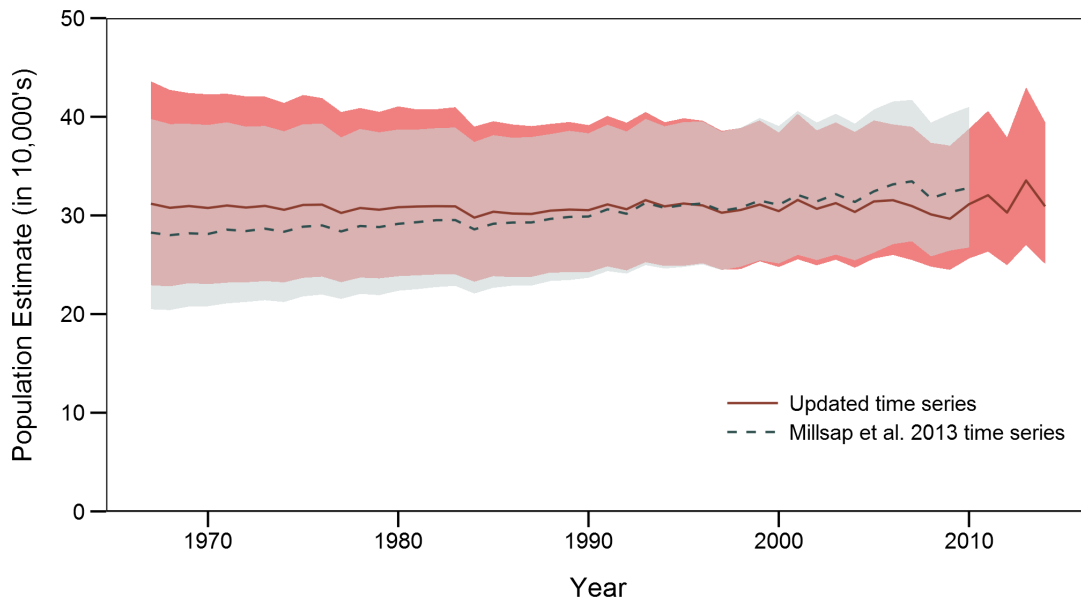


Figure 7. Comparison of time series for golden eagles in the western U.S. based on data from 1967–2010 (dashed line with blue shading, Millsap et al. 2013) and updated data for the period 1967–2014 (red line with red shading, Appendix A4). The lines are mean population estimates and colored shading represents the 95% credible intervals (CIs), with the 1967–2014 time series CIs shaded red, and the 1967–2010 time series CIs shaded blue. Note the large amount of overlap between the CIs.

Golden Eagle

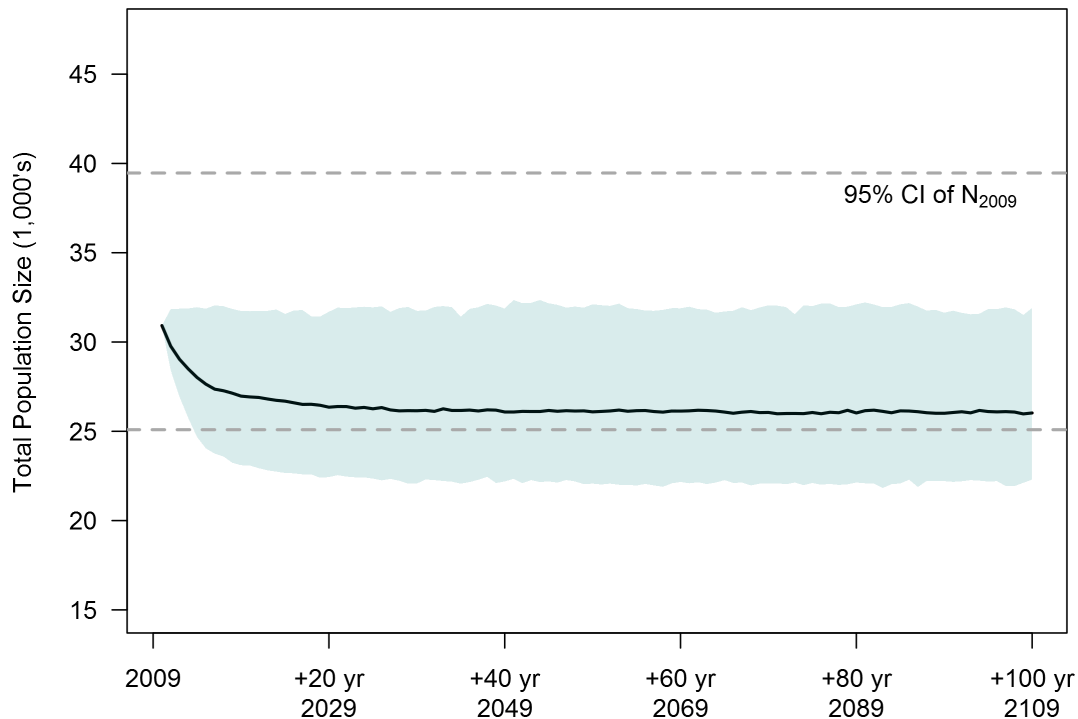


Figure 8. Population projection matrix trajectory for the golden eagle after model was adapted to allow for density-dependent effects on fecundity, as described in the text. The blue area represents the upper and lower 95% confidence limits, and the solid line is the median. The dashed lines represent the 95% credible intervals for the 2009 population estimate, which is the population objective.

total coterminous western U.S. population as 30,000 (20th quantile = 27,000) for 2009 (Figure A4-2). We combined the 2009 and 2014 western U.S. composite estimates with contemporary estimates of the eastern U.S. and Alaska golden eagle populations to calculate the population goal (N_{2009}) and harvest limits, respectively (see below).

Alaska

In 2014 and 2015, the Service funded aerial transect surveys over the same four-BCR area of the interior west in January to estimate mid-winter population size (Nielson and McManus 2014, Nielson et al. 2015). Golden eagles from natal areas above 60°N latitude are usually migratory (McIntyre et al. 2008), as are many individuals from the subarctic regions of Canada and Alaska (Kochert et al. 2002). Thus, the mid-winter population in the coterminous U.S. includes resident birds that remain in the coterminous U.S. year-round and migrants that occur at more northern latitudes in the summer, but migrate into the coterminous U.S. for the winter. Increases in counts from late summer to mid-winter likely provide a lower bound on the size of the northern migratory population of western golden eagles. The survey estimated increases in the number of golden eagles between late summer (August/September) and mid-winter (January) of 4,000 (95% credible interval = 3,800–4,100) in 2013–2014, and 17,000 (95% credible interval = 14,900–20,200) in 2014–2015. This mid-winter survey has not been conducted frequently enough to evaluate the meaning and significance of the annual variability in the change in numbers of eagles between late-summer and winter. However, these are the first data that allow approximation of the size of the high-latitude migratory golden eagle population

in western North America. Assuming the presumed northern migrant golden eagles are originating from natal areas in Canada (west of the 100th meridian) and Alaska in proportion to the relative area of those regions (76% Canada, 24% Alaska), then in 2013–2014 and 2014–2015 around 1,000–4,000 mid-winter migrants originated from Alaska. We used the larger estimate as our population size for Alaska for the liberal PEIS alternatives, and the midpoint (2,500) as the population estimate for the conservative PEIS alternatives. This assumes that all golden eagles in Alaska in the late summer are wintering in the coterminous U.S. In comparison, in 2009, the Service coarsely estimated the size of the Alaskan golden eagle population at 2,400 individuals (U.S. Fish and Wildlife Service 2009b).

Eastern U.S.

Golden eagles occur frequently in eastern North America, primarily as winter migrants from breeding and natal areas in eastern Canada (Morneau et al. 2015). Recently, the size of this population has been estimated at 5,000 (20th quantile = 4,000; Dennhardt et al. 2015), which corresponds with what is known about the number of occupied nesting territories in the breeding range of this population in eastern Canada (Morneau et al. 2015).

Total U.S.

We pooled western U.S., Alaska, and eastern U.S. population estimates to develop a total estimate of golden eagle population size for the U.S. in 2014 for the purpose of computing contemporary harvest limits (Table 10), presented in the Effects of Take section below. We used the 2009 estimate for the coterminous western U.S. and contemporary estimates for Alaska and the eastern U.S. as our population goal for the golden eagle (39,000, 20th quantile = 34,000).

Population Trajectory

The updated summer golden eagle population trend for the coterminous western U.S. from the composite model did not differ substantially from the trend reported by Millsap et al. (2013), with an annual rate-of-change of 1.0 (95% credible interval = 0.99–1.01) over the most recent decade (Figure 7). The annual rate-of-change from the demographic (population projection) model, however, averaged 0.998 (95% confidence interval 0.997–0.999), and suggested that golden eagles in the coterminous western U.S. might be gradually declining toward a new, lower equilibrium of about 26,000 individuals (Figure 8). Confidence limits for the demographic model projection broadly overlapped the credible interval for the composite model projection, so the results are generally consistent despite their differing ramifications. As noted previously, the validity of future predictions under both models are dependent on continuation of the biological and ecological conditions under which the vital rates were estimated.

Management Unit Comparison

To compare the different EMU configurations under consideration, we used 683 golden eagle band recovery records from the BBL data set from 1926–2014. As with the similar data set for bald eagles, we compared the frequency with which banded golden eagles were recovered within the same EMU as they were originally banded (640 of the 683 banded were recovered within the U.S.). We found that 73% (range = 0–86%) of golden eagles were banded and recovered within the same 2009 EMU (Table 11), whereas 84% (range = 50–87%) were in the same Flyway EMU (Table 12). Again, as with bald eagles, golden eagles were not banded systematically or randomly with respect to EMUs.

Table 10. Estimated total golden eagle population size in 2014 at the median (N) and 20th quantile (N_{20}) by potential eagle management unit (EMU). Estimated sustainable harvest rates (h) and harvest limits (H) are also presented for each quantile for each EMU. Harvest rates and limits are constrained by a management objective factor (F_0 , see text) such that take is consistent with the objective of maintaining the potential for an equilibrium population size greater than or equal to N_{2009} , 29,659, the population objective for the coterminous western U.S.

Management Unit	N	N_{20}	h	h_{20}	H	H_{20}	Source
Alaska	4,091	2,544	≈ 0	≈ 0	0	0	Nielson et al. 2014, 2015
Eastern	5,122	4,002	≈ 0	≈ 0	0	0	Dennhardt et al. 2015
BCR 5	189	114	≈ 0	≈ 0	0	0	Attachment 4
BCR 9	6,596	5,682	≈ 0	≈ 0	0	0	Attachment 4
BCR 10	5,675	4,851	≈ 0	≈ 0	0	0	Attachment 4
BCR 11	836	519	≈ 0	≈ 0	0	0	Attachment 4
BCR 15	72	38	≈ 0	≈ 0	0	0	Attachment 4
BCR 16	4,258	3,585	≈ 0	≈ 0	0	0	Attachment 4
BCR 17	9,837	8,091	≈ 0	≈ 0	0	0	Attachment 4
BCR 18	1,459	1,091	≈ 0	≈ 0	0	0	Attachment 4
BCR 32	718	549	≈ 0	≈ 0	0	0	Attachment 4
BCR 33	418	247	≈ 0	≈ 0	0	0	Attachment 4
BCR 34	411	229	≈ 0	≈ 0	0	0	Attachment 4
BCR 35	786	528	≈ 0	≈ 0	0	0	Attachment 4
Atlantic/Mississippi Flyways	5,122	4,002	≈ 0	≈ 0	0	0	Dennhardt et al. 2015
Central Flyway	15,327	13,210	≈ 0	≈ 0	0	0	Attachment 4
Pacific Flyway	15,927	14,437	≈ 0	≈ 0	0	0	Attachment 4
Total (US west)	31,254	30,191			0	0	
Total (Contiguous US and Alaska)	40,467	34,193			0	0	

Table 11. Golden eagle banding locations and subsequent location of mortality recoveries summarized by 2009 eagle management units (EMU), which are Bird Conservation Regions (BCRs) for the golden eagle (Figure 1). BCRs are referred to here by their BCR number. Band recovery is classified as the percentage of the total recovered bands that were recovered in the same BCR, the same BCR or an adjacent BCR, or a non-adjacent BCR (other) relative to the banding location. BCRs that did not have any banded eagles subsequently recovered were omitted.

BCR Banded	Total Recovered	Recovered (%)		
		Same BCR	Same or Adjacent BCR	Other BCR
3	5	0	0	100
4	10	30	30	70
5	3	0	100	0
6	3	0	67	33
8	3	0	67	33
9	245	86	98	2
10	53	64	94	6
11	82	78	91	9
12	2	50	50	50
13	4	0	75	25
14	1	0	0	100
16	52	65	100	0
17	23	65	100	0
18	52	77	96	4
19	13	54	92	8
20	1	0	100	0
22	3	67	100	0
23	4	25	100	0
24	4	25	75	25
28	21	76	90	10
29	3	0	67	33
30	1	0	100	0
32	40	85	93	8
33	5	60	100	0
34	4	50	100	0
36	3	0	0	100
Total	640	73	93	7

Table 12. Golden eagle banding locations and subsequent mortality recoveries summarized by administrative Flyway (Figure 1). Band recovery is classified as the percentage of the total recovered bands that were recovered in the same Flyway, the same Flyway or an adjacent Flyway, or a non-adjacent Flyway (other) relative to the banding location.

EMU Banded	Total Recovered	Recovered (%)		
		Same EMU	Same EMU or Adjacent EMU	Other EMU
Atlantic	23	74	91	9
Central	135	82	90	10
Mississippi	14	50	86	14
Pacific	408	87	90	10
Total	580	84	90	10

Effects of Take

The Service currently manages eagle take at two geographic scales, regional EMUs and the ‘local-area population’ (LAP). As noted previously, for the PEIS the Service is considering two alternative EMU configurations for the regional management scale—the EMUs established in 2009 and the four administrative flyways, which may better represent geographic use across seasons. Unlike EMUs, the LAP is unique to each prospective permit and is defined as the area of the permitted activity bounded by the 90th quantile of the natal dispersal distance for golden eagles, 109 mi, and the median female natal dispersal distance for bald eagles, 86 mi (Millsap et al. 2014). These values were adopted by the EMT based on recommendations by ETAT (Appendix A5) to update the values discussed in the Eagle Conservation Plan Guidance (ECPG) following updated analyses published by Millsap et al. (2014).

Eagle take at the EMU-scale is governed by a take rate that is compatible with maintaining an equilibrium population size equal to or greater than the population objective. Take limits at the LAP-scale, on the other hand, apply only to take permitted or authorized by the Service and, while they allow for local population declines under some situations, they are intended to prevent local extirpation of eagles—both breeding and non-breeding. The Service has acknowledged that some, perhaps even most, eagles taken at a permitted project will originate from natal areas outside the LAP. However, given fidelity to migration corridors and wintering areas by both bald and golden eagles (McIntyre et al. 2008, Mojica et al. 2008), limiting take at the LAP-scale has conservation benefits—which likely accrue to more than just eagles breeding within the LAP.

With take limits at both the EMU- and LAP-scales, across an EMU we would expect a landscape with some areas in proximity to permitted projects with comparatively high levels of authorized anthropogenic mortality, but offset by other areas where authorized anthropogenic take is low, averaging to a maximum across the EMU equal to or less than the EMU take limit. In cases where take exceeds the EMU take limit, all excessive take must be offset by mitigation that will commensurately reduce ongoing mortality from other sources, such that there is no authorized increase in net mortality (U.S. Fish and Wildlife Service 2009a).

Take Limits at the Scale of Eagle Management Units

Approach for Estimating Sustainable Take Limits

We used a potential biological removal (PBR) model to estimate sustainable lethal take rates for both species of eagle (Williams et al. 2002, Dillingham and Fletcher 2008, Runge et al. 2009). The PBR model produces an estimate of the sustainable harvest rate (h , hereafter take rate) using the formula:

$$h = \frac{r_{max}}{2} F_0,$$

where r_{max} = the maximum rate of population growth, sampled from the uncertainty distribution; and F_0 = a management objective factor, ranging from 0 (no harvest) to 2 (harvest rate = r_{max}), where a value of $F_0 = 1$ is the maximum sustainable yield (MSY) and a value of 0.5 is 1/2 MSY. MSY occurs at a population size of 1/2 carrying capacity.

The maximum rate of population growth (r_{max}) is rarely observed in nature, and no published estimates exist for bald or golden eagles or any closely related eagle species. Under these circumstances r_{max} is typically estimated from available demographic data using one of several approaches; we tested two such

approaches. First, following [Runge et al. \(2009\)](#), we estimated r_{max} using Slade’s equation ([Slade et al. 1998](#)):

$$1 = p\lambda^{-1} + l_{\alpha}b\lambda^{-\alpha} - \lambda_{\alpha}bp^{(\varpi-\alpha+1)}\lambda^{-\varpi+1},$$

where p = the annual adult survival rate from the Seber dead-recovery models, sampled as a uniform distribution between the lower and upper 95% credible limits on the estimates; l_{α} = the product of annual survival rates for ages 0–4 from the Seber dead-recovery models, sampled as uniform distributions between the lower and upper 95% credible limits; b = per individual fecundity, sampled as a uniform distribution from the mean to the upper 95% credible limit of the random-effect predictive distribution for fecundity; ϖ = estimated maximum lifespan, which we obtained by expanding the annual survival rates, and based on that expansion, sampled from a uniform distribution between 25–30 years; α = age-at-first breeding, estimated from Birds of North America accounts for each species ([Buehler 2000](#), [Kochert et al. 2002](#)) as 4–6 years sampled as a continuous uniform distribution; and λ (the intrinsic growth rate) = $r_{max} + 1$. The formula is solved for λ by optimizing a solution on that parameter. Using Monte Carlo methods, we simulated each parameter 10,000 times and solved for λ , which provided r_{max} . For the second approach, we parameterized our demographic population models with uniform samples between the mean and upper 95% credible intervals of the fecundity and survival distributions and solved for λ . Both approaches produced similar results, thus we used the demographic model estimates because they required fewer assumptions.

We used the sustainable take rate distribution, h , to calculate sustainable take limits (H) at time t using the formula:

$$H_t = hN_t,$$

where N_t = population size at time t , sampled from the uncertainty distribution for N_t as described previously. Thus, the estimate of H_t for both eagle species is represented by a distribution that accounts for uncertainty in both the sustainable take rate and population size at time t . The subscript is intended as a reminder that both H and N are not constants, and must be updated regularly with monitoring information (see [Population Monitoring](#)). Unless otherwise noted, we further constrained our estimates of h by setting F_0 to a value consistent with the objective of maintaining an equilibrium population size $\geq N_{2009}$ ([Figure 9](#)). For the liberal-alternative, we used the medians of the parameter distributions to estimate H . For the conservative-alternative, we accounted for uncertainty by estimating h_{20} using the 20th quantiles of r_{max} and carrying capacity, then multiplying by the 20th quantile of the appropriate population size estimate to obtain H_{20} . [Figure 9](#) provides an example of how h , H , h_{20} , and H_{20} , were determined relative to the population objective using standard harvest yield curve for the bald eagle ([Williams et al. 2002](#), [Runge et al. 2009](#)). Estimates of carrying capacity are a key component of the equilibrium harvest curve. We used our demographic model estimates of equilibrium population size as our values of carrying capacity in harvest rate analyses, though actual carrying capacity might occur at lower population levels if resources become limiting before demographic rates.

Ideally, r_{max} is calculated in the absence of anthropogenic mortality. We had no data to estimate current anthropogenic take for bald eagles. However, for golden eagles we were able to estimate the proportion of mortality in each age class attributable to anthropogenic versus natural causes ([Table 9](#)). We used this information to estimate “natural” r_{max} (r_{max} in the absence of anthropogenic mortality) for the golden eagle, as well as to quantify the amount of the potential sustainable take that is already occurring following an approach similar to that used by [Whitfield et al. \(2004a\)](#) for Scottish golden eagles.

Sustainable Take for Bald Eagles

Outside the Southwest region, we estimated that r_{max} for bald eagles with all current mortality was 20.6% (20th quantile = 18.4%), yielding $h = 10.3\%$ ($h_{20} = 9.2\%$). Our demographic-model estimate of carrying capacity was 227,800 (20th quantile = 197,500). To remain consistent with management objectives, we then adjusted h to a level compatible with maintaining an equilibrium population $\geq N_{2009}$ by setting set F_0 to 0.78

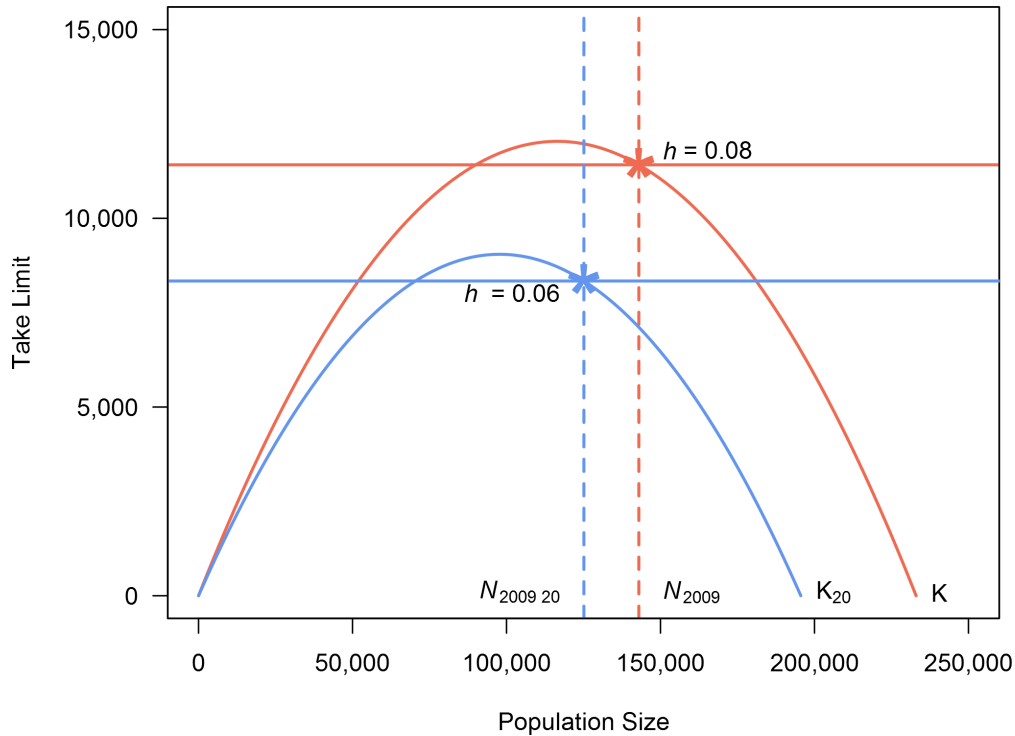


Figure 9. Harvest yield curve for the bald eagle in the U.S., excluding the Southwest region, under liberal (coral) and conservative (blue) alternatives. The sustainable take limit is the value of the y-axis at the intersection of the yield curve and the population size objective (after Runge et al. 2009). In the case of the liberal alternative, the sustainable harvest rate (h) at maximum sustainable yield (MSY) is 10%. However, we adjusted h to 8% using a management objective factor of 0.78, which corresponds with an equilibrium population size of 143,000, which is our management objective. We followed the same approach for the conservative alternative, but using the 20th quantiles of the relevant estimates rather than the medians. The take limits shown here do not correspond with the final take limits in Table 3 because the take limit for Alaska was further constrained to meet specific regional management objectives.

($F_{0_{20}} = 0.63$). Using this approach, $h = 8\%$ ($h_{20} = 6\%$) for the bald eagle outside the Southwest. In the Southwest, we estimated that $r_{max} = 17.9\%$ (20th quantile = 15%). Our demographic model suggested N_{2009} in the Southwest was less than $1/2$ demographic carrying capacity. As it was of interest to managers to allow for further bald eagle population growth in the Southwest, we set h to $1/2$ the harvest rate at MSY (4.5%), and h_{20} to the 20th quantile of $1/2$ the MSY harvest rate (3.75%), rather than the higher take rates associated with the 2009 population estimate. In Alaska, because of uncertainties in the population size estimate, managers opted to maintain H and H_{20} at approximately 500, as was recommended in 2009 (U.S. Fish and Wildlife Service 2009a); we adjusted F_0 accordingly. Collectively, across EMUs the estimated bald eagle take limits in the United States are 6,273 and 4,240 under the liberal and conservative alternatives, respectively (Table 3).

Sustainable Take for Golden Eagles

We estimated that natural r_{max} (calculated in the absence of anthropogenic mortality) for the golden eagle was 10.9% (20th quantile = 9.7%), yielding $h = 5.4\%$ ($h_{20} = 4.9\%$). Our demographic-model estimate of carrying capacity was 73,000 (20th quantile = 64,000). H at our population objective would be 2,000 ($h_{20} = 1,600$) and MSY under these conditions would be 2,200. However, we estimated that currently about 3,400 (95% credible interval = 935–9,253) golden eagles die annually from anthropogenic causes in the U.S. (Table 8). Despite the considerable uncertainty in estimates of both MSY and current levels of

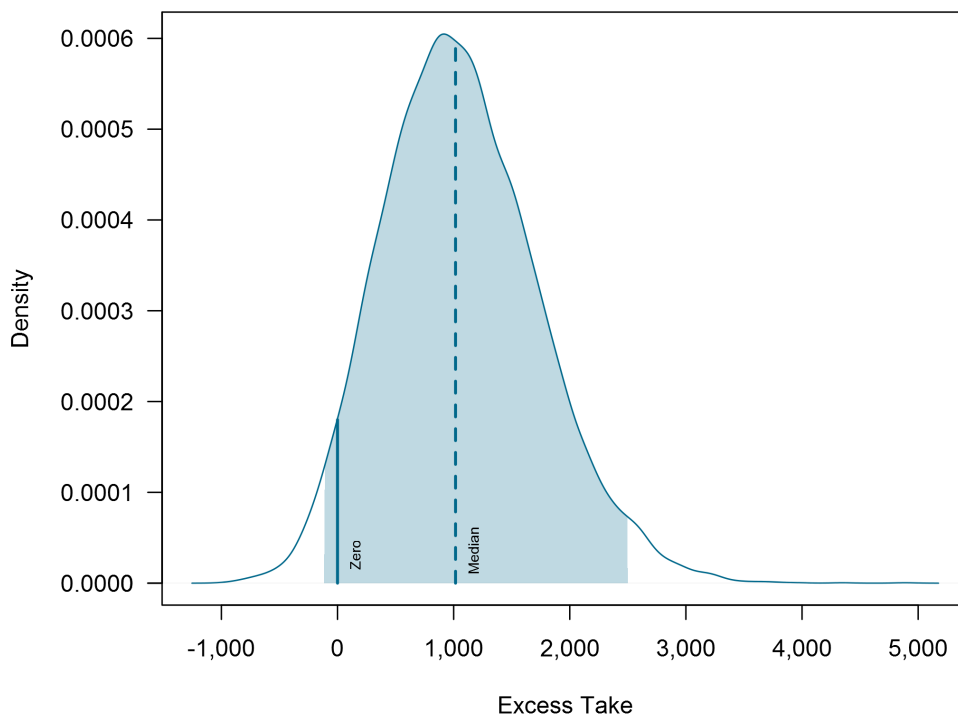


Figure 10. Histogram of the difference between the posterior distribution of the estimated current amount of human-caused mortality of golden eagles in the United States (see Table 8) and the posterior distribution of the estimated sustainable take rate at maximum sustainable yield. The blue-shaded area indicates the portion of the probability distribution that lies within the 95% credible interval, the dashed vertical blue line is the median, and the solid vertical blue line indicates zero. Note that most of the distribution lies above zero, which means estimated sustainable take is less than the estimated levels of current human-caused mortality.

mortality, these data suggest golden eagles in the U.S. are currently experiencing more take than can be sustained at the population objective or at MSY (Figure 10). This result is somewhat at odds with our estimate of the stable long-term population trend from the composite model. It is possible that golden eagles are compensating for the high unnatural mortality rate with increases in survival or fecundity to a greater degree than we have allowed for in our demographic model. Regardless, adding additional unmitigated mortality will either exacerbate the potential for declines, or steepen the rate of any decline that is presently occurring. To illustrate this further, we computed the effect of added take for the golden eagle in 1% increments up to a 10% harvest rate using the population model described previously (Figure 11). All added take resulted in population declines to new, lower equilibrium sizes. The upper 95% confidence intervals under all of the scenarios tested were below N_{2009} and therefore not consistent with the Service’s population objective. Given this, we use zero as the take value for both h and h_{20} for the golden eagle (Table 10).

Metrics for Take as a Result of Nest Disturbance

For disturbance to have a population-level effect, it has to result in a loss of potential productivity. In 2009, the Service used the EMU-specific productivity (mean number of young fledged per occupied nesting territory) for each species per year as the expected loss for each instance of authorized nest disturbance (U.S. Fish and Wildlife Service 2009b). Here we follow the same approach with updated take values from the appropriate random-effects predictive distributions from the productivity meta-analysis (Figures A2-1 and A2-4). We used the median values of the distributions for the liberal alternatives, and the 80th quantiles for

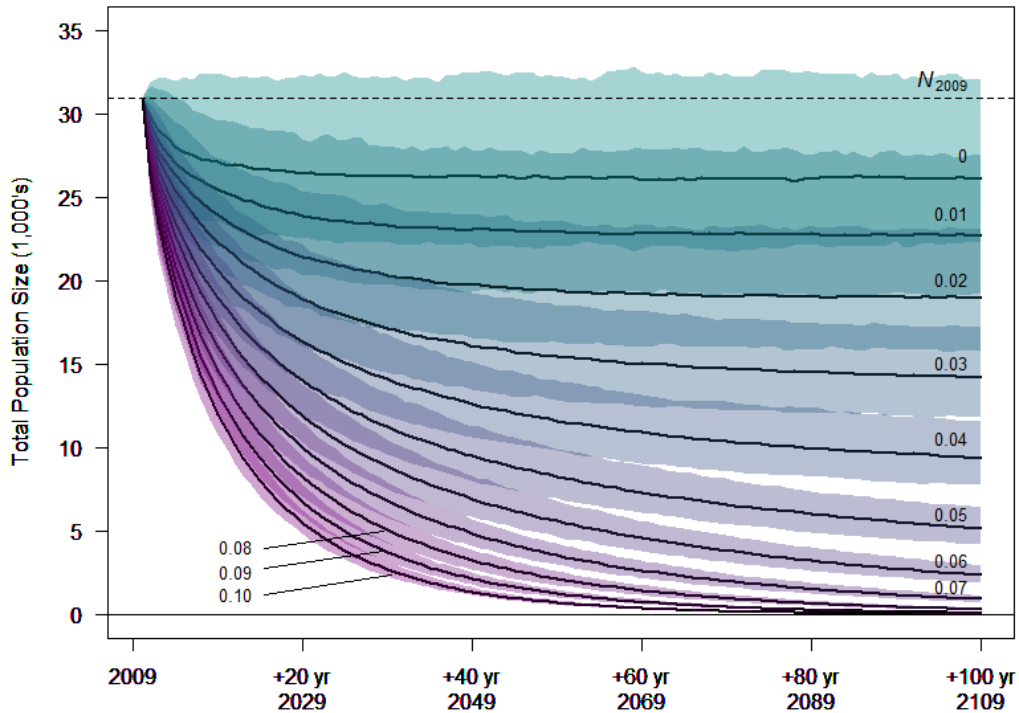


Figure 11. Simulated effect of added take on golden eagle populations. The top line is the projected golden eagle population trend with no additional take. Each line below the top line represents the population projection with increased take in 1% increments, up to a take rate of 10% in the bottom line. Shaded areas represent the 95% confidence intervals on the projections and transition from blue to purple as the take rate increases.

the conservative alternatives to maintain a protective 20% probability of underestimating the productivity potentially lost as a result of disturbance. Following this approach, for each instance of nest disturbance predicted to result in loss of productivity, take thresholds for bald eagles outside the Southwest are debited by 1.12 or 1.33 eagles, under the liberal and conservative alternatives respectively, per year that the disturbance occurs. For bald eagles in the Southwest, take thresholds are reduced by 0.73 or 0.95, and for golden eagles by 0.53 or 0.59, respectively (Tables 13 and 14).

Metrics for Take as a Result of Territory Loss

Loss of an occupied nesting territory results in the recurring loss of annual production from that territory. However, this loss of future production is difficult to estimate and account for in debiting take thresholds. In 2009, the Service quantified future production lost from loss of an occupied territory by comparing equilibrium population size with N and $N - 1$ nesting territories, then debiting EMU take limits by the difference (U.S. Fish and Wildlife Service 2009b). This approach assesses the effects of loss indirectly and relates it to a future equilibrium population size rather than the population objective. Here, for each instance of occupied territory loss, we subtract the mean annual per nesting-territory productivity from the EMU take limit annually for the generation time of the eagle species (Tables 13 and 14). We define generation time as the average age of breeders in the population (Caswell 2001, Bienvenu and Legendre 2015). Using this as the temporal scale over which we account for productivity lost is biologically relevant and sufficiently long to assure that potential longer-term effects can be accounted for by future adjustments to the EMU take limits based on reassessments of eagle populations (see Population Monitoring below).

Table 13. Take limits associated with take as a result of disturbance to nesting bald eagles, and the loss of occupied bald eagle nesting territories.

Eagle Management Unit	Debits to Take Limits				
	Per Instance Disturbance Take, 50 th Quantile	Per Instance Disturbance Take, 80 th Quantile	Generation Time (years)	Cumulative Per Territory, 50 th Quantile ^a	Cumulative Per Territory, 80 th Quantile ^a
Alaska ^a	1.12	1.33	10	11.20	13.30
Great Lakes	1.12	1.33	10	11.20	13.30
Lower Mississippi	1.12	1.33	10	11.20	13.30
Mid-Atlantic	1.12	1.33	10	11.20	13.30
New England	1.12	1.33	10	11.20	13.30
Northern Rocky Mountains	1.12	1.33	10	11.20	13.30
Pacific	1.12	1.33	10	11.20	13.30
Rocky Mountains and Plains	1.12	1.33	10	11.20	13.30
Southeast	1.12	1.33	10	11.20	13.30
Southwest	0.73	0.95	12	8.76	11.40
Alaska-FW ^a	1.12	1.33	10	11.20	13.30
Atlantic Flyway	1.12	1.33	10	11.20	13.30
Central Flyway	1.12	1.33	10	11.20	13.30
Mississippi Flyway	1.12	1.33	10	11.20	13.30
Pacific Flyway, South	0.73	0.95	12	8.76	11.40
Pacific Flyway, North	1.12	1.33	10	11.20	13.30

^a The per-instance take debit is applied annually for the generation time. These are the cumulative debits at the end of the specified generation time.

We recognize that for golden eagles in particular, nesting territories are often occupied by successive generations of individuals. Additionally, for both species, some nesting territories hold more value than others (Millsap et al. 2015, Watts 2015). Moreover, it is often difficult to predict in advance whether an activity will result in loss of a nesting territory, or simply the loss of a nest structure and cause a shift in use to an existing or new alternative nest—which may have little or no consequence to the eagle population (Watts 2015). For these reasons, each instance where loss of a nesting territory is a possible outcome requires additional review on the part of Service biologists. Permitting the loss of high-value nesting territories with a long history of occupancy and production could have greater population-level consequences.

We used the mean of the fertility rate schedule from the matrix demographic models (effectively the mean age of breeders in the population) as the generation time. Generation time is 12 years for bald eagles in the Southwest and 10 years for bald eagles in the rest of the U.S. Golden eagle generation time is 11 years. The corresponding debits to take limits by EMU are given in Tables 13 and 14.

Take Limits at the Scale of the Local Eagle Population

The objective of the LAP take limit is to regulate take such that local populations are protected from extirpation due to Service-authorized activities. Although the primary aim is to prevent extirpation of local nesting populations, there is increasing evidence of strong philopatry to non-breeding areas in both species of eagle (McIntyre et al. 2008, Mojica et al. 2008), and the LAP take limits also provide protection from overharvest of wintering and migrating eagles. As noted above, LAP take limits pertain only to take permitted or authorized by the Service, and are cumulative, taking into consideration all Service-authorized activities affecting the LAP.

Table 14. Take limits associated with take as a result of disturbance to nesting golden eagles, and the loss of occupied golden eagle nesting territories.

Eagle Management Unit	Debits to Take Limits				
	Per Instance Disturbance Take, 50 th Quantile	Per Instance Disturbance Take, 80 th Quantile	Generation Time years	Cumulative Per Territory, 50 th Quantile ^a	Cumulative Per Territory, 80 th Quantile ^a
Alaska	0.53	0.59	11	5.83	6.49
Eastern	0.53	0.59	11	5.83	6.49
BCR 5	0.53	0.59	11	5.83	6.49
BCR 9	0.53	0.59	11	5.83	6.49
BCR 10	0.53	0.59	11	5.83	6.49
BCR 11	0.53	0.59	11	5.83	6.49
BCR 15	0.53	0.59	11	5.83	6.49
BCR 16	0.53	0.59	11	5.83	6.49
BCR 17	0.53	0.59	11	5.83	6.49
BCR 18	0.53	0.59	11	5.83	6.49
BCR 32	0.53	0.59	11	5.83	6.49
BCR 33	0.53	0.59	11	5.83	6.49
BCR 34	0.53	0.59	11	5.83	6.49
BCR 35	0.53	0.59	11	5.83	6.49
Atlantic/Mississippi	0.53	0.59	11	5.83	6.49
Central Flyway	0.53	0.59	11	5.83	6.49
Pacific Flyway	0.53	0.59	11	5.83	6.49

^a The per-instance take debit is applied annually for the generation time. These are the cumulative debits at the end of the specified generation time.

The Service identified LAP take-rates of $\geq 1\%$ as being of concern, and rates of 5% being at the maximum of what should be considered (U.S. Fish and Wildlife Service 2013). Note that the take authorized (within the LAP take limits) is in addition to the average background rate of anthropogenic mortality—for golden eagles, this is about 10% (Table 9). Thus, total anthropogenic mortality for a golden eagle LAP experiencing the maximum permitted take rate of 5% is likely about 15%. We do not have similar mortality information for bald eagles, but the same principle (that take authorized would be in addition to an existing background rate of anthropogenic mortality) applies to that species as well. As part of the LAP analysis for both species, Service biologists also consider available information on unpermitted take occurring within the LAP area and carefully evaluate evidence of excessive unpermitted take during the permitting process.

The population size of the LAP is estimated by expanding the density estimates for EMUs to the LAP area. We acknowledge this approach is simplistic for at least two reasons: (1) given the eagle density estimates come from nesting or late-summer population surveys, they do not account for seasonal influxes of eagles that occur through migration and dispersal; (2) this approach assumes eagle density is uniform across the EMU, which we know is inaccurate. In most cases the first simplification leads to an underestimate of true density, particularly in core wintering areas during the non-breeding months. As such, this serves as an added buffer against over-harvest of local nesting eagles. The second assumption of uniform density leads to greater relative protection of areas with higher than average eagle density within an EMU, and less relative protection in areas of lower density. Over time, with better information on resource selection and factors accounting for variation in density (e.g., Tack and Fedy 2015), as well as improved knowledge of seasonal changes in eagle density and population-specific movement patterns, we can improve the LAP analysis to more realistically account for the true LAP impact by projects under consideration. For now, however, LAP take thresholds allow the Service to authorize limited take of eagles while favoring eagle conservation in the

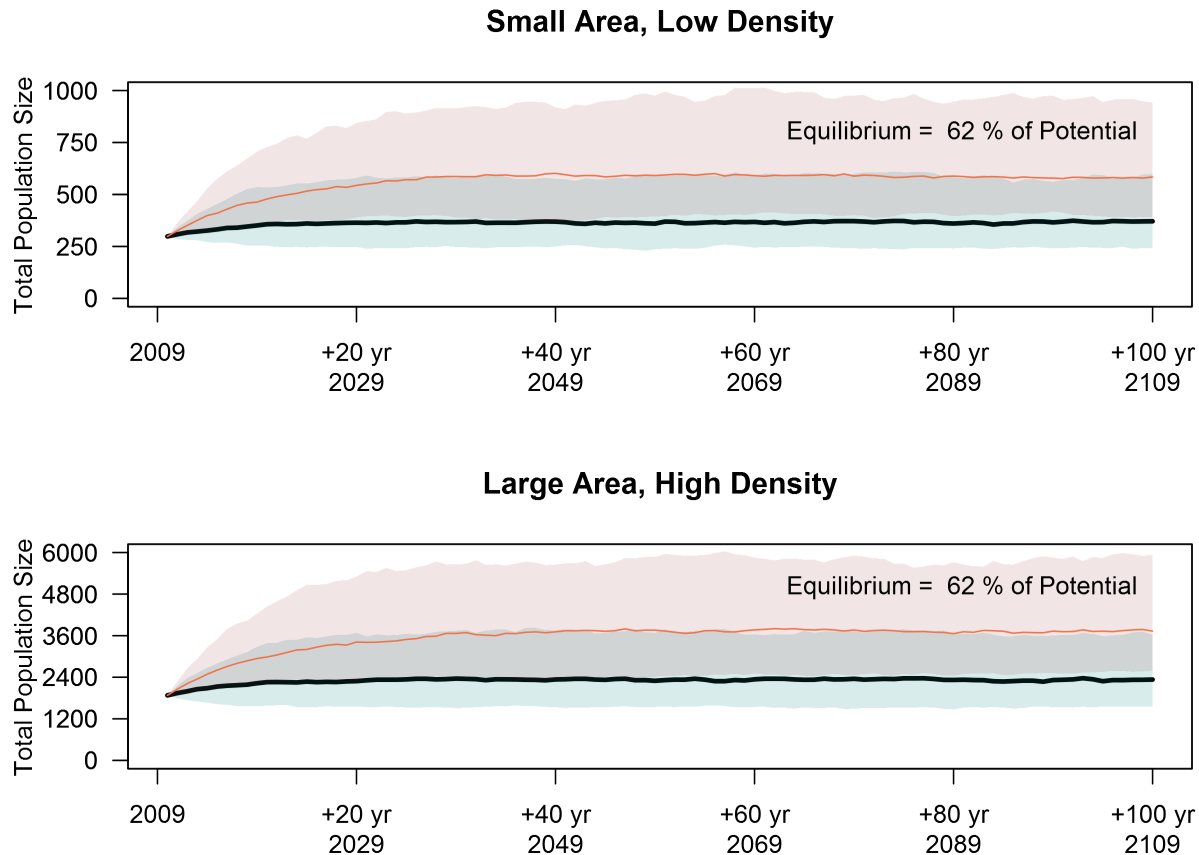


Figure 12. Effect on bald eagle local area populations (LAP) of a 5% additive harvest rate. The top figure is for a small project (6,870 km² local area) in a low-density EMU (0.008 bald eagles per km²), whereas the bottom figure is for a large project (37,551 km² local area) in a high-density EMU (0.05 bald eagles per km²). The black line and blue-shaded area shows the trajectory and 95% confidence interval at the 5% LAP take limit, the red line and pink-shaded area is the trajectory and 95% confidence interval in the absence of added take.

face of uncertainty.

To understand the potential consequence to the LAP of authorizing take up to the levels of the LAP take thresholds, we conducted a series of simulations using our demographic models to add a 5% take-rate to background take levels for a hypothetical LAP of both species of eagle (Figure 12, Figure 13). We looked at hypothetical large and small project footprints in high- and low-density EMUs. For the golden eagle, adding 5% take results in a decline in the LAP and eventually lowers the equilibrium as much as 80%. However, the LAP did not go to extirpation for the scenarios considered. For the bald eagle, an additive 5% take does not cause declines in projected LAPs, but reduces the size of the eventual equilibrium LAP by 38% from the equilibrium in the absence of added take.

The Role of Compensatory Mitigation

Authorized take above EMU take limits has to be offset by compensatory mitigation that will produce a commensurate decrease in a pre-existing mortality factor, or increase in carrying capacity (U.S. Fish and Wildlife Service 2009a, 2013). The effect of this mitigation must be that no net increase in mortality occurs within the EMU where the take is authorized. In the case of golden eagles, our analyses suggest even current

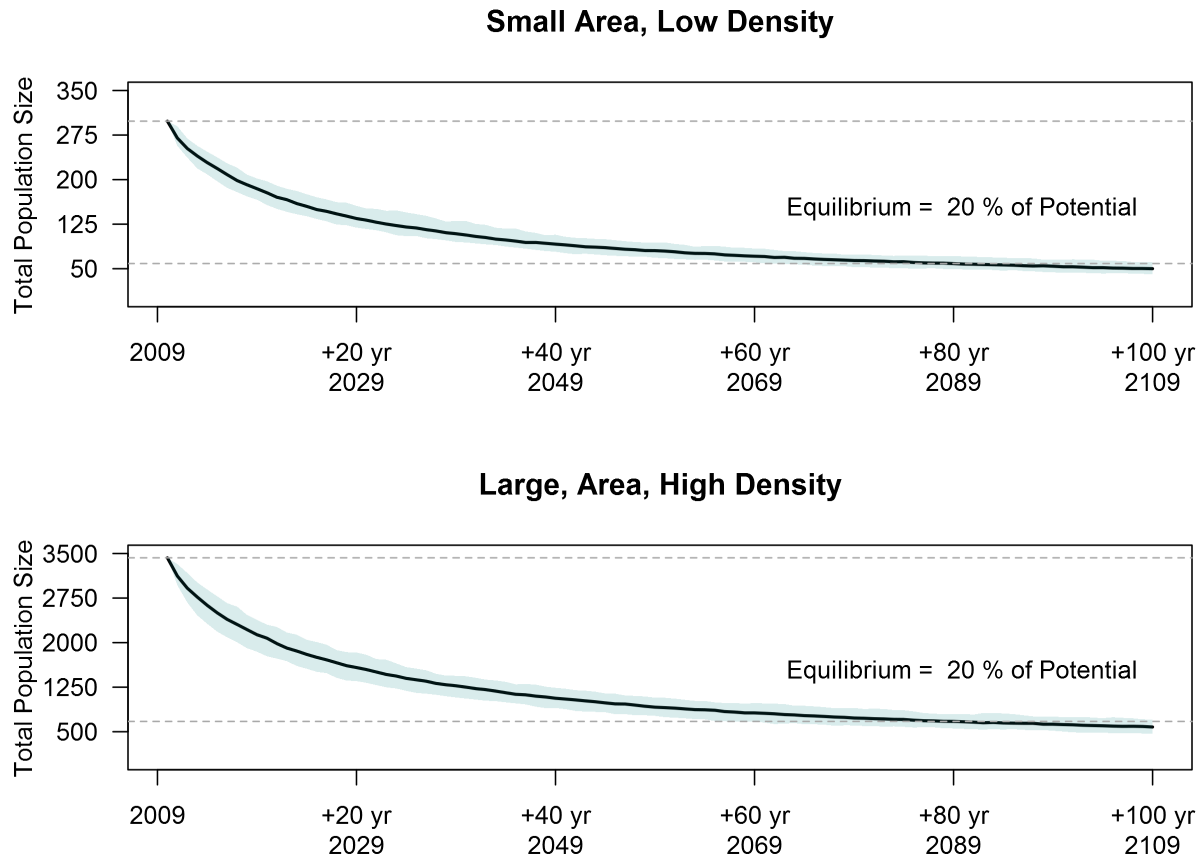


Figure 13. Effect on golden eagle local area populations (LAP) of a 5% additive harvest rate (above a background rate that is, on average, about 10%). The top figure is for a small project (37,299 km² local area) in a low-density EMU (0.008 golden eagles per km²), whereas the bottom figure is for a large project (126,950 km² local area) in a high-density EMU (0.027 golden eagles per km²). Dotted gray lines show the starting and equilibrium population levels, whereas black lines and blue shading denote the median population size and 95% confidence limits.

levels of take may not be sustainable. Offsetting mitigation for golden eagles at a rate of > 1:1 may be necessary to be compatible with the Service's population objective.

The factor that most limits how much golden eagle take the Service can permit is the amount of ongoing unpermitted take or natural mortality that can reasonably be expected to be offset. Quantifying the real effects of conservation actions in reducing mortality has proven difficult to date. Electric distribution power line retrofitting to reduce electrocutions ([Avian Power Line Interaction Committee \(APLIC\) 2006](#), [U.S. Fish and Wildlife Service 2013](#)) remains the best understood existing mortality source with thorough representation in the scientific literature, and our analyses suggest that if perfectly effected, about 500 (20th quantile = 280) golden eagle deaths could be offset through this approach annually ([Table 8](#)). Work to develop other approaches for implementing and quantifying the performance of other compensatory mitigation mechanisms is an area of active research, and promising advances are being made (e.g., [Cochrane et al. 2015](#)).

The Service currently requires that offsetting mitigation be undertaken in the same EMU where the take is authorized ([U.S. Fish and Wildlife Service 2013](#)). Our analysis suggests this spatial scale is still reasonable, especially under the Flyway EMU alternatives which take into account the full annual cycle of both eagle species. However, because a substantial proportion of the mortality of golden eagles originating in Alaska occurs on migration or during winter in the interior western coterminous U.S. and north-central Mexico ([McIntyre 2012](#)), effective mitigation for take of Alaskan golden eagles could occur in these areas as well.



Population Monitoring

As noted previously, the take limits are time-sensitive and require regularly updated estimates of population size. More generally, the Service has also implemented the eagle take permit process under a formal adaptive management framework, such that monitoring eagle populations and updating population estimates and take limits are critical parts of the adaptive management feedback loop (U.S. Fish and Wildlife Service 2013). For these reasons, the Service is interested in formalizing its eagle population monitoring commitments as part of the PEIS process.

The existing golden eagle assessment approach (using a hierarchical model to combine density estimates from the summer aerial-transect survey with BBS indices [Appendix A4]) provides reasonable information on golden eagle population size and trend at the coarse scale considered under the national permit program. The winter golden eagle survey also provides useful information on the number and distribution of golden eagles in the core of the species' range in winter. This information about wintering eagles is essential for more accurate accounting of the effects of take at different locations on different natal populations of golden eagles; pairing summer and winter surveys maximizes the opportunity to quantify wintering golden eagle populations that move into the coterminous U.S. from northern latitudes (see [Golden Eagle: Population Size](#)).

As part of future bald eagle nesting territory survey efforts, the Service will investigate the potential for combining the dual-frame survey estimates of occupied nesting territories with BBS indices to better link the dual-frame results to changes in total population size (expanding beyond the current focus on breeding numbers); additionally, this capitalizes on the rigorous and standardized data set from the BBS. Because our conversion of the dual-frame survey results to total population size estimates depends on accurate EMU-scale estimates of productivity, the Service will investigate adjustments to the dual-frame survey design that will provide information on nest success and brood sizes in a sample of occupied nesting territories.

There are several other areas of active inquiry that should improve the Service's ability to effectively manage eagles in the future. In particular, resource utilization functions have the potential to vastly improve the accuracy of LAP analyses. Additionally, surveys and studies to locate, map, and prioritize nesting territories are important in that they can serve to identify and direct projects away from important high-density areas or high-performing nesting territories. Service biologists will continue to look for ways to implement these surveys as efficiently and effectively as possible, including periodic reassessments of statistical power and reliability, and integrating other sources of information (e.g., Christmas Bird Counts) with ongoing surveys to improve power, representativeness, and to expand the scale of inference.

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Appendix A1. Review of Eagle Productivity Data

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Literature Review and Data Compilation

In March 2015, the Service conducted a comprehensive review of available peer-reviewed publications that included data on eagle productivity. The data were evaluated, filtered, and, if appropriate, used to update and improve estimates of eagle productivity—including variability and the range of observed productivity across the United States. The Service used the improved productivity estimates to update demographic population models and evaluate the effect of mortality rates under various management scenarios.

We conducted a thorough literature search through a variety of available databases and search tools (Google Scholar, Web of Science, BioOne, Biosis, and ProQuest) using various combinations of the following key words: eagle, bald eagle, golden eagle, productivity, nest productivity, nest success, nest monitoring, population analysis, and population status. We also searched prominent authors of eagle biology and ecology. We filtered the literature search results to focus on publications containing productivity data from 1995 to 2015. Given potential long-term fluctuations in productivity (e.g., due to changes in conservation practices, land use, or environmental factors), focusing on data from the last two decades helped ensure a contemporary estimate of eagle productivity. We further limited results to bald and golden eagle productivity observed in the conterminous United States and Alaska. We also initially selected references based on titles or abstracts, and later excluded publications that did not contain data sufficient to determine nest or territory occupancy and young fledged. We discovered 98 productivity references for bald eagles and 70 for golden eagles that met our criteria.

There was considerable variability in the reporting of nest or territory activity and inconsistency in the use of the terms “active” and “occupied” by different authors in describing breeding status. We specifically looked for publications that included the number of young fledged from occupied nesting territories based on criteria for occupancy initially described by [Postupalsky \(1974\)](#) and later by [Steenhof and Newton \(2007\)](#). Limiting our results to papers that specifically allowed determination of territorial occupancy for determining productivity further reduced the number of studies in our summary.

We defined productivity as the total young fledged per occupied nesting territory ([Steenhof and Newton 2007](#)). However, we found inconsistencies between the papers we reviewed in the way productivity was calculated; in some cases it was possible to base productivity estimates on the number of occupied nesting territories sampled, whereas in other cases productivity was clearly weighted towards successful nesting territories. When necessary we back-calculated productivity using the total occupied nesting territories and total number of young counted. We also excluded studies that had manipulative components (e.g., egg removal, experimental disturbance) since the manipulations could affect productivity.

Though we preferred to include studies that reported annual estimates of productivity (or the data that could be used to compute them), some studies only reported aggregated data or estimates for multiple years within a specific area or a subset of known nesting territories. We excluded any redundant data identified in the course of the review.

We found a greater number of publications reporting productivity for bald eagles than golden eagles, as well as uneven coverage of the known range of both species. We subsequently used State or other agency

monitoring reports for breeding eagles where the data available would provide more representative spatial coverage of the known breeding range. Most peer-reviewed studies focused on local sub-populations or specific nesting territories monitored over time.

The relatively small number of papers—18 for bald eagles and 12 for golden eagles—included in our final compilation of productivity data from the literature and subsequent meta-analysis (see [Appendix A2](#)) reflects our stringent data quality standards. Many of these final papers contained data spanning multiple years and included a large number of nesting territories. We believe these data are representative of available data on eagle productivity across the known bald and gold eagle breeding ranges in the U.S.

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Appendix A2. Bald and Golden Eagle Productivity Meta-Analysis

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Division of Migratory Bird Management

Abstract

As part of a larger effort to update and improve productivity parameters used in eagle population modeling efforts, Brennan and Millsap ([Appendix A1](#)) compiled a dataset of contemporary productivity information for bald and golden eagles, *Haliaeetus leucocephalus* and *Aguila chrysaetos* respectively, across the U.S. from 1995–2014. With these data, I used a random effects meta-analysis model and estimated the predictive distributions for bald eagle and golden eagle productivity. Bald eagle productivity differed by region with lower productivity in the Southwest (mean = 0.77, SE = 0.249) than in the rest of the continental U.S. (mean = 1.15, SE = 0.252), whereas golden eagle productivity did not differ by region (mean = 0.55, SE = 0.087). Apart from the fixed stratum differences for bald eagles, the best-supported models included standard errors for the random effects for study, area (bald eagles only), year given study, and overdispersion; the extent to which the random effect credible intervals overlapped zero varied by species.

Introduction

Meta-analyses combine results of different studies of the same subject in order provide stronger and more robust inferences ([Borenstein et al. 2010](#)). Meta-analyses are often used in the medical field to combine clinical trial data. Just as medical trials may have different methodologies, target populations, and sampling designs and selections, so do wildlife demographic studies ([Johnson 2002](#)). There are two main types of meta-analyses: fixed and random effect. Fixed effect models are used when the studies are thought to be functionally equivalent, whereas the random effect model assumes that they have common characteristics but are not the same ([Borenstein et al. 2010](#)). Thus a fixed effect model assumes that a single value is common to all studies, in contrast to a random effects model which assumes that the values belong to a common distribution ([Higgins et al. 2009](#)).

Summarizing a range of studies over different areas and time spans, accounting for the study, area, and annual components of variation also complicates the analysis with decisions on how to separate and characterize the different forms of variation. Rather than estimate the common value (in our case productivity), [Higgins et al. \(2009\)](#) recommend using predictive distributions. They also recommend for a small number of studies, using the t -distribution with k minus 4 degrees of freedom (k is the number of studies), instead of the normal distribution. Due to the combined complexities of deciding on the proper prediction variance and other model choices, I used the classic normal distribution for this analysis. The approach is similar to what is presented in [New et al. \(2015\)](#) for the prior parameters for the eagle example where the authors created a mixture distribution from the small number of projects available and estimated parameters for a common distribution from the mixture. My methods here similarly yield a common predictive distribution for productivity from the projects available.

Methods

Data

Brennan and Millsap (Appendix A1) searched the published literature for bald and golden eagle productivity data and compiled datasets for each species from studies within the U.S. from 1995–2014. I categorized the target populations for the included studies in terms of area and time span and accounted for separate values for multi-area and multi-year data. I used sample size (the number of nesting territories or nests), number of fledglings, productivity values, and standard errors reported in the studies. When not reported, I back-calculated sample size from number of fledglings and productivity. In one case where only the productivity value was reported, the sample size became the inverse of the productivity value—resulting in one fledgling in the study and the smallest weight possible given to that study.

There were 18 studies included in the bald eagle analysis: one multi-area study, nine multi-year studies, and two multi-area and multi-year studies. In cases where studies included multi-area or multi-year data, I used random effects for area or year nested within study. The data did not support interactions between area and year in the 2 multi-area and multi-year studies. There were 12 studies included in the golden eagle analysis: nine multi-year studies but no multi-area and multi-year studies. This limited the golden eagle analysis to only considering study-to-study and year-to-year variation.

Model

The productivity random effects model is a Poisson log-normal hierarchical model (although a gamma distribution could replace the log normal). The data are the number of successful fledglings in each study (with values separated by areas and years in multi-strata studies). The log sample sizes, S_{ijkl} number of nesting territories, are treated as offsets but are shown here on the original scale,

$$F_{ijkl} \sim \text{Poisson}(R_{ijkl}S_{ijkl}).$$

F_{ijkl} is the number of fledglings in the k^{th} area and l^{th} year of the j^{th} study in the i^{th} region. Not all subscripts are necessary if it is not multi-area and multi-year study. R_{ijkl} is the estimated random effect productivity estimate, and S_{ijkl} is the sample size in number of occupied nesting territories. Since the model conditions on occupied nesting territories, we only make the basic assumption that the likelihood occupied nesting territories were observed was not linked to the productivity rate. If the chances of detecting an occupied nesting territory early, even if it later fails, are good then the potential for such detection bias should be low. Log productivity is affected by the region, the study within that region, and if applicable a year within a given study.

$$\log(R_{ijkl}) = N(r_i + \psi_{j|i} + \alpha_{k|ij} + \tau_{l|ij}, \sigma_{\text{Overdispersion}}^2).$$

Study, $\psi_{j|i}$; area, $\alpha_{k|ij}$; and year, $\tau_{l|ij}$, are nested random effects, with study nested within region, and area and year nested within study; there were no multi-region studies. The overdispersion variance is $\sigma_{\text{Overdispersion}}^2$.

The random effects use an Ottomert transformation that converts n random variables into n centered variables with the same standard deviation and the same correlations among all the effects. The transformation corrects for the under-estimation of the standard deviation caused by generating and centering n random variables. The area in multi-area random effects and year in multi-year random effects are nested within study, so their effects are centered within each study.

$$\begin{aligned}\psi_{j|i} &= \text{Ottomert}(Study_j | Region_i) N(0, \sigma_{\text{Study}}^2) \\ \alpha_{k|ij} &= \text{Ottomert}(Area_k | Study_j) N(0, \sigma_{\text{Area}}^2) \\ \tau_{l|ij} &= \text{Ottomert}(Year_l | Study_j) N(0, \sigma_{\text{Year}}^2)\end{aligned}$$

Because studies are nested within region, the area and year effects are also nested within region. I assumed the study, area, and year variation were the same across regions. For bald eagles, there was only one multi-year study representing the Southwest and only one study representing the East for golden eagles.

I ran glm and glmer models in R (R Core Team 2014) to discriminate among models using AIC and then estimated the best-supported models using Stan (Stan Development Team 2015), which is equivalent to a Bayesian estimation with non-informative priors. I included overdispersion by adding a random effect where the effect was different for every observation and tested models for an overall mean only and an overall mean with overdispersion. I also calculated simple estimates of productivity by aggregating the fledged and occupied territory counts for each area and year of each study by region then taking the ratio.

Results

Overdispersion gave a vast reduction in AIC (Δ AIC) for both the bald eagle and golden eagle models (118.49 and 35.37, respectively). For bald eagles, the best-supported model was a random effects model with overdispersion that included a fixed effect for region (separating the Southwest from the rest of the U.S.; Table A2-1). The Southwest had lower overall productivity (Table A2-2) than the rest of the U.S., but there was wide overlap between the predictive distributions (Figure A2-1). Both prediction distributions are right skewed and leptokurtic, therefore the best way to use the productivity information as part of a demographic model is to sample the posterior simulations.

All of the random effects (study, area, year, and overdispersion) were important to the model (25.68); the estimates of the standard errors for the random effects are in Table A2-3. The random effects from the final model were more spread out than the simple estimates for both regional distributions (Figure A2-1). Normally we would expect random effects estimates to shrink or be less spread out, but the differences are small and likely due to separating the study, area, and year random effects. The study, area, and year effects are all significant and the credible intervals do not overlap zero. The total random effect variance is the sum of the variances of all the random effects (Table A2-2).

Figure A2-2 shows the percent change due to the study random effects apart from regional differences and the percent change due to the year given study. The productivity estimates by study from the random effects model include the region effects and the study, area, and year random effects (Table A2-4); they vary from 0.48 to 0.57. The model random effect estimates by study, area, and year are presented in Table A2-5.

For golden eagles, the best-supported model was the random effects model with overdispersion (Table A2-1). The study and year random effects were important to the model (4.02), but there were no multi-area studies so I did not include area random effects. I explored models with a regional effect (e.g., Eastern U.S., Western U.S., Alaska) but there was no support for including any regional differences (3.56 for 4 degrees of freedom; Table A2-6) so the final model estimated an overall productivity for the entire U.S. including Alaska (Table A2-2). The overall prediction estimate along with the 95% prediction intervals is shown in Figure A2-4. The estimates from the final model are a bit lower than simple estimates taken by aggregating the fledged and occupied territory counts then taking the ratio (Figure A2-4). Explaining this will require further exploration. The distribution is right skewed, skewness = 2.09, and is highly leptokurtic, kurtosis = 22.59, therefore sampling the posterior simulations is the best way to use the productivity estimates in other models, since they do not fit a common distribution.

The estimates of the standard errors for the random effects are in Table A2-2. The study and year random effects had low variation (medians 0.1 and 0.29, respectively), and all random effect credible intervals overlap zero (Figure A2-5). The non-significance of the study and year effects and the significance of the overdispersion reinforce the AIC differences observed in the model comparisons. The productivity estimates from the random effects model by study include the region effects and the study random effects; they vary from 0.48 to 0.57 (Table A2-5). The random effect model estimates which include the study, area, and

year random effects for each study and year combination are included in Table A2-6 along with the simple productivity ratios.

Discussion

I conducted this modeling effort with the specific goal of rapidly producing a usable predictive distribution for productivity that could be used in subsequent population modeling efforts. Though the approach was logical, there were a number of decisions that could be explored further. I only included the study random effect variances in the predictions. Though this is consistent with most meta-analysis models, it is unusual to have the additional complexities of multi-area and multi-year studies. An alternative approach may be to include both random effects and overdispersion in the prediction variation along with the additional consideration of using a *t*-distribution instead of a normal distribution. However all of this would make the already large prediction intervals larger, possibly to the point of no longer being useful. The current approach used to estimate the predictive distribution is consistent with other meta-analysis models and sampling the posterior simulations will provide reasonable productivity estimates given the data available.

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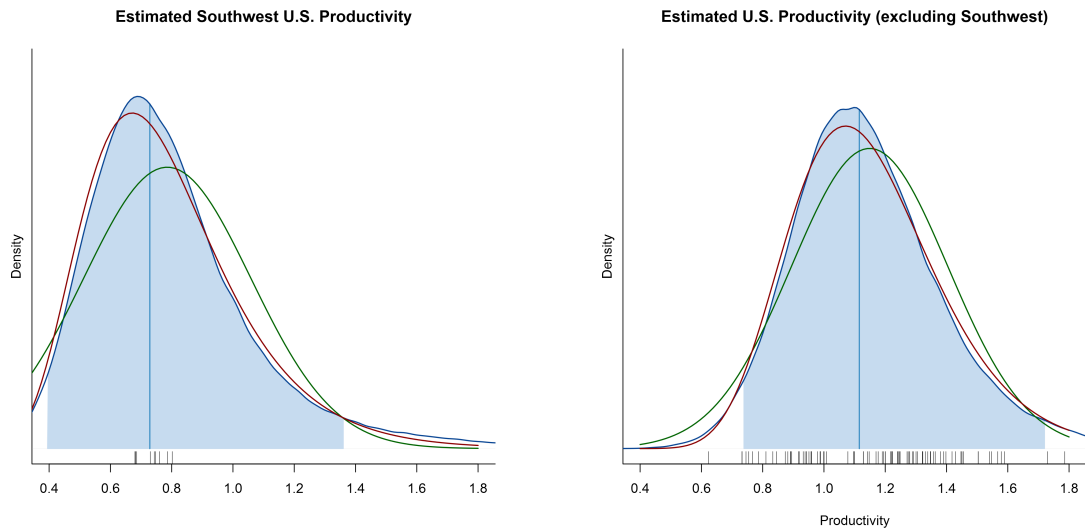


Figure A2-1. Bald eagle productivity for the Southwest U.S.(left) and the U.S. excluding the Southwest (right). The blue curve is the empirical density distribution of the estimates—which are shown via the rug just above the x-axis. The vertical blue line is the median with the area within the 95% credible intervals shaded blue. The red and green curves represent the log normal and normal distributions (respectively) defined by the estimated means and standard deviations.

Table A2-1. AIC values for bald and golden eagles from glm and glmer models which included overdispersion, study, area, year, and region effects with a mean. Region included Alaska, the Southwest (SW), the conterminous U.S. excluding the Southwest (Lower 48) and the entire continental U.S. (Overall) for bald eagles. Region included Eastern U.S. (East), Western U.S. (West), Alaska (AK) and Overall for golden eagles.

Species	Overdispersion	Fixed-Effects	Random-Effects	Difference-DoF	AIC
Bald Eagle		Overall	None	1	1,449.50
	x	Overall	None	2	924.08
	x	Alaska+Lower48+SW	None	4	907.00
		Alaska+Lower48+SW	All	6	999.81
	x	Overall	All	4	883.57
	x	Lower48+SW	All	6	881.75
	x	Alaska+Lower48+SW	All	7	881.32
Golden Eagle		Overall	None	1	688.50
	x	Overall	None	2	506.56
	x	Alaska+East+West	None	4	510.01
		Alaska+East+West	All	5	541.47
	x	Overall	All	4	502.53
	x	East+West	All	5	504.20
	x	Alaska+East+West	All	6	506.09

Table A2-2. Regional prediction means, standard errors (SE), medians, and lower and upper limits (LCL, UCL) of the 95% credible intervals from the random effects models for bald and golden eagle productivity. The bald eagle model included a fixed effect for region and estimated productivity for the U.S. excluding the Southwest (U.S.–SW) and the Southwest (SW). The golden eagle model is an overall random effects model.

Species	Region	Mean	SE	Median	(LCL–UCL)
Bald eagle	U.S.–SW	1.15	0.252	1.12	(0.73–1.72)
	SW	0.77	0.249	0.73	(0.40–1.36)
Golden eagle	Overall	0.55	0.087	0.54	(0.40–0.75)

Table A2-3. Productivity model random effect standard errors and lower and upper limits (LCL, UCL) of the 95% credible intervals for a) bald eagles and b) golden eagles. The total standard error is the square root of the sum of all the random effect variances.

(a) Bald Eagle

Random Effect	SE	Mean	SE	Median	(LCL–UCL)
Study	0.21	0.047	0.20	(0.14–0.32)	
Area	0.13	0.056	0.12	(0.05–0.26)	
Year	0.14	0.020	0.14	(0.11–0.18)	
Overdispersion	0.02	0.016	0.02	(0.01–0.06)	
Total	0.26	0.041	0.25	(0.19–0.35)	

(b) Golden Eagle

Random Effect	SE	Mean	SE	Median	(LCL–UCL)
Study	0.11	0.079	0.10	(0.00–0.30)	
Year	0.27	0.132	0.29	(0.02–0.49)	
Overdispersion	0.31	0.12	0.32	(0.07–0.51)	
Total	0.47	0.059	0.46	(0.36–0.60)	

Percent Year Effect Change

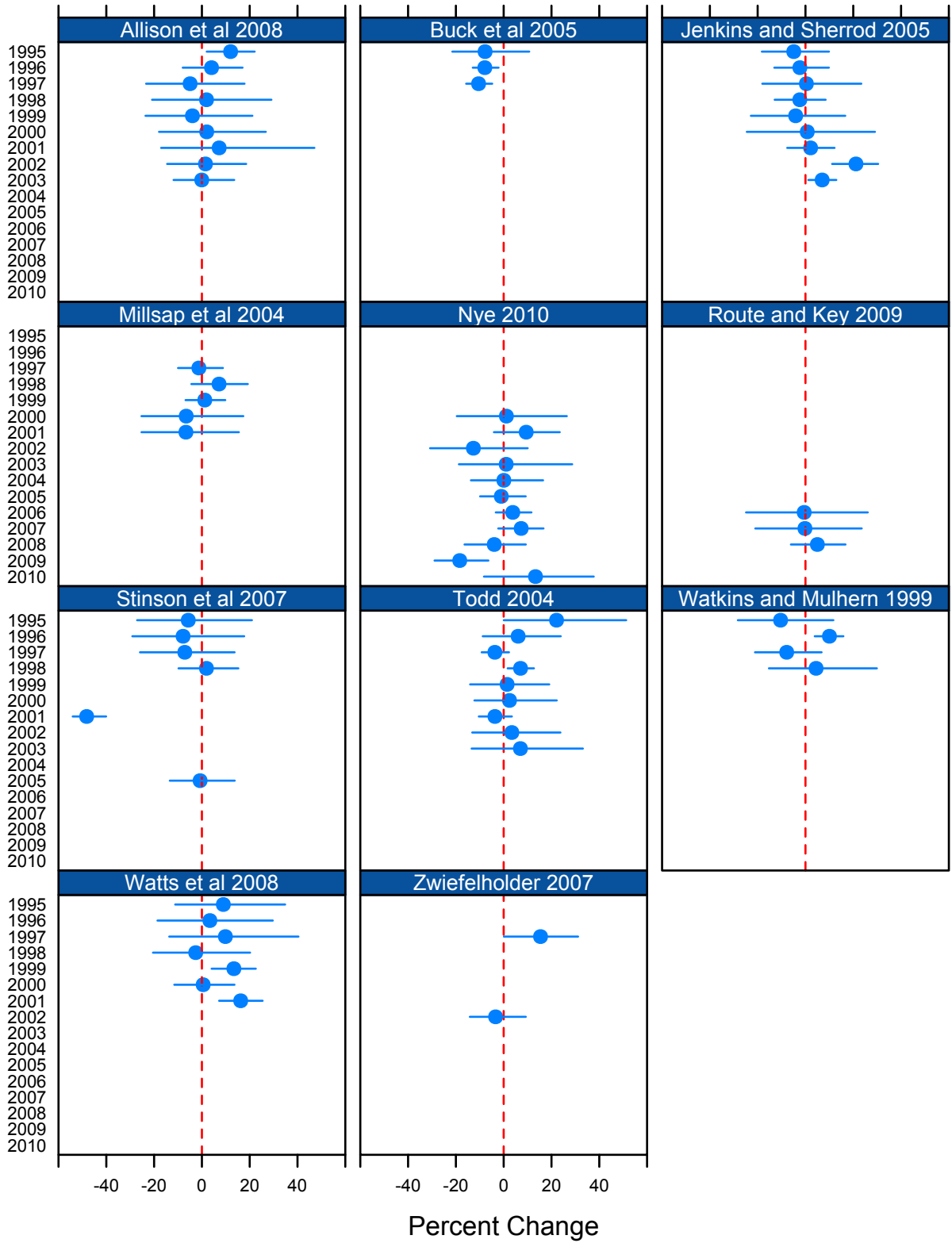


Figure A2-2. Bald eagle productivity model random effects (percent change) and 95% credible intervals due to year given study.

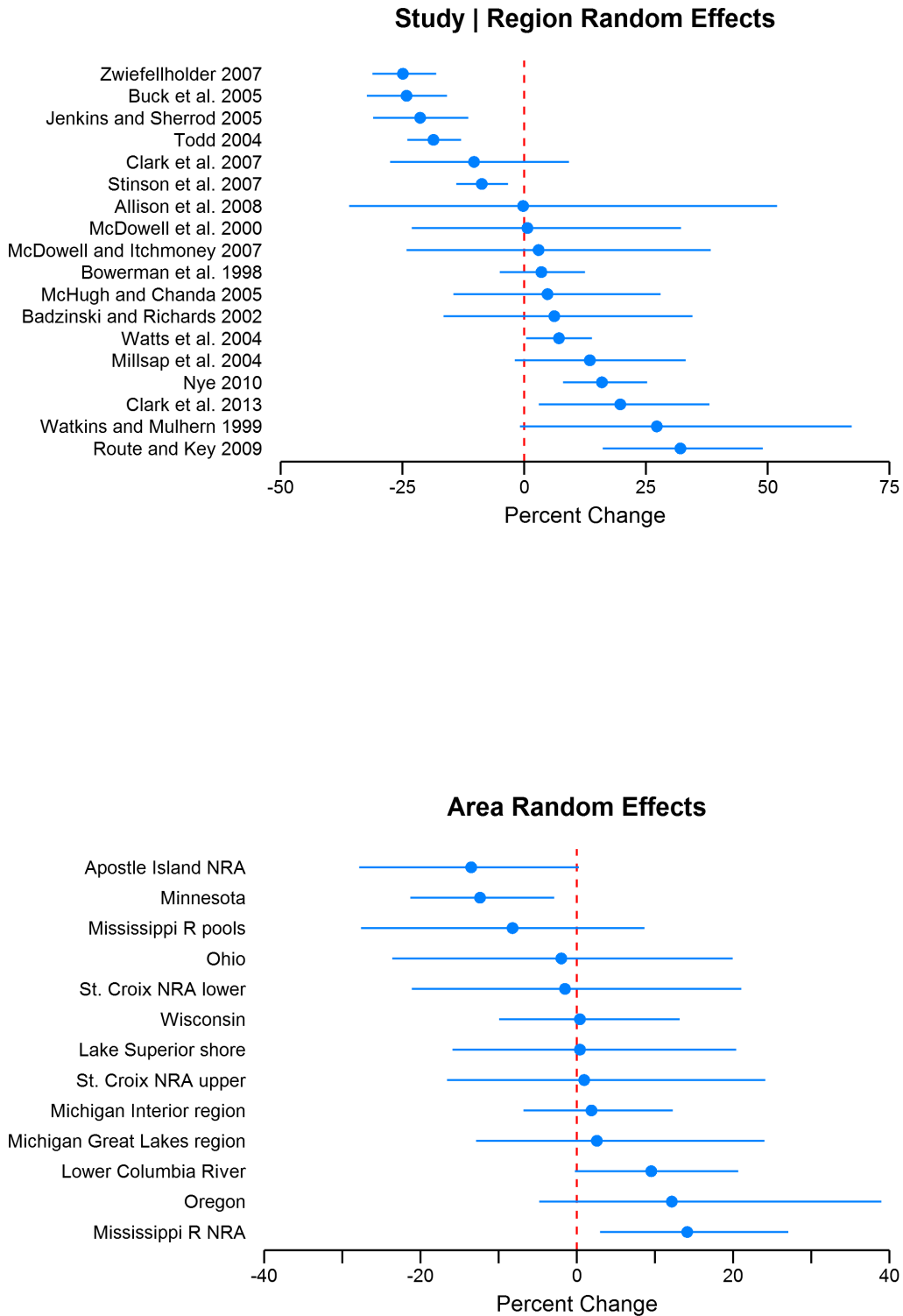


Figure A2-3. Bald eagle productivity model random effects (percent change) and 95% credible intervals due to study after accounting for regional (top) and area (bottom) differences in the model (see [Table A2-6](#) for the full list of studies with the associated region, area, and year). The effects have wide credible intervals, some of which do not overlap zero.

Table A2-4. Bald eagle productivity model median random effects (and lower and upper limits, LCL and UCL, of the 95% credible intervals) apart from region. The effect medians are presented in descending order. Fledged and nest counts are aggregated over all areas and years for each study. Ratio is the simple ratio of the total fledged to the total occupied nesting territories across all areas and years.

Study	Ratio	Median	(LCL–UCL)	Fledged	Occupied Nesting Territories
Allison et al. 2008	0.74	0.73	(0.64–0.83)	234	317
Zwiefelholder 2007	0.84	0.84	(0.78–0.92)	836	998
Buck et al. 2005	0.93	0.85	(0.76–0.94)	766	828
Jenkins and Sherrod 2005	0.88	0.88	(0.78–0.99)	241	274
Todd 2004	0.92	0.91	(0.87–0.96)	1,916	2,091
Clark et al. 2007	0.97	1.00	(0.81–1.24)	62	64
Stinson et al. 2007	1.04	1.03	(1.00–1.05)	8,074	7,784
McDowell et al. 2000	1.16	1.12	(0.84–1.49)	29	25
McDowell and Itchmoney 1997	1.21	1.16	(0.84–1.57)	17	14
Bowerman et al. 1998	1.21	1.16	(1.08–1.25)	1,817	1,497
McHugh and Chanda 2005	1.21	1.17	(0.95–1.45)	64	53
Badzinski and Richards 2002	1.24	1.20	(0.93–1.53)	41	33
Watts et al. 2008	1.26	1.20	(1.16–1.25)	4,001	3,181
Millsap et al. 2004	1.32	1.28	(1.10–1.49)	158	120
Nye 2010	1.31	1.30	(1.22–1.37)	1,540	1,178
Clark et al. 2013	1.38	1.35	(1.15–1.55)	177	128
Watkins and Mulhern 1999	1.71	1.42	(1.09–1.86)	41	24
Route and Key 2009	1.55	1.48	(1.30–1.67)	254	164

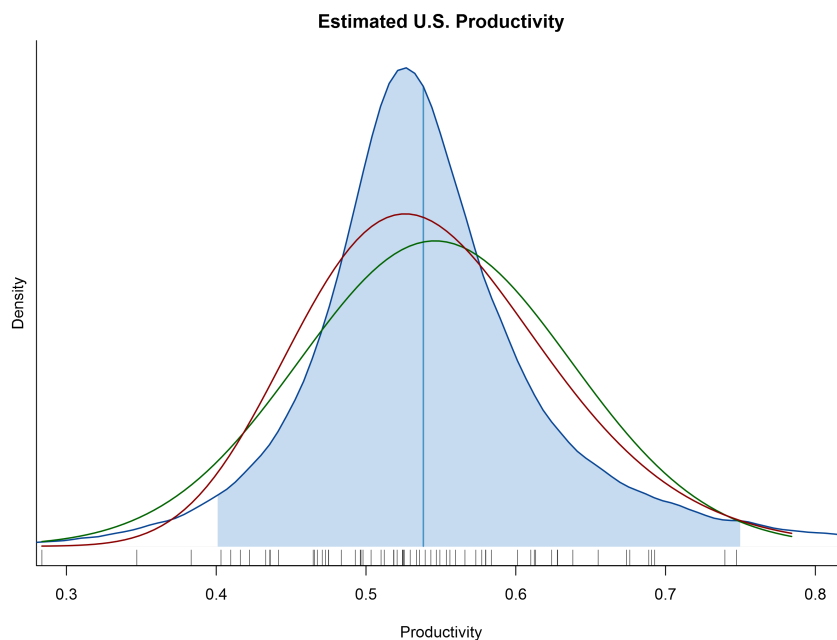


Figure A2-4. Golden eagle productivity for the U.S. The blue curve is the empirical density distribution of the estimates—which are shown via the rug just above the x-axis. The vertical blue line is the median with the area within the 95% credible intervals shaded blue. The red and green curves represent the log normal and normal distributions (respectively) defined by the estimated mean and standard deviation.

Table A2-5. Golden eagle productivity model median random effects (and lower and upper limits, LCL and UCL, of the 95% credible intervals) apart from region. The effect medians are presented in descending order. Fledged and nest counts are aggregated over all areas and years for each study. Ratio is the simple ratio of the total fledged to the total occupied nesting territories across all areas and years.

Productivity random effects	Ratio	Median	(LCL–UCL)	Fledged	Occupied Nesting Territories
Hopi Navajo 2013	0.51	0.49	(0.40–0.58)	362	715
Hawkwatch International 2009 ^a	0.50	0.51	(0.41–0.60)	257	510
Morneau et al. 2012	0.49	0.53	(0.40–0.67)	24	49
Hawks Aloft 2002	0.50	0.53	(0.41–0.68)	38	76
McIntyre and Schmidt 2012	0.61	0.53	(0.46–0.62)	692	1,140
Preston 2014	0.56	0.54	(0.44–0.65)	149	264
Hawks Aloft 2006	0.64	0.54	(0.43–0.75)	27	42
McIntyre and Adams 1999	0.61	0.54	(0.44–0.69)	112	184
Isaacs 2011	0.60	0.54	(0.43–0.73)	169	280
Berengia 2014	0.60	0.55	(0.45–0.69)	117	196
Ritchie et al. 2003	1.18	0.56	(0.45–0.87)	13	11
Hawkwatch International 2009 ^b	0.92	0.58	(0.47–0.84)	85	92

Percent Year Change Effect | Study Effects

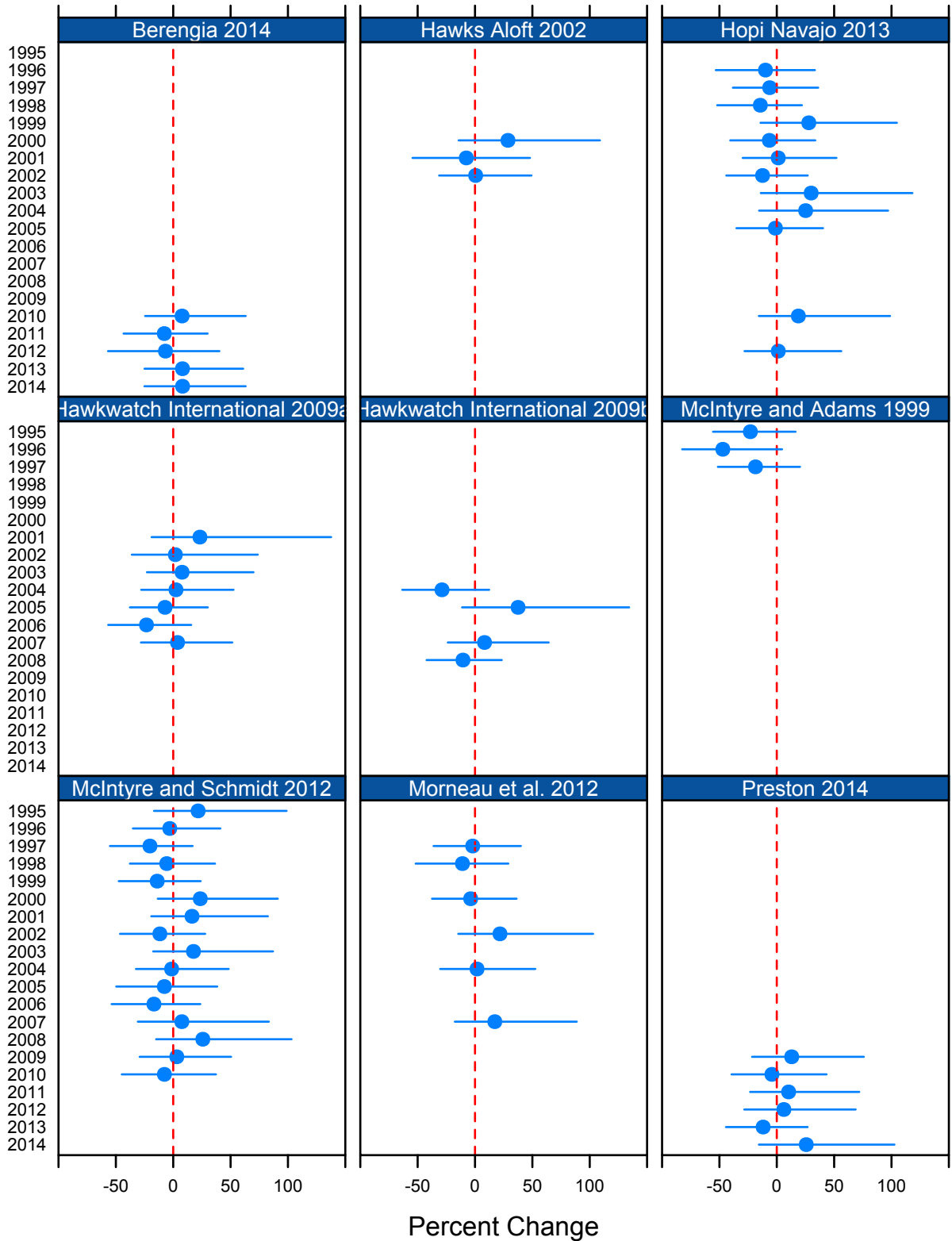


Figure A2-5. Golden eagle productivity model random effects (percent change) and 95% credible intervals due to year given study. The random effect credible intervals all overlap zero.

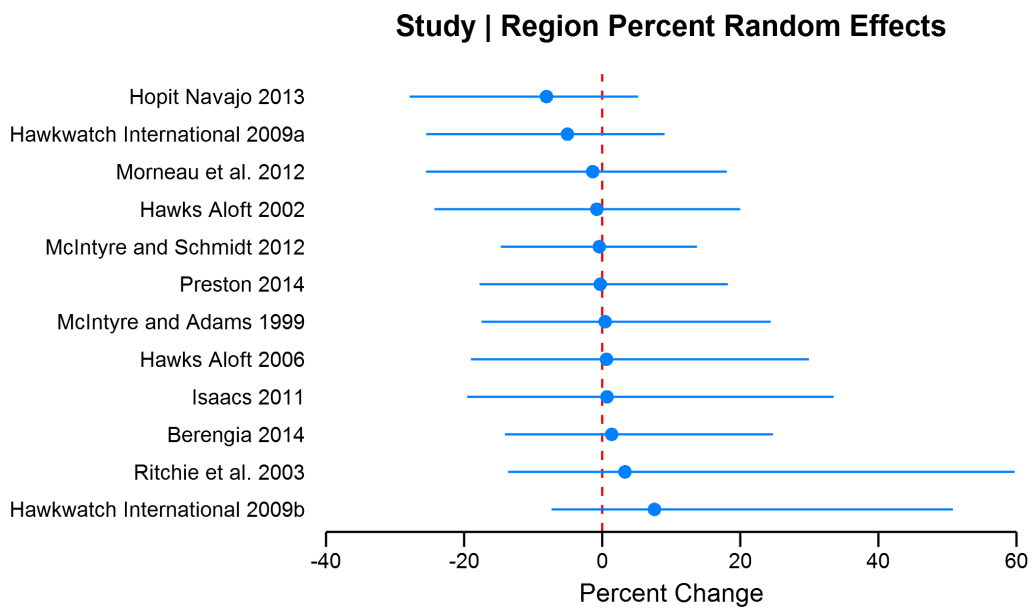


Figure A2-6. Golden eagle productivity model random effects (percent change) and 95% credible intervals due to study after accounting for regional differences in the model (see [Table A2-7](#) for the full list of studies with the associated region and year). The credible intervals all overlap zero.

Table A2-6. Bald eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each area and year combination of the studies.

Study	Region	Area	Year	Fledged	Sample Size	Productivity	Model Median
Watts et al. 2008	Other		1997	227	416	1.40	0.624
Allison et al. 2008	SW		2003	25	42	0.60	0.677
Allison et al. 2008	SW		2000	23	38	0.61	0.680
Allison et al. 2008	SW		1998	21	34	0.62	0.692
Allison et al. 2008	SW		1997	23	32	0.72	0.722
Buck et al. 2005	Other	Lower Columbia River	1997	32	54	0.59	0.737
Buck et al. 2005	Other	Lower Columbia River	1996	39	48	0.81	0.741
Allison et al. 2008	SW		1996	23	30	0.77	0.742
Todd 2004	Other		1996	141	203	0.69	0.747
Allison et al. 2008	SW		2001	28	36	0.78	0.748
Buck et al. 2005	Other	Lower Columbia River	1995	22	35	0.63	0.759
Allison et al. 2008	SW		1995	23	28	0.82	0.764
Jenkins and Sherrod 2005	Other		1996	13	25	0.52	0.778
Allison et al. 2008	SW		1999	31	36	0.86	0.780
Jenkins and Sherrod 2005	Other		1999	20	32	0.63	0.796
Allison et al. 2008	SW		2002	37	41	0.90	0.800
Zwiefelholder 2007	Other		1997	368	460	0.80	0.812
Jenkins and Sherrod 2005	Other		1998	21	28	0.75	0.836
Jenkins and Sherrod 2005	Other		1997	19	26	0.73	0.837
Zwiefelholder 2007	Other		2002	468	538	0.87	0.875
Todd 2004	Other		2000	205	234	0.88	0.884
Jenkins and Sherrod 2005	Other		2000	29	33	0.88	0.891
Todd 2004	Other		2003	273	309	0.88	0.891
Jenkins and Sherrod 2005	Other		1995	17	19	0.89	0.895
Todd 2004	Other		1995	176	192	0.92	0.916
Jenkins and Sherrod 2005	Other		2001	31	32	0.97	0.917
Stinson et al. 2007	Other		1995	509	558	0.91	0.917
Stinson et al. 2007	Other		1995	509	558	0.91	0.917
Todd 2004	Other		1998	189	202	0.94	0.933

Table A2-6. Bald eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each area and year combination of the studies. (*continued*)

Study	Region	Area	Year	Fledged	Sample Size	Productivity	Model Median
Jenkins and Sherrod 2005	Other		2002	38	38	1.00	0.945
Stinson et al. 2007	Other		1996	564	599	0.94	0.946
Stinson et al. 2007	Other		1996	564	599	0.94	0.946
Todd 2004	Other		1999	207	216	0.96	0.950
Buck et al. 2005	Other	Oregon	1997	244	248	0.98	0.960
Todd 2004	Other		2002	280	290	0.97	0.962
Buck et al. 2005	Other	Oregon	1996	215	230	0.93	0.964
Todd 2004	Other		2001	266	269	0.99	0.979
Stinson et al. 2007	Other		1997	565	574	0.98	0.986
Stinson et al. 2007	Other		1997	565	574	0.98	0.986
Buck et al. 2005	Other	Oregon	1995	214	213	1.00	0.989
Todd 2004	Other		1997	179	176	1.02	0.993
Clark et al. 2007	Other			62	64	0.97	1.005
Bowerman et al. 1998	Other	Michigan Great Lakes		81	90	0.90	1.006
Jenkins and Sherrod 2005	Other		2003	53	41	1.29	1.083
Stinson et al. 2007	Other		1998	713	648	1.10	1.097
Stinson et al. 2007	Other		1998	713	648	1.10	1.097
Stinson et al. 2007	Other		2005	925	840	1.10	1.098
Stinson et al. 2007	Other		2005	925	840	1.10	1.098
Stinson et al. 2007	Other		2001	761	673	1.13	1.127
Stinson et al. 2007	Other		2001	761	673	1.13	1.127
McDowell et al. 2000	Other			29	25	1.16	1.129
McDowell and Itchmoney 1997	Other			17	14	1.21	1.157
Bowerman et al. 1998	Other	Michigan Interior		207	176	1.19	1.168
McHugh and Chanda 2005	Other			64	53	1.21	1.174
Bowerman et al. 1998	Other	Wisconsin		694	583	1.19	1.186
Badzinski and Richards 2002	Other			41	33	1.20	1.193
Bowerman et al. 1998	Other	Ohio		38	30	1.27	1.195
Nye 2010	Other		2003	87	75	1.16	1.208

Table A2-6. Bald eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each area and year combination of the studies. (*continued*)

Study	Region	Area	Year	Fledged	Sample Size	Productivity	Model Median
Millsap et al. 2004	Other		1998	12	12	1.00	1.213
Millsap et al. 2004	Other		1998	16	12	1.33	1.213
Watts et al. 2008	Other		1998	563	462	1.20	1.216
Millsap et al. 2004	Other		1999	13	12	1.08	1.222
Millsap et al. 2004	Other		1999	15	12	1.25	1.222
Nye 2010	Other		2005	112	92	1.22	1.235
Nye 2010	Other		2007	153	124	1.24	1.245
Millsap et al. 2004	Other		2000	15	12	1.25	1.246
Millsap et al. 2004	Other		2000	15	12	1.25	1.246
Route and Key 2009	Other	Apostle Island NRA	2007	10	9	1.10	1.249
Bowerman et al. 1998	Other	Minnesota		797	618	1.29	1.276
Nye 2010	Other		2010	244	192	1.27	1.278
Millsap et al. 2004	Other		2001	13	12	1.08	1.280
Millsap et al. 2004	Other		2001	19	12	1.58	1.280
Nye 2010	Other		2009	223	173	1.29	1.286
Watts et al. 2008	Other		1996	490	377	1.30	1.291
Nye 2010	Other		2004	111	84	1.32	1.310
Nye 2010	Other		2008	190	145	1.31	1.311
Nye 2010	Other		2001	83	62	1.34	1.317
Watkins and Mulhern 1999	Other		1995	5	5	1.00	1.327
Nye 2010	Other		2002	94	70	1.34	1.329
Route and Key 2009	Other	Lake Superior shore	2007	18	14	1.30	1.330
Clark et al. 2013	Other			177	128	1.38	1.343
Nye 2010	Other		2000	71	51	1.35	1.347
Watts et al. 2008	Other		1995	464	340	1.40	1.347
Watts et al. 2008	Other		1999	650	472	1.40	1.362
Route and Key 2009	Other	St. Croix NRA upper	2007	28	19	1.50	1.372
Route and Key 2009	Other	St. Croix NRA lower	2007	6	4	1.50	1.378
Route and Key 2009	Other	Apostle Island NRA	2008	8	8	1.00	1.386

Table A2-6. Bald eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each area and year combination of the studies. (*continued*)

Study	Region	Area	Year	Fledged	Sample Size	Productivity	Model Median
Watts et al. 2008	Other		2001	849	601	1.40	1.398
Watkins and Mulhern 1999	Other		1998	11	7	1.57	1.405
Millsap et al. 2004	Other		1997	17	12	1.42	1.426
Millsap et al. 2004	Other		1997	23	12	1.91	1.426
Route and Key 2009	Other	Apostle Island NRA	2006	16	11	1.50	1.440
Watkins and Mulhern 1999	Other		1996	9	5	1.80	1.449
Watts et al. 2008	Other		2000	758	513	1.50	1.454
Route and Key 2009	Other	Mississippi River pools	2008	36	24	1.50	1.494
Nye 2010	Other		2006	172	110	1.56	1.494
Route and Key 2009	Other	Mississippi River NRA	2007	22	15	1.50	1.532
Route and Key 2009	Other	St. Croix NRA lower	2008	16	10	1.60	1.534
Watkins and Mulhern 1999	Other		1997	16	7	2.29	1.556
Route and Key 2009	Other	St. Croix NRA upper	2006	31	20	1.60	1.582
Route and Key 2009	Other	St. Croix NRA lower	2006	9	5	1.80	1.589
Route and Key 2009	Other	Mississippi River NRA	2008	30	14	2.10	1.704
Route and Key 2009	Other	Mississippi River NRA	2006	24	11	2.20	1.770

Table A2-7. Golden eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each year of the studies.

Study	Region	Year	Fledged	Sample Size	Productivity	Model Median
McIntyre and Schmidt 2012	AK	2002	4	73	0.05	0.296
Hopi Navajo 2013	West	2003	9	60	0.15	0.358
Hawkwatch International 2009a	West	2001	13	60	0.22	0.397
Hopi Navajo 2013	West	2010	6	29	0.21	0.415
Hopi Navajo 2013	West	2002	20	71	0.28	0.417
McIntyre and Schmidt 2012	AK	2003	19	71	0.27	0.420
McIntyre and Schmidt 2012	AK	2004	20	73	0.27	0.424
Hopi Navajo 2013	West	1997	7	25	0.28	0.445
Preston 2014	West	2012	15	48	0.31	0.447
Hawkwatch International 2009a	West	2003	26	78	0.33	0.448
Hawkwatch International 2009a	West	2002	23	68	0.34	0.456
Hopi Navajo 2013	West	2012	23	60	0.38	0.468
Hawks Aloft 2002	West	2002	12	33	0.36	0.474
Morneau et al. 2012	East	2007	4	14	0.29	0.474
Preston 2014	West	2011	17	44	0.39	0.480
Morneau et al. 2012	East	2002	2	8	0.25	0.481
McIntyre and Adams 1999	AK	1995	25	59	0.42	0.484
Preston 2014	West	2013	17	42	0.41	0.485
Morneau et al. 2012	East	1997	2	7	0.29	0.492
Hawkwatch International 2009b	West	2008	8	16	0.50	0.494
Hopi Navajo 2013	West	2001	35	75	0.47	0.496
McIntyre and Schmidt 2012	AK	1995	24	56	0.43	0.497
McIntyre and Adams 1999	AK	1996	30	62	0.48	0.504
McIntyre and Schmidt 2012	AK	2001	31	68	0.46	0.505
McIntyre and Schmidt 2012	AK	1996	28	61	0.46	0.506
McIntyre and Schmidt 2012	AK	1998	33	66	0.50	0.518
McIntyre and Schmidt 2012	AK	2005	38	76	0.50	0.520
Hawkwatch International 2009a	West	2004	42	84	0.50	0.520
Hopi Navajo 2013	West	1996	14	26	0.54	0.521
Berengia 2014	West	2012	21	41	0.51	0.521
Hawkwatch International 2009a	West	2007	34	67	0.51	0.522
Preston 2014	West	2014	29	54	0.54	0.530
Hopi Navajo 2013	West	1999	39	70	0.56	0.531
Hopi Navajo 2013	West	2004	43	76	0.57	0.535
Berengia 2014	West	2011	23	42	0.55	0.535
Hawks Aloft 2002	West	2000	12	22	0.55	0.539
Morneau et al. 2012	East	1998	3	5	0.60	0.540
Hawks Aloft 2006	West		27	42	0.64	0.542
Isaacs 2011	West		169	280	0.60	0.544
Hawkwatch International 2009b	West	2007	8	11	0.73	0.546
Morneau et al. 2012	East	2004	6	9	0.67	0.557
Ritchie et al. 2003	AK		13	11	1.18	0.558

Table A2-7. Golden eagle studies included in the analysis, their simple productivity ratios, and the final random effect model median estimates for each year of the studies. (*continued*)

Study	Region	Year	Fledged	Sample Size	Productivity	Model Median
Berengia 2014	West	2013	26	41	0.63	0.561
Berengia 2014	West	2014	26	41	0.63	0.564
Berengia 2014	West	2010	21	31	0.68	0.572
McIntyre and Schmidt 2012	AK	2010	49	75	0.65	0.575
Hawks Aloft 2002	West	2001	14	21	0.66	0.579
Hopi Navajo 2013	West	2005	59	84	0.70	0.583
Hopi Navajo 2013	West	2000	55	76	0.72	0.584
McIntyre and Schmidt 2012	AK	2008	52	75	0.69	0.585
McIntyre and Schmidt 2012	AK	2000	51	70	0.73	0.596
Preston 2014	West	2010	34	43	0.79	0.615
Hopi Navajo 2013	West	1998	52	63	0.83	0.621
Hawkwatch International 2009a	West	2005	67	87	0.77	0.621
Hawkwatch International 2009a	West	2006	52	66	0.79	0.624
Morneau et al. 2012	East	2000	7	6	1.17	0.633
McIntyre and Schmidt 2012	AK	1997	58	69	0.84	0.636
Hawkwatch International 2009b	West	2005	32	35	0.91	0.640
McIntyre and Schmidt 2012	AK	2007	73	81	0.90	0.658
McIntyre and Schmidt 2012	AK	2009	67	74	0.91	0.660
McIntyre and Schmidt 2012	AK	2006	76	80	0.95	0.675
McIntyre and Adams 1999	AK	1997	57	63	0.90	0.675
McIntyre and Schmidt 2012	AK	1999	69	72	0.96	0.677
Preston 2014	West	2009	37	33	1.11	0.718
Hawkwatch International 2009b	West	2004	37	30	1.23	0.732

Appendix A3. The 2009 National Bald Eagle Post-Delisting Survey and Estimation Results

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Abstract

In 2009, with assistance from many States, Tribes, and other collaborators, the U.S. Fish and Wildlife Service conducted a national-scale survey to estimate the total number of occupied bald eagle nesting territories in the coterminous U.S. The survey followed a dual-frame sampling design developed by [Haines and Pollock \(1998\)](#) that we modified to account for detectability of nest structures and tested with pilot studies in several states from 2004–2006. We estimate 16,048 (SE 727) occupied bald eagle nesting territories in the coterminous U.S. in 2009. The estimate includes 13,025 occupied nesting territories estimated with the dual-frame design (overall coefficient of variation of 6) for areas with a high abundance of nest structures, and 3,023 occupied nesting territories, which we did not estimate with the dual-frame design and consider a minimum count, in areas with a low abundance of nest structures. The dual-frame approach illustrates the difficulties of adequately estimating the true number of nests from existing nest lists and the challenges of effectively sampling across a large geographic extent. The substantial increase in the number of occupied nesting territories from previous estimates reflects improvements in sampling design and estimation, as well as increases in the overall number of breeding pairs of bald eagles.

Survey Overview

In the ‘Post-Delisting Monitoring Plan for the Bald Eagle’ (Post-Delisting Plan), the U.S. Fish and Wildlife Service (Service) describes the post-delisting survey (PDS), a national-scale survey designed to monitor the status of the bald eagle by collecting occupancy data on nest structures (nests) over a 20-year period, beginning in early 2009 with a baseline survey ([U.S. Fish and Wildlife Service 2009](#)). In this case, occupied nests are equivalent to occupied nesting territories since there is only one occupied nest structure per nesting territory (see [Steenhof and Newton 2007](#)). The baseline PDS design provides an efficient and unbiased estimate of the number of occupied nests. It was designed, in part, to meet specific objectives for detecting population change with future surveys which affected survey sampling effort. The survey design takes advantage of the efficiency of surveying known nest locations with using randomly selected area plot searches to account for nests not included on the lists, while also accounting for detectability of nests using a modification of [Haines and Pollock \(1998\)](#) dual-frame survey approach.

Dual-frame Survey Design

Many States, Tribes, or other partner organizations maintain lists of current, and often historical, bald eagle nest locations within states and the status of those nests in certain years (e.g., whether or not they were used by a nesting pair of bald eagles). The effort put toward checking the status of these known nests varies depending on the State and the resources available and ranges from intensive State-wide censuses to opportunistic checks or reports by members of the public. The recent recovery success of bald eagles

combined with budget limitations has caused many states to discontinue long-term annual nest occupancy or production surveys. The cumulative information from these nest lists, though impressive, is incomplete (not all nests were found and checked), inconsistent (information recorded differs among states), may be biased (in particular in areas where nests are found opportunistically rather than with a dedicated search effort), and frequently out-of-date in terms of occupancy information. Because of these limitations, it is difficult to tease out unbiased estimates of nest occupancy and there is only a limited ability to extrapolate beyond the lists themselves to an overall breeding population of bald eagles.

The dual frame survey design provides a means of using the existing data on the number of occupied nests available from State lists and estimating the number of additional occupied nests not accounted for by the lists. This requires sampling survey plots for occupied nests, removing known nests (those on the lists) from the samples, and estimating the number of occupied nests that are not on the lists. Those numbers are then added to the list data to get the total number of occupied nests. Complications arise in this estimation because 1) not all states have updated their lists, so additional sampling of the list nests was necessary in some areas to estimate occupancy status for the lists; 2) not all nests are seen during sampling, necessitating the estimation of detectability of nests (occupied and unoccupied) during the area sampling using a multiple-observer procedure; 3) the occupancy status of observed nests is not certain; and 4) bald eagle abundance varies greatly within the coterminous U.S. and plot surveys could only be implemented in areas where bald eagle nesting populations could be surveyed with reasonable effort. The methods described in this paper accommodate all of these issues.

In 2004–2006, USFWS and the U.S. Geological Survey (USGS) conducted several pilot studies to test the dual-frame sampling approach by combining methods traditionally used by states to monitor occupied nests, standardizing the survey protocols, and adding area-based sample plot searches for nests not included on the nest lists (U.S. Fish and Wildlife Service 2009). Area sample plots also included a multiple-observer detectability component, which allowed the analysis to include an estimate of the proportion of nests missed during the area-based plot search surveys. These pilot surveys confirmed the utility of the dual-frame design for eagle nest surveys and provided the basis for the baseline PDS design, including the expected nest list coverages (percent of the total nests represented on State nest lists) used to design appropriate sampling levels given the survey objectives (U.S. Fish and Wildlife Service 2009).

Stratification for the 2009 Survey and Analysis

Because bald eagles are still rare in some portions of the United States, the PDS was implemented within strata defined by physiographic regions (Figure A3-1). Strata sampled as part of the dual frame survey design were only in regions where eagles are relatively abundant (Table A3-1). The final strata were based on a cluster analysis of bald eagle nest densities within Bird Conservation Regions (BCRs) and States with additional editing in consultation with survey biologists to incorporate knowledge of local natural history and habitat information (see U.S. Fish and Wildlife Service [2009] and U.S. North American Bird Conservation Initiative Committee [2000]). Area strata may include all or part of multiple State list frames since nest lists are nearly always maintained at individual State levels (Table A3-1). State list-frames are thus divided and combined as appropriate to create a strata-specific list-frame. The strata list-frame estimate is then combined with the area-frame survey estimate (corrected for detectability of nests based on multiple observer procedures) to compute the dual-frame estimates by strata.

Methods

Sample Unit Selection and Sample Size

We selected sample plots for the survey using the Generalized Random Tessellation Stratification (GRTS) methodology (Kincaid and Olsen 2013) as described in the Post-Delisting Plan. The dual-frame sample is

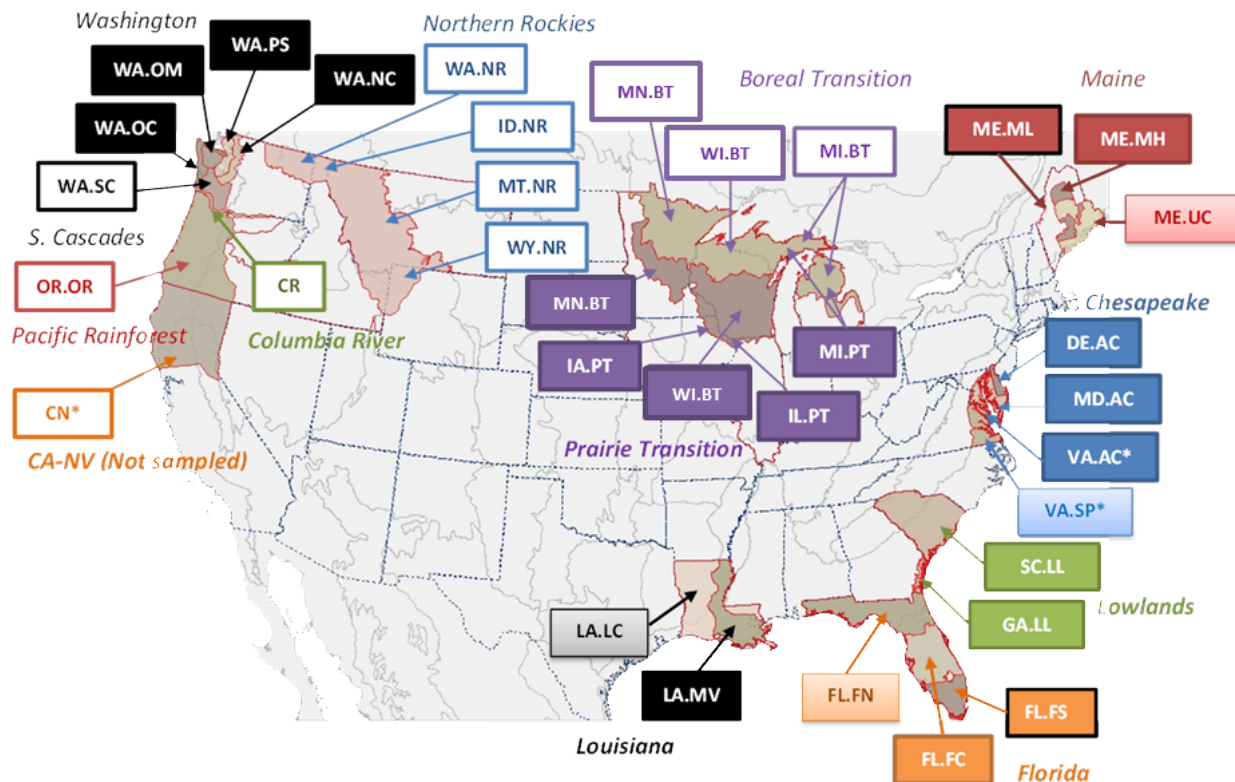


Figure A3-1. Survey strata for the 2009 bald eagle post-delisting survey. We defined the strata for area plot samples in terms of Bird Conservation Regions (U.S. North American Bird Conservation Initiative Committee 2000) and densities of known nests. We further subdivided the area-frame strata to accommodate differences in nest lists maintained by the component states or to meet information needs of cooperators. Geographic areas are thus identified as AA.BB, where AA defines the list sample frame and BB defines the area sample frame. We did not sample light grey areas with area plot surveys and estimated occupied nests only from list frame data.

based on an area sample that is augmented with information from known nest locations (the list). Because list information is often not current, additional sampling may be required to estimate the total number of occupied nests in the list. Here, we used GRTS to select plots for the area sampling, but also selected plots to estimate the number of the occupied nests in the list. Sample size determination for both the area sampling and list sampling are described in detail in the Post-Delisting Plan; list sampling included any known (list) nests included in selected area sample plots and the additional list sample plots. In list sample plots we only assessed the status of known nests, whereas in area plots we searched the entire plot area for any previously unknown (new) nests and also assessed the occupancy status of any known nests (U.S. Fish and Wildlife Service 2009). Known nests were removed from the area sample and made part of the list sample prior to data analyses. We drew a 20% over-sample for each sampling frame by strata; these were replacements for selected plots that could not be flown for a variety of access, safety, or logistical issues.

Surveys

We conducted survey flights in the early part of the breeding season when the majority of bald eagle breeding pairs are closely tied to nests and detectability should be highest (before leaf-out). Survey crews recorded GPS locations and nest status observations following protocols described in the Post-Delisting Plan. Post-survey, we used flight tracks and nest information to reconcile nest observations with known nests and

Table A3-1. Geographic sampling areas with the associated list frame (LL), area frame (AA), dual-frame (LL.AA) and a short locality description based on the associated Bird Conservation Region (BCR).

Sampling Area	List Frame	Area Frame	Dual Frame	BCR Description
Northwest (NW)	WA	SC	WA.SC	South Cascades
	WA	WC	WA.WC	Olympic Sound & Northeast Cascades
	CR	CR	CR.CR	Columbia River
	OR	OR	OR.OR	Pacific Rainforest
Northern Rockies (NR)	NR	NR	NR.NR	Northern Rockies
Great Lakes (GL)	WI		WI.BT	Boreal Hardwood Transition
	MI	BT	MI.BT	
	OT		OT.BT	
	WI	PT	WI.PT	Pine Hardwood Transition
	OT		OT.PT	
Louisiana (LA)	LA	MV	LA.MV	Mississippi Alluvial Valley
	LA	LC	LA.LC	Western Gulf Coastal Plain
Maine (ME)	ME	MH	ME.MH	Maine Upper Middle Coast & Highlands
	ME	ML	ME.ML	Maine including Aroostook
	ME	UC	ME.UC	Maine Down East
Chesapeake Bay (CB)	CH	CH	CH.CH	Chesapeake (Mid-Atlantic Coast)
Coastal Plain (CP)	SC		SC.LL	Southeastern Coastal Plain
	GA	LL	GA.LL	
Florida (FL)	FL	FC	FL.FC	Central Florida
	FL	FN	FL.FN	Northern Florida

determine the sampling frame assignment and status for each nest.

Estimating Occupied Nests

List Estimation

Due to the variation in list quality, we used several different methods to determine the number of occupied nests in the lists. Some states conducted a census of their lists in 2009 and could provide the number of known, occupied nests directly; in other states list sampling was conducted by the Service in conjunction with the area plot surveys to estimate the proportion of nests in the lists that were occupied. Finally, some states only record occupancy by territory rather than by nest, requiring additional assumptions (e.g., that other known nests within the territory were unoccupied) or adjustments in order to estimate the total number of occupied nests.

The proportion of occupied nests is:

$$p_i^{Occ} = \frac{\sum_i^{n_i^L} y_{ij}^L}{n_i^L},$$

where n_i^L is the number of sampled list nests for the i^{th} stratum. To simplify the estimate, we are ignoring that the list sample unit is a plot with list nests, rather than the nests themselves. y_{ij}^L is a variable to indicate if the j^{th} nest in the stratum is occupied. The total number of occupied nests is:

$$Y_i^L = N_i^L p_i^{Occ} ,$$

where the proportion of occupied nests is multiplied by N_i^L , the total number of list nests in the i^{th} stratum. The variance is:

$$Var(Y_i^L) = N_i^L (N_i^L - n_i^L) \frac{(S_i^L)^2}{n_i^L} ,$$

where sample standard deviation of the proportion is:

$$S_i^L = \sqrt{\frac{n_i^L}{(N_i^L - 1)} p_i^{Occ} (1 - p_i^{Occ})} .$$

Finally, the stratum estimates and variances are added to get the survey total occupied list nests and its variance. The square root of the variance is the standard error of the total occupied list nests.

Area Estimation: Correcting for Detection Probability With Multiple Observers

We used a capture-recapture based approach for estimating detectability of nests by observers when surveying plots. For each nest naively observed (that is the observers did not have knowledge of the nest location at the time they recorded the observation), we used information about which observers detected the nest to create a capture history (in this context “capture” is an observation state). The capture histories provided sufficient information to estimate detection as described in the Post-DeListing Plan. Pilots were included in the capture histories even though they were not observing at all times by including a “not looking” code in the capture histories in addition to the ‘detected’ and ‘did not detect’ codes. Recorded ‘not looking’ observations were considered non-detections for that observer in the analysis. We did not record capture histories for nests where the observers were aware of the nest location prior to their observations.

We developed a Bayesian capture-recapture model with non-informative priors based on [Link and Barker \(2010\)](#) in JAGS ([Plummer 2003](#)) to estimate detection probabilities among two or three independent observers. We summarized capture histories by combinations of observers in the pilot, front and rear seats ([Table A3-2](#)). Captures have three possible states: seen (1), not seen (0), and not looking (x). Capture histories for each nest observation are presented as 3-tuples (pilot observation, front seat observation, rear seat observation) and can have seven possible values—for example: a nest seen only by the front observer (010), only by the rear observer (001), or by all of the observers (111). The model accounted for observer, seat (pilot, front, or rear), and platform (fixed-wing airplane or helicopter) as random-effects off an overall mean, μ , on the logit scale:

$$\text{logit}(p_{ijk}^{det}) = \mu + \text{platform}_i + \text{seat}_j + \text{observer}_k$$

Each platform-observer-combination was fit with a multinomial for seven combinations of detections. The multinomial fits the number of capture histories of a given observer combination in a given platform and a given seat position. The probabilities are conditioned on the nest being observed. We estimated the probability of at least one of the three observers seeing a nest, irrespective of nest status, for each platform and observer combination (i.e., group-specific detection rates), along with 95% credible estimates. We then adjusted nest counts for detectability using the estimated detection probabilities for the observer-seat-platform combination that surveyed each plot.

Table A3-2. Survey capture history summaries and estimated detection probabilities by observer combination, seat, and platform. Seat positions include, in order, pilot, front seat observer (Front), and rear seat observer (Rear). Survey platform is either fixed-wing plane (FW) or helicopter (Hel). ‘0’ in the capture history indicates the nest was ‘not seen’ by the observer and a ‘1’ indicates ‘seen’. Detection probability is the probability at least one of the observers (given observers, platform and seats) would see a nest, and includes the estimated standard error.

Platform	Pilot	Front	Rear	001	010	011	100	101	110	111	Detection	
											Probability	SE
FW	JKB	DED	CST	7	1	2	0	0	0	0	90	3.0
FW	JKB	CST	DED	9	28	20	0	0	6	9	91	2.9
FW	JKB	CST	KS	0	2	0	0	0	0	0	89	4.3
FW	VRB	DJ	DSP	3	1	12	8	0	2	6	90	3.5
FW	VRB	DSP		0	0	0	1	0	0	0	80	4.7
FW	BBD	SAN	JHW	8	4	4	5	13	1	15	97	0.8
FW	MDK	CAK	JK	0	6	3	4	1	1	1	85	4.0
FW	MDK	HHO	RDR	3	1	5	6	1	4	1	78	5.6
FW	TSL	CAK	JGM	0	1	1	0	0	0	2	86	3.9
FW	TSL	CAK	RDR	10	21	6	2	2	4	3	82	3.5
FW	TSL	JCO	JB	0	1	1	0	0	0	0	85	10.1
FW	FHR	SPE	JGM	1	5	2	0	0	2	6	94	3.2
FW	FHR	GEM	TJH	2	0	0	1	0	1	2	89	5.5
FW	FHR	GEM	CWJ	2	7	2	1	0	1	2	85	5.4
FW	FHR	JGM		0	0	0	0	0	0	0	76	7.3
FW	FHR	MCO		0	3	0	8	0	5	0	56	7.1
FW	FHR	MS	MCO	1	1	1	1	0	0	1	76	9.2
FW	WER	CH	GR	0	2	2	0	0	0	0	85	10.4
FW	WER	CAK	ERB	1	15	4	2	0	0	0	80	3.8
FW	WER	CAK	KH	0	0	3	0	0	0	0	87	6.7
FW	WER	CAK	RDR	4	11	9	0	0	0	0	78	3.9
FW	DJS		SMP	2	0	0	0	0	0	0	64	18.2
FW	JWS	ERB		0	3	0	3	0	2	0	66	6.9
FW	JWS	JWG	ERB	4	4	3	1	0	1	1	82	6.0
FW	PPT	DSB	NK	0	5	2	1	0	0	1	84	5.6
FW	PPT	DSB	RM	4	0	2	0	0	0	1	93	3.3
FW	PPT	DSB		0	6	0	1	0	2	0	77	6.8
FW	PPT	NK	DSB	0	0	1	1	0	0	0	83	6.1
FW	PPT	RM	DSB	3	4	13	1	0	0	0	93	3.4
FW	RT	SAN	JHW	3	4	8	2	7	2	31	98	0.7
Hel	MBS	DSP	ERB	6	13	24	3	0	0	5	96	1.1
Hel	MBS	DSP	VRB	2	2	2	3	0	7	27	97	0.8
Hel	MBS	DSP	MCO	1	13	4	3	0	4	3	94	1.5
Hel	MBS	DSP		0	14	0	5	0	25	0	90	2.0

Area Estimation: New Occupied Nest Estimation

We removed any known nests (list nests) that were located in area survey plots from the area-frame data set prior to the analysis in order to ensure independence of the two data frames. The area-frame estimates thus represent the number of new occupied nests. By ‘new’, we mean new to the list, not necessarily newly constructed (e.g., one could observe a well-established nest that is clearly ‘old’ but previously unknown, especially in areas not traditionally included in State surveys). New nests found in 2009 were added to the list frame for future surveys.

Thompson (1992, pg. 168 equation 9) provides formulas for estimating population size under simple random sampling, accounting for detection uncertainty which can be applied independently to each stratum. The estimator assumes detection is constant over the stratum, however, detection can vary by observer, seat, and platform. To estimate population (nest) totals, we obtained the mean or density of occupied new nests and the variance for each stratum given a detection probability,

$$\bar{y}_i^N | p^{Det} = \left(\sum_j^{N_i^A} W_{ij} \right)^{-1} \sum_j^{N_i^A} \frac{w_{ij} y_{ij}^N}{p_{ij}^{Det}}$$

$$Var(y_i^N | p^{Det}) = \frac{n_i^A}{(N_i^A - 1)} \left(\sum_j^{N_i^A} w_{ij} \right)^{-1} \sum_j^{N_i^A} (y_{ij}^N - p_{ij}^{Det} (\bar{y}_i^N | p^{Det}))^2,$$

where the weights are the ratio of the total number of plots in the stratum over the number of sample plots, $w_{ij} = N_i^A / n_i^A$ for all j . Then, we used Monte Carlo integration (Givens and Hoeting 2005, pg. 144) over the distributions of the detections to get the unconditional population densities and variances using formulas 1.6 and 1.7 in Gelman et al. (1995, pg. 20). The distributions of the detections were simulated from the Bayesian model by platform-observer-seat combination.

The totals and variances were obtained by expanding the densities,

$$\bar{y}_i^N = E_{p^{Det}} y_i^N = E_{p^{Det}} (\bar{y}_i^N | p^{Det}) = E_{p^{Det}} (E(y_i^N | p^{Det}))$$

$$Var(y_i^N) = Var_{p^{Det}} (E_{p^{Det}} (y_i^N | p^{Det})) + E_{p^{Det}} (Var(y_i^N | p^{Det}))$$

which were applied to get the unconditional totals and variances. We report the standard errors by taking the square root of the variances.

$$Y_I^N = N_i^A \bar{y}_i^N$$

$$Var(Y_i^N) = N_i^A (N_i^A - n_i^A) \frac{Var(y_i^N)}{n_i^A},$$

Area Estimation: Area-Only Estimation

We also generated an estimate of the area-plot counts (as though there were no list data) for comparison. We used all nests observed in the area plots (including list nests) and the observer-detection correction to calculate the totals and standard errors as described above and compared estimates to the list-only and dual-frame estimates. Applying the observer-detection correction to the known list nests may make estimates slightly high since detection rates may be higher than for unknown nests. It is also important to include an equivalent number of sampled area plots relative to the effort put into the list sample in order to make a reasonable comparison between the estimates.

Dual-Frame Estimation

Since the estimates of occupied, known nests from the list frame and the estimates of occupied, new nests from the area-frame are independent, we sum the totals and their variances to get the dual-frame estimates (see [U.S. Fish and Wildlife Service \[2009\]](#) for additional details). We added the list-frame and area-frame stratum estimates to get the national total following [Haines and Pollock \(1998\)](#):

$$\hat{Y}_{DF} = \sum_i^{I_L} \hat{Y}_{Li} + \sum_i^{I_A} \hat{Y}_{Ni}$$

We similarly added the variances to get the variance of the total,

$$Var(\hat{Y}_{DF}) = \sum_i^{I_L} Var(\hat{Y}_{Li}) + \sum_i^{I_A} Var(\hat{Y}_{Ni})$$

List Coverage

The list coverage is the percent of the total estimated nests (irrespective of occupancy status) that were represented in the list frame; it highlights the relative number of nests missing from the list. The number of new nests is estimated using

$$(List\ Coverage\ \%)_i = \frac{N_i^L}{N_i^L + \hat{N}_i^N}$$

The number of nests in the list is fixed, but the number of new nests is estimated. We simulated the new nest distribution with a log normal with the log mean of the new nest estimate and its relative variance. The means and standard errors are taken from simulations of the above ratios.

Results and Discussion

The 2009 PDS was designed to 1) estimate the total number of occupied nesting territories for bald eagles in the coterminous U.S. using occupied nests as a measure of occupied nesting territories, 2) ensure estimates are unbiased and account for uncertainty, and 3) collect data such that the estimates allow for detection of population decline with future surveys as specified in the Post-DeListing Plan. We achieved these objectives using a dual-frame analysis to estimate the number of occupied nests and the uncertainty of the estimates. This approach allowed us to combine information from State nest lists (the list-frame), which was efficient to sample but was not standardized across States and can be biased, with information about nests that are not represented on the lists (the area-frame), which was less efficient to collect but used standardized protocols and accounted for detectability of nests. This approach also allowed for a direct comparison of estimates based solely on sampling the list-frame or the area-frame to the dual-frame estimates. Stratification using known nest densities and physiographic boundaries helped ensure representative and efficient sampling and estimation which are important given the PDS goals and highly variable bald eagle abundance within the U.S.

List-Frame Estimates

For list-frame estimates, when appropriate, we used State survey data for list nests rather than list-frame sample data collected specifically for the PDS. Often States surveyed a larger proportion of their total list and, in cases where we used State survey data, the observed proportion of list nests that were occupied was comparable to what we observed in the PDS sample ([Table A3-3](#)). We included observations of 7,461 nests

Table A3-3. List-frame estimates and nest observation totals of occupied bald eagle nests by strata. [Table A3-1](#) describes the strata. 'Survey Data' indicates whether the data are from a State survey or the post-delisting survey.

Sampling Area	Area Strata	List Strata	Survey Data	Occupied Seen	Total Seen	Occupied (%)	SE	Total Not Seen	Total Occupied	SE	CV (%)
NW	SC	WA	PDS	7	11	62	13	513	103	20.5	20
	WC	WA	PDS	32	87	37	5	2,277	873	117.0	13
	CR	CR	State	117	275	43	3	150	181	4.5	2
	OR	OR	PDS	57	130	44	4	1,181	575	51.0	9
NR	NR	NR	PDS	14	18	76	10	645	506	61.3	12
GL		WI	State	683	2,100	33	1	583	873	6.0	1
	BT	MI	State	528	2,104	25	1	230	586	2.2	0
		OT	PDS	22	54	41	7	1,069	459	70.2	15
	PT	WI	State	383	732	54	2	184	492	3.4	1
		OT	PDS	5	11	46	14	610	285	84.3	30
LA	MV	LA	PDS	25	36	69	8	448	334	33.6	10
	LC	LA	PDS	1	5	25	46	25	7	4.1	56
ME	MH	ME	State	44	127	35	4	23	52	1.0	2
	ML	ME	State	48	91	53	5	2	49	0.1	0
	UC	ME	State	408	1,145	36	1	60	429	0.8	0
CB	CH	CH	PDS	36	79	46	6	1,011	497	56.0	11
CP		SC	State	252	294	86	2	4	255	0.1	0
	LL	GA	State	42	62	67	6	0	42	0.0	0
FL	FC	FL	PDS	57	85	67	5	1,279	912	64.5	7
	FN	FL	PDS	10	15	66	12	599	403	69.0	17

Table A3-4. Estimates of new, occupied nests by area-frame strata. Plot densities are not corrected for observer detection, but new occupied nest totals are corrected for detection by the observer-seat-platform combination used to sample the plot.

Region	Area Stratum	New Occupied	Sampled Plots	Plot Density	SE	Total Plots	Total New Occupied	SE	CV (%)
NW	SC	1	5	0.20	0.20	203	42	40.5	96
	WC	10	18	0.56	0.15	452	262	65.7	25
	CR	1	6	0.17	0.17	115	20	19.1	93
	OR	3	22	0.14	0.07	1,244	127	81.6	64
NR	NR	2	14	0.14	0.10	2,603	401	250.7	62
GL	BT	13	27	0.48	0.17	2,549	1,614	461.3	29
	PT	3	9	0.33	0.17	1,775	660	291.8	44
LA	MV	9	15	0.60	0.32	610	429	198.3	46
	LC	0	5	0.00	0.00	849	0	0.0	0
ME	MH	1	13	0.08	0.08	268	23	20.6	90
	ML	0	7	0.00	0.00	418	0	0.0	0
	UC	1	15	0.07	0.07	411	31	27.4	90
CB	CH	33	33	1.00	0.30	570	702	172.4	25
CP	LL	4	17	0.24	0.14	991	280	139.6	50
FL	FC	8	19	0.42	0.12	613	264	71.4	27
	FN	5	16	0.31	0.15	801	257	120.6	47

(2,781 occupied nests) out of 17,994 known nests on State nest lists for the high-density strata. Occupancy rates ranged from 25–86% among strata and averaged approximately 50%. Within strata, we extrapolated the occupancy rates and standard errors for the sampled nests to the nests not included in the sample to get the total occupied nests for the list-frame—7,913 (SE 727) occupied nests.

Area-Frame Estimates

Table A3-4 shows the total new nests observed in area plots by stratum and the resulting plot densities which ranged from 0–1 new nests per plot (average 0.3). We applied detection probabilities that accounted for observer combination, seat, and platform to the plot densities to get the area-frame estimates. New nest estimates ranged from 0 new nests in parts of Maine and Louisiana where nest lists were actively updated through 2009 to 1,614 new nests in the Boreal Transition area, which is the northern part of the Great Lakes region.

Detection rates varied by observer-seat-platform combination with the probability of at least one observer detecting a nest ranging from as low as 56% (SE 7.1%) to as high as 98% (SE 0.7%). Platform had a noticeable impact on detectability, with helicopter crew detection rates of 90% (SE 0.8%) to 97% (SE 1.5%). This was consistent with the general impression of survey crews that searching for nest in helicopters was more accommodating in terms of general maneuverability and visibility in spite the helicopters often being used for plots with habitat that would be considered to have lower detectability (conifer-dominant tree stands).

Table A3-5. List-frame (List), area-frame (Area), and dual-frame (DF) estimates of occupied nests and standard errors by stratum for the 2009 National post-delisting survey. The list-frame estimates are known nests, the area-frame estimates are only new nests (known nests excluded), and the dual-frame estimates combine both new and known nests from the two independent sampling frames.

Region	Area Stratum	List	SE	Area	SE	DF	SE
Northwest	SC	103	20.5	42.0	40.5	145	45.4
	WC	873	116.5	262.0	65.7	1,135	133.7
	CR	181	4.5	20.0	19.1	201	19.6
	OR	575	51.0	127.0	81.6	702	96.2
Northern Rockies	NR	506	61.3	401.0	250.7	907	258.1
Great Lakes	BT	1,918	70.5	1,614.0	461.3	3,532	466.7
	PT	776	84.4	660.0	291.8	1,436	303.8
Louisiana	MV	334	33.6	429.0	198.3	762	201.1
	LC	7	4.1	0.0	0.0	7	4.1
Maine	MH	52	1.0	23.0	20.6	75	20.6
	ML	49	0.1	0.0	0.0	49	0.1
	UC	429	0.8	31.0	27.4	460	27.4
Chesapeake	CH	497	56.0	702.0	172.4	1,200	181.3
Coastal Plain	LL	297	0.1	280.0	139.6	577	139.6
Florida	FC	912	64.5	264.0	71.4	1,177	96.2
	FN	403	69.0	257.0	120.6	660	139.0

Dual-Frame Estimates

The dual-frame estimates represent the total occupied nests for each stratum (Table A3-5), and ranged from only 7 occupied nests in the Western Gulf Coastal Plain of Louisiana to 3,532 occupied nests in the Boreal Transition area in the northern portion of the Great Lakes region (SE 4.1 and 466.8, respectively). The overall estimate for all of the high-density strata combined was 13,025 occupied nests (SE 727). We combine this with the minimum number of occupied nests for the low-density strata to get the total national estimate of occupied nests for the coterminous U.S.

The Low-Density Strata

The occupied nest estimates for the low-density strata are based entirely on the list data for the States (or portions of States) included geographically in the strata (Figure A3-1). We totaled the number of occupied nests on each of the State nest lists that best represented the number of occupied nests in 2009 (Table A3-6). There were 3,023 additional occupied nests in the low-density portions of the coterminous U.S. which when added to the 13,025 occupied nests estimated for the high-density areas, yields a national total of 16,048 occupied nests or breeding pairs of bald eagles in 2009.

Area-Only Estimates

To evaluate the efficacy of the dual-frame survey and better understand the contribution of each sample frame to the analysis, we also analyzed the area survey plot data as though we only conducted the area

Table A3-6. The recorded number of occupied nests (considered minimums) for low-density areas (areas in the coterminous U.S. not included in the high-density strata in [Figure A3-1](#)). Reporting years other than 2009 are shown in parentheses. ‘Low’ after the State name indicates the low-density portion of a state partially included in a high-density area.

State List	Occupied Nests	State List	Occupied Nests
Alabama (2006)	77	Nevada	3
Arizona	50	New Hampshire	11
Arkansas (2008)	110	New Jersey	69
California	64	New Mexico	5
Colorado	51	New York	173
Connecticut (2010)	18	North Carolina	113
Florida South	120	North Dakota	67
Idaho Low	84	Ohio	151
Illinois Low	59	Oklahoma	72
Indiana	194	Oregon Low	20
Iowa Low	199	Pennsylvania	173
Kansas	33	Rhode Island	1
Kentucky	56	South Dakota (2012)	128
Massachusetts (2010)	30	Tennessee	130
Michigan Low	77	Texas (2005)	160
Minnesota Low	61	Utah	11
Mississippi	31	Vermont	3
Missouri (2011)	165	Washington Low	18
Montana Low	97	West Virginia (2010)	36
Nebraska	48	Wisconsin Low	5
		Wyoming Low (2005)	50
		Total	3,023

Table A3-7. ‘Area-only’ estimates of occupied nests, which use only information collected during area plot surveys (assumes no prior knowledge of nest locations from nest lists), by area-frame stratum.

Region	Area Stratum	Occupied	SE	CV (%)
Northwest	SC	169	162	96
	WC	969	224	23
	CR	84	38	45
	OR	1,221	619	51
Northern Rockies	NR	2,462	980	40
Great Lakes	BT	3,110	744	24
	PT	869	302	35
Louisiana	MV	1,002	507	51
	LC	0	0	0
Maine	MH	115	65	56
	ML	0	0	0
	UC	515	150	29
Chesapeake	CH	1,477	293	20
Coastal Plain	LL	349	147	42
Florida	FC	1,288	333	26
	FN	462	146	32

Table A3-8. The overall list-only, area-only, and dual-frame estimates of occupied nests and standard errors for the high-density strata.

Type	Occupied Nests	SE	CV (%)
List-only	7,913	214	3
Area-only	14,091	1,610	11
Dual Frame	13,025	727	6

Table A3-9. The number list nests (List), new nests (New), and percent list coverage (proportion of the total estimated nests that were represented on the nest list) by area-frame stratum. The list nest and new nest columns include all nests, regardless of occupancy status. We present the means and standard error for each stratum.

Area	Area Stratum	List	New	List Coverage (%)	SE
Northwest	SC	164	42	76	15.5
	WC	2,364	367	86	3.0
	CR	425	42	90	5.3
	OR	1,311	127	90	6.0
Northern Rockies	NR	663	823	45	9.4
Great Lakes	BT	6,140	2,621	70	6.2
	PT	1,537	1,352	53	10.4
Louisiana	MV	484	524	48	10.8
	LC	30	0	100	0.0
Maine	MH	150	23	83	11.5
	ML	93	0	100	0.0
	UC	1,205	31	96	3.3
Chesapeake	CH	1,090	1,001	52	4.9
Coastal Plain	LL	360	280	56	11.6
Florida	FC	1,364	264	83	3.7
	FN	614	257	70	9.5
Overall		17,994	7,752	68	3.1

plot survey; we refer to this as the ‘area-only’ analysis (Table A3-7). The area-only estimates are generally consistent with the dual-frame estimates but with much greater uncertainty around the estimates (Table A3-8). The greatest disparity in the area-only and dual-frame estimates occurs in strata where we were able to include data from a large sample of the nest list (e.g., the Columbia River strata [CR] in the Northwest). There is also disparity where no occupied nests were detected in the sample plots, as in parts of Maine (ML), though this is an artifact of the estimation of a zero variance where there should be some chance of a non-zero estimate even when the estimate itself is zero. In all strata, however, the inclusion of the list frame in the dual-frame analysis generally reduced the uncertainty in the estimates by 5%.

List Coverage

By estimating the total number of nests, regardless of status, using the same dual-frame approach for the high-density strata we can also evaluate how well State nest lists represent the true number of nests on the landscape (Table A3-9). The percent of the total estimated nests that were included on the nest lists, the list coverage, ranged from 48–100%, with 68% coverage overall (SE 10.8%, 0, and 3.1%, respectively). Known nest locations are generally efficient to sample but nest lists can be problematic since the lists are often not constructed using unbiased or geographically representative sampling and therefore the lists tend to be biased in unpredictable ways. A good example of this is Louisiana, where we found no new nests in the Western

Gulf Coastal Plain (100% list coverage) but estimated 52% more nests in the Mississippi Alluvial Valley strata. The State nest list provided a better representation of the numbers of nests in certain areas of the State than others, however, without the area plot sampling data to reveal the bias, there would have been no way to account for the difference.

Based on data from many other sources such as the USGS Breeding Bird Survey (BBS, Sauer et al. 2014) and many State monitoring reports, bald eagle populations continue to thrive in many areas. This survey was intended to provide a greatly-improved estimate of occupied nests to inform on-going management assumptions and efforts, but also to serve as an important baseline for comparison with similar estimates from future monitoring efforts. We estimated a total of 16,048 bald eagle breeding pairs in the coterminous U.S. in 2009. These numbers are higher than previous estimates, but are not out of scale with the levels of population increase seen with the BBS.

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Appendix A4. Updated Golden Eagle Population Size Estimate in the Western U.S.

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Summary

Millsap et al. (2013) derived population size estimates for golden eagles in 12 Bird Conservation Regions (BCRs) throughout the western US. In their analysis they combined indices from the breeding bird survey (BBS, Pardieck et al. 2015) and population size estimates from the Western EcoSystems Technology summer golden eagle survey (WGES, Nielson et al. 2012) where the two surveys overlapped, BCRs 9, 10, 16, and 17 (Figure A4-1), to 1) combine information from the two surveys into composite trend and population size estimates, and 2) estimate a scaling factor that could adjust BBS indices in non-overlapping BCRs for estimating a population size in those regions. Millsap et al. (2013) included data from 1967–2010 and provided population time series for those years, and trend estimates for 1968–2010.

We used the methods described by Millsap et al. (2013) to update golden eagle population sizes throughout the western U.S. with four additional years (2011–2014) of BBS and WGES data. The WGES has conducted distance sampling-based surveys since 2006 in the 4 overlap BCRs, except in 2011 when they did not conduct surveys in BCR 17. Therefore, we did not consider BCR 17 an overlap area in 2011 and adjusted the BBS index using the BCR 17 scaling factor for that year.

We compared our updated western U.S. time series, and short-term (1990–2014) and long-term (1968–2014) BCR-specific trends to those estimated by Millsap et al. (2013). In addition, we explored the feasibility of Flyway-scales as potential units for management. We estimated Central and Pacific Flyway population estimates by post-stratifying BCR-specific populations into the Flyways based on the proportion of area of each BCR within each Flyway (Table A4-1). We provide population estimates for each BCR (Table A4-3), Flyway (Table A4-2), and the entire western U.S. from 1967–2014.

The updated trend estimates for all BCRs were closer to stable than those reported by Millsap et al. (2013), with the exception of BCRs 32 and 33 which were slightly more negative. The BCR-specific trends were not statistically different based on the overlap of the 95% credible intervals (Figure A4-2). We detected a slightly increasing trend in the Central Flyway and slightly decreasing trend in the Pacific Flyway, but these were not statistically different than a stable population and were not different than the overall estimate for the entire western U.S. (Figure A4-3). The estimated time series indicated a slightly higher population size in the Pacific Flyway compared to the Central Flyway, but this difference was not significant (Figure A4-4). The western U.S. population estimates were similar to those reported in Millsap et al. (2013), except that the slight increasing population trend they discussed was not evident in the updated time series (Figure A4-5). We also note the additional years of data do not change the scaling factors for the individual BCRs or the overall average (Figure A4-6).

Literature Cited

Millsap, B. A., G. S. Zimmerman, J. R. Sauer, R. M. Nielson, M. C. Otto, E. Bjerre, and R. Murphy. 2013. Golden eagle population trends in the western United States: 1968–2010. *Journal of Wildlife Management* 77:1436–1448.

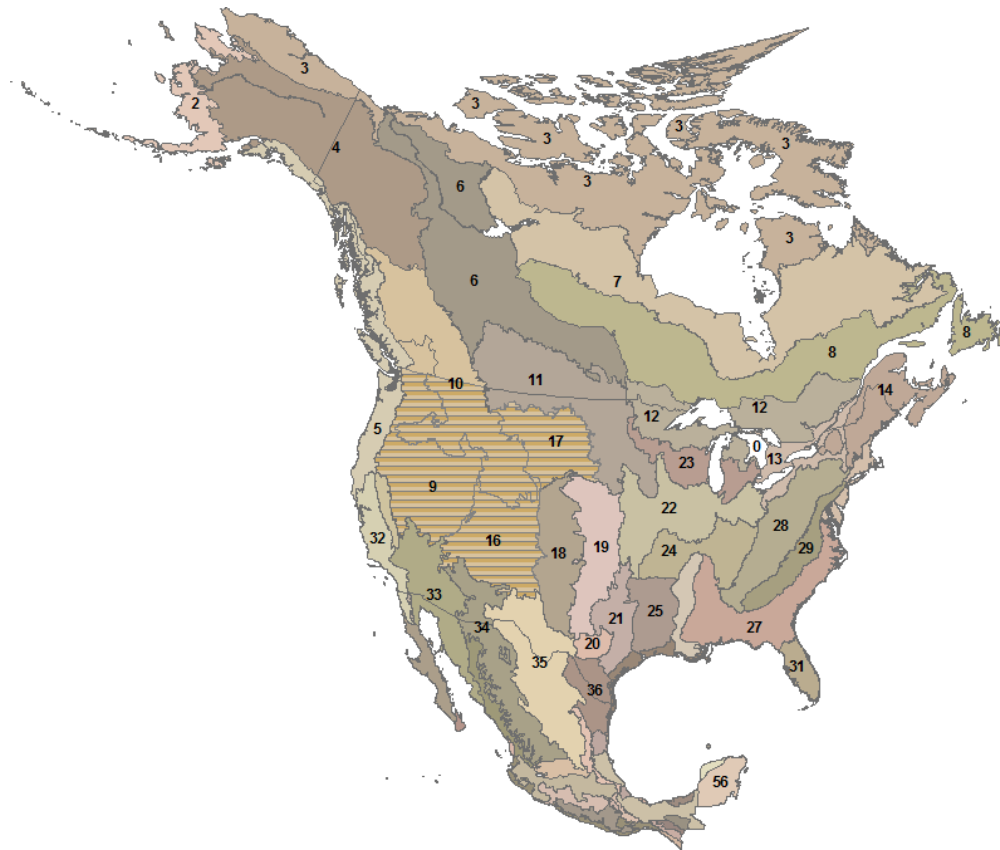
Nielson, R. M., L. McManus, T. Rintz, and L. L. McDonald. 2012. A survey of golden eagles (*Aquila chrysaetos*) in the western US, 2006–2012. Western EcoSystems Technology, Inc., Cheyenne, Wyoming, USA.

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Table A4-1. Proportion of BCRs in the Pacific and Central Flyways used to post-stratify BCR-specific golden eagle population size to a Flyway scale, 1968–2014.

BCR	Proportion of BCR	
	Pacific Flyway	Central Flyway
5	1	0
9	1	0
10	0.78	0.22
11 ^a	0.08	0.92
15	1	0
16	0.69	0.31
17	0.01	0.99
18	0	1
32	1	0
33	1	0
34	0.91	0.09
35	0.04	0.96

^a Approximately 24% of BCR occurred in the Mississippi Flyway; for the purposes of this analysis, we included that with the Central Flyway



BCR	Name	BCR	Name
1	Aleutian/Bering Sea Islands	20	Edwards Plateau
2	Western Alaska	21	Oaks and Prairies
3	Arctic Plains and Mountains	22	Eastern Tallgrass Prairie
4	Northwestern Interior Forest	23	Prairie Hardwood Transition
5	Northern Pacific Rainforest	24	Central Hardwoods
6	Boreal Taiga Plains	25	West Gulf Coastal Plain/Ouachitas
7	Taiga Shield and Hudson Plains	26	Mississippi Alluvial Valley
8	Boreal Softwood Shield	27	Southeastern Coastal Plain
9	Great Basin	28	Appalachian Mountains
10	Northern Rockies	29	Piedmont
11	Prairie Potholes	30	New England/Mid-Atlantic Coast
12	Boreal Hardwood Transition	31	Peninsular Florida
13	Lower Great Lakes/St. Lawrence Plain	32	Coastal California
14	Atlantic Northern Forest	33	Sonoran and Mohave Deserts
15	Sierra Nevada	34	Sierra Madre Occidental
16	Southern Rockies/Colorado Plateau	35	Chihuahuan Desert
17	Badlands and Prairies	36	Tamaulipan Brushlands
18	Shortgrass Prairie	37	Gulf Coastal Prairie
19	Central Mixed-grass Prairie		

Figure A4-1. North American Bird Conservation Initiative (NABCI) bird conservation regions (BCRs). The horizontal lines indicate the 4 BCRs where the breeding bird survey (BBS) and Western EcoSystems Technology summer golden eagle survey (WGES) overlap.

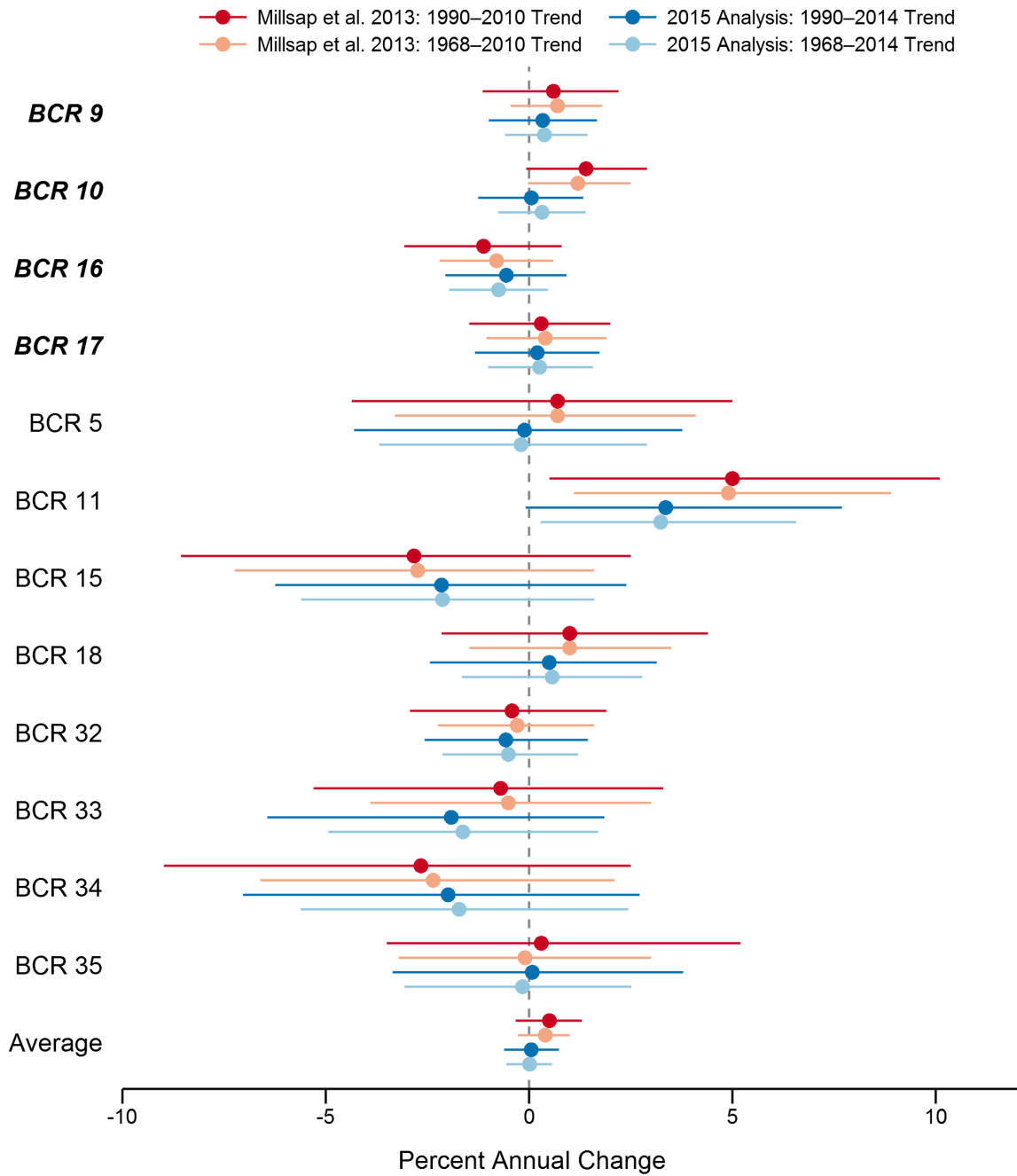


Figure A4-2. Updated golden eagle BCR-specific trends compared to those estimated by Millsap et al. (2013). The BCRs where both breeding bird survey (BBS) and Western EcoSystems Technology summer golden eagle survey (WGES) data were collected are at the top in bold.

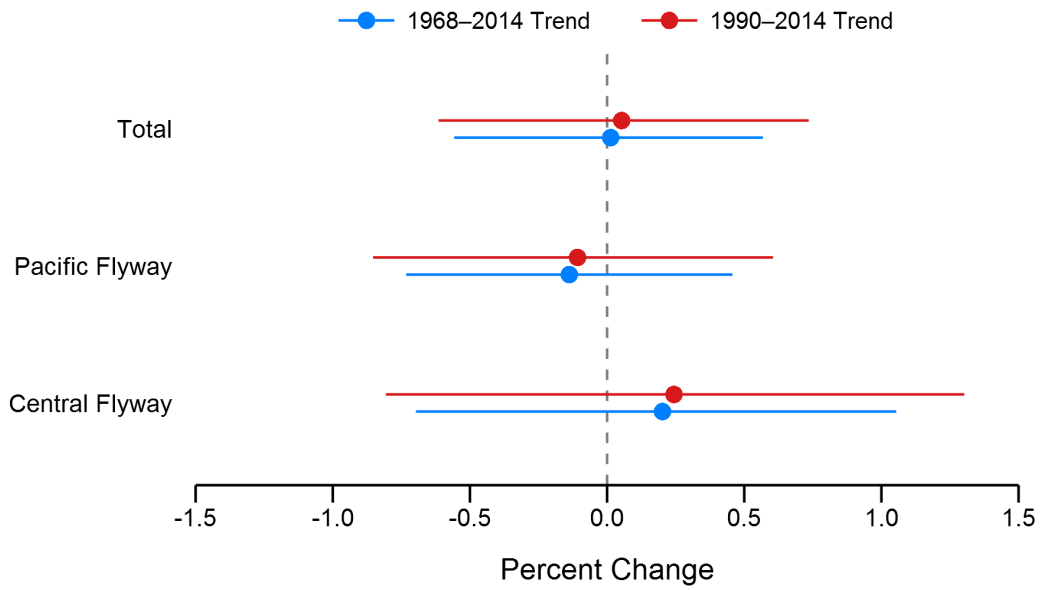


Figure A4-3. Comparison of golden eagle trends in the Pacific and Central Flyways and the total trend for the entire western U.S.

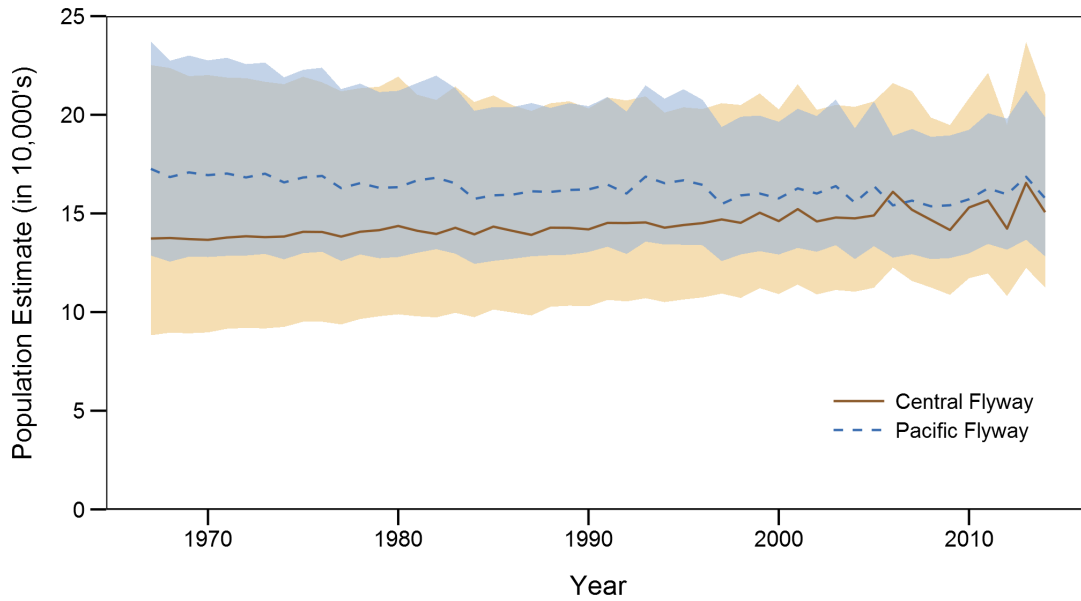


Figure A4-4. Comparison of Flyway-specific time series of golden eagle population size estimates from 1967–2014.

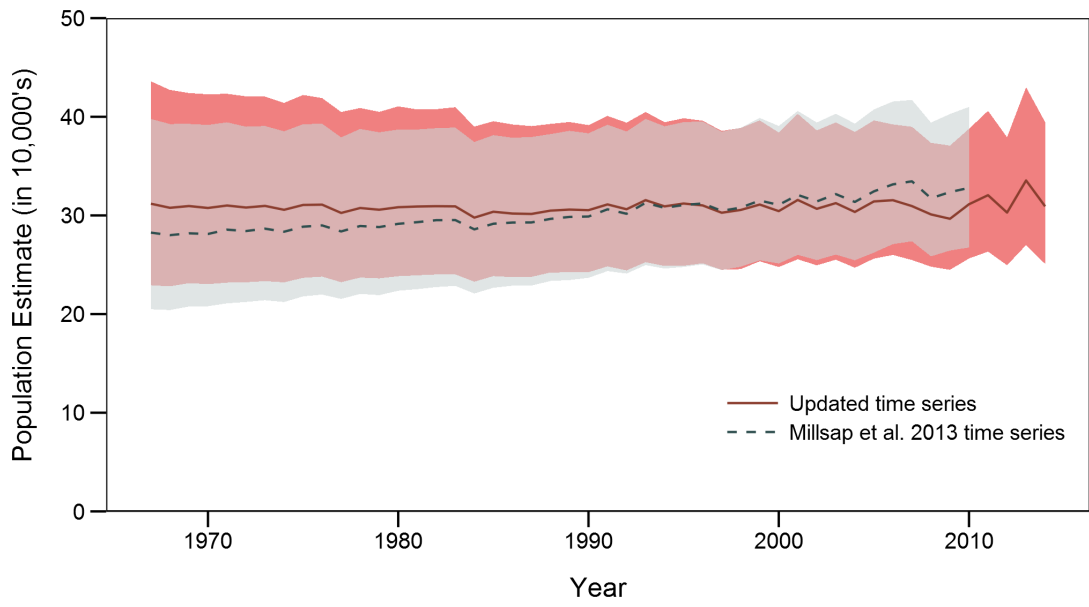


Figure A4-5. Comparison of time series for golden eagles in the western U.S. based on data from 1967–2010 (Millsap et al. 2013) and updated data (1967–2014).

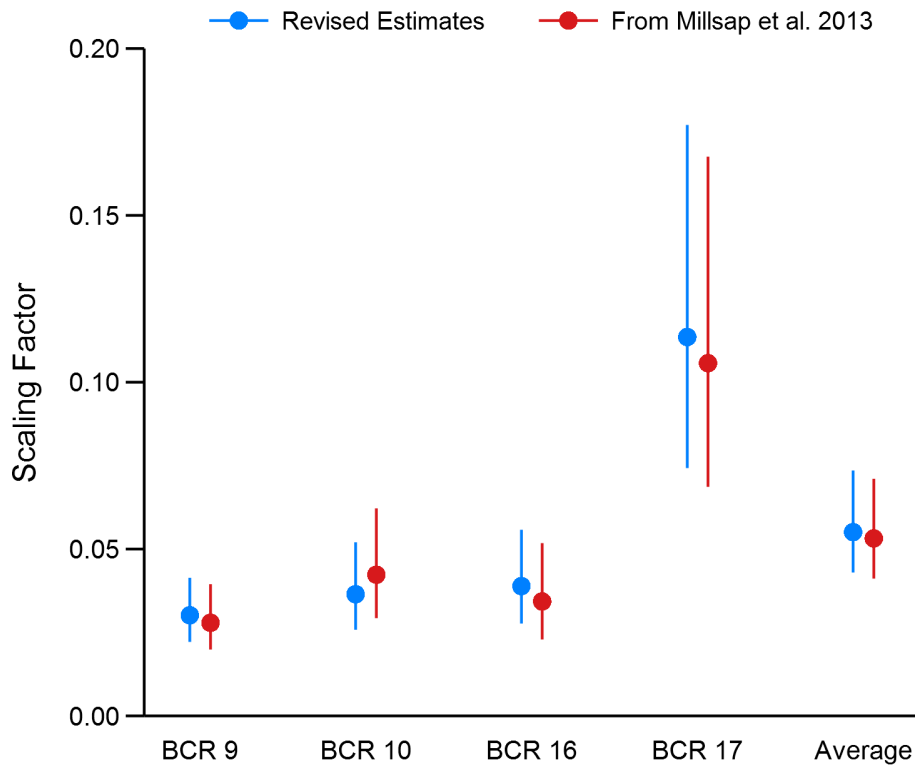


Figure A4-6. Comparison between Millsap et al. (2013) and updated scaling factors used to adjust BBS indices to a population estimate for golden eagles.

Table A4-2. Population size estimates for golden eagles in the Central and Pacific Flyways, and the total western U.S., 1967–2014.

Region	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
CF	1967	14,272	3,457	8,752	13,869	22,322
CF	1968	14,299	3,392	8,949	13,874	22,211
CF	1969	14,210	3,329	8,925	13,786	21,878
CF	1970	14,188	3,295	9,004	13,795	21,960
CF	1971	14,321	3,249	9,146	13,924	21,785
CF	1972	14,292	3,210	9,106	13,908	21,701
CF	1973	14,293	3,184	9,174	13,944	21,686
CF	1974	14,326	3,141	9,319	13,961	21,616
CF	1975	14,552	3,160	9,540	14,182	21,858
CF	1976	14,585	3,112	9,639	14,218	21,824
CF	1977	14,282	2,989	9,398	13,960	21,168
CF	1978	14,528	2,982	9,711	14,152	21,340
CF	1979	14,621	3,016	9,765	14,244	21,599
CF	1980	14,868	3,050	10,050	14,477	21,917
CF	1981	14,554	2,883	9,866	14,232	21,180
CF	1982	14,344	2,791	9,750	14,037	20,802
CF	1983	14,749	2,861	10,169	14,402	21,276
CF	1984	14,359	2,742	9,852	14,037	20,558
CF	1985	14,723	2,761	10,221	14,415	21,125
CF	1986	14,522	2,686	9,992	14,251	20,629
CF	1987	14,375	2,621	9,967	14,103	20,259
CF	1988	14,697	2,622	10,343	14,416	20,663
CF	1989	14,707	2,613	10,364	14,443	20,595
CF	1990	14,643	2,578	10,383	14,352	20,520
CF	1991	15,048	2,606	10,814	14,743	21,036
CF	1992	14,934	2,577	10,750	14,650	20,846
CF	1993	15,059	2,573	10,893	14,747	20,938
CF	1994	14,723	2,448	10,581	14,480	20,202
CF	1995	14,863	2,460	10,767	14,600	20,469
CF	1996	14,947	2,419	10,865	14,716	20,390
CF	1997	15,093	2,411	11,040	14,829	20,650
CF	1998	14,976	2,432	10,957	14,699	20,475
CF	1999	15,532	2,506	11,385	15,245	21,182
CF	2000	15,044	2,394	11,025	14,811	20,404
CF	2001	15,683	2,517	11,636	15,366	21,454
CF	2002	15,036	2,365	11,020	14,810	20,297
CF	2003	15,254	2,365	11,315	15,001	20,690
CF	2004	15,285	2,412	11,304	15,030	20,720
CF	2005	15,323	2,371	11,383	15,058	20,580
CF	2006	16,529	2,352	12,578	16,319	21,735
CF	2007	15,769	2,401	11,789	15,531	21,250

Table A4-2. Population size estimates for golden eagles in the Central and Pacific Flyways, and the total western U.S., 1967–2014. (*continued*)

Region	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
CF	2008	15,107	2,194	11,465	14,901	20,035
CF	2009	14,582	2,137	11,017	14,363	19,385
CF	2010	15,780	2,324	11,878	15,566	21,034
CF	2011	16,218	2,558	12,080	15,914	22,195
CF	2012	14,629	2,197	10,966	14,391	19,611
CF	2013	17,229	2,904	12,503	16,884	23,871
CF	2014	15,554	2,475	11,523	15,276	21,156
PF	1967	17,541	2,760	12,855	17,285	23,604
PF	1968	17,097	2,621	12,600	16,856	22,840
PF	1969	17,317	2,620	12,846	17,113	23,032
PF	1970	17,191	2,546	12,821	16,982	22,702
PF	1971	17,315	2,533	12,977	17,129	22,839
PF	1972	17,117	2,457	12,831	16,925	22,411
PF	1973	17,227	2,465	12,989	17,016	22,594
PF	1974	16,826	2,336	12,728	16,646	21,844
PF	1975	17,011	2,356	12,953	16,846	22,039
PF	1976	17,128	2,368	13,004	16,952	22,242
PF	1977	16,505	2,229	12,602	16,333	21,259
PF	1978	16,758	2,228	12,881	16,598	21,544
PF	1979	16,506	2,141	12,769	16,371	20,995
PF	1980	16,540	2,129	12,814	16,388	21,067
PF	1981	16,853	2,195	13,017	16,698	21,553
PF	1982	17,045	2,223	13,228	16,863	21,884
PF	1983	16,686	2,106	13,020	16,518	21,378
PF	1984	15,948	1,977	12,478	15,821	20,212
PF	1985	16,039	1,973	12,607	15,905	20,246
PF	1986	16,162	1,946	12,765	16,034	20,426
PF	1987	16,248	1,921	12,905	16,108	20,380
PF	1988	16,220	1,891	12,885	16,070	20,255
PF	1989	16,365	1,927	13,033	16,220	20,471
PF	1990	16,352	1,878	13,053	16,229	20,362
PF	1991	16,683	1,908	13,318	16,529	20,756
PF	1992	16,221	1,825	12,993	16,078	20,078
PF	1993	17,135	2,002	13,630	16,993	21,532
PF	1994	16,692	1,876	13,419	16,546	20,695
PF	1995	16,872	1,983	13,438	16,733	21,137
PF	1996	16,586	1,806	13,412	16,475	20,398
PF	1997	15,617	1,701	12,629	15,488	19,331
PF	1998	16,097	1,764	13,004	15,975	19,905
PF	1999	16,198	1,729	13,120	16,090	19,857
PF	2000	15,916	1,689	12,889	15,829	19,461

Table A4-2. Population size estimates for golden eagles in the Central and Pacific Flyways, and the total western U.S., 1967–2014. (continued)

Region	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
PF	2001	16,375	1,754	13,306	16,242	20,164
PF	2002	16,109	1,724	13,092	15,985	19,725
PF	2003	16,606	1,780	13,502	16,468	20,413
PF	2004	15,728	1,689	12,736	15,599	19,364
PF	2005	16,611	1,842	13,424	16,430	20,732
PF	2006	15,574	1,574	12,775	15,475	18,995
PF	2007	15,825	1,612	13,032	15,720	19,340
PF	2008	15,499	1,573	12,729	15,404	18,876
PF	2009	15,595	1,553	12,904	15,480	18,952
PF	2010	15,886	1,605	13,069	15,765	19,366
PF	2011	16,428	1,660	13,465	16,322	20,003
PF	2012	16,117	1,652	13,184	16,011	19,651
PF	2013	17,023	1,878	13,728	16,879	21,080
PF	2014	15,924	1,759	12,846	15,806	19,737
Total	1967	31,813	5,173	22,947	31,398	43,304
Total	1968	31,396	5,030	22,890	30,976	42,687
Total	1969	31,528	4,962	23,036	31,069	42,287
Total	1970	31,380	4,854	23,049	30,990	42,164
Total	1971	31,637	4,801	23,416	31,179	42,322
Total	1972	31,409	4,748	23,210	31,049	41,831
Total	1973	31,520	4,725	23,376	31,119	41,972
Total	1974	31,152	4,570	23,260	30,772	41,118
Total	1975	31,563	4,612	23,671	31,178	41,662
Total	1976	31,713	4,564	23,796	31,322	41,796
Total	1977	30,787	4,389	23,159	30,435	40,383
Total	1978	31,286	4,373	23,805	30,931	40,746
Total	1979	31,126	4,314	23,774	30,762	40,474
Total	1980	31,408	4,337	23,990	31,025	40,830
Total	1981	31,407	4,255	24,007	31,069	40,617
Total	1982	31,389	4,201	24,163	31,061	40,696
Total	1983	31,435	4,140	24,236	31,057	40,406
Total	1984	30,307	3,984	23,432	29,947	39,148
Total	1985	30,761	3,988	23,867	30,429	39,611
Total	1986	30,684	3,908	23,853	30,354	39,142
Total	1987	30,623	3,817	23,947	30,358	38,776
Total	1988	30,917	3,814	24,279	30,631	39,051
Total	1989	31,072	3,844	24,414	30,790	39,437
Total	1990	30,995	3,776	24,429	30,723	39,077
Total	1991	31,731	3,817	25,176	31,411	40,094
Total	1992	31,155	3,707	24,829	30,832	39,131
Total	1993	32,194	3,814	25,641	31,892	40,519

Table A4-2. Population size estimates for golden eagles in the Central and Pacific Flyways, and the total western U.S., 1967–2014. (*continued*)

Region	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
Total	1994	31,415	3,646	25,051	31,172	39,250
Total	1995	31,735	3,732	25,149	31,447	39,750
Total	1996	31,533	3,575	25,192	31,315	39,130
Total	1997	30,710	3,488	24,716	30,459	38,358
Total	1998	31,073	3,549	24,950	30,814	38,699
Total	1999	31,731	3,579	25,457	31,474	39,427
Total	2000	30,960	3,488	24,881	30,732	38,533
Total	2001	32,058	3,597	25,932	31,757	40,106
Total	2002	31,145	3,465	25,075	30,875	38,620
Total	2003	31,861	3,509	25,789	31,619	39,582
Total	2004	31,013	3,481	24,973	30,705	38,702
Total	2005	31,934	3,528	25,791	31,659	39,649
Total	2006	32,104	3,368	26,228	31,852	39,479
Total	2007	31,594	3,396	25,769	31,307	39,147
Total	2008	30,606	3,182	25,076	30,395	37,537
Total	2009	30,177	3,153	24,720	29,917	37,060
Total	2010	31,666	3,343	25,832	31,418	39,067
Total	2011	32,646	3,556	26,591	32,334	40,668
Total	2012	30,746	3,276	25,102	30,492	37,845
Total	2013	34,252	3,998	27,382	33,910	42,956
Total	2014	31,477	3,562	25,293	31,182	39,334

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014.

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
5	1967	190	120	59	165	467
5	1968	217	155	69	182	593
5	1969	183	97	59	164	424
5	1970	182	97	60	162	414
5	1971	182	95	61	163	411
5	1972	171	80	57	157	370
5	1973	180	84	67	165	385
5	1974	170	76	59	156	355
5	1975	189	88	75	172	408
5	1976	191	90	77	172	420
5	1977	179	79	72	165	372
5	1978	170	71	66	159	341
5	1979	180	75	75	166	361
5	1980	201	90	91	181	424
5	1981	181	74	80	168	361
5	1982	171	66	69	162	329
5	1983	172	66	72	163	326
5	1984	185	72	85	171	362
5	1985	182	72	83	168	359
5	1986	165	62	64	156	312
5	1987	166	63	61	157	310
5	1988	166	62	67	158	308
5	1989	171	64	69	161	322
5	1990	190	77	90	174	384
5	1991	199	98	95	179	424
5	1992	198	84	95	180	412
5	1993	169	63	69	161	314
5	1994	200	92	95	180	429
5	1995	171	63	69	162	316
5	1996	177	66	78	166	336
5	1997	169	63	67	161	314
5	1998	202	91	91	183	431
5	1999	176	67	76	166	336
5	2000	196	84	89	179	406
5	2001	196	84	90	179	402
5	2002	174	67	71	164	335
5	2003	197	85	88	181	407
5	2004	179	72	73	167	355
5	2005	188	80	79	174	382
5	2006	193	84	81	177	402
5	2007	207	103	86	185	445
5	2008	171	76	56	160	349

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
5	2009	173	77	56	160	363
5	2010	183	83	65	167	386
5	2011	184	89	64	168	395
5	2012	174	83	53	160	374
5	2013	184	89	64	167	399
5	2014	195	97	68	175	438
9	1967	5,962	1,547	3,559	5,743	9,630
9	1968	5,647	1,415	3,363	5,475	8,938
9	1969	5,988	1,464	3,672	5,806	9,418
9	1970	6,039	1,462	3,729	5,851	9,418
9	1971	6,240	1,512	3,891	6,044	9,834
9	1972	6,072	1,397	3,842	5,908	9,308
9	1973	6,112	1,394	3,889	5,956	9,360
9	1974	6,071	1,358	3,909	5,909	9,198
9	1975	6,051	1,376	3,850	5,873	9,252
9	1976	6,264	1,415	4,025	6,080	9,622
9	1977	5,719	1,280	3,595	5,599	8,631
9	1978	5,938	1,291	3,816	5,786	8,882
9	1979	5,854	1,248	3,807	5,736	8,708
9	1980	5,952	1,259	3,871	5,816	8,800
9	1981	6,193	1,320	4,071	6,046	9,210
9	1982	6,505	1,421	4,292	6,314	9,855
9	1983	6,233	1,292	4,182	6,081	9,252
9	1984	5,626	1,164	3,637	5,525	8,179
9	1985	5,679	1,156	3,707	5,581	8,238
9	1986	5,841	1,162	3,871	5,740	8,482
9	1987	5,865	1,146	3,921	5,759	8,463
9	1988	5,922	1,129	4,024	5,826	8,419
9	1989	6,263	1,202	4,298	6,129	9,016
9	1990	6,134	1,138	4,215	6,023	8,671
9	1991	6,332	1,183	4,395	6,202	8,995
9	1992	6,170	1,113	4,291	6,067	8,653
9	1993	7,068	1,383	4,865	6,890	10,308
9	1994	6,559	1,198	4,577	6,421	9,266
9	1995	7,049	1,379	4,852	6,863	10,233
9	1996	6,220	1,096	4,368	6,121	8,673
9	1997	5,652	1,026	3,882	5,571	7,887
9	1998	6,420	1,123	4,572	6,309	8,981
9	1999	6,304	1,090	4,446	6,207	8,748
9	2000	6,040	1,028	4,256	5,954	8,294
9	2001	6,355	1,086	4,533	6,247	8,762
9	2002	6,387	1,081	4,523	6,282	8,780

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
9	2003	6,703	1,153	4,783	6,590	9,278
9	2004	6,098	1,039	4,323	6,011	8,393
9	2005	7,009	1,272	4,997	6,834	9,931
9	2006	5,599	864	4,112	5,528	7,487
9	2007	6,211	984	4,559	6,112	8,394
9	2008	5,880	910	4,271	5,821	7,871
9	2009	6,112	928	4,518	6,026	8,149
9	2010	6,038	918	4,462	5,955	8,062
9	2011	6,757	1,009	5,015	6,680	8,989
9	2012	6,426	980	4,730	6,344	8,603
9	2013	7,083	1,196	5,064	6,961	9,746
9	2014	6,578	1,098	4,739	6,467	9,011
10	1967	4,955	1,346	2,868	4,783	8,052
10	1968	4,988	1,327	2,888	4,819	8,034
10	1969	4,974	1,295	2,946	4,825	7,921
10	1970	4,986	1,292	2,920	4,835	7,936
10	1971	5,046	1,279	2,985	4,914	7,974
10	1972	5,045	1,252	3,020	4,906	7,875
10	1973	5,097	1,251	3,074	4,964	7,929
10	1974	5,102	1,218	3,107	4,973	7,818
10	1975	5,125	1,199	3,171	5,001	7,802
10	1976	5,145	1,196	3,162	5,017	7,834
10	1977	5,310	1,198	3,335	5,194	7,980
10	1978	5,329	1,187	3,364	5,206	8,041
10	1979	5,339	1,164	3,416	5,227	7,988
10	1980	5,323	1,138	3,444	5,214	7,829
10	1981	5,384	1,135	3,518	5,274	7,936
10	1982	5,373	1,106	3,498	5,272	7,817
10	1983	5,385	1,110	3,513	5,271	7,847
10	1984	5,415	1,090	3,552	5,314	7,836
10	1985	5,410	1,075	3,581	5,317	7,818
10	1986	5,576	1,089	3,737	5,467	7,994
10	1987	5,627	1,058	3,830	5,537	8,001
10	1988	5,643	1,056	3,863	5,553	7,942
10	1989	5,639	1,030	3,889	5,537	7,925
10	1990	5,610	1,016	3,897	5,526	7,849
10	1991	5,876	1,058	4,077	5,778	8,254
10	1992	5,841	1,029	4,100	5,758	8,092
10	1993	5,841	1,013	4,105	5,754	8,039
10	1994	5,708	987	4,004	5,624	7,856
10	1995	5,773	972	4,096	5,696	7,855
10	1996	5,983	1,001	4,267	5,892	8,147

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
10	1997	5,906	975	4,197	5,819	8,045
10	1998	5,751	948	4,074	5,683	7,771
10	1999	5,911	951	4,270	5,835	7,968
10	2000	6,097	969	4,422	6,018	8,170
10	2001	6,152	981	4,465	6,064	8,266
10	2002	5,938	942	4,267	5,868	7,936
10	2003	6,310	987	4,632	6,205	8,453
10	2004	6,154	968	4,452	6,071	8,241
10	2005	6,080	945	4,414	6,017	8,090
10	2006	6,435	983	4,713	6,355	8,574
10	2007	6,265	963	4,593	6,203	8,341
10	2008	6,642	1,022	4,872	6,545	8,855
10	2009	6,466	974	4,791	6,383	8,569
10	2010	6,870	1,032	5,113	6,789	9,097
11	1967	198	124	45	170	515
11	1968	202	126	46	175	530
11	1969	205	124	47	178	519
11	1970	206	123	49	182	505
11	1971	234	136	67	205	574
11	1972	219	121	55	195	519
11	1973	275	166	87	236	680
11	1974	234	129	62	209	543
11	1975	243	132	66	219	564
11	1976	272	148	83	242	641
11	1977	263	139	77	235	606
11	1978	273	154	72	244	636
11	1979	260	126	68	239	556
11	1980	322	185	110	283	787
11	1981	299	158	91	269	685
11	1982	294	145	82	270	642
11	1983	298	141	85	275	637
11	1984	330	168	99	300	738
11	1985	343	160	124	315	734
11	1986	342	156	118	315	723
11	1987	357	164	127	328	743
11	1988	331	151	86	311	671
11	1989	404	186	163	369	850
11	1990	362	156	123	340	724
11	1991	412	185	148	379	861
11	1992	396	167	130	373	786
11	1993	503	244	206	449	1,141
11	1994	536	262	220	477	1,207

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
11	1995	568	285	232	505	1,340
11	1996	531	256	213	479	1,172
11	1997	507	214	202	469	1,027
11	1998	497	216	183	462	1,010
11	1999	709	386	282	607	1,762
11	2000	608	279	249	551	1,320
11	2001	564	253	202	523	1,174
11	2002	748	398	300	653	1,782
11	2003	638	310	235	581	1,388
11	2004	773	387	316	685	1,730
11	2005	655	296	239	602	1,382
11	2006	691	308	268	632	1,439
11	2007	631	292	197	587	1,321
11	2008	661	315	192	613	1,391
11	2009	747	386	243	677	1,674
11	2010	760	374	251	690	1,649
11	2011	807	385	297	735	1,742
11	2011	6,575	1,008	4,830	6,491	8,775
11	2012	684	342	179	630	1,491
11	2013	902	441	325	816	2,013
11	2014	840	399	289	768	1,812
12	2012	6,216	971	4,544	6,138	8,363
13	2013	6,472	1,134	4,558	6,362	9,001
14	2014	5,683	1,003	3,977	5,599	7,924
15	1967	193	110	59	169	457
15	1968	187	114	58	165	436
15	1969	182	102	59	161	421
15	1970	177	114	58	158	396
15	1971	171	84	59	155	374
15	1972	164	76	59	149	346
15	1973	169	78	65	154	364
15	1974	154	68	58	141	321
15	1975	160	70	66	147	326
15	1976	147	63	58	136	301
15	1977	143	60	56	133	288
15	1978	140	57	55	130	277
15	1979	147	66	65	135	300
15	1980	135	54	59	127	258
15	1981	129	49	54	121	246
15	1982	132	53	59	123	256
15	1983	123	46	52	117	230
15	1984	121	44	54	114	225

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
15	1985	133	60	62	121	270
15	1986	116	43	51	110	216
15	1987	122	48	58	113	238
15	1988	110	39	48	105	201
15	1989	121	50	58	111	235
15	1990	109	39	52	103	201
15	1991	103	38	45	98	190
15	1992	101	36	44	96	187
15	1993	99	36	43	94	183
15	1994	98	37	43	92	182
15	1995	95	35	40	90	176
15	1996	98	39	44	92	188
15	1997	93	36	41	88	179
15	1998	96	39	43	89	189
15	1999	94	38	41	87	184
15	2000	86	35	34	81	171
15	2001	90	38	38	83	183
15	2002	83	35	31	77	167
15	2003	81	35	30	76	166
15	2004	82	37	30	76	172
15	2005	78	35	27	72	165
15	2006	83	41	31	75	182
15	2007	78	37	27	71	169
15	2008	77	38	25	70	171
15	2009	76	37	25	68	170
15	2010	73	37	23	65	167
15	2011	78	43	25	69	185
15	2012	70	38	20	62	167
15	2013	72	39	21	63	170
15	2014	70	39	20	62	170
16	1967	6,311	1,955	3,349	6,046	10,818
16	1968	6,218	1,880	3,362	5,946	10,671
16	1969	6,067	1,804	3,299	5,819	10,210
16	1970	6,007	1,744	3,317	5,755	10,055
16	1971	5,834	1,680	3,239	5,605	9,710
16	1972	5,944	1,696	3,366	5,715	9,850
16	1973	5,896	1,648	3,373	5,667	9,697
16	1974	5,658	1,547	3,212	5,468	9,172
16	1975	5,729	1,535	3,341	5,530	9,248
16	1976	5,710	1,523	3,366	5,501	9,172
16	1977	5,480	1,427	3,221	5,299	8,672
16	1978	5,677	1,450	3,424	5,473	9,021

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
16	1979	5,419	1,355	3,269	5,246	8,501
16	1980	5,372	1,334	3,240	5,208	8,443
16	1981	5,391	1,312	3,315	5,231	8,464
16	1982	5,294	1,259	3,281	5,130	8,209
16	1983	5,236	1,225	3,263	5,102	8,070
16	1984	5,081	1,168	3,196	4,955	7,671
16	1985	5,106	1,149	3,263	4,966	7,746
16	1986	5,087	1,139	3,269	4,945	7,690
16	1987	4,932	1,057	3,211	4,810	7,307
16	1988	5,006	1,068	3,275	4,870	7,467
16	1989	4,688	995	3,031	4,591	6,919
16	1990	4,870	1,006	3,228	4,744	7,216
16	1991	4,883	993	3,298	4,762	7,117
16	1992	4,597	902	3,073	4,504	6,622
16	1993	4,598	882	3,142	4,503	6,581
16	1994	4,744	919	3,252	4,636	6,857
16	1995	4,374	832	2,968	4,287	6,216
16	1996	4,837	949	3,365	4,707	7,117
16	1997	4,559	844	3,180	4,460	6,450
16	1998	4,282	780	2,946	4,211	5,992
16	1999	4,377	774	3,087	4,293	6,078
16	2000	4,214	742	2,954	4,147	5,854
16	2001	4,238	739	2,982	4,163	5,864
16	2002	4,120	724	2,890	4,055	5,740
16	2003	4,042	697	2,826	3,981	5,586
16	2004	3,926	681	2,722	3,876	5,398
16	2005	4,031	671	2,887	3,966	5,534
16	2006	4,069	716	2,874	3,992	5,673
16	2007	3,787	627	2,745	3,725	5,173
16	2008	3,471	566	2,516	3,417	4,737
16	2009	3,556	566	2,598	3,506	4,817
16	2010	3,612	580	2,624	3,559	4,876
16	2011	3,580	551	2,641	3,535	4,800
16	2012	4,129	690	2,955	4,073	5,632
16	2013	4,200	794	2,934	4,098	6,087
16	2014	4,223	762	2,969	4,146	5,924
17	1967	9,060	3,080	4,387	8,610	16,393
17	1968	9,075	3,002	4,497	8,621	16,106
17	1969	9,045	2,966	4,509	8,634	15,987
17	1970	9,143	2,944	4,712	8,727	16,045
17	1971	9,131	2,880	4,704	8,739	15,775
17	1972	9,126	2,838	4,706	8,740	15,741

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
17	1973	9,129	2,816	4,784	8,749	15,708
17	1974	9,235	2,789	4,910	8,838	15,790
17	1975	9,459	2,810	5,110	9,088	16,056
17	1976	9,472	2,757	5,309	9,125	16,016
17	1977	9,200	2,623	5,052	8,865	15,295
17	1978	9,415	2,638	5,296	9,055	15,545
17	1979	9,541	2,670	5,366	9,170	15,696
17	1980	9,803	2,707	5,680	9,425	16,033
17	1981	9,390	2,516	5,399	9,079	15,230
17	1982	9,161	2,415	5,297	8,874	14,820
17	1983	9,693	2,509	5,804	9,357	15,535
17	1984	9,312	2,388	5,416	9,025	14,790
17	1985	9,546	2,394	5,780	9,233	15,166
17	1986	9,280	2,305	5,506	9,013	14,551
17	1987	9,212	2,252	5,496	8,968	14,258
17	1988	9,570	2,249	5,944	9,312	14,721
17	1989	9,526	2,218	5,922	9,271	14,632
17	1990	9,577	2,211	6,037	9,302	14,627
17	1991	9,848	2,232	6,318	9,555	15,088
17	1992	9,863	2,218	6,353	9,586	14,955
17	1993	9,924	2,201	6,450	9,638	15,041
17	1994	9,511	2,063	6,105	9,275	14,128
17	1995	9,665	2,061	6,302	9,448	14,389
17	1996	9,537	2,021	6,211	9,330	14,102
17	1997	9,716	2,007	6,377	9,508	14,307
17	1998	9,606	1,994	6,303	9,385	14,044
17	1999	10,075	2,099	6,711	9,816	15,005
17	2000	9,655	1,968	6,336	9,440	14,118
17	2001	10,311	2,134	7,004	10,034	15,362
17	2002	9,655	1,941	6,362	9,466	14,053
17	2003	9,822	1,946	6,572	9,605	14,323
17	2004	9,915	1,982	6,623	9,689	14,447
17	2005	10,012	1,976	6,755	9,780	14,494
17	2006	10,968	1,945	7,728	10,765	15,384
17	2007	10,406	2,009	7,125	10,202	15,118
17	2008	9,721	1,775	6,807	9,518	13,728
17	2009	9,066	1,677	6,283	8,877	12,879
17	2010	10,133	1,875	7,001	9,943	14,403
17	2011	10,624	2,138	7,241	10,329	15,633
17	2012	9,113	1,755	6,266	8,900	13,183
17	2013	11,437	2,491	7,469	11,103	17,214
17	2014	10,012	2,030	6,755	9,764	14,675

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
18	1967	1,170	470	501	1,093	2,303
18	1968	1,151	442	506	1,080	2,208
18	1969	1,166	434	527	1,096	2,194
18	1970	1,148	421	524	1,085	2,137
18	1971	1,212	439	573	1,141	2,274
18	1972	1,215	435	581	1,146	2,252
18	1973	1,206	420	588	1,138	2,207
18	1974	1,164	402	562	1,104	2,103
18	1975	1,239	416	620	1,170	2,217
18	1976	1,198	390	610	1,139	2,107
18	1977	1,229	395	626	1,175	2,166
18	1978	1,216	379	638	1,165	2,091
18	1979	1,248	383	661	1,192	2,184
18	1980	1,214	355	648	1,166	2,037
18	1981	1,257	361	701	1,207	2,102
18	1982	1,377	440	764	1,298	2,427
18	1983	1,197	343	626	1,157	1,965
18	1984	1,235	342	675	1,196	2,002
18	1985	1,326	376	759	1,270	2,213
18	1986	1,338	362	791	1,284	2,204
18	1987	1,234	327	672	1,205	1,966
18	1988	1,268	330	730	1,230	2,032
18	1989	1,403	373	845	1,341	2,291
18	1990	1,296	332	752	1,257	2,050
18	1991	1,254	322	701	1,223	1,952
18	1992	1,313	324	783	1,275	2,046
18	1993	1,248	315	684	1,222	1,932
18	1994	1,333	318	801	1,298	2,059
18	1995	1,374	335	848	1,331	2,150
18	1996	1,393	341	858	1,348	2,185
18	1997	1,386	337	855	1,342	2,155
18	1998	1,350	330	793	1,315	2,089
18	1999	1,483	375	917	1,426	2,368
18	2000	1,437	364	865	1,387	2,296
18	2001	1,499	392	906	1,438	2,434
18	2002	1,417	358	834	1,374	2,244
18	2003	1,442	367	852	1,398	2,283
18	2004	1,358	359	731	1,328	2,150
18	2005	1,458	376	858	1,410	2,346
18	2006	1,456	382	834	1,415	2,339
18	2007	1,492	397	852	1,444	2,411
18	2008	1,377	381	719	1,344	2,210

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
18	2009	1,598	466	908	1,529	2,701
18	2010	1,561	436	876	1,500	2,580
18	2011	1,645	500	917	1,564	2,850
18	2012	1,444	423	730	1,398	2,401
18	2013	1,501	441	789	1,449	2,502
18	2014	1,502	450	772	1,449	2,517
32	1967	934	269	513	901	1,551
32	1968	913	257	500	884	1,499
32	1969	927	252	525	899	1,506
32	1970	918	244	529	890	1,473
32	1971	874	228	491	853	1,388
32	1972	894	227	526	869	1,417
32	1973	905	225	539	881	1,416
32	1974	848	210	489	830	1,312
32	1975	911	222	559	882	1,423
32	1976	863	205	523	840	1,340
32	1977	878	207	539	854	1,346
32	1978	872	204	540	849	1,327
32	1979	852	197	526	833	1,287
32	1980	891	210	556	865	1,380
32	1981	848	187	534	829	1,263
32	1982	862	190	552	839	1,309
32	1983	807	179	497	791	1,199
32	1984	840	184	539	821	1,263
32	1985	837	185	531	816	1,262
32	1986	819	174	524	801	1,208
32	1987	831	179	530	811	1,236
32	1988	796	167	505	781	1,168
32	1989	815	173	531	798	1,209
32	1990	817	180	526	797	1,225
32	1991	792	167	510	776	1,166
32	1992	757	164	464	748	1,108
32	1993	817	178	530	795	1,233
32	1994	799	170	521	780	1,192
32	1995	768	164	490	753	1,136
32	1996	809	180	523	786	1,228
32	1997	772	169	494	756	1,162
32	1998	750	165	470	737	1,116
32	1999	772	171	491	754	1,158
32	2000	767	173	480	750	1,167
32	2001	779	180	490	759	1,196
32	2002	766	178	476	745	1,166

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
32	2003	784	191	482	760	1,229
32	2004	737	176	445	720	1,140
32	2005	748	182	451	727	1,168
32	2006	742	183	443	722	1,161
32	2007	734	185	433	712	1,156
32	2008	700	178	401	684	1,099
32	2009	715	185	412	695	1,141
32	2010	711	189	401	692	1,147
32	2011	728	196	419	702	1,189
32	2012	715	193	402	692	1,154
32	2013	699	194	386	675	1,156
32	2014	719	205	401	690	1,204
33	1967	947	538	287	826	2,285
33	1968	911	492	280	809	2,133
33	1969	925	492	308	822	2,159
33	1970	841	418	277	761	1,893
33	1971	888	440	315	798	1,985
33	1972	810	380	284	742	1,771
33	1973	823	392	302	749	1,769
33	1974	785	358	285	724	1,671
33	1975	783	343	304	724	1,623
33	1976	805	353	321	739	1,672
33	1977	777	328	320	720	1,574
33	1978	723	304	271	676	1,460
33	1979	745	308	307	694	1,508
33	1980	692	275	272	652	1,357
33	1981	698	278	293	654	1,365
33	1982	707	274	309	662	1,367
33	1983	772	352	356	703	1,614
33	1984	663	245	283	628	1,243
33	1985	696	271	325	646	1,362
33	1986	622	226	263	593	1,157
33	1987	671	251	324	626	1,282
33	1988	656	240	313	616	1,245
33	1989	628	221	299	594	1,150
33	1990	635	227	310	596	1,186
33	1991	592	202	282	563	1,071
33	1992	562	196	243	534	1,020
33	1993	584	202	284	554	1,053
33	1994	615	226	307	575	1,173
33	1995	604	223	297	563	1,148
33	1996	598	216	292	560	1,129

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
33	1997	533	185	247	506	977
33	1998	555	200	260	522	1,041
33	1999	568	219	266	530	1,095
33	2000	507	189	208	482	950
33	2001	527	202	233	495	1,006
33	2002	512	198	219	480	992
33	2003	488	192	199	460	938
33	2004	496	198	208	462	974
33	2005	474	192	181	446	930
33	2006	509	223	210	466	1,039
33	2007	494	214	202	453	1,027
33	2008	485	217	187	443	1,021
33	2009	451	204	160	417	937
33	2010	458	214	165	418	979
33	2011	449	212	158	410	962
33	2012	449	221	153	405	979
33	2013	439	219	141	396	989
33	2014	435	219	138	391	985
34	1967	1,050	789	216	848	3,030
34	1968	987	689	212	819	2,793
34	1969	994	711	230	814	2,790
34	1970	945	633	209	794	2,599
34	1971	938	608	232	790	2,493
34	1972	909	585	214	775	2,405
34	1973	931	600	254	793	2,428
34	1974	843	487	227	740	2,080
34	1975	902	537	264	779	2,286
34	1976	831	464	251	734	2,030
34	1977	814	431	262	723	1,911
34	1978	762	399	233	684	1,750
34	1979	743	380	225	669	1,704
34	1980	719	352	236	656	1,581
34	1981	806	450	288	708	1,903
34	1982	744	379	264	670	1,651
34	1983	670	305	222	619	1,424
34	1984	695	317	267	639	1,459
34	1985	674	303	260	621	1,410
34	1986	644	277	245	598	1,332
34	1987	708	352	289	635	1,568
34	1988	614	250	243	574	1,208
34	1989	632	261	261	587	1,301
34	1990	633	275	269	579	1,306

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
34	1991	600	245	252	559	1,201
34	1992	599	237	263	559	1,170
34	1993	547	221	216	518	1,034
34	1994	580	228	262	542	1,135
34	1995	541	202	230	514	1,026
34	1996	560	227	246	523	1,093
34	1997	519	195	217	490	970
34	1998	500	200	187	475	957
34	1999	513	205	215	482	1,006
34	2000	524	210	229	488	1,034
34	2001	579	282	246	517	1,285
34	2002	576	304	239	509	1,329
34	2003	503	210	209	466	1,011
34	2004	476	209	184	440	984
34	2005	447	196	151	418	909
34	2006	462	206	173	428	941
34	2007	455	209	168	417	960
34	2008	463	221	169	421	1,000
34	2009	427	205	141	391	916
34	2010	414	204	132	378	896
34	2011	476	250	164	425	1,082
34	2012	445	239	144	392	1,050
34	2013	418	226	124	372	981
34	2014	418	230	123	369	983
35	1967	843	433	284	763	1,853
35	1968	900	480	322	803	2,037
35	1969	872	409	321	794	1,875
35	1970	787	355	274	726	1,620
35	1971	887	418	347	807	1,901
35	1972	840	380	321	768	1,739
35	1973	795	345	290	739	1,612
35	1974	887	386	368	814	1,827
35	1975	770	313	284	725	1,505
35	1976	815	328	330	763	1,614
35	1977	795	320	325	744	1,551
35	1978	771	301	303	727	1,471
35	1979	800	310	337	757	1,494
35	1980	782	294	325	742	1,469
35	1981	831	310	374	784	1,584
35	1982	768	284	321	732	1,429
35	1983	848	323	388	797	1,616
35	1984	805	297	363	762	1,493

Table A4-3. Population size estimates of golden eagles in each BCR in the western U.S., 1967–2014. (*continued*)

BCR	Year	Mean	SD	Quantiles		
				2.50%	50%	97.50%
35	1985	830	294	386	782	1,535
35	1986	855	322	408	800	1,634
35	1987	898	341	444	831	1,751
35	1988	834	303	403	788	1,542
35	1989	783	265	353	752	1,396
35	1990	763	269	326	733	1,397
35	1991	839	295	410	791	1,547
35	1992	759	250	335	734	1,324
35	1993	796	264	373	763	1,421
35	1994	733	258	275	710	1,300
35	1995	753	249	329	729	1,312
35	1996	789	260	371	756	1,375
35	1997	899	324	461	839	1,675
35	1998	1,064	484	521	940	2,352
35	1999	750	251	318	723	1,329
35	2000	827	283	397	784	1,478
35	2001	768	262	332	738	1,363
35	2002	767	263	335	736	1,370
35	2003	849	308	412	797	1,593
35	2004	819	292	375	775	1,498
35	2005	753	265	314	726	1,362
35	2006	897	333	435	836	1,727
35	2007	834	306	386	783	1,559
35	2008	957	403	459	872	1,963
35	2009	790	300	336	745	1,494
35	2010	855	334	386	796	1,655
35	2011	744	300	271	703	1,442
35	2012	882	348	396	819	1,728
35	2013	846	338	366	787	1,644
35	2014	802	323	326	748	1,586

Appendix A5. USFWS Policy Update to the Use of Eagle Natal Dispersal Distances in Permitting Decisions

A recommendation from the U.S. Fish and Wildlife Service's¹ Eagle Technical Assessment Team for consideration by the U.S. Fish and Wildlife Service's Eagle Management Team

¹ U.S. Fish and Wildlife Service, Division of Migratory Bird Management

Background

The Service's approach to managing eagle populations, as outlined in its 2009 Eagle Permit Rule Final Environmental Assessment (hereafter, 2009 FEA), is to do so primarily at broad, regional eagle management unit (EMU) scales. EMUs for golden eagles are Bird Conservation Regions. For bald eagles, EMUs are based on nest densities and correspond with Service Region boundaries, with slight modifications. However, the Service also is concerned with possible declines in eagle breeding populations at smaller geographic scales. Within a stable regional population there could be a local area where substantial mortality or decreased productivity develops. Initially, there may be enough floaters in this local area population (LAP) to fill breeding area vacancies and compensate for a loss of recruits, but as losses accelerate some vacancies may not be filled. Thus, the potential "rescue effect" attributed to eagles recruiting into the LAP by returning to breed at sites near their natal areas probably is diminished, i.e., probability of recruiting likely decreases with increasing distance from the center of the LAP. In this way, the population is functionally closed to ingress of individuals. This potential depression in resiliency of LAPs is of concern, so the Service uses a biological basis for defining LAPs. Specifically, the Service uses metrics based on natal dispersal distance (NDD), defined as the linear gap between a bird's location of origin and its first breeding or potential breeding location (Greenwood and Harvey 1982, *Annual Review of Ecology, Evolution, and Systematics* 13:1–21). An NDD metric is used as a buffer distance from a given site or project of concern due to potential for take of eagles; the landscape encompassed by the buffer thus represents the LAP area associated with the site or project. Determination of LAP size and benchmark levels is described in Appendix F of the Service's 2012 *Eagle Conservation Plan Guidance, Module 1—Land-based Wind Energy*.

Results reported in the recent paper, *Natal Dispersal Distance of Bald and Golden Eagles Originating in the Coterminous United States as Inferred from Band Encounters*, by Millsap et al. (2014, *Journal of Raptor Research* 48:13–23) should prompt the Service to update the NDD criteria currently used for determining geographic boundaries and estimating size of LAPs of golden eagles and bald eagles. Using a refined dataset and improved analytical methods, the authors re-analyzed NDDs of golden eagles and bald eagles based on band recoveries. The key statement in Millsap et al. (2014) on implications of their analysis of natal dispersal of each eagle species follows:

"Depending on the management policy and circumstances, choice of a natal dispersal value in the range of the 50th to 90th quantile of the distribution as an effect-distance for breeding populations of both species of eagle seems reasonable. For Golden Eagles, this range is 46–175 km [29–109 mi], and for Bald Eagles, 69–346 km [43–215 mi]. This range includes the natal dispersal value of 69 km [42 mi] currently in use for Bald Eagles, but the 90th quantile for Golden Eagles is slightly less than the 225 km [140 mi] currently in use for this species (U.S. Fish and Wildlife Service 2013)."

Of additional importance in the updated analysis by Millsap et al. (2014) is documentation of a difference in NDD between sexes of bald eagles, with females having significantly longer NDDs than males. This is typical across Class Aves and is likely the case for golden eagles as well, though there were not enough recoveries of bands from known-sex golden eagles for the authors to conduct an analysis of sex-biased NDD for the species.

Another issue should be addressed in this update. During this review, an inconsistency was discovered in the 2009 FEA with respect to the values used for NDDs of bald eagles and golden eagles. Of note in the 2009 FEA's "Definitions and Interpretations Used. . ." is this: "Natal dispersal distance—extent of movement between the place of birth and place of first breeding". The following, excerpted directly from the 2009 FEA, demonstrate the inconsistent application of NDD:

(p. 24) "We used natal populations (eagles within the *median* [emphasis added] natal dispersal range of each other) in our evaluation in order to look at distribution across the landscape"

. . . which is consistent with this for bald eagles:

(p. 24) "We used natal populations (eagles within the median natal dispersal range of each other) in our evaluation. . ."

. . . but not consistent with this for golden eagles:

(p. 26) ". . . 90% of mature golden eagles reencountered during the breeding season were within 140 miles [emphasis added] of their natal site. We will consider the natal dispersal distance of golden eagles when evaluating effects to local area populations."

then, finally. . .

(p. 37) "For overall permit management, we will consider local area population effects within the species specific natal dispersal distances (43 miles for bald eagles, 140 miles for golden eagles)."

Per the above passages from pages 26 and 37, the intent of the 2009 FEA was to use the 90th quantile value from the (normal) distribution of NDD records available for golden eagles. Reference to the point value used for NDD as a "median" from page 24 was incorrect for this species. The intent of the 2009 FEA was to also use the 90th quantile for bald eagles (not reflected in the above excerpts) and thus address NDD consistently between species. Unfortunately, the median output from a related analysis was shifted inadvertently to the text for bald eagles in the Draft Environmental Assessment stage and the discrepancy was overlooked during completion of the 2009 FEA.

Decision Point

Technical Issue

Technical issues to be considered are the updated distributions of NDDs for each eagle species, and the finding that natal dispersal differs by sex for at least the bald eagle.¹ The latter is not an insignificant issue; for bald eagles, the current practice of pooling data from both sexes to estimate the median NDD ignores 72% of the distance over which female recruits would originate, per the update by Millsap et al. (2014) Use of either the median of the distribution of female NDDs (Table A5-1, option 4) or the 90th quantile of the

¹Important note: NDD criteria considered herein and their implications for estimating LAP size in decisions regarding "benchmark" levels of take of golden eagles and bald eagles are summarized, using a hypothetical example, in Table A5-1.

Table A5-1. Examples of bald and golden eagle natal dispersal distance criteria and the implications for take benchmarks.

Species	NDD metric ^a	NDD value (mi)	LAP area (mi ²) ^b	LAP size (n eagles, example) ^c	5% benchmark (n eagles)
Golden eagle	<i>current 2009 FEA pooled 90th quantile</i>	140	61,575	1,000	50
	(option 1) revised–pooled median	29	2,642	43	2
	(2) revised–pooled 90 th quantile	109	37,325	606	30
Bald eagle	<i>current 2009 FEA pooled median</i>	43	5,809	500	25
	(option 1) revised–pooled median	42	5,542	477	24
	(2) revised–pooled 90 th quantile	215	145,220	12,497	625
	(3) revised–male, median ^d	37	4,301	370	19
	(4) revised–female, median	86	23,235	2,000	100
	(5) revised–male, 90 th quantile ^e	200	125,664	10,814	541
(6) revised–female, 90 th quantile ^e	419	551,541	47,465	2,373	

^a Currently used values are derived from a normal distribution. Revised values are from lognormal distributions in [Millsap et al. \(2014\)](#).

^b Area calculation is based on a circle with the respective NDD value as its radius, extending from a central point instead of from a polygon as normally would be the case.

^c LAP size is based on simple extrapolation of density estimates to Eagle Management Units, as described in the 2009 FEA: (1) for golden eagles, density estimates for Bird Conservation Regions were from Partners In Flight; (2) for bald eagles, density estimates for Service Regions were derived from minimum number of occupied breeding areas at the time of delisting. Because this example is hypothetical, no EMUs are identified.

^d The sample of bald eagle band recoveries used by [Millsap et al. \(2014\)](#) was large enough to assess NDD for each sex. This was not the case for golden eagles, however.

^e Sex-specific, 90th quantile NDD values for bald eagles were not directly conveyed in [Millsap et al. \(2014\)](#), but were calculated by Brian A. Millsap for this document based on the dataset and approach used in the publication.

pooled distribution of NDDs (option 2) would capture more of this area, resulting in a LAP size about 4 and 25 times greater, respectively, than currently used (column 5). The median of the distribution of male NDDs (option 3) would capture slightly less than the area currently used.

Under Service policy for benchmark guidelines authorizing take of up to 1% and 5% of the size of a given LAP, an increase in LAP size allows for correspondingly greater levels of take under permit. An increase in authorized take of bald eagles seems appropriate, given the species' current robust population status across its range in the U.S. When the median distribution of the female NDD is used, a 4-fold increase in take authorized at the 5% benchmark levels seems reasonable (example in [Table A5-1](#), column 6: increase from current 25 to 100 eagles under option 4). However, a 25-fold increase resulting from use of a 90th quantile of

the pooled distribution of NDDs (column 6: increase from current 29 to 625 under option 2) seems excessive, at least until uncertainty surrounding NDD (Millsap et al. 2014), including regional variation, is reduced. The 90th quantile NDD value for male bald eagles is nearly as excessive, while that for females is far more excessive (options 5 and 6).

Again, for golden eagles, current data are too few to discern whether NDD differs by sex, though a difference likely occurs. Use of the pooled 90th quantile is consistent with the Service's current approach for the golden eagle. Moreover, the smaller LAP area, population size, and 5% benchmark level that result (40% decrease in example in Table 1, option 2, columns 4–6) seem appropriate, given the species' more tentative population status. However, this status does not warrant use of the median (option 1), which could lead to almost no take at the 5% benchmark level.

Inconsistent Terms Issue

In current LAP analyses, the Service uses the median and the 90th quantile of the distribution of NDDs for bald eagles and golden eagles, respectively, per the 2009 FEA. In both cases, data from sexes are pooled. As noted above, this stems from an undetected error in the 2009 FEA; the intent at the time was to use the 90th quantile of the distribution of NDD data, pooled from both sexes, for each species.

Recommendations

There are four reasonable alternatives for addressing these two issues:

Alternative 1a (ETAT-recommended alternative)

Acknowledge and reconcile errors in use of terms in the 2009 FEA; the original intent was to use the 90th quantile of the distribution of NDDs for both species. Be consistent with the 2009 FEA's approach to NDD criteria for golden eagles by using the updated 90th quantile NDD value. For bald eagles, be reasonably consistent by using the updated median NDD value but base this on data from females to better account for their significantly greater NDDs. This approach is recommended because it would incorporate new evidence of sex-biased NDD of bald eagles reported by Millsap et al. (2014), acknowledging that use of a pooled value is no longer consistent with best available science for this species. Use of the females' greater NDD is recommended because it is a more liberal approach, appropriate given the species' current robust status; use of the males' NDD would, in contrast, result in reduced levels of authorized take. Under this alternative, the Service would complete this policy revision at the earliest possible date to incorporate the best and most recent information in decision-making.

Alternative 1b

Same as Alternative 1a, except inconsistent terminology in the 2009 FEA would be reconciled during the NEPA process associated with the ongoing Rule revision rather than as an earlier policy decision.

Alternative 2a

Acknowledge and reconcile errors in use of terms in the 2009 FEA, just as in Alternative 1a; again, the original intent was to use the 90th quantile of the distribution of NDDs for both species. Adopt as the updated NDD criterion the 90th quantile values for the pooled NDD distributions for both of the respective species, reported by Millsap et al. (2014). However, the bald eagle's new NDD under this alternative would result in an excessively large local area size, exceeding the area encompassed by regional EMUs in some cases. Moreover,

this approach would not incorporate new evidence of sex-biased NDD of bald eagles reported by [Millsap et al. \(2014\)](#) and thus would fail to incorporate the latest, best available information. Regardless, under this alternative, the Service would complete this policy revision at the earliest possible date to incorporate the best and most recent information in decision-making.

Alternative 2b

Same as Alternative 2a, except inconsistent terminology in the 2009 FEA would be reconciled during the NEPA process associated with the ongoing Rule revision rather than as an earlier policy decision.

Decision

The EMT supported the ETAT-recommended alternative (1a) and the process to update the ECPG appendix is underway. This approach to updating the NDD will acknowledge and reconcile errors the errors in the 2009 FEA, it will be consistent with the 2009 FEA's approach to NDD criteria for golden eagles by using the updated 90th quantile NDD value, it will be reasonably consistent for bald eagles by using the updated median NDD value for females (thereby incorporating the new evidence of sex-biased NDD of bald eagles), it is a more liberal approach for bald eagles which is appropriate given the species' current robust status, and a slightly more conservative approach with golden eagles as the revised NDD would result in reduced levels of authorized take. Under this alternative, the Service will complete this policy revision at the earliest possible date by revising the ECPG ([U.S. Fish and Wildlife Service 2013](#), Appendix F) to reflect the incorporation of the best and most recent information in decision-making.

Literature Cited

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