

Running title: *Grizzly bears in the southwestern U.S.A.*

Restoring an extirpated species: grizzly bears in the southwestern U.S.A.?

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Abstract

We appraised the biophysical suitability of Arizona and New Mexico, U.S.A., for restoration of grizzly bears *Ursus arctos horribilis*. We extended and integrated existing models of habitat capability and remoteness from humans, and used historical and recent sightings of grizzly bears to calibrate model metrics to our study area conditions. We applied previously published standards or new standards based on established concepts to identify areas productive enough and remote enough from humans to sustain grizzly bears locally, as well as habitat complexes that were capable of supporting robust grizzly bear populations because of large size and high quality. We identified a promising complex of habitat in the south-central part of our study area (the Gila/Mogollon complex) that was larger than 20,000 km², overlapped extensively with oak *Quercus* spp. dominated vegetation and wapiti *Cervus elaphus* range, and that otherwise exhibited high suitability. We also used seven socio-economic indicators to assess the potential acceptance of grizzly bears by local human residents as well as the similarity of counties with potential restoration areas to other counties in the U.S.A. currently with grizzly bears.

Keywords: Arizona; Extirpation; Grizzly bears; Habitat capability; Habitat suitability; Human dimensions; New Mexico; Restoration; Scale; Southwest; *Ursus arctos horribilis*

1. Introduction

Increasingly, restoration is a focus for those promoting ecological conservation. For at-risk animal species that experienced widespread extirpation owing to historical human activities, restoration often includes reintroduction to former range. At a practical and operational level, this kind of restoration is similar to the introduction of game species to former range or to potentially suitable areas outside historical distributions. However, restoration of at-risk species differs from establishment of game populations by demanding greater odds of success (Breitenmoser et al., 2001). By definition, at-risk species offer fewer robust populations as potential sources of translocated animals. In contrast to game species, human acceptance of translocated at-risk species is also often less assured in restoration areas. Overall, restoration of at-risk species requires more attention to human dimensions as well as more reliable appraisals of biophysical conditions in potential restoration areas (Miller et al., 1999; Breitenmoser et al., 2001).

One obvious difficulty in assessing restoration prospects for extirpated animal species arises because there are no *in situ* populations that, through scientific study, can provide site-specific answers to site-specific questions. Appraisal of potential restoration prospects necessarily involves the application of ecological theory to analysis of historical data together with extrapolation of ecological relations observed for extant populations elsewhere. In general, a few key questions need to be addressed in any appraisal of restoration prospects: What factors caused initial extirpations?; What is the current status of these stressors in potential restoration areas, especially in contrast to times when extirpations occurred?; What are the current ecological effects of these stressors?; and, as a bottom line, Does enough suitable habitat exist to support a robust restored population? (Yalden, 1993; Miller et al., 1999; Simberloff et al., 1999;

Breitenmoser et al., 2001). Given that the spatial configuration of habitat conditions affects the population dynamics of virtually all species, these questions need to be answered in a spatially explicit manner. Thus, data need to be spatially referenced, spatially comprehensive, and adequate surrogates for important ecological factors. Where funding is limited and potential restoration areas are extensive, data also need to be available at a modest price and with modest expenditure of effort (Yalden, 1993; Miller et al., 1999; Haight et al., 2000). These constraints make appraisal of restoration prospects for extirpated species one of the greatest challenges of applied ecology (Simberloff et al., 1999).

Grizzly bears *Ursus arctos horribilis* once occupied most of the western U.S.A. (US), including non-desert areas of Arizona and New Mexico (Mattson and Merrill, In press). Between 1850 and 1950 they were extirpated from about 98% of their former range, including all of the southwestern US and adjacent Mexico. Compared to Eurasia during the same period, these extirpations were rapid and extensive (Mattson, 1990). There is no mystery why grizzly bears almost disappeared from the contiguous US. They died because humans – primarily rapidly spreading European settlers – killed them (Storer and Tevis, 1955; Brown, 1985). Grizzly bears continue to die in the US almost solely of human causes (Mattson et al., 1996a; McLellan et al., 1999), at rates that are determined by how often they encounter humans and the probability that the encounter will turn lethal (Mattson et al., 1996b). Between 1850 and 1920 humans were highly lethal to grizzly bears, with rates of extirpation modified by landscape features that affected how often humans encountered bears (e.g., the distribution of attractive habitats and the presence of mountainous terrain; Merrill et al., 1999; Mattson and Merrill, In press). Within the last 70 yrs humans have become much more accepting of grizzly bears in the contiguous US (Kellert et al., 1996), with greatest reductions in human lethality occurring since

the institution of protections under the US Endangered Species Act (ESA) in 1974 (Mattson and Merrill, In press).

Grizzly bears are ideal candidates for restoration to parts of their former ranges in the contiguous US. We suspect that grizzly bears are currently absent from many otherwise suitable areas solely as an historical artifact of the rapid pace of extirpations and the extreme lethality of humans between 1850 and 1950 (Mattson, 1990). Moreover, humans are not only much more benign, on average, but motivated to restore species like grizzly bears by national conservation policies such as the US ESA as well as non-governmental programs like the Wildlands Project and Yellowstone-to-Yukon Conservation Initiative (Soulé and Terborgh, 1999). We also know a great deal about the ecology and broad-scale habitat relations of grizzly bears in the contiguous US. Mace et al. (1999), Merrill et al. (1999), Carroll et al. (2001) and Mattson and Merrill (In press) developed broad-scale models that explain landscape-level relations and predict the location and extent of potential habitat. In addition, Merrill et al. (1999) developed a method for representing habitat relations at an appropriate scale and for predicting areas of potential conflict between humans and grizzly bears; Mattson and Merrill (In press) developed size and shape criteria for judging the robustness of grizzly bear ranges. This previous research provides many of the tools needed for a reliable appraisal of restoration prospects for grizzly bears in areas where they have been extirpated.

During the last 20 yrs there have been major advances in describing and understanding how humans view wildlife and in relating these views to potential explanatory factors like gender, age, income, employment, and education. Kellert (1985) developed a schematic for describing worldviews that he and others have applied to surveys of residents of the interior western US, in areas that contain current or potential grizzly bear range. From this and other

information, Kellert (1985a, 1985b, 1989), Reading and Kellert (1993), and Reading et al. (1994) elaborated on how this schematic of worldviews relates to predators – including grizzly bears. Thus, this research into human dimensions provides a means of bridging from demographic, economic, and educational characteristics to probable views of wildlife and nature and, from that, to probable acceptance of grizzly bears. In general, this research provides a sound basis for identifying human-related factors most relevant to judging how well humans will accept and live with potentially problematic species like grizzly bears.

Here we appraise the prospects for restoring grizzly bears in the southwestern US states of Arizona and New Mexico. To do this, we extended and integrated previous models and metrics for appraising biophysical conditions, calibrated these models to historical and contemporaneous data, and analyzed human dimensions using schematics that describe how humans view wildlife and nature. Our appraisal addresses the following hierarchical questions: (1) What areas are the most remote from humans?; (2) What areas exhibit the greatest intrinsic biophysical capability of supporting grizzly bears?; (3) What are cutpoints for identifying areas sufficiently remote and sufficiently productive to allow bears to survive and replace themselves?; (4) What is the size, shape, and juxtaposition of areas where grizzly bears are most likely to persist?; (5) Which areas or complex of areas are biophysical capable of supporting a robust grizzly bear population?; (6) Which potential restoration areas are occupied by humans most likely to accept grizzly bears and exhibiting demographic, economic, and educational traits most like humans in areas with extant grizzly bear populations?; and (7) Based on biophysical and human conditions, which area(s) are the best candidates for restoration of grizzly bears in this region?

2. Study Area

Our 608,666-km² study area consists of the states of Arizona and New Mexico, US, and extends south to north from 31° 20' to 37° 00' N latitude and east to west from 103° 00' to 115° 00' W longitude (Fig. 1). In Arizona, elevations range from 30 m to 3860 m and, in New Mexico, from 915 m to 4011 m. Most of the north-central part of our study area consists of the deeply incised Colorado Plateau, surrounded by higher elevations of the San Juan, Sangre de Cristo, and Mogollon Mountains in New Mexico and the Kaibab and Mogollon Plateaus and their escarpments in Arizona. The southern and eastern parts of the study area consist of broad plains or valleys broken by isolated peaks and mountain ranges such as the San Mateo, San Andreas, Sacramento, and Guadalupe Mountains in New Mexico. The large size and elevational amplitude of the study area results in a broad spectrum of climates ranging from alpine on the highest peaks to hot arid desert in the lowest plains and valleys. During the last 20 yrs annual precipitation and temperatures averaged about 90–200 mm and 22–24 °C in the hottest driest deserts and about 400–800 mm and 5–10 °C at the coldest wettest weather stations, excluding the highest mountains. Annual snowfall at elevations >2100 m often exceeded 250 cm.

The diverse climates, substrates, and topography of our study area support diverse vegetation (Brown, 1994). The high plains of eastern New Mexico are dominated by short grasses, including buffalo grass *Buchlöe dactyloides* and grama grasses *Bouteloua* spp. Shrubs, most commonly mesquite *Prosopis* spp. and creosote brush *Larrea tridentata*, characterize the Chihuahuan desert of southwestern New Mexico. The Sonoran and Mohave deserts of southern and western Arizona also support abundant creosote brush, but are typified by bursage *Ambrosia* spp. and cholla cacti *Opuntia* spp. as well as visually prominent species like saguaro *Carnegiea gigantea*, paloverde *Cercidium* spp., and Joshua tree *Yucca brevifolia*. To the north, the

Colorado Plateau is dominated by Great Basin desert scrub and conifer woodlands characterized by species of saltbrush or shadscale *Atriplex* spp., sagebrush *Artemisia* spp., and juniper *Juniperus* spp. Piñon pines, *Pinus edulis* and *P. monophylla*, are also common. At higher elevations, shrubby thicket-forming species such as Gambel and live oak, *Quercus gambelli* and *Q. turbinella*, become more abundant before being replaced at higher elevations yet by forests of ponderosa pine *P. ponderosa*. Conifer forests of true firs *Abies* spp., spruces *Picea* spp., pines, and Douglas-fir *Pseudotsuga menziesii* dominate the highest elevations below small isolated areas of alpine tundra.

As of 2000, about 6,950,000 people lived in the study area, concentrated primarily in the cities of Phoenix, Tucson, Flagstaff, and Yuma in Arizona and Albuquerque, Santa Fe, Las Cruces, and Roswell in New Mexico. About 39% of the study area is owned by the U.S. federal government. Of this federally owned land, 24,900 km² is in officially designated wilderness areas. Much larger areas are classified as “roadless” but without binding legal protections.

3. Methods

3.1. Data

We obtained known locations of grizzly bears in Arizona and New Mexico from Brown (1985:42–43) and known locations of grizzly bears in northern Idaho from Layser (1978), Zager (1983), and Kasworm and Their (1994; Merrill et al., 1999). We also obtained delineations of grizzly bear ranges in Arizona and New Mexico circa 1918 from Brown (1985:134–135). Information in Brown (1985) was derived from historical records for 1825 through 1935, the last year any grizzly bear was known to be alive in either New Mexico or Arizona. Locations in northern Idaho were derived from historical and contemporary records for 1950 through 1993.

We scanned maps of locations or ranges from each of these sources, edited out extraneous digital information, and registered the remaining points or areas to a 1-km²-resolution grid.

We used existing sources of digital data to develop a geographic information system for our analysis. We used US Census Bureau TIGER files (scale 1:100,000) to calculate road densities. We converted TIGER line files to raster format with a cell size of 50 m. Cells representing roads and nonroads were coded 1 and 0, respectively. We calculated density using a circular moving window to sum cell values within a 1 km² area. We obtained spatially referenced information on human population in Arizona and New Mexico from updated 1990 US Census Bureau data summed at the block group level. We did not use results of the US 2000 Population Census because much of the information we needed was not yet published. We obtained distribution maps for piñon pine from Critchfield and Little (1966) and for oak-dominated vegetation types from Küchler (1964).

We obtained human-related information at the county level from results of the US 1997 Census of Agriculture and the US 1990 Population Census. For each county that contained prospective core grizzly bear habitat in our study area (see below) or that contained extant grizzly bear populations elsewhere in the contiguous US, we tallied (1) area in agriculture production, (2) area in croplands, (3) number of cattle, (4) number of domesticated sheep, (5) number of people who owned or otherwise operated agricultural enterprises, (6) percent of persons employed in agriculture, (7) median family income (in US \$), (8) percent of families with incomes below the officially designated poverty level, (9) percent of persons with High School degrees, (10) percent with college degrees, and (11) percent ≥ 65 yrs of age. We converted (1), (2), and (5) to percentages of either total county area or population and (3) and (4)

to densities (n/km^2). We reduced the number of variables used in our analyses to seven ([1], [3], [4], [6], [7], [10], and [11]) by eliminating highly correlated ($r \geq 0.5$) variables.

3.2. *Modeling remoteness from humans*

We used methods described in more detail by Merrill et al. (1999) to model the remoteness of a given map pixel from human activity. The metric that we term “remoteness” here was termed “habitat effectiveness” (*HE*) by Merrill et al. (1999). Remoteness is a function of road density (*ACCESS*) within a 2.8-km radius and the potential amount of human activity on these roads (*H'*). Potential human activity is a function of regional (i.e., 80 km-radius) human population size, the presence (or absence) of a National Park, and distances to and population sizes of all surrounding censused units (e.g., census blocks or townships). The effects of regional population size and nearness of a National Park on levels of backcountry activity were estimated empirically by Merrill et al. (1999) from records of activity in US National Forests. The effects of site-specific populations and distances from them were modeled by inverse distance weighting interpolation, with the scaling (i.e., power) parameter derived from observed frequency distributions of trip distances by recreationists in the western US.

3.3. *Modeling biophysical habitat capability*

We modeled the intrinsic biophysical capability of map pixels to support grizzly bears from the results of Mattson and Merrill (In press). Mattson and Merrill (In press) developed models that related estimated distributions of grizzly bears in the western US circa 1850 to landscape features that included the distributions of probable high quality bear foods (e.g., bison [*Bos bison*], oaks, and piñon pines), aboriginal humans and their sedentary corn-based cultures, and broad-scale types based on topography and dominant vegetation (i.e., “ecoregions”). The model explaining mid-1800’s distributions of grizzly bears in the southwestern US included effects of

oak-dominated vegetation types, piñon pine range, densities of aboriginal humans, occurrence of corn-based cultures, hot deserts, and coastal or Rocky Mountain ecoregions. The model was developed at a resolution of 900-km², the approximate size of a female grizzly bear life range (see below; Mattson and Merrill, In press). To derive an estimate of intrinsic habitat capability (reckoned as the log-odds [$\text{logit}(p)$] that an area would have been core grizzly bear range in 1850), we set the effects of aboriginal humans to zero, yielding the following model:

$$\text{Logit}(p) = -0.67 + 1.2RM + 0.71CM - 0.21HD + 0.50OAK + 0.003PP \quad (1)$$

where RM is the extent of the Rocky Mountain ecoregion, CM is the extent of the Coastal Mountain ecoregion, HD is the extent of the Hot Desert ecoregion, OAK is the extent of oak-dominated vegetation types, and PP the extent of piñon pine range, each reckoned as number of km² in a 900-km² grid cell.

3.4. *Biophysical habitat suitability*

We calculated an index of habitat suitability that was analogous to the site-specific potential for increase by a restored grizzly bear population (Merrill et al., 1999). We scaled the index of remoteness and the index of habitat capability each to range from 0 to 1. We used untransformed logits for habitat capability with a constant added such that the minimum logit value equaled 0. Viewing the inverse of remoteness as an analog of potential death rate and habitat capability as an analog of potential birth rate (Merrill et al., 1999), habitat suitability was simply the standardized index of habitat capability minus 1 minus the standardized index of remoteness.

3.5. *Scaling modeling results*

Representations of habitat capability or suitability should match the spatial scale at which focal animals move, live, and die, as well as the scale appropriate to management time frames

(Schonewald-Cox et al., 1991; Ruggerio et al., 1994). In this analysis we addressed outcomes relevant to populations spanning the lifetimes of many animals (i.e., population viability or robustness). The appropriate proximal scale for generalizing map information was thus the life-range, and given the demographic importance of females, the life-range of a female bear (Merrill et al., 1999). This parameter is known for grizzly bears in the Yellowstone region of Wyoming, Montana, and Idaho (Blanchard and Knight, 1991), where females, on average, use about 900 km² during a lifetime. We adopted this value for our analysis in the southwestern US because we judged conditions here to be similar to those in the Yellowstone region; i.e., both areas are mountainous, relatively arid, and produce few fleshy fruits (Mattson and Merrill, In press).

We rescaled our results by recalculating the modeled values of remoteness and habitat capability for each map pixel as the average of values within a surrounding 900-km² area. The resulting surface of values was smoothed in comparison to values calculated at the resolution of our digital data (i.e., the grain of the results was considerably increased), resulting in the spatial aggregation of areas with high or low average values. We applied this moving-window analysis to the results of our habitat capability model despite the fact that this value was already calculated at a 900-km² resolution. This recalculation corrected for potential discrepancies between our arbitrary placement of the 900-km²-resolution grid for initial calculations and the opportunity for real grizzly bears to center their ranges anywhere on the landscape.

3.5. Calibration and establishment of thresholds

We adopted a threshold previously used to identify suitable grizzly bear habitat in Idaho (Merrill et al., 1999) for application to our index of remoteness. This threshold was expressed in terms of a probability ($p = 0.012$), specifically the probability that a 1-km² grid cell in northern Idaho would contain a bear location ($n = 124$; see 3.1.) versus not as a function of habitat

suitability. To apply this threshold to our index of remoteness, we calculated remoteness for 1-km² grid cells in northern Idaho, specified a relation between remoteness and the logit-transformed probability that a cell would or would not contain a bear location (by logistic regression; Hosmer and Lemeshow, 1989; Demaris, 1992), and then, based on this relation, identified the remoteness value corresponding to the threshold probability (Merrill et al., 1999). Thus, we deemed areas where remoteness was greater than the threshold to be sufficiently remote from humans to support grizzly bears. The extrapolation of this threshold from northern Idaho to the southwestern US was possible because the data sources and model used to calculate remoteness were identical for the two areas.

We developed a threshold for biophysical habitat capability that was specific to our southwestern US study area. This was necessary because the model of habitat capability and the data that motivated it were unique to the Southwest. We adopted an *a priori* threshold of $p = 0.5$; that is, that a given 1-km² grid cell in the Southwest would be equally likely to contain a historical grizzly bear location (see 3.1.) or a randomly located point. This was analogous in concept to a drug dose that is lethal 50% of administrations (Woodroffe and Ginsberg, 1998). The challenge was to develop a modeling approach that captured this concept. We used weighted logistic regression (Hosmer and Lemeshow, 1989) and an arbitrarily large number of random points (1,315), each weighted ($= 0.0707$) so that the total frequency of random points used to specify the model equaled the total number of bear locations ($n = 93$). The resulting model allowed for direct translation of the p -value threshold to a corresponding habitat capability value. Thus, we deemed areas where habitat capability was greater than the threshold to be intrinsically capable of supporting grizzly bears, all human-related factors set equal. We adopted

the same modeling approach to relate the distribution of historical grizzly bear locations to current habitat suitability (see 3.4.).

3.6. Identifying biophysically suitable restoration areas

We identified areas biophysically suited to restoration of grizzly bears by application of remoteness and habitat capability thresholds as well as broader-scale design criteria. We first identified areas (i.e., patches) that exceeded both the remoteness and habitat capability thresholds; that were both intrinsically capable of supporting grizzly bears and that were remote enough from humans to impart a high probability of survival to resident bears. We then excluded all patches that were less than the approximate size of an adult female grizzly bear's *annual* range (300 km²; Blanchard and Knight, 1991). We assumed that bears ending up in these patches would have low odds of surviving a single year. We then identified patches that were larger than a female life range (900 km²). We considered these areas to be potential demographic sources of dispersing animals; that is, "source" patches (Merrill et al., 1999). We then identified groups of patches all >300 km² in size that, if >900 km² in size, potentially functioned as a complex by virtue of being within reciprocal dispersal distances of each other or, if all but one were ≤900 km², functioned as a complex by virtue of being within potential dispersal distance of the common source patch. We adopted 20 km as our threshold for potential dispersal, which is well within the range observed for young males, but at the upper limits observed for young females (Blanchard and Knight, 1991; McLellan and Hovey, 2001). We then calculated the total area (km²) and an edge index for each patch and complex of patches. We calculated the edge index as the ratio of observed edge length (km) to length expected if the patch were a circle. Values of this index ranged from 1 to 4 in our study area. As a final step, we judged the long-term biophysical suitability of each patch or complex for restoration of

grizzly bears by relating them to size and shape criteria derived from an analysis of historical extirpations in the contiguous US. Most grizzly bears in ranges $>20,000 \text{ km}^2$ in size, or in ranges $7,000\text{--}20,000 \text{ km}^2$ in size and with an edge index <2 , survived adverse conditions between 1920 and 1970 (Mattson and Merrill, In press). We judged any patch or complex that exceeded these criteria to be biophysically suited for restoration.

3.7. *Appraising human dimensions in potential restoration areas*

We used discriminant analysis to appraise the similarity of human dimensions between counties in our study area containing prospective restoration areas and counties elsewhere in the contiguous US containing extant grizzly bear populations. We assumed that there would be novel conservation issues in counties with dissimilar human characteristics as well as related greater uncertainty about restoration prospects. We used a relatively uncorrelated set of relevant human-related measures (see 3.1.) and a non-parametric kernel-based approach to discriminant analysis (Habbema et al., 1974; Hand, 1982) in which we employed normal kernels and a radius of 1.5 (Mahalanobis distance) for density estimation. Based on the model, we used cross-validation to assign probabilities of being currently “with” or “without” grizzly bears to each county. We used the “with” probability value to score counties that contained prospective restoration areas in terms of relative similarity to counties currently supporting grizzly bears.

We used a deterministic algorithm to score counties in terms of prospective acceptance of grizzly bears by resident humans (*Accept*). We developed the algorithm based on previous work that identified factors associated with hostility to potential predators and the relative prevalence of negative attitudes among different human classes. In particular, older age, employment in agriculture, and lack of college education have been associated with a higher frequency of negative attitudes (Kellert, 1985a, 1985b, 1989; Reading and Kellert, 1993; Reading et al.,

1994). Kellert (1985a) and Reading et al. (1994) provided proportional frequencies of negative attitudes broken down specifically in terms of age, employment, and education. We averaged these not dissimilar proportions between the two studies to derive weighting factors for each class. We multiplied the proportion of people in each county that were ≥ 65 yrs of age ($P65$), that were employed in agriculture ($P\text{AgEmp}$), and that had a college degree ($PColl$) by their corresponding weights and integrated the resulting values in a linear additive model:

$$Accept = -0.53P65 - 0.60P\text{AgEmp} + 0.69PColl. \quad (2)$$

Higher values of *Accept* indicate greater potential acceptance of grizzly bears.

We identified areas where there was high potential for conflict between humans and grizzly bears based on overlap between habitat with intrinsically high habitat capability and areas in private ownership or with potentially high levels of human activity. We considered potential conflict areas to be where habitat capability was greater than and remoteness less than their respective medians, constrained to candidate restoration areas plus a 20-km buffer (Merrill et al., 1999); in other words, where humans and restored bears were more likely than not to both be present. We also considered private lands inside or within 20 km of candidate restoration areas to be likely sites of conflict because of potential incompatibilities between private property rights and management favoring grizzly bears.

4. Results

4.1. Remoteness and habitat capability thresholds

The index of remoteness (*Remote*) was strongly related to logit-transformed probabilities that a given 1-km² cell contained a grizzly bear location, versus not, in northern Idaho ($\text{Logit}[p]_{\text{NI}}$). The best fit was obtained with a polynomial (Fig. 2a) because incidental sightings of bears by humans declined in the most remote areas (Merrill et al., 1999). The relation most

useful for our purposes was described by a single-term model (Fig. 2a) that was nearly as informative as the polynomial (sample-size-corrected Akaike Information Criterion [AIC_c] = 1194 and 1188, respectively):

$$\text{Logit}(p)_{\text{NI}} = -5.71 + 3.57\text{Remote} \quad (3)$$

Somer's $D = 0.37$, $R^2_L = 0.87$, $df = 760$, $G^2 = 399$, and $P = 1.00$ for the model. The test statistics are for goodness-of-fit. Based on this relation, $p = 0.012$ (our *a priori* threshold probability; Merrill et al., 1999) corresponded to $\text{Remote} = 0.35$.

The indices of habitat capability (HC) and current suitability ($Suit$) were strongly positively related to logit-transformed probabilities that a location was that of an historical grizzly bear sighting versus a random point in our southwestern study area ($\text{Logit}[p]_{\text{SW}}$; Fig. 2b). The model describing the relation for HC was:

$$\text{Logit}(p)_{\text{SW}} = -1.17 + 3.96\ln(HC + 1) \quad (4)$$

Somer's $D = 0.48$, $R^2_L = 0.12$, $df = 1 \cdot 10^3$, $G^2 = 217$, and $P = 1.00$ for this model. The model describing the relation for $Suit$ was only slightly less informative than the model for HC ($AIC_c = 231$ versus 230, respectively), and was described by the relation:

$$\text{Logit}(p)_{\text{SW}} = -1.49 + 3.85\text{Suit} \quad (5)$$

Somer's $D = 0.45$, $R^2_L = 0.12$, $df = 1 \cdot 10^3$, $G^2 = 226$, and $P = 1.00$ for this model. Based on the relation with HC , $p = 0.5$ (our *a priori* threshold probability; Woodroffe and Ginsberg, 1998) corresponded to $HC = 0.35$.

4.2. Patterns of habitat loss, 1850–present

Loss of habitat capability between 1850 and the present (i.e., proportional reductions owing to loss of remoteness) was not uniform in joint terms of categorized capability under pristine conditions and categorized extremity ($df = 1$, Mantel-Haenszel $\chi^2 = 14.6$, $P = 0.0001$; Table 1).

The smallest proportional losses were concentrated in the least productive habitat; i.e., 78% of the smallest losses ($\leq 50\%$) were concentrated in habitats with below-median capability under pristine conditions. However, overall, 88% of the study area experienced $>50\%$ loss of habitat capability since 1850 with near equal losses in the intrinsically most and least capable habitats. This pervasive heavy loss of habitat capability explains the strong positive relation, spatially, between intrinsic habitat capability and current habitat suitability ($r^2 = 0.71$, $df = 1$ and 1313 , $F = 3133$, $P < 0.0001$).

4.3. Extent and location of remote and capable habitat

Areas that exceeded our remoteness and habitat capability thresholds were extensive. A total of $159,051 \text{ km}^2$ in 39 areas exceeded our remoteness threshold and $187,347 \text{ km}^2$ in two areas exceeded our habitat capability threshold (Fig. 3). The intersection of sufficiently capable and sufficiently remote habitats amounted to $42,150 \text{ km}^2$ in 29 areas. After removing areas (i.e., patches) $<300 \text{ km}^2$ in size, total area was reduced to $41,563 \text{ km}^2$ in 14 patches. This biophysically suitable habitat comprised 6.8% of the total study area.

4.4. Biophysical suitability of potential restoration areas

Of the $41,563 \text{ km}^2$ of biophysically suitable habitat, $39,854 \text{ km}^2$ occurred in 9 potential source patches (i.e., $>900 \text{ km}^2$ in size; Fig. 4; Table 2). Five source patches grouped in two complexes were within potential reciprocal dispersal of each other: the Canadian River complex (Vermejo and South Canadian patches), and the Gila/Mogollon complex (the Gila/Mogollon, Pine Mountain, and Cibola patches). The Gila/Mogollon complex exceeded $20,000 \text{ km}^2$ in size (Fig. 5a). Individually, the Gila/Mogollon and Galenas patches exhibited the most promise as restoration areas on the basis of size and shape criteria.

The Gila/Mogollon, Vermejo, and Galenas patches exhibited the highest median capability and suitability (Fig. 5b). The extent of overlap with wapiti range and oak dominated vegetation was also greatest in the Gila/ Mogollon complex (Fig. 6; Table 2), although relative overlap with wapiti range was also high in the Vermejo and Mt. Taylor source patches. The South Canadian and Cibola patches exhibited the least overlap with either wapiti or oaks.

4.5. Potential compatibility with resident humans

Overall, socio-economic indicators for counties containing prospective grizzly bear restoration areas in New Mexico and Arizona differed from indicators for counties in the contiguous US currently with grizzly bears (Wilk's $\Lambda = 0.44$, $df = 7$ and 40 , $F = 7.31$, $P < 0.0001$). Of the individual indicators, % of the populace employed in agriculture, median family income, % of populace with a college degree, and cattle densities were all substantially lower in Arizona and New Mexico (Table 3). Discriminating counties inhabited by grizzly bears from counties with prospective habitat in the Southwest, total-sample standardized canonical coefficients for the first (and only) canonical variable were 0.94 (% of populace in agricultural employment), 0.63 (cattle density), 0.45 (% of populace with a college degree), 0.23 (median family income), 0.04 (sheep density), -0.28 (% of populace older than 64), and -0.82 (% of land in private agriculture). The squared canonical correlation = 0.56 and the discriminant function error rate based on cross-validation = 12.5%. Of the Southwest counties, Greenlee (AZ), Catron (NM), Chaves (NM), Otero (NM), and Socorro (NM) were most like counties currently with grizzly bears (Table 3; Fig. 7a).

Indexed potential acceptance of grizzly bears did not differ substantially between counties currently with grizzly bears and counties in the Southwest containing prospective grizzly bear restoration areas (Table 3). In the Southwest, Coconino (AZ), Graham (AZ), Yavapai (AZ),

Grant (NM), San Miguel (NM) and Socorro (NM) Counties were potentially most accepting of grizzly bears. Gila (AZ), Catron (NM), Colfax (NM), Guadalupe (NM), Lincoln (NM), Sierra (NM), and Torrance (NM) Counties were potentially least accepting. Potentially hostile counties in New Mexico were arrayed in a contiguous arc through the south-central part of the state that included parts of the Gila/Mogollon complex and the Galenas source patch (Fig. 7b). Counties potentially more accepting of grizzly bears were scattered (Fig. 7a), but included substantial parts of the Gila/Mogollon complex as well.

Most of the Canadian complex and the Coconino, Mt. Taylor, and Galenas patches consisted of conditions where conflict with humans was likely. Areas of above-median capability and below-median remoteness were common throughout all of these areas (Fig. 8a). The Canadian complex and Galenas patch were also distinguished by prevalent private lands (Fig. 8b). The Gila/Mogollon complex contained the largest areas free of high potential for conflict, although private lands and areas where high capability juxtaposed with relatively high potential for human activity were common in the northern part of this complex in New Mexico, in Catron County.

Discussion

5.1. General issues

Restorations of extirpated species have been most successful where undertaken in areas large enough to sustain large populations, where the causes of historic extirpations were rectified, and where habitat was highly productive (Smith and Clark, 1996; Wolf et al., 1998; Miller et al., 1999; Breitenmoser et al., 2001). There also is evidence that success rates were higher with omnivores and where undertaken in the core of historic range (Wolf et al., 1998). Grizzly bears are omnivores and thus benefit from dietary flexibility that, in theory, buffers them

from environmental vicissitudes of potential restoration sites. On the other hand, our study area is not near the core of historic North American grizzly bear range, although it is not clear what being “near the core” means functionally (Lomolino and Channell, 1998). Grizzly bears were extirpated deterministically at scales considerably finer than the scale of their North American range (Mattson and Merrill, In press). Consistent with Lomolino and Channell (1995), such a pattern tends to discount the importance of being near the core of historic range rather than elsewhere. Regardless, the most robust features of past successes pertained to the extent, productivity, and current hostility of restoration areas. Our challenge was to bridge from these generalities to a meaningful site-specific assessment.

Our standards and model metrics explicitly addressed productivity, hostility, and extent of prospective restoration areas in New Mexico and Arizona. Given that humans are the primary cause of historical extirpations and the current cause of almost all grizzly bear deaths (Mattson et al. 1996a; McLellan et al., 1999; Mattson and Merrill, In press), we addressed hostility by site-specific remoteness from humans (i.e., potential frequency of contact) and potential for conflict, and by county-level potential acceptance of bears by residents (i.e., potential lethality of humans). Habitat capability addressed the intrinsic ability of a site to support bears. Standards for size and shape addressed the broader-scale sufficiency of potential restoration areas. Thus, in concept, we addressed the three key dimensions of potential restoration areas. However, the dimensionless indices that were model outputs needed to be translated into some meaningful measure of potential grizzly bear presence and persistence.

We calibrated our landscape metrics to the historical presence of grizzly bears in our study area (i.e., habitat capability) or, where calibration was to the presence of grizzly bears elsewhere, we used metrics with consistent meaning across regions (i.e., remoteness). Partly we

took this approach because we had no resident animals from which to estimate *in situ* vital rates. However, we had other compelling reasons. Vital rates are normally calculated on an annual basis for specific animals and thereby vary in time and space as a function of annual variation in environmental conditions at the scale of annual ranges. This makes extrapolation of vital rates from other times and places particularly uncertain and risky, and ostensibly necessitates landscape metrics for calibration that have a temporal resolution of years and a spatial resolution of annual ranges if not finer. Vital rates also need to be translated by demographic models into probabilistic estimates of long-term population growth. Such an exercise requires, at a minimum, that environmental variation and density dependence be specified either on the basis of empirical estimates or assumptions about population processes (Boyce et al., 2001). We know little about such phenomena among bears, but enough to know that they have potentially major effects on the performance of demographic models and related uncertainties of population projections (Mills et al., 1996; Boyce et al., 2001). Thus, vital rates estimated elsewhere combined with demographic models that contain momentous assumptions create large uncertainties (Boyce et al., 2001; Breitenmoser et al., 2001).

The approach we took to calibration and setting standards was, by comparison, more likely to be robust. By generalizing and calibrating our landscape metrics to and at the scale of decades and life-ranges, we subsumed irrelevant details of finer-scale variation. Moreover, we calibrated to the presence of grizzly bears documented at comparably broad temporal and spatial scales as well as during times of duress. The scale of model and data matched and the resulting calibration and thresholds reflected broad-scale long-term persistence of grizzly bears under onerous conditions – during times when human persecution was often intense. Thus, our appraisal is relevant to long time frames and more likely to be conservative than liberal with

respect to the risk of over-estimating suitability of biophysical conditions for grizzly bears in the southwestern US.

5.2. Prospects for restoration

Acorns and tissue from wapiti are potentially important grizzly bear foods. Both are abundant in our study area (about 24,000 wapiti [Bryant and Maser, 1982] and extensive stands of numerous species of oaks [Brown, 1994]). Diets of grizzly bears are distinguished from diets of black bears, *Ursus americanus* and *U. thibetanus*, in similar habitats by greater consumption of tissue from ungulates (Mattson, 1998; Jacoby et al., 1999). Moreover, tissue from ungulates is known to be an important source of energy for grizzly bears where wapiti and bison are currently numerous as well as historically in the southwestern US (Mattson, 1997a; Jacoby et al., 1999). The importance of high-fat-content fruits and seeds from trees is well documented for both black and grizzly bears. Consumption of seeds produced by whitebark pine *Pinus albicaulis* has major positive effects on reproductive success of female grizzly bears in the Yellowstone region of the western US (Mattson, 2000). In the Southwest, black bears heavily consume acorns of Gambel and live oaks whenever and wherever they are available (LeCount et al., 1984; Beck, 1991; Costello et al., 2001), also with major positive effects on the reproductive success of females (Costello et al., 2001). Our model of habitat capability explicitly incorporates effects of oak-dominated vegetation, and so overlap of prospective restoration areas with this landscape feature merely confirms model calculations. However, for lack of fine-scale information on wapiti distributions during 1850, our model of habitat capability does not incorporate effects of this potential grizzly bear food (Mattson and Merrill, In press). Thus, current overlap between wapiti range and prospective restoration areas is important added information.

Biophysically, prospects for restoration of grizzly bears in the southwestern US are best in the Gila/Mogollon complex, in the south-central part of our study area. Moreover, by standards of grizzly bear persistence in northern Idaho between 1950 and 1993 and in the contiguous US between 1920 and 1970, prospects for restoration are good. The Gila/Mogollon complex exceeds 20,000 km² in size and, compared to other potential source areas, contains habitat with some of the highest median suitability and capability, overlaps most extensively with ranges of wapiti and oak-dominated vegetation, and contains the most extensive area free of high potential for conflict with resident humans. Suitable habitat may also extend beyond the boundaries that we have identified, to the south and west in Arizona. These potential additions are sufficiently remote from humans, contain extensive areas of oak-dominated vegetation, and currently support some of the highest densities of black bears in the Southwest (LeCount, 1982).

Human residents of some counties containing the Gila/Mogollon complex are also potentially accepting of large carnivores like grizzly bears (i.e., Graham [AZ], Grant [NM], and Socorro [NM]). On the other hand, prospects are compromised by the attenuated shape of this complex and the potential hostility of residents in other counties (i.e., Gila [AZ], Catron [NM], Sierra [NM]). Attenuation allows for greater intrusion of negative edge effects (Woodroffe and Ginsberg, 1998; Mattson and Merrill, In press). Potential hostility is related to a mix of high levels of employment in agriculture (Catron County), large numbers of elderly residents (Sierra County), and low levels of education (Gila and Sierra Counties). Relatively high rates of exposure to humans and conflict with private property rights are also potential problems in parts of this complex, especially in Catron County.

Other areas besides the Gila/Mogollon complex may have potential for restoration of grizzly bears. In particular, the Galenas patch has potential because of its relatively large size

(>7,000 km²) and relative nearness to the Guadalupe patch. With restoration of intervening habitat, connectivity between these two patches is a possibility. On the other hand, the Galenas patch is elongate, contained almost wholly in counties where residents are likely to be quite hostile to restoration of grizzly bears (Sierra, Lincoln, and Guadalupe Counties), and typified by a high potential for conflict between humans and bears because of extensive private property and relatively high exposure rates. The Canadian River complex also exhibits potential. It is relatively isolated from other patches in our study area but is favored by high median suitability and capability, high overlap with wapiti range, greater potential neutrality among resident humans, and moderately large size. Moreover, this complex may be larger than shown by our analysis because it is contiguous with currently unappraised habitat in Colorado. A large privately owned population of bison also resides in the Vermejo patch on a large tract of private land where grizzly bears may be accepted (M. Phillips, Turner Endangered Species Fund, Personal Communication). This is significant because bison are, *per capita*, scavenged by grizzly bears more frequently and heavily than any other type of ungulate in the Yellowstone region, where moose *Alces alces*, wapiti, bison, and deer *Odocoileus* spp. are all present in substantial numbers (Green et al., 1997; Mattson, 1997a).

5.3. *Potential next steps*

This broad-scale analysis provides important information for those debating the merits of restoring grizzly bears to Arizona and New Mexico. Debate often founders on basic issues such as the existence of biophysically suitable habitat (Merrill et al., 1999). We can state with confidence that habitat suitable for restoration of grizzly bears does exist in the southwestern US – in the Gila/Mogollon complex. With this in mind, future discussion can hopefully focus on issues related to economics, policy, and human values as well as the merits of further more

focused analysis. If there are those who see value in restoring grizzly bears to biophysically suitable areas in this region, then a finer-scale analysis focused on complexes and patches with the greatest potential is a necessary next step.

Before proceeding with restoration planning, additional information should be generated on potential seasonal distributions of restored bears, potential distribution and types of conflict sites, and the nature and potential alleviation of human concerns and issues. Models exist that can be used to appraise the value of potential grizzly bear foods and, from that and other information on food distribution and abundance, predict seasonal distributions of grizzly bears (e.g.; for grizzly bear foods in general, Mattson et al. [1999]; for fleshy fruits, Welch et al. [1997]; for grazed herbaceous foods, Rode et al. [2001]; for ants, Mattson [2001]; for roots, Mattson [1997b], and for ungulates, Mattson [1997a] and Green et al. [1997]). This information can be combined with spatially explicit inventories of potentially problematic human-related features to identify sites with high potential for conflicts and, moreover, identify the reasons why. Perhaps most important, residents of potential restoration areas need to be surveyed to clarify their questions, concerns, and level of acceptance. From this, situations can be identified where sharing of information or participation in restoration and development of management strategies might legitimately gain greater acceptance of grizzly bears.

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Table 1

Percent losses of the capability of habitat to support grizzly bears between 1850 and the present in Arizona and New Mexico, U.S.A., by categories of loss severity (proportional) and categories of habitat capability under pristine conditions.

Pristine habitat capability	Proportional loss of habitat capability			
	0–0.25	0.26–0.5	0.51–0.75	0.76–1
0–0.047	0.4	4.0	7.9	14.2
0.048–0.123	0.9	4.3	8.1	14.5
0.123–0.503	0.2	1.4	6.8	13.8
0.504–1	0.3	0.8	9.3	12.9
Total %	1.8	10.6	32.2	55.5

Table 2

Name, size, shape index, median productivity and suitability, and percent overlap with ranges of wapiti *Cervus elaphus* and oak-dominated vegetation for source patches and complexes potentially suited for grizzly bear restoration in Arizona and New Mexico, U.S.A.

Complex or patch	Area (km ²)	Shape index	Median capability	Median suitability	% wapiti range	% oak- dominated
Canadian River	4 495	2.52	0.61	0.61	50	0
Vermejo	2 234	1.72	0.76	0.68	100	0
South Canadian	2 261	1.84	0.51	0.55	1	0
Coconino	1 453	1.48	0.50	0.54	31	0
Mt. Taylor	1 842	1.54	0.69	0.68	95	0
Galenas	7 504	2.31	0.74	0.68	32	0
Gila/Mogollon	21 717	3.99	0.72	0.70	60	15
Cibola	1 431	1.54	0.59	0.61	8	0
Pine Mountain	2 058	1.49	0.58	0.68	7	53
Gila/Mogollon	18 228	3.42	0.74	0.71	70	12
Guadalupe	1 640	1.66	0.56	0.57	28	0

Figure captions

Fig. 1. The states of Arizona and New Mexico, U.S.A., showing the largest cities in this region and major topographic features within or near the range of prospective grizzly bear restoration areas. Lower elevations are shown as progressively darker shading.

Fig. 2. (a) Relations between the probability that a 1-km² grid cell in northern Idaho, U.S.A., contained a grizzly bear sighting, versus not, and indexed current remoteness from humans. The dashed line shows the best-fit polynomial model, the solid line the best-fit single term model. The threshold *p* value (0.012) and corresponding value of remoteness (0.35) are shown. (b) Relations between the probability that a location was that of a historical grizzly bear sighting versus a random point in Arizona and New Mexico, U.S.A., and indices of habitat capability (solid line and solid circles) and current habitat suitability (dashed line and open circles). The threshold *p* value (0.5) and corresponding value of habitat capability (0.35) are shown. In both (a) and (b) circles and associated standard error bars are for quintiles or quartiles of the data and are shown to illustrate goodness-of-fit.

Fig. 3. (a) Remoteness from people and (b) intrinsic grizzly bear habitat capability for Arizona and New Mexico, U.S.A. Progressively greater remoteness or capability is indicated by progressively darker shading. Areas exceeding habitat capability and remoteness thresholds are delineated by dashed lines in (a) and (b); triangles denote historical grizzly bear sightings. Areas in (c) exceeding only the remoteness threshold are shown in dark gray, areas exceeding only the habitat capability threshold are shown in light gray, and areas exceeding both thresholds are shown in black.

Fig. 4. Potential source patches and their names, for grizzly bears in Arizona and New Mexico, U.S.A. Potential source patches are delineated by a solid line and 20 km buffers corresponding to zones of potential dispersal from source patches by a dashed line. Progressively darker shading indicates increasing habitat suitability within potential source patches.

Fig. 5. (a) Plot of potential source patches and source patch complexes for grizzly bears in Arizona and New Mexico, U.S.A., relative to shape index and size. Areas within the plot denoted by '1', '2', and '3' identify ranges of combinations of size and shape with progressively lower odds of sustaining a grizzly bear population (from Mattson and Merrill, In press). (b) Plot of potential source patches and sources patch complexes relative to their median habitat capability and suitability. In both (a) and (b), individual patches and complexes are denoted by gray and black circles, respectively.

Fig. 6. Joint distribution of potential source patches for grizzly bears in Arizona and New Mexico, U.S.A., and ranges of wapiti and oak-dominated vegetation. A thick solid line delineates source patches. Ranges of wapiti are shaded light gray, of oak-dominated vegetation are shaded dark gray, and of both are shaded black.

Fig. 7. Potential source patches for grizzly bears in Arizona and New Mexico, U.S.A., and: (a) counties either potentially most accepting of grizzly bears or most similar by socio-economic indicators to counties currently with grizzly bears in the contiguous U.S.A.; (b) counties either potentially least accepting of grizzly bears or most dissimilar to counties currently with grizzly bears. A dashed line delineates potential source patches. The most similar counties are denoted by right diagonal cross-hatching, the most dissimilar by

horizontal cross-hatching, the potentially most accepting by left diagonal cross-hatching, and the potentially least accepting by vertical cross-hatching.

Fig. 8. Areas with high potential for conflict between humans and restored grizzly bears in Arizona and New Mexico, U.S.A., relative to potential grizzly bear source patches and dispersal areas. Areas with high potential for conflict by virtue of (a) above median habitat capability and below median remoteness or (b) overlap of privately owned land with potential source patches and dispersal areas are shown in dark gray. Source patches are delineated by a thin solid line and surrounding potential dispersal areas by a thick dashed line.















