Appendix A. Chronological List of the Grizzly Bear Recovery Process for the Greater Yellowstone Area

I. Grizzly Bear Recovery Plan revision (1993)

II. Workshop on habitat-based recovery criteria

Development of habitat-based recovery criteria draft for Yellowstone area

Agency review

Public comment

Incorporation of comments

Draft final Habitat Criteria

III. Conservation Strategy development for the Yellowstone area, including habitat-based recovery criteria

Agency review

Public comment

Incorporation of comments

Draft final Conservation Strategy

MOU to implement the Conservation Strategy signed by all agencies

IV. Achievement of recovery targets in the Recovery Plan for demographic values and for habitat criteria specified for that grizzly bear population

V. Formal consideration of status change

VI. Preparation of Proposed Rule, if warranted.

Publication of Proposed Rule in the Federal Register. Proposed Rule documents the status of the population according to the five factors in ESA Section 4(a)(1) including population and habitat status, and references Conservation Strategy for documentation of the existence of adequate regulatory mechanisms and consideration of DPS policy.

VII. Public comment period with public hearings

VIII. Consideration and incorporation of public comments and any new information developed as a result of the comment period

IX. Publication of Final Rule in the Federal Register of status change or continuation of listed status in conjunction with release of the final Conservation Strategy, final Habitat Criteria, and final DPS analysis

Appendix B. Methods to Calculate the Total Numbers of Adult Females from Counts of Unduplicated Females with Cubs

Estimating Numbers of Females with Cubs-of-the-Year in the Yellowstone Grizzly Bear Population

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Abstract: For grizzly bears (*Ursus arctos horribilis*) in the Greater Yellowstone Ecosystem (GYE), minimum population size and allowable numbers of human-caused mortalities have been calculated as a function of the number of unique females with cubs-of-the-year (F_{CUB}) seen during a 3-year period. This approach underestimates the total number of F_{CUB}, thereby biasing estimates of population size and sustainable mortality. Also, it does not permit calculation of valid confidence bounds. Many statistical methods can resolve or mitigate these problems, but there is no universal best method. Instead, relative performances of different methods can vary with population size, sample size, and degree of heterogeneity among sighting probabilities for individual animals. We compared 7 nonparametric estimators, using Monte Carlo techniques to assess performances over the range of sampling conditions deemed plausible for the Yellowstone population. Our goal was to estimate the number of F_{CUB} present in the population each year. Our evaluation differed from previous comparisons of such estimators by including sample coverage methods and by treating individual sightings, rather than sample periods, as the sample unit. Consequently, our conclusions also differ from earlier studies. Recommendations regarding estimators and necessary sample sizes are presented, together with estimates of annual numbers of F_{CUB} in the Yellowstone population with bootstrap confidence bounds.

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Criteria for recovering the grizzly bear in the lower United States include annual limits on mortalities (U.S. Fish and Wildlife Service 1993). Since 1993, these limits have been calculated as a function of the number of F_{CUB} present in the population, as estimated during 6-year running periods. Currently, the number of F_{CUB} present each year (N) is estimated as the number of such animals actually observed ($\hat{N}_{\text{Obs}}$). To the extent that criteria for distinguishing family groups are conservative (see Knight et al. 1995), and because it is highly unlikely that all such animals are seen, $\hat{N}_{\text{Obs}}$ almost certainly underestimates N. This helps ensure that mortality limits are conservative, but precludes calculation of valid confidence bounds. Moreover, use of a biased estimator like $\hat{N}_{\text{Obs}}$ effectively removes decisions regarding the appropriate degree of conservatism from the purview of managers. This is not a trivial issue because the magnitudes of biases and uncertainties inherent in $\hat{N}_{\text{Obs}}$ may be biologically and managerially significant. Efforts to calculate statistically sound estimates of N have focused on parametric approaches. Eberhardt and Knight (1996) applied the Peterson-type estimators of Chapman and Bailey (Seber 1982), and Boyce et al. (1999, Cumulative counts of unique individuals for estimating population size, U.S. Fish and Wildlife Service, Missoula, Montana, USA) recommended the maximum likelihood method of Lewontin and Prout (1956). These methods assume that each family group has an equal probability of being sighted. Because this assumption is untenable for the Yellowstone data (Keating et al., 1999, Estimating the number of females with cubs-of-the-year in the Yellowstone grizzly bear population: are maximum-likelihood estimates that assume equal sightability conservative? U.S. Fish and Wildlife Service, Missoula, Montana, USA.), estimates based on these methods will be negatively biased. Seeking a more robust approach, Boyce et al. (2001) recommended joint estimation of N over all years using an estimator derived from the zero-truncated negative binomial distribution. This estimator can be traced to

Greenwood and Yule (1920), with early applications to wildlife population estimation by Tanton (1965, 1969) and Taylor (1966). The sampling model assumed by the negative binomial estimator allows for heterogeneous sighting probabilities among individuals and, thus, is equivalent to model M_h of Otis et al. (1978). Unfortunately, Boyce et al. (2001) found that the negative binomial estimator gave reasonable results only when the coefficient of variation among individual sighting probabilities (CV) was assumed to be constant over time. This assumption is difficult to justify for grizzly bears in Yellowstone, where year-to-year differences in distributions and abundances of foods affect bear movement patterns and, in turn, the likelihood of seeing particular bears (Picton et al. 1986). Such differences almost certainly affect heterogeneity among individual sighting probabilities, implying that CV varies among years. Also, because the size, distribution, and behavior of bear populations may interact in ways that affect sightability (Keating 1986), CV likely changes with N. The claim of an increased bear population in Yellowstone (Boyce et al. 2001), therefore, is inconsistent with the assumption of a constant CV. The joint estimation procedure recommended by Boyce et al. (2001) suffers other drawbacks as well. Most seriously, estimates of N from previous years may change retrospectively as new data are added — a property that is justifiable only if CV is truly constant over time. Overall, problems with the parametric methods used to date argue for considering other alternatives.

Many nonparametric estimators might apply to this problem (e.g., Otis et al. 1978, Bunge and Fitzpatrick 1993, Lee and Chao 1994). Indeed, when estimating N under model M_h, many studies have favored non-parametric methods such as the jackknife (Burnham and Overton 1978, 1979), Chao (Chao 1984, 1989), and sample coverage estimators (Chao and Lee 1992, Lee and Chao 1994). Among the nonparametric methods available, however, there is no universal best choice, as relative performances can vary with N, CV, or sample size (Burnham

and Overton 1979, Smith and van Belle 1984, Chao 1988). What we require is an estimator that is reasonably robust to variations in these parameters over the range of values experienced when sampling the Yellowstone grizzly bear population. To identify such an estimator, we used Monte Carlo methods to compare performances of 7 nonparametric methods when sampling from a range of conditions that encompassed those deemed plausible for observations of F_{CUB} in the GYE.

Methods

General Problem and Notation

The sampling model we used approximates the true sampling scheme, in which reports of F_{CUB} come from observers using various sampling methods (ground-based observation, trapping, systematic fixed-wing observations, or fixed-wing observations made incidental to other work). Because the sampling period associated with each of these methods varies considerably (or, in some cases, is undefined) we used the sighting of an individual F_{CUB} as the sample unit. The problem of estimating population size from repeated sightings of unique individuals may then be phrased as a special case of the more general model in which multiple individuals may be sighted during a given sampling period (e.g., Otis et al. 1978).

Suppose that, during a given year, after recording *n* independent random sightings of individuals from a closed population of size *N* (where *N* is unknown), we observe *m* unique animals. The average probability that any particular sighting will be of the *i*th individual is p_i, and probabilities for all *N* individuals are given by **p** = $(p_1, p_2, ..., p_N)$, where $\sum_{i=1}^{N} p_i = 1$. Because the model allows for heterogeneous p_i values, temporal or spatial differences in habitat use or sampling effort are incorporated into **p**, as are differences in probabilities of reporting and recording sightings of particular animals. We assume all individuals are correctly identified (consequences of misidentification are considered below). In our sample, individuals were observed with

frequency **n** = $(n_1, n_2, ..., n_N)$, which is multinomially distributed with cell probabilities $(p_1, p_2, ..., p_N)$. However, we do not know the identities of the *N* - *m* animals for which $n_i = 0$. The number of different individuals observed exactly *j* times was f_j, and $\mathbf{f} = (f_0, f_1, f_2, ..., f_n)$ is fully observable except for f_0, the number of bears not observed in our sample. Important relationships include $n = \sum_{i=1}^{N} n_i = \sum_{j=1}^{n} j f_j$, $m = \sum_{j=1}^{n} f_j$, and $N - m = f_0$. The problem is to estimate *N* (or, equivalently, f_0) using only the observable information in **f** and *n*.

In this idealized model, all information about population size is obtained from the *n* randomly sighted individuals. For the Yellowstone grizzly bear population, observations of radiomarked F_{CUB} made during radiorelocation flights provide additional information from non-randomly sighted individuals. In particular, observations of otherwise unobserved F_{CUB} may be added to *m* to improve the estimate of minimum population size, yielding $\hat{N}_{\text{Obs}} \geq m$. $\hat{N}_{\text{Obs}}$ provides a natural lower bound for estimating *N* and is the estimator that has been used previously to set annual mortality limits. Overall, we seek an estimator that improves upon $\hat{N}_{\text{Obs}}$ while minimizing the risk of overestimating *N*.

The Estimators

In addition to *m* and $\hat{N}_{\text{Obs}}$, which we included in our analyses for comparative purposes, we evaluated 7 non-parametric estimators (see Table 1 for example calculations). The first 5 methods we considered estimate *N* as $\hat{N} = m + \hat{f}_0$, where $\hat{f}_0$ is an estimate of the number of unobserved individuals.

We first examined Chao's (1984) estimator,

$$\hat{N}_{\text{Chao1}} = m + \frac{f_1^2}{2f_2}. \qquad (1)$$

In Eq. (1), $\hat{f}_0 = f_1^2/(2f_2)$. Using $\hat{N}_{\text{Chao1}}$, the statistical expectation for the estimate, $E(\hat{N})$, equals *N* only when sighting probabilities are the same for all animals; i.e., when CV = 0. Theoretically, when CV > 0, $E(\hat{N}) < N$ (Chao 1984). This does not ensure $\hat{N}_{\text{Chao1}} \leq N$ in all cases, but does suggest that $\hat{N}_{\text{Chao1}}$ might provide an inherently conservative approach to estimating *N*. We also considered a similar bias-corrected form of this estimator, developed by Chao (1989). Where the sample unit is the individual animal, Chao's (1989) estimator is given by (Wilson and Collins 1992),

$$\hat{N}_{\text{Chao2}} = m + \frac{f_1^2 - f_1}{2(f_2 + 1)}.$$

Here, $\hat{f}_0 = (f_1^2 - f_1)/[2(f_2 + 1)]$. Unlike $\hat{N}_{\text{Chao1}}$, $\hat{N}_{\text{Chao2}}$ will yield an estimate even when $f_2 = 0$.

Burnham and Overton (1978, 1979) devised a jackknife estimator $(\hat{N}_{Jk})$, of the general form

$$\hat{N}_{\mathrm{J}k} = m + \sum_{j=1}^{k} \alpha_{jk} f_j,$$

where α_{jk} is a coefficient in terms of *n*, and α_{jk} = 0 when *j* > *k* (see Table 2). Here, f_0 is estimated as the series $\sum_{j=1}^{k} \alpha_{jk} f_j$. Theoretically, jackknife estimates of order *k* = 1 to *n* could be calculated, but variance increases rapidly with *k* so that, in practice, *k* is small (Burnham and Overton 1979). We considered the first- and second-order jackknife estimators ($\hat{N}_{\mathrm{J}1}$ and $\hat{N}_{\mathrm{J}2}$, respectively; Table 2), as well as a best *k*th-order jackknife estimator. Burnham and Overton (1979) suggested 2 methods for choosing a best value for *k* for a particular study. Because previous work showed little difference between them (K.A. Keating unpublished data), we considered only their first method, which evaluates estimates of order *k* = 1 to 5 (Table 2). The method is as follows. Beginning with *k* = 1 and proceeding to subsequently higher values of *k*, test the null hypothesis that $E(\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k}) = 0$ versus the alternative hypothesis that $E(\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k}) \neq 0$. If the observed difference is not significant, testing ends and $\hat{N}_{\mathrm{J}k}$ is taken as the best jackknife estimate. We reference the resulting *k*th-order estimate as $\hat{N}_{\mathrm{J}k1}$. The test is based on the statistic

$$T_k = \frac{\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k}}{[\hat{\mathrm{var}}(\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k} \mid m)]^{1/2}},$$

$$\hat{\mathrm{var}}(\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k} \mid m) = \frac{m}{m-1}\left[\sum_{j=1}^{n} (b_j)^2 f_j - \frac{(\hat{N}_{\mathrm{J},k+1} - \hat{N}_{\mathrm{J}k})^2}{m}\right],$$

where

and $b_j = \alpha_{j,k+1} - \alpha_{jk}$. T_k was evaluated at $\alpha = 0.05$ using *P* values determined from the standard normal distribution.

Chao and Lee (1992) proposed an estimator based on sample coverage ©, where *C* is the sum of the p_i values for the *m* individuals actually observed in the sample. Lee and Chao (1994) offered 2 estimators of *C* that, in the notation of our sampling model, are given by

$$\hat{C}_1 = 1 - \frac{f_1}{n}, \qquad (2)$$

and

$$\hat{C}_2 = 1 - \frac{f_1 - 2f_2/(n-1)}{n}. \qquad (3)$$

In Eqs. (2) and (3), the quantities f_1/n and $[f_1 - 2f_2/(n-1)]/n$, respectively, estimate the sum of the p_i values for the f_0 unobserved animals. For our model (equivalent to model M_h of Otis et al. [1978]), Lee and Chao (1994) then estimated *N* as

$$\begin{aligned} \hat{N}_{\mathrm{SC}j} &= \frac{m}{\hat{C}_j} + \frac{f_1}{\hat{C}_j}\hat{\gamma}^2 \\ &= \frac{m + f_1\hat{\gamma}^2}{\hat{C}_j}, \end{aligned} \qquad (4)$$

where *j* = 1 or 2, and γ is a measure of the coefficient of variation of the p_i's. Essentially, Eq. (4) begins with a Peterson-type estimator ($m/\hat{C}_j$) that assumes equal sightability (i.e., all $p_i = 1/N$; Darroch and Ratcliff 1980), then adds a bias correction term $(f_1\hat{\gamma}^2/\hat{C}_j)$ that increases with heterogeneity, as estimated by $\hat{\gamma}^2$. Put another way, the quantity $f_1\hat{\gamma}^2$ estimates the number of additional individuals that would have been observed if **p** had, in fact, been homogeneous. Adding this to *m* then dividing by the estimated coverage estimates *N*. Where the sample unit is the sighting of an individual animal, $\hat{\gamma}^2$ is calculated as (Chao and Lee 1992),

$$\hat{\gamma}^2 = \max\left\{ \hat{N}\sum_{j=1}^{n} \frac{j(j-1)f_j}{n(n-1)} - 1, \quad 0 \right\}. \qquad (5)$$

Calculation of $\hat{\gamma}^2$ requires an initial estimate of *N*. Following Chao and Lee (1992), we used $\hat{N} = m/\hat{C}_j$. We considered but did not use the partitioned sample coverage estimator of Chao et al. (1993, 2000) because preliminary Monte Carlo results showed the method offered no advantage over $\hat{N}_{\mathrm{SC}j}$ when applied to our field data.

Monte Carlo Comparisons

Estimator performances were compared using Monte Carlo methods. Parameters for the Monte Carlo sampling were chosen to encompass the range of values deemed plausible when sampling F_{CUB} in the GYE. Overall, we simulated 15 populations, including all combinations of *N* = 20, 40, and 60 animals, where the coefficient of variation among the p_i values was set to CV = 0.0, 0.25, 0.50, 0.75, or 1.0. We calculated p_i as the integral of a standard beta distribution over the interval (*i* – 1)/*N* to *i/N*; i.e.,

$$p_i = I_{i/N}(a,b) - I_{(i-1)/N}(a,b), \quad (6)$$

where $I_x(a,b)$ is the incomplete beta function ratio with parameters *a* and *b* (Johnson et al. 1995). We used a downhill simplex (Press et al. 1992) to select values for *a* and *b* (Table 3) that gave the desired CV among the p_i values. We then sampled each population, with replacement, by generating *n* pseudorandom numbers from the specified beta distribution and tallying each as a sighting of the *i*th animal if it fell within the interval (*i* – 1)/*N* to *i/N*. We chose *n* so that the number of sightings per individual in the population (*n*/*N*) was equal to 0.5, 1.0, 1.5, 2.0, 2.5, 3.0, 3.5, or 4.0. After each sampling bout, we estimated *N* using each of the estimators described above. This process was repeated 1,000 times for each parameterization of the

model. For each parameterization and estimator, performance was summarized as the bias and root mean square error (RMSE) of the estimator, where RMSE = $\sqrt{\text{bias}^2 + \text{SD}^2}$. In addition, 2 estimators ($\hat{N}_{\text{SC1}}$ and $\hat{N}_{\text{SC2}}$) yielded explicit estimates of CV, in the form of $\hat{\gamma}$ (Eq. 5).

Following the above analyses, the most promising estimator was selected. Confidence bounds for estimates based on the best method were calculated using the method of Boyce et al. (2001), in which bootstrap samples were drawn from the distribution of individual sighting frequencies implied by $\hat{N}$ (i.e., from the estimate of the vector **n**). Details are as follows. A model population with $\hat{N}$ individuals was constructed and the first *m* individuals were assigned sighting frequencies $\mathbf{n}^* = (n_1^*,\ n_2^*,\ ...,\ n_m^*)$, corresponding to the actual sighting frequencies (n_i values) for the *m* animals observed in the original sample. The remaining $\hat{N}$ - *m* individuals were assigned sighting frequencies of 0. A bootstrap sample of $\hat{N}$ (rounded to the nearest integer) individual sighting frequencies (n_i^* values) was then randomly drawn with replacement from $\mathbf{n}^*$. The number of samples for which $n_i^* = j$ was tabulated as f_j^* , giving the bootstrap sighting frequency vector $\mathbf{f}^* = (f_1^*, f_2^*, ..., f_n^*)$, and the bootstrap number of sightings $n^* = \sum_{j=1}^{n} j f_j^*$. The estimate was then recalculated using the information in $\mathbf{f}^*$ and n^* . This procedure was repeated 1,000 times for each estimate. Confidence bounds were calculated using both the percentile and bias-corrected-and-accelerated (BCA) methods (Efron and Tibshirani 1993). We assessed performances of the 2 methods by comparing observed versus nominal coverages.

Although 90 or 95% confidence bounds are normal for scientific hypothesis testing, managers may appropriately choose a higher level of risk. Thus, we compared coverages for lower, 1-tailed 70, 80, 90, and 95% confidence bounds. Earlier studies reported 2-tailed confidence

bounds (e.g., Eberhardt and Knight 1996, Boyce et al. 2001). However, we believe 2-tailed bounds are inappropriate for this problem because managers charged with recovering the Yellowstone grizzly bear population are concerned with possible overharvest, not underharvest. Thus, they seek assurance that the true population size is greater than or equal to the estimated size. It follows that lower, 1-tailed confidence bounds provide the appropriate measure of uncertainty.

Field Data

Sightings of F_{CUB} were examined for 1986–2001. We considered only sightings from within the grizzly bear recovery zone and the surrounding 10-mile buffer area because calculated mortality limits only apply to human-caused mortalities within this area. Boyce et al. (2001) considered sightings throughout the GYE. Consequently, sample sizes (*n* values) and numbers of unique, randomly observed F_{CUB} (*m* values) reported herein differ slightly from values reported by Boyce et al. (2001).

For each year, unique family groups were distinguished as per Knight et al. (1995). Observations of radiocollared animals made during radiolocation flights were included when calculating the minimum number of F_{CUB} known to exist in the population each year $(\hat{N}_{\text{Obs}})$, but were excluded from statistical estimates of *N* because such sightings were non-random. Sightings were summarized by year as the number of unique family groups seen once, twice, etc. Total numbers of F_{CUB} for each year were then estimated using the method selected following our Monte Carlo comparisons. Lower, 1-tailed confidence bounds were calculated using the selected bootstrap procedure.

Results

Monte Carlo Comparisons

Patterns of estimator performance varied little with population size. For brevity, therefore, we discuss only results for model populations with N = 40 individuals.

Population Estimates.—All estimates tended to converge toward *N* as relative sample size (*n*/*N*) increased, but rate of convergence and direction of bias at small to moderate sample sizes varied considerably among estimators and with CV (Fig. 1). Contrary to expectations, Chao's (1984) estimator, $\hat{N}_{\text{Chao1}}$, was postively biased when CV was small. This bias was especially pronounced when *n*/*N* also was small. However, $\hat{N}_{\text{Chao1}}$ was among the least biased estimators when CV was large, regardless of sample size. As predicted by theory (Chao 1989), $\hat{N}_{\text{Chao2}}$ was nearly unbiased when CV = 0, but became increasingly and negatively biased as CV increased. The jackknife estimators ($\hat{N}_{\text{J1}}$, $\hat{N}_{\text{J2}}$, and $\hat{N}_{\text{J}k1}$) were all negatively biased when *n*/*N* < 1.0, but tended to overestimate *N* at sample sizes where $1.0 < n/N \leq 3.0$, particularly when CV was small. The jackknife estimators also did not converge toward *N* as quickly as other estimators as sample size increased. Patterns for the 2 sample coverage estimators were similar: both tended to overestimate *N* when *n*/*N* and CV were small, but converged relatively quickly toward *N* as *n*/*N* exceeded 1.0, particularly when $0.25 \leq \text{CV} \leq 0.75$.

With some methods, it was not always possible to estimate *N*. Over the full range of conditions modeled, $\hat{N}_{\text{Chao1}}$, $\hat{N}_{\text{J}k1}$, $\hat{N}_{\text{SC1}}$, and $\hat{N}_{\text{SC2}}$ failed to yield estimates in 0.2% of the cases (range = 0.0–29.0% for $\hat{N}_{\text{Chao1}}$; range = 0.0–6.6% for $\hat{N}_{\text{J}k1}$, $\hat{N}_{\text{SC1}}$, and $\hat{N}_{\text{SC2}}$). Reasons for failures varied. For $\hat{N}_{\text{Chao1}}$, no estimate is possible when $f_2 = 0$ because this leads to division by zero (Eq. 1). For $\hat{N}_{\text{J}k1}$, the selection process was aborted if a best jackknife estimate was not selected from the estimates $\hat{N}_{\text{J1}} - \hat{N}_{\text{J5}}$. Using $\hat{N}_{\text{J}k1}$, Burnham and Overton (1979) similarly failed to identify a best estimate in 3.7% of their trials. For $\hat{N}_{\text{SC1}}$ and $\hat{N}_{\text{SC2}}$, no population estimate is possible if the estimated sample coverage is zero, as this also leads to division by zero (Eq. 4). This occurs

when individuals in the sample are seen only once each, so that $f_1 = n$ and $f_2 = 0$ (Eqs. 2 and 3). For all of these methods, failure rates declined as sample size and, hence, information content increased.

For $\hat{N}_{\text{Chao1}}$, $\hat{N}_{\text{Chao2}}$, $\hat{N}_{\text{SC1}}$, and $\hat{N}_{\text{SC2}}$, RMSE declined monotonically toward zero as *n*/*N* increased (Fig. 2). Patterns of decline were indistinguishable for $\hat{N}_{\text{SC1}}$ and $\hat{N}_{\text{SC2}}$, and RMSE converged more quickly toward zero for these estimators than for $\hat{N}_{\text{Chao1}}$ or $\hat{N}_{\text{Chao2}}$. Also for these 4 estimators, RMSE increased with CV when $n/N \geq 1$. When *n*/*N* was small, $\hat{N}_{\text{J1}}$, $\hat{N}_{\text{J2}}$, and $\hat{N}_{\text{J}k1}$ exhibited the lowest RMSEs of the estimators we evaluated. However, rate of convergence toward zero as sample size increased was slow compared to other methods; indeed, RMSE for the jackknife estimators often increased with sample size when $0.5 \leq n/N \leq 2.0$. Also, relatively low RMSEs, especially for $\hat{N}_{\text{J1}}$, often were due to low standard deviations overcompensating for high bias. This suggested that $\hat{N}_{\text{J1}}$ may yield narrow confidence bounds, but that those bounds will be centered around highly biased estimates, likely resulting in poor coverage.

Of the methods we compared, our overall choice was the second-order sample coverage estimator, $\hat{N}_{\text{SC2}}$ (see Discussion). Comparing observed versus nominal lower, 1-tailed confidence bounds for $\hat{N}_{\text{SC2}}$ showed that coverage was affected by *n*/*N* and CV, and by the method used to calculate confidence bounds (Figs. 3 and 4). Disparities between observed and nominal coverages generally increased with CV, but declined as the nominal confidence level increased. Results varied most noticeably with *n*/*N* when CV $\geq$ 0.75. Using the percentile bootstrap method, nominal values sometimes overstated the true coverage when CV = 0.0, but tended to either closely approximate or understate true coverage when $0.25 \leq \text{CV} \leq 1.0$ (Fig. 3). Using the

BCA bootstrap method, nominal values more closely approximated observed coverages when CV = 0.0, and tended to either approximate or understate true coverage when 0.25 ≤ CV ≤ 0.75. For CV = 1.0, however, nominal values tended to overstate true coverage by a large margin when $n/N \geq 2.0$. Overall, we chose the percentile bootstrap method for calculating confidence bounds because, with CV = 0.0 unlikely in natural populations, we believe that it better minimizes the risk of overestimating *N*.

Estimates of n/N and CV.—In our Monte Carlo study, *n*/*N* and CV were important determinants of performance for our estimator of choice, $\hat{N}_{SC2}$. Estimates of these values are given by $n/\hat{N}_{SC2}$ and $\hat{\gamma}$ (Eq. 5), respectively. Presumably, such estimates might be used to ask whether actual values of *n*/*N* and CV in our field studies were within the range of values in which $\hat{N}_{SC2}$ performed well. First, however, it is prudent to ask whether $n/\hat{N}_{SC2}$ and $\hat{\gamma}$ themselves provide good estimates. Comparisons showed that $n/\hat{N}_{SC2}$ provided nearly unbiased estimates of *n*/*N* throughout the range of conditions we modeled (Fig. 5a). However, $\hat{\gamma}$ was a biased estimator of CV, overestimating the true value when CV = 0.0 and underestimating in all other cases (Fig. 5b). The degree to which $\hat{\gamma}$ underestimated CV when $\mathrm{CV} \geq 0.25$ was influenced by relative sample size. When *n*/*N* = 3.0, $\hat{\gamma}$ tended to underestimate CV by about 0.07–0.14. When *n*/*N* = 0.5, $\hat{\gamma}$ tended to underestimate CV by about 0.10–0.59.

Field Data

Observation frequencies for F_{CUB} in Yellowstone's grizzly bear recovery area and the surrounding 10-mile buffer zone were tabulated for 1986–2001 (Table 4). Sample sizes ranged from 20 observations in 1987 to 94 in 1999. Using $\hat{N}_{SC2}$ and rounding to the nearest integer, estimated numbers of F_{CUB} in the Yellowstone population ranged from 20 animals in 1987 and

1989 to 60 in 2000 (Table 5). Estimated relative sample size ($n/\hat{N}_{SC2}$) averaged 1.5 and ranged from 0.5 in 1995 to 2.6 in 1986 and 1999, with $n/\hat{N}_{SC2} \geq 1.0$ for 14 of the 16 years examined (Table 5). The estimated coefficient of variation among individual sighting probabilities ($\hat{\gamma}$) averaged 0.46 and ranged from 0.0 in 1990, 1993 and 1994 to 0.90 in 2000 (Table 5).

The total number of unique F_{CUB} actually observed ($\hat{N}_{Obs}$) ranged from 13 in 1987 to 42 in 2001 (Table 5). This included animals that would not have been detected without radiotelemetry. The number of unique F_{CUB} detected through random sightings alone (*m*) ranged from 12 in 1987 to 39 in 2001 (Table 5). On average, additional information provided by radiotelemetry increased the number of unique F_{CUB} observed by 2.1 animals/year (range = 0–5 animals). For every year, $\hat{N}_{SC2}$ exceeded $\hat{N}_{Obs}$ (Table 5). However, when rounded to the nearest integer, the lower, 1-tailed 95 and 90% confidence bounds for $\hat{N}_{SC2}$ were less than $\hat{N}_{Obs}$ for 10 and 5 of the years, respectively (Table 5). Lower, 1-tailed 70 and 80% confidence bounds were $\geq \hat{N}_{Obs}$ for all years except 1990 (Table 5).

Discussion

Whether Yellowstone's grizzly bears are removed from the threatened species list depends, in part, on whether human-caused mortalities are within calculated limits. Because mortality limits are computed as a function of the number of F_{CUB} present in the population, statistically sound estimates of annual numbers of F_{CUB} (*N*) are needed. Parametric methods proposed by Eberhardt and Knight (1996) and Boyce et al. (2001; unpublished report,1999) improved on the practice of basing mortality limits on a minimum estimate for *N*, determined as the number of unique F_{CUB} observed in a given year $(\hat{N}_{Obs})$. However, these methods require untenable

assumptions about the form and constancy of distributions of individual sighting probabilities. At best, these assumptions leave unnecessary room for dispute, potentially undermining the credibility of results and diverting attention from other important issues. At worst, they can cause serious biases.

Nonparametric approaches are free of assumptions about distributions of sighting probabilities, but have not previously been applied to this problem. Nor should they be applied uncritically, as both absolute and relative performances of different estimators can vary with sampling conditions. In this study, we sought a nonparametric method that performs well over the range of sampling conditions deemed plausible for sightings of F_{CUB} in the GYE. Comparing 7 variations of the Chao (Chao 1984, 1989), jackknife (Burnham and Overton 1978, 1979), and sample coverage (Chao and Lee 1992, Lee and Chao 1994) methods, our provisional choice for estimating numbers of F_{CUB} in the Yellowstone population was the second-order sample coverage estimator, $\hat{N}_{SC2}$. Differences between $\hat{N}_{SC2}$ and the first-order sample coverage estimator, $\hat{N}_{SC1}$, were minor, with both methods converging more rapidly toward *N* as sample size increased than did other estimators. For both estimators, however, the coefficient of variation among individual sighting probabilities (CV) affected performance. Over all CV values, $\hat{N}_{SC2}$ exhibited a slightly better balance than $\hat{N}_{SC1}$ between tendencies to overestimate and underestimate when relative sample size (*n*/*N*) was in the range of $1.0 < n/N \leq 2.0$ (Fig. 1). Performance under these conditions was seen as particularly important because estimates of *n*/*N* for our field study were within this range most years (Table 5).

Chao's (1984) estimator ($\hat{N}_{Chao1}$) showed a greater tendency toward positive bias and exhibited somewhat larger RMSEs than $\hat{N}_{SC2}$ (Figs. 1, 2), but otherwise performed well. Because the most

serious biases were associated with model populations where CV = 0 (an unlikely situation in nature), $\hat{N}_{\text{Chao1}}$ may be a suitable alternative to the sample coverage estimators. However, we cannot recommend the other methods we compared. Over all CV values, RMSEs for $\hat{N}_{\text{Chao2}}$ were lower than for $\hat{N}_{\text{SC2}}$ (Fig. 2), but $\hat{N}_{\text{Chao2}}$ became increasingly and negatively biased as CV increased (Fig. 1). Because individual animals clearly are not equally sightable, use of such an estimator would introduce a chronic, negative bias into estimates of population size and sustainable mortality. Jackknife estimates oscillated, being negatively biased when *n*/*N* was small, positively biased at moderate values of *n*/*N*, and converging toward *N* only as *n*/*N* increased beyond values observed in our field study (Fig. 1). Neither bias nor RMSE declined monotonically with sample size for any of the jackknife estimators. This suggested that, relative to the other methods examined, larger sample sizes would be needed to achieve comparably accurate estimates and that increased sample size might actually lead to increased bias in some situations. The latter problem was particularly pronounced in the range of $1.0 < n/N \leq 2.0$ (Figs. 1, 2).

In a similar analysis, Mowat and Strobeck (2000) evaluated nonparametric estimators available in the program CAPTURE (Otis et al. 1978, White et al. 1982, Rexstad and Burnham 1991). They selected Burnham and Overton's (1979) best-order jackknife method ($\hat{N}_{Jk1}$) for estimating numbers of grizzly bears in 2 Canadian populations that showed evidence of "relatively weak heterogeneity" among individual capture probabilities (Mowat and Strobeck 2000:191). Our study differed in important respects. First, all else being equal, the underlying distribution of sighting probabilities should be more heterogeneous in our study (i.e., CV should be larger) because our sample unit consisted of a single sighting rather than a sample period. Second, because our sampling universe included only F_{CUB}, population size appeared to be smaller than

the 74 and 262 animals estimated by Mowat and Strobeck (2000). Although population size was not a major determinant of estimator performance in our study, we considered only a narrow range of values (N = 20, 40, and 60 animals). Over a larger range, N might emerge as a more important factor. Third, we considered sample coverage estimators (Chao and Lee 1992, Lee and Chao 1994) not available in CAPTURE. Fourth, Mowat and Strobeck (2000), apparently, did not vary sampling effort in a way that would have revealed the oscillatory pattern we observed for the jackknife estimators.

Like all estimators we examined, performance of $\hat{N}_{SC2}$ varied with n/N. As expected, the largest biases and RMSEs were associated with the smallest relative sample size, n/N = 0.5. Performance improved dramatically, however, with even modest increases in n/N, leading us to recommend a minimum sample size of n/N = 1. A nearly unbiased estimate of n/N was $n/\hat{N}_{SC2}$ (Fig. 5a). Observed values for $n/\hat{N}_{SC2}$ met or exceeded our recommended minimum for all but 2 years during 1986–2001 (Table 5). This suggested that observed sample sizes were large enough in most years to support fairly good estimates of N (Fig. 1). At this minimal level of sampling effort, however, confidence bounds were sometimes undesirably broad (Table 5). To narrow confidence bounds, we suggest that n/N = 2 is a reasonable and achievable goal. Based on estimates of N for 1996–2001 (Table 5), such a goal would translate into target sample sizes of about 80–120 independent random sightings of F_{CUB} per year. This compares with observed sample sizes of 45–94 sightings/year during that same period and indicates a need for increased support for this aspect of the Yellowstone grizzly bear monitoring effort.

Performance of $\hat{N}_{SC2}$ also varied with the degree of heterogeneity among individual sighting probabilities, as measured by CV. However, such variation was dramatic only when n/N = 0.5.

When $n/N \geq 1$, $\hat{N}_{SC2}$ was fairly robust to variations in CV, especially in the range of $0.0 \leq \mathrm{CV} \leq 0.75$ (Fig. 1). Even when CV = 1.0, bias was <10%, regardless of *n*/*N* (Fig. 1). An advantage of $\hat{N}_{SC2}$ is that CV is estimated ($\hat{\gamma}$, Eq. 5) as part of the calculation. For 1986–2001, $\hat{\gamma}$ averaged 0.46 and ranged from 0.0–0.9, suggesting that actual CVs were within the range of values in which $\hat{N}_{SC2}$ performs well. Our Monte Carlo study demonstrated, however, that $\hat{\gamma}$ was negatively biased when $\mathrm{CV} \geq 0.25$, particularly when *n*/*N* is small (Fig. 5). Using calculated values for *n*/ $\hat{N}_{SC2}$ and $\hat{\gamma}$ (Table 5), rough corrections for such biases can be inferred from Fig. 5. For example, when *n*/*N* = 1.0 and CV = 0.4, $\hat{\gamma}$ tended to underestimate CV by about 0.2 (Fig. 5). Given *n*/ $\hat{N}_{SC2}$ = 1.5 and $\hat{\gamma}$ = 0.58 for 2001 (Table 5), this suggests an unbiased estimate for CV of about 0.85 for that year. Similar inferences for other years yielded a maximum estimated CV of around 1.3 in 2000, but suggested that, overall, CV rarely was much greater than 1. Thus, we believe that actual CVs for sighting probabilities of F_{CUB} in the Yellowstone population typically are within the range of values in which $\hat{N}_{SC2}$ performs well.

Regardless of method, there is an inherent risk of overestimating *N* that, in turn, could lead to setting mortality limits at unsustainably high levels. To minimize this risk, we believe it is prudent to base management on some lower, 1-tailed confidence bound. This would provide a specified level of assurance that the population of F_{CUB} is at least as large as estimated. For example, calculated confidence bounds indicated that we can be 95% certain there were at least 42 F_{CUB} in the Yellowstone grizzly bear population in 2001, and 80% certain there were at least 47 (Table 5). To determine whether such bounds accurately depict the risk of overestimating *N*, we compared nominal versus observed sample coverages using both the BCA and percentile bootstrap methods (Efron and Tibshirani 1993). The BCA method, theoretically, is superior to

the percentile method (Efron and Tibshirani 1993). Nonetheless, we recommend the percentile method for this application because the BCA method substantially overstated true coverage under conditions that might reasonably occur in field studies; i.e., when CV= 1.0 and $n/N \geq 2.0$ (see Table 5). Such an error would cause us to understate the true risk of overestimating *N*. Although the percentile method overstated true coverage when CV = 0.0 and nomimal coverage was 70 or 80%, we view this as less serious because it is not reasonable to expect that CV = 0.0 for natural populations.

In general, we believe $\hat{N}_{\text{SC2}}$ is superior to $\hat{N}_{\text{Obs}}$ as a basis for calculating mortality limits for Yellowstone's grizzly bears, particularly if lower, 1-tailed confidence bounds are used to minimize the risk of overestimation. In some years, however, depending on the confidence level that is chosen, $\hat{N}_{\text{Obs}}$ may be the better alternative. For example, $\hat{N}_{\text{Obs}}$ equaled or exceeded the lower, 1-tailed 90% confidence bound for $\hat{N}_{\text{SC2}}$ (rounded to the nearest integer) in 8 of the 16 years examined (1986–1990, 1993, 1994, 1998, and 1999; Table 5), yet is unburdened by the same risk of overestimation. Thus, it offers a superior estimate of a lower bound for *N* for those years. This situation occurs largely because $\hat{N}_{\text{Obs}}$ incorporates additional information from non-random sightings of radiocollared animals; information that cannot legitimately be used when calculating $\hat{N}_{\text{SC2}}$ or its confidence bounds.

Overall, we sought a reliable statistical method for estimating numbers of F_{CUB} because such estimates are essential for setting mortality limits for grizzly bears in the GYE. Given recommended sample sizes, we scrutiny. Thus, we have refrained from using estimates generated in this study to project total population size or infer acceptable levels of mortality, believing that the remaining issues should be addressed first. An important issue is the

assumption that every sighting was correctly believe $\hat{N}_{SC2}$ is a reasonable choice for this purpose and that it improves on earlier approaches. We emphasize, however, that knowledge of the number of F_{CUB} is not, by itself, sufficient for setting mortality limits. Other calculations and assumptions are involved that merit additional and comparable identified to individual. Misidentifications undoubtedly occurred, leading to errors of Type I (sightings of the same animal mistakenly classified as sightings of different animals) or Type II (sightings of different animals mistakenly classified as sightings of the same animal). Our experience in applying the rule set of Knight et al. (1995) suggests that Type II errors are much more likely. Such a bias would cause a tendency to undercount the number of unique animals actually seen (*m*), while also inflating sighting frequencies (n_i values) for the $\hat{m}$ animals estimated to have been seen. In turn, this would lead to estimates of *N* that are more negatively biased than depicted in our Monte Carlo results, regardless of the estimator that is used. Such a bias, although undesirable, is not by itself inconsistent with our goal of improving on $\hat{N}_{Obs}$ while minimizing the risk of overestimating *N*. Effects of misidentification on precision are less clear, however. Misidentification introduces uncertainty in sighting frequencies and, thus, would increase uncertainty in estimates based on those frequencies. Our lower, 1-tailed confidence bounds did not incorporate this additional uncertainty and, thus, were probably higher than they would have been if effects of misidentification had been fully accounted for. The tendency toward positive bias in the lower confidence bound would have been countered to some degree by 2 factors. First, any negative bias in $\hat{N}$ resulting from misidentification would necessarily have been accompanied by a similar bias in the confidence bounds surrounding $\hat{N}$. Second, our lower, 1-tailed confidence bounds already were biased low within the range of conditions most often experienced in this study (Fig. 3). Overall, effects of misidentifications on precision would be

mitigated, but to an unknown degree. Additional work to better define the nature, magnitude, and consequences of identification errors is needed and has been undertaken. In the meantime, we offer this work as the first in what we hope will be a series of refinements that better ensure reliable estimates of allowable mortality, while minimizing the risk of error.

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Table 1. Example calculations for the 7 non-parametric estimators compared in this study, using 1997 grizzly bear sighting data from the Greater Yellowstone Ecosystem (see Tables 4 and 5). For 1997, n = 65 sightings of females with cubs-of-the-year (F_{CUB}) were made via means other than radiotelemetry. Distinguishing individuals as per Knight et al. (1995), m = 29 unique animals were seen; 13 were seen once (f_1 = 13), 7 were seen twice (f_2 = 7), 4 were seen 3 times (f_3 = 4), 1 was seen 4 times (f_4 = 1), 3 were seen 5 times (f_5 = 3), and 1 was seen 7 times (f_7 = 1). Two additional and otherwise unobserved F_{CUB} were seen only as a result of using radiotelemetry. Because all calculations were carried out in double precision, rounding errors are evident in some of the examples.

Estimator	Example calculation
Unique F_{CUB} observed via random sightings	$m = 29$
Unique F_{CUB} observed via random sightings and radiotelemetry	$\hat{N}_{\text{Obs}} = 29 + 2 = 31$
Chao	$\hat{N}_{\text{Chao1}} = m + \frac{f_1^2}{2f_2} = 29 + \frac{13^2}{2(7)} \approx 41.1$
Bias-corrected Chao	$\hat{N}_{\text{Chao2}} = m + \frac{f_1^2 - f_1}{2(f_2+1)} = 29 + \frac{13^2 - 13}{2(7+1)} \approx 38.8$
First-order jackknife	$\hat{N}_{\text{J1}} = m + \left(\frac{n-1}{n}\right) f_1 = 29 + \left(\frac{65-1}{65}\right) 13 = 41.8$

Table 1 (continued).

Estimator	Example Calculation
Second-order jackknife	$\hat{N}_{\mathrm{J2}} = m + \left(\frac{2n-3}{n}\right) f_1 - \left(\frac{(n-2)^2}{n(n-1)}\right) f_2 = 29 + \left(\frac{2(65)-3}{65}\right) 13 - \left(\frac{(65-2)^2}{65(65-1)}\right) 7 \approx 47.7$
Best-order jackknife	$\hat{N}_{\mathrm{J}k1} = \hat{N}_{\mathrm{J1}} = 41.8$ was selected because $\mathrm{T}_1 = \frac{\hat{N}_{\mathrm{J2}} - \hat{N}_{\mathrm{J1}}}{[\hat{\mathrm{var}}(\hat{N}_{\mathrm{J2}} - \hat{N}_{\mathrm{J1}} \mid m)]^{1/2}} \approx \frac{47.7-41.8}{[17.996]^{1/2}} \approx 1.396 < 1.960,$ where $\hat{\mathrm{var}}(\hat{N}_{\mathrm{J2}} - \hat{N}_{\mathrm{J1}} \mid m) = \frac{m}{m-1}\left[\sum_{j=1}^{2} (b_j)^2 f_j - \frac{(\hat{N}_{\mathrm{J2}} - \hat{N}_{\mathrm{J1}})^2}{m}\right]$ $\approx \frac{29}{29-1}\left[\left(\frac{2(65)-3}{65} - \frac{65-1}{65}\right)^2 13 + \left(\frac{(65-2)^2}{65(65-1)}\right)^2 7 - \frac{(47.7-41.8)^2}{29}\right] \approx 17.996$
First-order sample coverage	$\hat{N}_{\mathrm{SC1}} = \frac{m + f_1\hat{\gamma}^2}{\hat{C}_1} = \frac{29 + 13(0.325)}{0.800} \approx 41.5,$ where $\hat{C}_1 = 1 - \frac{f_1}{n} = 1 - \frac{13}{65} = 0.800$ and $\hat{\gamma}^2 = \max\left\{\frac{m}{\hat{C}_1}\sum_{j=1}^{n}\frac{j(j-1)f_j}{n(n-1)} - 1,\ 0\right\} = \max\left\{\frac{29}{0.800}\left(\frac{2(7)+6(4)+12(1)+20(3)+42(1)}{65(65-1)}\right) - 1,\ 0\right\} \approx 0.325$

Second-order sample coverage

$$\hat{N}_{\mathrm{SC2}} = \frac{m + f_1\hat{\gamma}^2}{\hat{C}_2} = \frac{29 + 13(0.319)}{0.803} \approx 41.3,$$

$$\text{where } \hat{C}_2 = 1 - \frac{f_1 - 2f_2/(n-1)}{n} = 1 - \frac{13 - 2(7)/(65-1)}{65} \approx 0.803$$

$$\text{and } \hat{\gamma}^2 = \max\left\{\frac{m}{\hat{C}_2}\sum_{j=1}^{n}\frac{j(j-1)f_j}{n(n-1)} - 1,\ 0\right\} = \max\left\{\frac{29}{0.803}\left(\frac{2(7)+6(4)+12(1)+20(3)+42(1)}{65(65-1)}\right) - 1,\ 0\right\} \approx 0.319$$

Table 2. Jackknife estimators of population size, $\hat{N}_{Jk}$, for order k = 1–5, where m is the number of unique individuals observed after n samples, and f_i is the number of individuals observed exactly i times (after Burnham and Overton 1979).

$$\hat{N}_{J1} = m + \left(\frac{n-1}{n}\right) f_1$$

$$\hat{N}_{J2} = m + \left(\frac{2n-3}{n}\right) f_1 - \left(\frac{(n-2)^2}{n(n-1)}\right) f_2$$

$$\hat{N}_{J3} = m + \left(\frac{3n-6}{n}\right) f_1 - \left(\frac{3n^2-15n+19}{n(n-1)}\right) f_2 + \left(\frac{(n-3)^3}{n(n-1)(n-2)}\right) f_3$$

$$\hat{N}_{J4} = m + \left(\frac{4n-10}{n}\right) f_1 - \left(\frac{6n^2-36n+55}{n(n-1)}\right) f_2 + \left(\frac{4n^3-42n^2+148n-175}{n(n-1)(n-2)}\right) f_3 - \left(\frac{(n-4)^4}{n(n-1)(n-2)(n-3)}\right) f_4$$

$$\hat{N}_{J5} = m + \left(\frac{5n-15}{n}\right) f_1 - \left(\frac{10n^2-70n+125}{n(n-1)}\right) f_2 + \left(\frac{10n^3-120n^2+485n-660}{n(n-1)(n-2)}\right) f_3 - \left(\frac{(n-4)^5-(n-5)^5}{n(n-1)(n-2)(n-3)}\right) f_4 + \left(\frac{(n-5)^5}{n(n-1)(n-2)(n-3)(n-4)}\right) f_5$$

Table 3. Values of the parameters (*a*, *b*) of the standard beta distributions used to model p = $(p_1, p_2, ..., p_N)$, where p_i is the probability that a particular sighting will be of the *i*th animal. Values are listed by size (*N*) of the model population and the coefficient of variation (CV) among the p_i values.

N	**(*a*, *b*)**				
	CV = 0.00	**CV = 0.25**	**CV = 0.50**	**CV = 0.75**	**CV = 1.00**
20	**(1.000, 1.000)**	**(0.955, 1.270)**	**(0.791, 1.380)**	**(0.664, 1.446)**	**(0.589, 1.600)**
40	**(1.000, 1.000)**	**(1.084, 1.398)**	**(0.797, 1.382)**	**(0.686, 1.477)**	**(0.593, 1.512)**
60	**(1.000, 1.000)**	**(1.173, 1.449)**	**(0.794, 1.369)**	**(0.688, 1.462)**	**(0.611, 1.559)**

Table 4. Observation frequency (f_j) by year, where f_j is the number of unique females with cubs-of-the-year (F_{CUB}) that were seen exactly j times during that year. Total number of observations is given by $n = \sum_{j=1}^{\infty} jf_j$. Only observations made without the benefit of radiotelemetry and within or <10 miles of the designated grizzly bear recovery zone were included.

Year	n	Observation frequency														
		f_1	f_2	f_3	f_4	f_5	f_6	f_7	f_8	f_9	f_{10}	f_{11}	f_{12}	f_{13}	f_{14}	f_{15}
1986	82	7	5	6	1	1	0	1	2	0	0	0	0	0	0	1
1987	20	7	3	1	1	0	0	0	0	0	0	0	0	0	0	0
1988	36	7	4	4	1	1	0	0	0	0	0	0	0	0	0	0
1989	27	6	5	0	1	0	0	1	0	0	0	0	0	0	0	0
1990	49	7	6	7	1	1	0	0	0	0	0	0	0	0	0	0
1991	62	11	3	3	3	1	2	1	0	0	0	0	0	0	0	0
1992	37	15	5	1	1	1	0	0	0	0	0	0	0	0	0	0
1993	29	7	8	2	0	0	0	0	0	0	0	0	0	0	0	0
1994	29	9	7	2	0	0	0	0	0	0	0	0	0	0	0	0
1995	25	13	2	1	0	1	0	0	0	0	0	0	0	0	0	0
1996	45	15	10	2	1	0	0	0	0	0	0	0	0	0	0	0
1997	65	13	7	4	1	3	0	1	0	0	0	0	0	0	0	0
1998	75	11	13	5	1	1	0	2	0	0	0	0	0	0	0	0
1999	94	9	4	6	2	4	2	0	1	0	0	1	0	0	0	0
2000	72	17	8	1	2	1	0	2	0	1	0	0	0	0	0	0
2001	84	16	12	8	0	1	1	0	0	1	0	0	0	0	0	0

Table 5. Estimates of annual numbers ($\hat{N}_{\text{Obs}}$)of females with cubs-of-the-year (F_{CUB}) in the Yellowstone grizzly bear population, 1986–2001. $\hat{N}_{\text{Obs}}$ gives the number of unique F_{CUB} actually observed, including those located using radiotelemetry; *m* gives the number of unique F_{CUB} observed using random sightings only; and $\hat{N}_{\text{SC2}}$ gives the second-order sample coverage estimates, per Lee and Chao (1994; Eqs. 3–5). Lower, 1-tailed confidence bounds are for $\hat{N}_{\text{SC2}}$ and were calculated using Efron and Tibshirani's (1993) percentile bootstrap method. Also included are annual estimates of relative sample size ($n/\hat{N}_{\text{SC2}}$, where *n* is the total number of observations of F_{CUB}) and of the coefficient of variation among sighting probabilities for individual animals ($\hat{\gamma}$, Eq. 5).

Year	$\hat{N}_{\text{Obs}}$	*m*	$\hat{N}_{\text{SC2}}$	Lower 1-tailed confidence bounds				$n/\hat{N}_{\text{SC2}}$	$\hat{\gamma}$
				70%	80%	90%	95%		
1986	25	24	31.9	28.4	27.0	25.1	23.5	2.6	0.86
1987	13	12	19.5	16.8	15.2	13.3	11.7	1.0	0.37
1988	19	17	21.5	20.1	19.1	17.7	16.7	1.7	0.25
1989	15	13	20.2	16.9	15.3	13.7	12.3	1.3	0.71
1990	25	22	25.5	24.4	23.5	22.2	21.3	1.9	0.00
1991	24	24	34.5	31.1	29.3	27.0	25.2	1.8	0.63
1992	25	23	47.6	40.0	36.4	32.1	28.9	0.8	0.61
1993	19	17	21.8	20.1	19.0	17.9	16.3	1.3	0.00
1994	20	18	25.5	23.4	21.8	19.9	18.8	1.1	0.00
1995	17	17	54.9	41.2	35.9	28.8	24.7	0.5	0.86
1996	33	28	41.4	38.7	36.6	34.0	31.8	1.1	0.00
1997	31	29	41.3	37.5	35.5	33.0	31.1	1.6	0.57
1998	35	33	40.9	38.4	37.1	35.1	33.7	1.8	0.44
1999	32	29	35.7	33.3	32.1	30.4	29.0	2.6	0.61
2000	35	32	59.7	51.8	48.2	43.8	40.3	1.2	0.90
2001	42	39	54.6	49.5	47.3	44.6	42.2	1.5	0.58

Fig. 1. Percent bias of population estimates calculated using the Chao ($\hat{N}_{\text{Chao1}}$), bias-corrected Chao ($\hat{N}_{\text{Chao2}}$), first-order jackknife ($\hat{N}_{\text{J1}}$), second-order jackknife ($\hat{N}_{\text{J2}}$), best-order jackknife ($\hat{N}_{\text{J}k1}$), first-order sample coverage ($\hat{N}_{\text{SC1}}$), and second-order sample coverage ($\hat{N}_{\text{SC2}}$) estimators. Number of unique individuals observed (*m*) is shown for comparison. Each point represents the mean of 1,000 Monte Carlo replicates; in each, calculations were based on *n* random sightings drawn from a model population with *N* = 40 individuals. CV gives the coefficient of variation among sighting probabilities for the 40 individuals. CV = 0.0 indicates equal sightability.

Fig. 2. Root mean square error (RMSE) of population estimates calculated using the Chao ($\hat{N}_{\text{Chao1}}$), bias-corrected Chao ($\hat{N}_{\text{Chao2}}$), first-order jackknife ($\hat{N}_{\text{J1}}$), second-order jackknife ($\hat{N}_{\text{J2}}$), best-order jackknife ($\hat{N}_{\text{J}k1}$), first-order sample coverage ($\hat{N}_{\text{SC1}}$), and second-order sample coverage ($\hat{N}_{\text{SC2}}$) estimators. Number of unique individuals observed (*m*) is shown for comparison. Each data point represents the mean of 1,000 Monte Carlo replicates; in each, calculations were based on *n* random sightings drawn from a model population with *N* = 40 individuals. CV gives the coefficient of variation among sighting probabilities for the 40 individuals. CV = 0.0 indicates equal sightability.

Fig. 3. Observed versus nominal coverages of lower, 1-tailed confidence bounds for second-order sample coverage estimates ($\hat{N}_{SC2}$), calculated using the percentile bootstrap method (Efron and Tibshirani 1993). Points above the dashed line indicate that mean observed coverage was greater than nominal coverage, so confidence bounds tended to be conservative. Each data point represents the mean of 1,000 Monte Carlo replicates; in each, calculations were based on *n* random sightings drawn from a model population with *N* = 40 individuals. CV gives the coefficient of variation among sighting probabilities for the 40 individuals. CV = 0.0 indicates equal sightability.

Fig. 4. Observed versus nominal coverages of lower, 1-tailed confidence bounds for second-order sample coverage estimates ($\hat{N}_{SC2}$), calculated using the bias corrected and accelerated bootstrap method (Efron and Tibshirani 1993). Points above the dashed line indicate that mean observed coverage was greater than nominal coverage, so that confidence bounds tended to be conservative. Each data point represents the mean of 1,000 Monte Carlo replicates; in each, calculations were based on *n* random sightings drawn from a model population with *N* = 40 individuals. CV gives the coefficient of variation among sighting probabilities for the 40 individuals. CV = 0.0 indicates equal sightability.

Fig. 5. Estimated ($n/\hat{N}_{SC2}$) versus observed (n/N) relative sample sizes (A), and estimated ($\hat{\gamma}$) versus observed (CV) values for the coefficient of variation among individual sighting probabilities (B). In both (A) and (B), each point represents the mean value, based on 1,000 Monte Carlo replicates; in each, calculations were based on *n* random sightings drawn from a model population with *N* = 40 individuals.

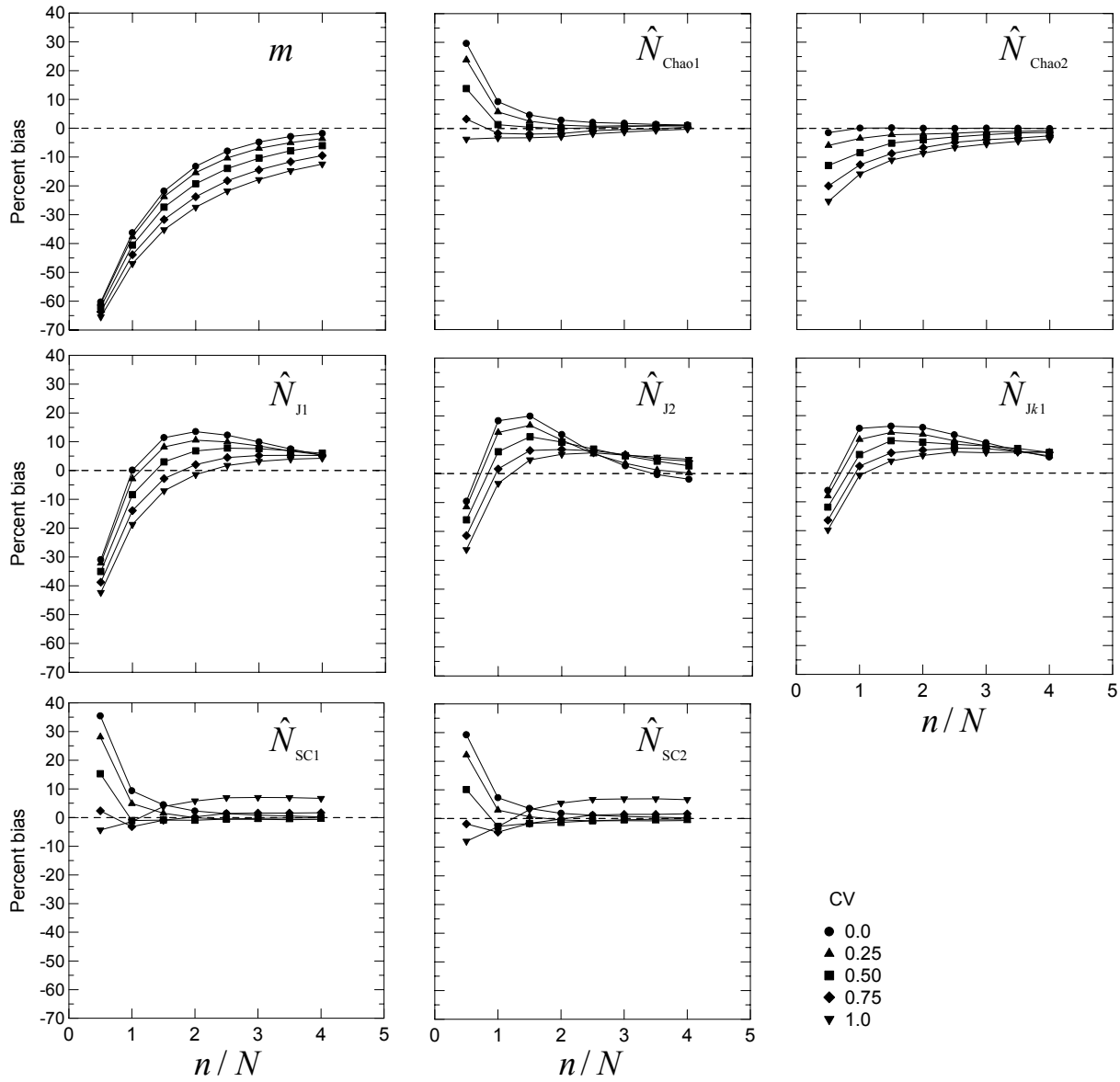
m
$\hat{N}_{Chao1}$
$\hat{N}_{Chao2}$
$\hat{N}_{J1}$
$\hat{N}_{J2}$
$\hat{N}_{Jk1}$
$\hat{N}_{SC1}$
$\hat{N}_{SC2}$
Percent bias
n / N
CV
0.0
0.25
0.50
0.75
1.0

Fig.1

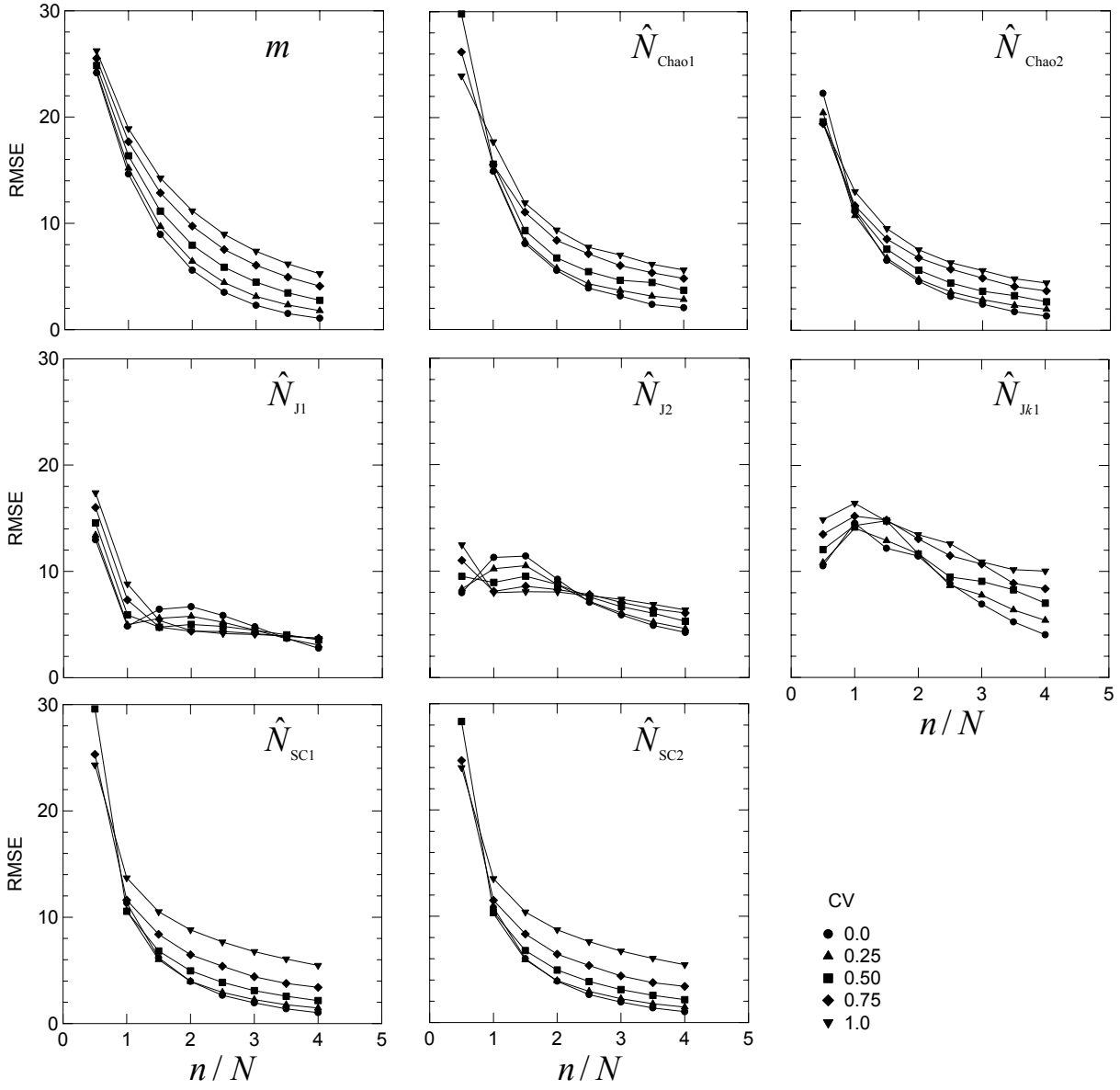
m
$\hat{N}_{Chao1}$
$\hat{N}_{Chao2}$
$\hat{N}_{J1}$
$\hat{N}_{J2}$
$\hat{N}_{Jk1}$
$\hat{N}_{SC1}$
$\hat{N}_{SC2}$
RMSE
n / N
CV
0.0
0.25
0.50
0.75
1.0

Fig. 2

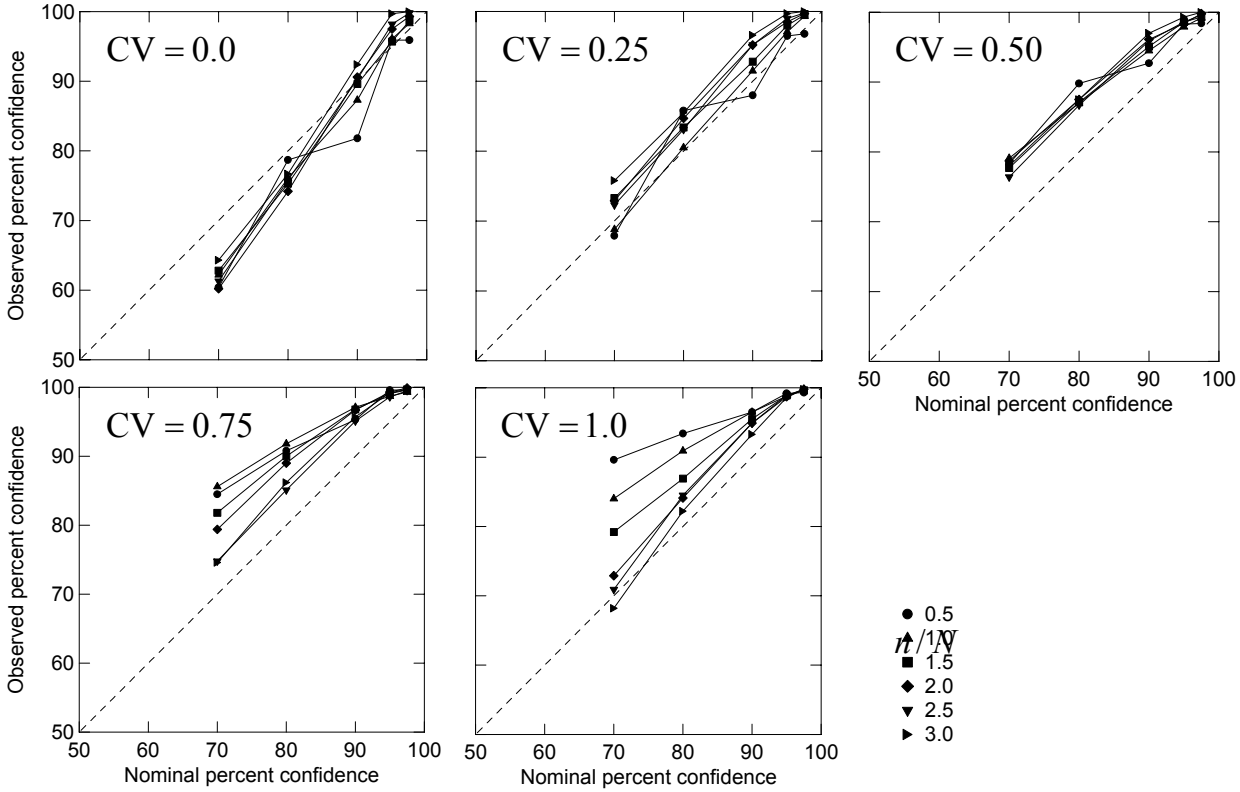
CV = 0.0
CV = 0.25
CV = 0.50
CV = 0.75
CV = 1.0
Observed percent confidence
Nominal percent confidence
50
60
70
80
90
100
0.5
1.0
1.5
2.0
2.5
3.0

Fig. 3

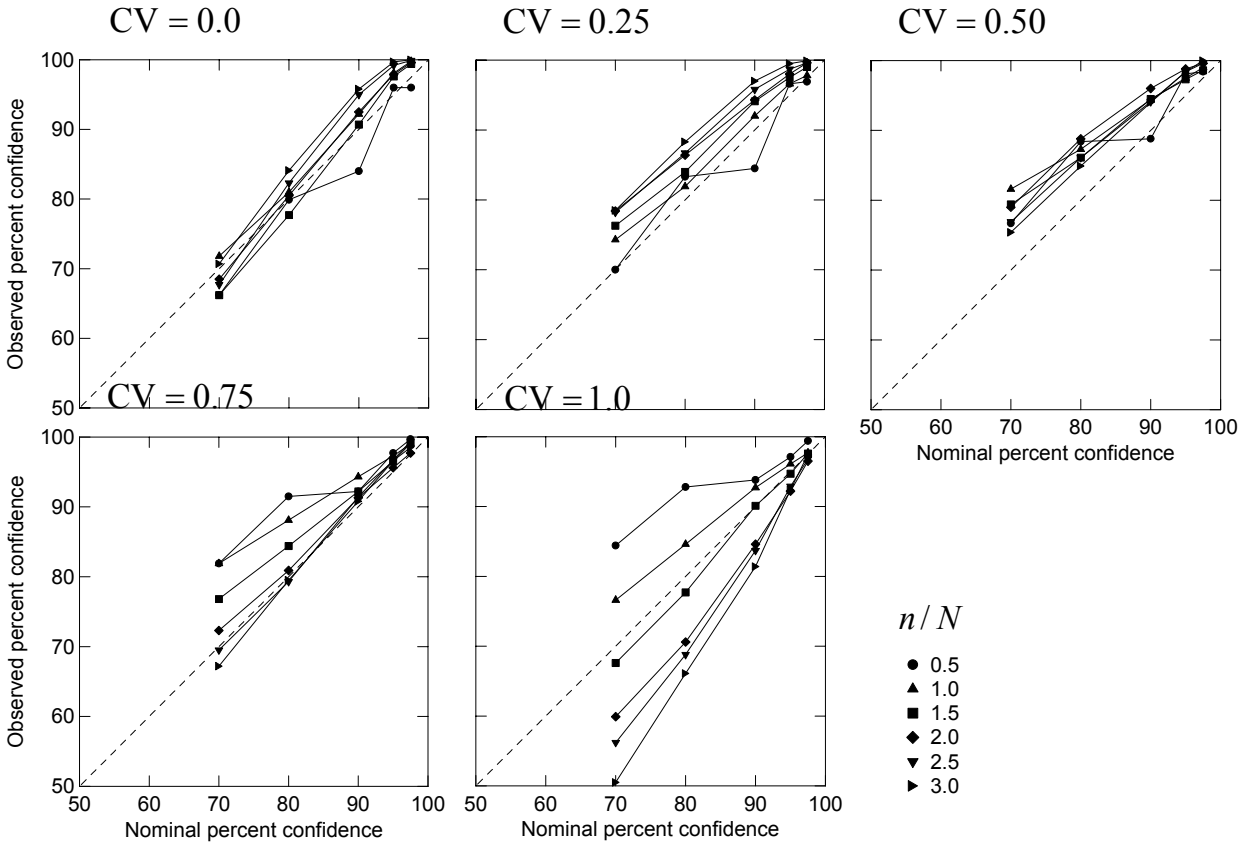
CV = 0.0
CV = 0.25
CV = 0.50
CV = 0.75
CV = 1.0
Observed percent confidence
Nominal percent confidence
n / N
0.5
1.0
1.5
2.0
2.5
3.0

Fig. 4

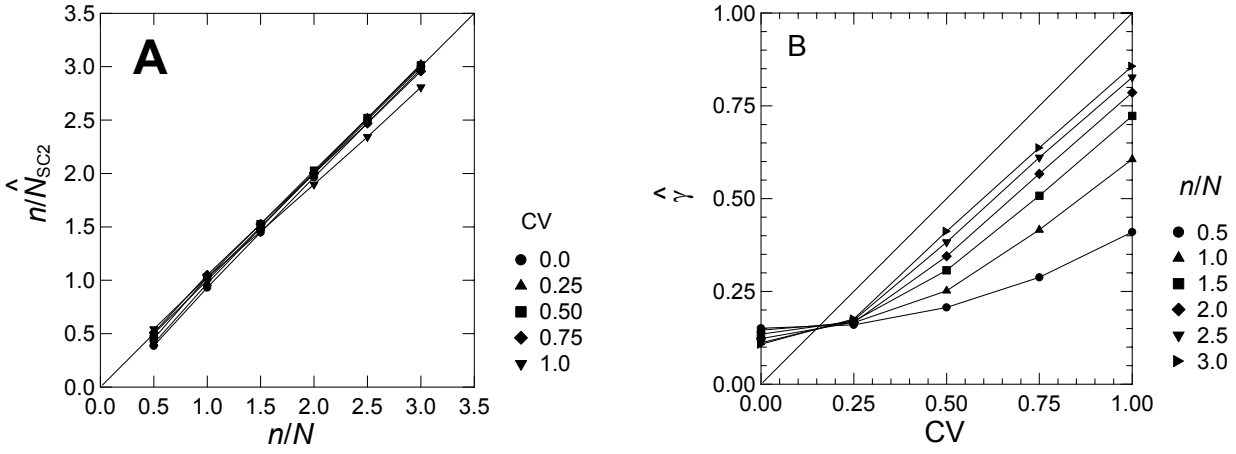
A
$\hat{n}/N_{SC2}$
n/N
CV
0.0
0.25
0.50
0.75
1.0
B
$\hat{\gamma}$
CV
n/N
0.5
1.0
1.5
2.0
2.5
3.0

Fig. 5

Appendix C. Calculation of Total Population Size and Mortality Limits

(These calculations will be updated with new information as it becomes available)

In a recovered population, a limited number of animals can be removed without jeopardizing the viability of the population, and without precluding growth. To assure that human-caused mortality will remain closely regulated, the management agencies will conduct a management review if the running 6 year average of total known and probable human-caused grizzly bear mortalities in the GYA exceeds 4%[1] (rounded to a whole number) of the most recent total population estimate, or if the running 6 year average of the most recent known and probable female human-caused mortalities in the GYA exceeds 30% (rounded to a whole number) of this 4% limit.

The following assumptions and methods will be used to derive the four percent limit on known and probable human-caused mortality:

a) The grizzly bear population in the Yellowstone Area is assumed to be 50 percent adults and 50 percent subadults (Interagency Grizzly Bear Committee, 1987, pp. 47-59).

b) The grizzly population in the Yellowstone Area is assumed to be 45.2 percent male and 54.8 percent female, including both adults and subadults (Eberhardt and Knight 1996).

c) The proportion of adult females in the Yellowstone Area grizzly bear population will be recalculated as new data become available as per the methods in Knight et al. (1993). The most recent data on the percentage of adult females in the population is 27.4% (Eberhardt and Knight 1996).

d) Calculation of the annual mortality limits is based on the total number of females with cubs in each year calculated using sightings and resightings of females with cubs referenced in Appendix B. The total population estimate will be based on the sum of the most recent 3 years of estimates of the total number of females per year, minus the number of known and probable adult (age 5 years and above) female mortalities, and dividing the resulting number by 0.274, which is the most recent percentage of adult females in the population (Eberhardt and Knight 1996). This method will count all unduplicated sightings of females with cubs in the GYA.

   - The mortality limits for the GYA will be derived by multiplying the total population estimate by 4 percent for total mortality limits, and that 4 percent figure by 30 percent to establish female mortality limits. These mortality limits are conservative because:
   - According to Harris (1986), a grizzly bear population can sustain 6.5 percent total human-caused mortality without population decline and; and
   - It is recognized that if known and probable human-caused grizzly mortality reaches 4 percent (rounded to a whole number) of the population estimate, total mortality will exceed this level because some unknown, illegal mortality is likely to occur. The 4 percent mortality limit (rounded to a whole number), and 30 percent female mortality limit are applied to conservative estimators of total population size (see Appendix B).

[1] Note: This number is based on the best available information. If new information based on the best available science becomes available it will be adjusted as necessary.

EXAMPLE: The following data are theoretical for the region of the Yellowstone Area, and are presented to provide a mathematical example for the calculations. It is important to realize that the mortality limits will change each year in response to changes in annual sightings of unduplicated females with cubs.

Year	*1995*	*1996*	*1997*
Annual total estimate for FWCs from resightings	*25*	*35*	*31*
Adult (5+) female mortalities	*3*	*2*	*2*

25 + 35 + 31 = 91 – 7 = 84 ÷ 0.274 = 307 as a total population size in this example

The total known mortality limit would be 307 X 0.04 = 12 in this example

The known female mortality limit would be 13 X 0.30 = 3.6 in this example

Appendix D. Background on Genetic Management of the GYA Grizzly Population

Data indicates that current levels of genetic diversity in the Yellowstone population are not resulting in deleterious effects. An ongoing loss below current levels of genetic diversity could lead to detrimental conditions, therefore maintenance of existing levels of diversity at a minimum are desirable. Table 1 shows the current level of genetic diversity in the NCDE and Yellowstone as compared to some other healthy North American brown bear populations (from Waits, et al. 1998).

Genetic diversity has declined slightly in YE since early in the 20th century (Miller and Waits 2003). The data suggest a gradual decline of diversity over the last century, not an acute drop following dump closure and associated high mortality levels in the late 1960s. Miller and Waits further state that the rate of inbreeding for the Yellowstone grizzly population appears to have been much less severe than originally hypothesized. Comparisons between the NCDE grizzly population and the Yellowstone population show that even as early as 1910 the Yellowstone grizzly population was not as genetically diverse as the population in the NCDE. They further report that a bottleneck test for heterozygosity excess failed to find any evidence for a bottleneck in the 1910s sample (p=0.37; 40). Thus, for whatever reason, it is likely that genetic diversity was historically low in Yellowstone area grizzly bears and that recent lower diversity levels for Yellowstone bears are not a result of recent (over the last 90+ years) management events.

Table 1. Genetic variability within healthy North American brown bear populations based on nuclear DNA microsatellite analysis averaged over 8 loci (from Waits et al, 1998).

Population	Alleles	Diversity	Sample size
Kodiak Island, Alaska	2.1	26.5%	34
Kluane National Park, Canada	7.4	76.2%	24
East Slope, Alberta, Canada	6.4	65.6%	30
NCDE, Montana, USA	6.8	70.3%	35
Yellowstone, USA	4.4	55.5%	46

Diversity is calculated by $h=(1-x_i^2)n/(n-1)$, where x_i is the frequency of the ith lineage (allele) and n is the sample size.

Miller and Waits (2003) calculated the effective population size (N_e) for the Yellowstone population based on recent data on changes in allele frequencies using three maximum likelihood estimators and a moment estimator. The result was a N_e/N ratio of 0.27. This means that approximately 27% of the Yellowstone population is contributing genetic material to future generations by being involved in reproduction. N_e and the N_e/N ratio are important numbers when considering the minimum size of an isolated population for genetic issues.

It has been argued that N_e should remain above 50 to avoid the negative short-term effects of inbreeding and above 500 to maintain long-term evolutionary potential (Franklin 1980). Trend data suggest that the Yellowstone area grizzly population is larger now than it has been in the past three decades (Servheen 1999, Eberhardt and Knight 1996, Boyce et al. 2001, IGBST 2001) implying that N_e is probably $\geq$ 50. Miller and Waits (2003) state: "If recent evidence that *N*

is at least 400 is accurate (IGBST 1998, 1999), then N_e is likely to be near or greater than 100 (0.27x400=108). Hence, it is unlikely that YE is in immediate genetic danger."

It is unlikely, however, that N_e in an isolated Yellowstone population will ever near or exceed 500 (if N_e/N=0.27, N would need to be $\geq$ 1850). Furthermore, there is no guarantee that N could not decline in the future to result in an N_e less than 50 (a total population size of $\leq$185) where inbreeding becomes a concern. Genetic variability can only be maintained over long time spans and inbreeding safely avoided through gene flow. For several reasons, NCDE is a good candidate source population due to its relative proximity and the fact that it currently has a relatively large population. It is the nearest geographical population to the Yellowstone area, NCDE has high levels of diversity and historically bears occupied intervening regions. Interestingly, Miller and Waits report (2003) that pairwise F_{ST} comparisons between the contemporary NCDE sample and all three Yellowstone area samples are all $\approx$ 0.12 (44), suggesting that the two populations were not historically panmictic. If N_e in the NCDE is and remains large, one effective migrant per generation (a generation for grizzly bears is defined as 10 years or the average time it takes an adult female to replace herself) into YE will maintain the current levels of diversity.

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Appendix E. Existing Bear Foods and Related Monitoring Programs

Winter-killed Ungulate Carcass and Associated Bear Use Survey

During April and May of each year, YNP and IGBST personnel conduct ungulate carcass surveys along 126.5 miles of survey routes on the Northern Winter Range, 82.5 miles of survey routes in the Firehole River drainage, 17 miles of survey routes in the Norris Geyser Basin, and 27 miles of survey routes in the Heart Lake area. Survey routes are hiked, snowshoed, or skied by teams of two people. All ungulate carcasses as well as bears and bear sign (tracks, scat, feeding sign) observed from the survey routes are recorded. Data collected include species, sex, and age class of ungulate carcasses found, estimated date and cause of death, scavenging by bears, species of bear using the carcass, use of carcass by other scavengers, and UTM location.

Cutthroat Trout Spawning Stream And Associated Bear Use Surveys

Beginning 1 May each year, 8 frontcountry streams (Lodge Cr., Hotel Cr., Hatchery Cr., Incinerator Cr., Wells Cr., Bridge Cr., Weasel Cr., and Sand Point Cr.) within or near the Lake Developed area, and 5 frontcountry streams (Sandy Cr., Sewer Cr. Little Thumb Cr., Arnica Cr., and 1167 Cr.) within or near the Grant Village development are checked daily to detect the presence of adult cutthroat trout (Andrascik 1992, Olliff 1992). Once adult trout are found (i.e., onset of spawning), weekly surveys of cutthroat trout on these streams and on an additional 8 backcountry streams (Cub Cr., Clear Cr., Columbine, Flat Mountain Arm Cr., Delusion Lake Outlet, Trail Cr., and 1150 Cr.) are conducted. In each stream on each sample day, two people walk upstream from the stream mouth and record the number of adult trout observed. Sampling continues one day per week until most adult trout return to the lake (i.e., end of spawning). Counts are used to estimate the peak periods, relative magnitude and duration of spawning runs (Reinhart 1990). While making fish counts, observers record bear sign (e.g., bear sightings, fish parts, hair, scats, and tracks) and collect hair from DNA hair collection corrals. Track measurements and DNA from collected hair are used to determine the number, species, and association of family groups of bears.

Cutthroat Trout Population Monitoring Programs

Since the discovery of lake trout in Yellowstone Lake in 1994, park biologists have been developing and refining control techniques for lake trout removal and for assessing potential impacts to native Yellowstone cutthroat trout. The cutthroat trout population is monitored using four methods including fish traps, spawning stream surveys, largemesh gillnetting, and hydroacoustic technology.

Fish Trap Surveys. Information on the numbers of upstream and downstream migrants, and the size and age class of the cutthroat trout spawning migration are collected annually from weirs with fish traps erected each spring at the mouths of Clear Creek, Arnica Creek, and Bridge Creek, three tributaries to Yellowstone Lake (Koel 2001). The fish traps are generally installed during the month of May, the exact date depending on winter snow accumulation, weather conditions and spring snow melt. Fish passage, enumeration, and sampling occur through dip-netting trout that enter the upstream and downstream trap boxes and/or visually counting trout as they swim through wooden chutes attached to the traps. An electronic fish counter is also periodically used. Other data collected include weights, lengths, sex and ages (based on collected scales) of captured fish. Daily instream flows and water temperatures are also collected. Continued operation of the Clear Creek, Arnica Creek, and Bridge Creek fish traps may be used for long term monitoring of the potential impacts of lake trout on the Yellowstone Lake cutthroat trout population.

Spawning Stream Surveys. Beginning 1 May each year, 8 frontcountry streams (Lodge Cr., Hotel Cr., Hatchery Cr., Incinerator Cr., Wells Cr., Bridge Cr., Weasel Cr., and Sand Point Cr.) within or near the Lake Village developed area, and 5 frontcountry streams (Sandy Cr., Sewer Cr. Little Thumb Cr., Arnica Cr., and 1167 Cr.) within or near the Grant Village development are checked daily to detect the presence of adult cutthroat trout (Andrascik 1992, Olliff 1992). Once adult trout are found (i.e., onset of spawning), weekly surveys of cutthroat trout on these streams are conducted. In each stream on each sample day, two people walk upstream from the stream mouth and record the number of adult trout observed. Sampling continues one day per week until most adult trout return to the lake (i.e., end of spawning). Counts are used to estimate the peak periods, relative magnitude and duration of spawning runs (Reinhart 1990). While making fish counts, observers record bear sign (e.g., bear sightings, fish parts, hair, scats, and tracks). Track measurements are used to estimate the number, species, and association of family groups of bears frequenting spawning streams.

Largemesh Gillnetting Surveys. A largemesh gillnetting program is also used to monitor the population structure of cutthroat trout in Yellowstone Lake. At each of 11 sampling sites around Yellowstone Lake, 5 38.1 x 1.8 m monofilament gillnets spaced 100m apart, are set overnight in 2 - 6 m of water (Koel 2001). Length, weight, sex, stage of maturity, and scales for aging are collected for each captured fish. Continuation of this gillnetting operation may be used for long term monitoring of the potential impacts of lake trout on the Yellowstone Lake cutthroat trout population.

Hydroacoustic Surveys. Cutthroat trout density data will be gathered lakewide on Yellowstone Lake using hydroacoustic survey techniques (Koel 2001). One survey requires approximately 4 field days for a 2-person crew. Data analysis would require an additional 4 to 10 days of a trained biologists time for each survey. Approximately three surveys will be conducted annually.

Whitebark Pine Surveys

Nineteen whitebark pine transects are currently visited annually. Each transect contains 10 marked trees. Cones are counted on each marked tree between July 15 and August 15 depending on annual phenology. The objective is to count cones after maturation, but before cones and seeds have been collected by red squirrels (*Tamiasciurus hudsonicus*) and Clark's nutcrackers (*Nucifraga columbiana*). Data is recorded on standard field forms and sent to the IGBST. The IGBST maintains the official ecosystem database. The presence or absence of blister rust and beetle infestations as well as grizzly bear, black bear, red squirrel, and Clark's nutcracker activity are noted for each transect.

Army Cutworm Moths

IGBST Monitoring Program. The IGBST and Wyoming Game and Fish Department currently monitor bear use of moth aggregation sites during radio tracking and annual grizzly bear observation flights. When army cutworm moths are present on the high elevation talus slopes, concentrations of grizzly bears are observed at the moth aggregation sites during these flights. The presence of bears at the aggregation sites is used as an indirect measure of the presence or absence of moths during a given year. This monitoring program does not provide direct information on the relative abundance of moths.

State of Montana Monitoring Program. Army cutworm moth larvae are agricultural pests which eat a wide range of host plants including small grains, alfalfa and sugar beets (Blodgett 1997). Moth outbreaks occur sporadically, when insect population potential is high and environmental factors are favorable to the insects' survival (Blodgett 1997). Because army cutworm moths are an agricultural pest, the State of Montana has a cutworm moth monitoring and forecasting program. The forecasting method employed by county extension agents entails trapping for army cutworm moths in agricultural areas between August and October. Extension agents set two

army cutworm pheromone traps per county (G. Johnson, Montana State University, pers. commun.). Trap sites are located in agricultural areas often where soil has been tilled to seed winter wheat in the fall as moth larvae prefer such soft soils (G. Johnson, MSU, pers. commun.). Extension faculty find the amount of fall moth activity can be indicative of moth egg lay (Blodgett 1997). When trap catches exceed 800 moths during the August through October trapping period, extension agents forecast potentially damaging larvae populations may appear the following spring (G. Johnson, MSU, pers. commun.).

Many factors can affect moth larval development. Abundant precipitation from May through July is harmful for the worms and can reduce local cutworm populations (Blodgett, MSU, pers. commun.). Army cutworm moth outbreaks have been noted in warm and dry years when rainfall from 1 May through 31 July was less than 4 inches (Blodgett 1997). If serious cutworm problems are suspected, agents see crop damage by the first of April. Fewer adult moths are trapped after warm and dry weather patterns with mild winters when there is a lack of early spring snow cover to insulate and protect larvae from freezing (G. Johnson, MSU, pers. commun.). Dry weather in the fall also contributes to the mortality of moth eggs and larvae (G. Johnson, MSU, pers. commun.). Pesticides also affect larval recruitment. Warrior, a synthetic pyrethroid, is an EPA registered army cutworm moth pesticide for use on wheat crops. Currently, pesticide companies are in the process of registering this pesticide for use on barley crops as well (G. Johnson, MSU, pers. commun.).

Since 1992, a statewide army cutworm moth pheromone trapping program has been conducted in Montana. Twenty counties in Montana participated in the program in 1997 (Blodgett 19970). In fall 1998, MSU extension agents plan to coordinate with extension agents at universities in Wyoming, Colorado and Nebraska to expand the moth trapping program to include county trapping efforts in their respective States. In addition to trapping for moths, extension agents plan to gather daily weather and temperature data to improve their forecasting technique (G. Johnson, MSU, pers. commun.). The IGBST, WGF, and YNP are currently evaluating methods for incorporating State army cutworm moth monitoring programs into existing grizzly bear foods monitoring programs.

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Appendix F. Habitat Baseline 1998

The 1998 baseline represents the most approximate estimate of the habitat standards within the Primary Conservation Area as of 1998. That estimate relied on the best data available of what was known to be on the ground at the time. Baseline data establishes a set of information against which future improvements and /or impacts can be assessed. As new information is available, the database will be adjusted and will serve as a tracking system for monitoring improvements and evaluating habitat conditions and the need for mitigation measures in the future. Any new features identified that are not included in the existing 1998 baseline identified here will be reviewed as to their actual status in 1998. If the feature was indeed there in 1998 it will be added to the baseline tables or maps, otherwise it will be dealt with as per the standards identified in this document.

Baseline information is also included here for Habitat Effectiveness (HE) and Habitat Value (HV) outputs from the Yellowstone Grizzly Bear Cumulative Effects Model. The HV information is presented to demonstrate the HV of secure habitats as compared to the entire subunit. HE will be monitored over time and the data presented here will serve as the baseline on which to evaluate changes in HE. No baseline was established for other habitat monitoring items.

Secure Habitat and Motorized Access Route Density

Using Geographic Information System databases created by each administrative unit the percent secure habitat, open motorized access route density > 1 mile per square mile and total motorized access route density > 2 mile/square mile was estimated as of 1998 for each BMU subunit (Table 2). OMARD is evaluated for each of two seasons, as access routes may be restricted in one season and not another. TMARD and secure habitat are single values by definition and do not vary by season. The contribution of private roads and state and county highways was also evaluated for each BMU subunit (Table 3). These values represent a minimum percent for OMARD, TMARD and a maximum percent for secure habitat even if all motorized access features administered by the land management agencies were obliterated or decommissioned on public lands. A standardized program (AML) that runs in the ARC/INFO software environment was used to make the calculations. The buffer command in ARC/INFO is used to buffer all relevant motorized access features by 500 meters. The area outside of this buffer is secure habitat. Motorized access route density is calculated using a moving windows process with 30-meter cells and a 1-mile square window.

The 1997 Targhee Forest Plan standards and guidelines, when fully implemented and adopted, meet the intent of the habitat standard for motorized access for this Conservation Strategy. Secure habitat percentages in tables 1 and 2 for those portions of the subunits on the Targhee National Forest (Henry's Lake 1 and 2, Plateau 1 and 2 and Bechler 1) are calculated using the process identified above. However, the actual standard for secure habitat is identified in the associated management area prescription in the 1997 Targhee Forest Plan. The values presented here are used for consistency with other administrative units and are unlikely to change significantly as the 1997 Targhee Forest Plan also has a standard for Open Motorized Access Route Densities and Total Motorized Access Route Densities..

Developed Sites on Public Lands

Developed sites include all sites on public land developed or improved for human use or resource development such as campgrounds, trailheads, lodges, administrative sites, service stations, summer homes, restaurants, visitor's centers, and permitted resource development sites such as oil and gas exploratory wells, production wells, plans of operation for minerals activities, work camps, etc. Developed sites on public lands are currently inventoried in existing GIS databases and are an input item to the Yellowstone Grizzly Bear Cumulative Effects Model

(CEM). Table 4 displays the number of developed sites for each administrative unit by BMU subunit as of 1998.

Activities based in statutory rights, such oil and gas leases and mining claims under the 1872 General Mining Law are also tracked as part of the developed site monitoring effort. Mining claims and or oil and gas leases do not in and of themselves constitute a site development, but have the potential to be developed sometime in the future. There were no oil and gas leases inside the PCA as of 1998 and 1,354 mining claims in ten of the subunits inside the PCA (Table 4). It is important to note that one mining claim does not necessarily mean a potential for one operating plan. Claims are often staked around known mineral deposits to protect the original claim and operating plans can sometimes encompass hundreds of claims. In addition, there are always a number of claims filed that after more detailed exploration do not prove to have enough mineralization to be economically developed. Claims or claim groups with approved operating plans are included in the developed site baseline (Table 3).

Bear Management Subunit	Administrative Unit				
	Gallatin NF	Custer NF	Caribou-Targhee NF	Shoshone NF	Bridger-Teton NF
Boulder/Slough #1	8	144			
Buffalo/Spead Creek #1					14
Buffalo/Spread Creek #2					6
Hellroaring/Bear #1	653				
Henrys Lake #1			5		
Henrys Lake #2			3		
Lamar #1	429	42			
Shoshone #3				16	
South Absaroka #2				28	
South Absaroka #3				6	
Total	1,090	186	8	50	20

Table 1. Number of Mining claims as of 1998 in bear management subunits in the Yellowstone Primary Conservation Area.

Table 2. The 1998 baseline values for secure habitat, open motorized access route density > 1 mile per square mile (OMARD), and total motorized access route density > 2 miles per square mile (TMARD) for the 40 Bear Management (BMU) Subunits in the Greater Yellowstone Area. Includes USFS, NPS, BLM, state, county, and private motorized access routes.

SUBUNIT NAME	BMU #	OMARD % > 1 (mi / mi^2)		TMARD% > 2 (mi / mi^2)	% Secure Habitat	SIZE[1] (mi^2)
		S1	S2			
Bechler/Teton	18	12.7	12.7	4.7	78.1	534
Boulder/Slough #1	4	2.2	2.2	0.1	96.6	282
Boulder/Slough #2	4	1.0	1.0	0	97.7	232
Buffalo/Spread Creek #1	17	10.1	10.2	4.1	88.3	222 (220)
Buffalo/Spread Creek #2	17	13.3	14.5	10.4	81.1	508
Crandall/Sunlight #1	6	11.9	16.2	4.0	81.1	130
Crandall/Sunlight #2	6	13.6	14.6	8.9	82.3	316
Crandall/Sunlight #3	6	12.8	16.6	8.2	80.4	222
Firehole/Hayden #1	10	6.3	6.3	1.2	88.4	339
Firehole/Hayden #2	10	6.3	6.3	0.9	88.4	177
Gallatin #1	2	1.6	1.6	0.1	96.3	128
Gallatin #2	2	7.8	7.8	3.8	90.2	155
Gallatin #3	2	41.5	42.6	16.9	55.3	218
Hellroaring/Bear #1	3	20.8	21.5	13.5	77.0	185
Hellroaring/Bear #2	3	0.6	0.6	0.2	99.5	229
Henrys Lake #1	12	44.7	44.7	25.9	45.4	201 (191)
Henrys Lake #2	12	46.1	46.1	28.1	45.7	153 (140)
Hilgard #1	1	25.1	25.1	12.5	69.8	201
Hilgard #2	1	16.0	16.0	10.3	71.5	141
Lamar #1	5	7.0	7.0	3.3	89.4	300
Lamar #2	5	0	0	0	100	181
Madison #1	11	24.2	24.5	10.2	71.5	227
Madison #2	11	31.7	31.7	22.3	66.5	157 (149)
Pelican/Clear #1	8	1.3	1.3	0.4	97.8	108
Pelican/Clear #2	8	3.0	3.0	0.2	94.1	257
Plateau #1	13	19.0	19.2	9.8	68.9	286
Plateau #2	13	6.1	6.1	2.4	88.7	(420)
Shoshone #1	7	1.5	1.5	0.9	98.5	122
Shoshone #2	7	1.1	1.1	0.4	98.8	132
Shoshone #3	7	3.4	3.4	1.3	97.0	141
Shoshone #4	7	3.9	4.6	2.0	94.9	189
South Absaroka #1	16	0.4	0.4	0	99.2	163
South Absaroka #2	16	0	0	0	99.9	191
South Absaroka #3	16	2.1	2.1	2.3	96.8	348
Thorofare #1	15	0	0	0	100	274
Thorofare #2	15	0	0	0	100	180
Two Ocean/Lake #1	14	1.8	1.8	0.1	96.3	485 (372)
Two Ocean/Lake #2	14	0	0	0	100	143 (125)
Washburn #1	9	12.4	12.4	2.9	83.0	178
Washburn #2	9	3.6	3.6	0.7	92.0	144
Mean for PCA/Total square miles		**10.4**	**10.7**	**5.3**	**86.2**	**9209 (9035)**

1 Lakes > 1 square mile in size were removed from subunit totals, OMARD, TMARD and secure habitat calculations. Numbers in parenthesis are square miles of subunit without these lakes.

Table 3. The 1998 baseline values for secure habitat, open motorized access route density > 1 mi per square mile (OMARD), and total motorized access route density > 2 miles per square mile (TMARD) for the 40 Bear Management (BMU) Subunits in the Greater Yellowstone Area. Includes only private roads and state and county highways. (These motorized features are not subject to management under this strategy and the values in this table represent a minimum percent for OMARD, TMARD and a maximum percent for secure habitat even if all motorized access features administered by the land management agencies were obliterated or decommissioned on public lands.)

SUBUNIT NAME	BMU #	OMARD % > 1 (mi / mi^2)		TMARD% > 2 (mi / mi^2)	% Secure Habitat	SIZE[1] (mi^2)
		S1	S2			
Bechler/Teton	18	0	0	0	99	534
Boulder/Slough #1	4	2	2	0	97	282
Boulder/Slough #2	4	0	0	0	100	232
Buffalo/Spread Creek #1	17	0	0	0	99	222 (220)
Buffalo/Spread Creek #2	17	2	2	0	95	508
Crandall/Sunlight #1	6	6	6	1	92	130
Crandall/Sunlight #2	6	8	8	1	89	316
Crandall/Sunlight #3	6	5	5	1	93	222
Firehole/Hayden #1	10	0	0	0	100	339
Firehole/Hayden #2	10	0	0	0	100	177
Gallatin #1	2	0	0	0	99	128
Gallatin #2	2	1	1	0	99	155
Gallatin #3	2	16	16	8	81	218
Hellroaring/Bear #1	3	9	9	4	91	185
Hellroaring/Bear #2	3	0	0	0	100	229
Henrys Lake #1	12	31	31	16	67	201 (191)
Henrys Lake #2	12	14	14	7	85	153 (140)
Hilgard #1	1	6	6	2	91	201
Hilgard #2	1	2	2	3	92	141
Lamar #1	5	2	2	1	97	300
Lamar #2	5	0	0	0	100	181
Madison #1	11	6	6	3	94	227
Madison #2	11	8	8	4	90	157 (149)
Pelican/Clear #1	8	0	0	0	100	108
Pelican/Clear #2	8	0	0	0	100	257
Plateau #1	13	2	2	1	95	286
Plateau #2	13	0	0	0	99	431 (420)
Shoshone #1	7	1	1	0	99	122
Shoshone #2	7	0	0	0	99	132
Shoshone #3	7	1	1	0	98	141
Shoshone #4	7	1	1	0	96	189
South Absaroka #1	16	0	0	0	99	163
South Absaroka #2	16	0	0	0	100	191
South Absaroka #3	16	0	0	0	100	348
Thorofare #1	15	0	0	0	100	274
Thorofare #2	15	0	0	0	100	180
Two Ocean/Lake #1	14	0	0	0	100	485 (372)
Two Ocean/Lake #2	14	0	0	0	100	143 (125)
Washburn #1	9	0	0	0	100	178
Washburn #2	9	0	0	0	100	144
Mean for PCA/Total square miles		3	3	1.3	96	**9209 (9035)**

[1] Lakes > 1 square mile in size were removed from subunit totals, OMARD, TMARD and secure habitat calculations. Numbers in parenthesis are square miles of subunit without these lakes.

Table 4. The 1998 baseline for numbers of developed sites on public lands within each of the Bear Management Subunits in the GYA.

Bear Management Subunit	Area (mi^2) [1]	Administrative Units	Summer Home Complexes	Developed Campgrounds	Trail Heads	Major Developed Sites and Lodges	Administrative or Maintenance Sites	Other Developed Sites	Plans of Operation for Minerals Activities[2]
Bechler/Teton	534	TNF	0	1	9	2	2	6	0
		YNP	0	0	2	0	2	2	0
		GTNP	0	8	3	1	3	10	0
Boulder/Slough #1	282	CNF	0	0	0	0	0	0	7
		GNF	0	1	6	0	1	2	2
Boulder/Slough #2	232	GNF	0	0	0	0	2	0	0
		YNP	0	1	3	0	2	1	0
Buffalo/Spread Creek #1	222	BTNF	1	3	3	3	2	5	0
	(220)	GTNP	0	0	7	2	2	3	0
Buffalo/Spread Creek #2	508	BTNF	0	0	3	0	1	4	0
Crandall/Sunlight #1	130	SNF	0	2	5	1	1	4	0
		GNF	0	0	3	0	0	5	0
Crandall/Sunlight #2	316	SNF	0	4	7	1	2	3	0
		GNF	0	1	0	0	0	0	0
Crandall/Sunlight #3	222	SNF	0	4	4	0	2	2	0
Firehole/Hayden #1	339	YNP	0	1	5	1	6	13	0
Firehole/Hayden #2	172	YNP	0	1	3	1	2	8	0
Gallatin #1	128	YNP	0	0	3	0	1	0	0
Gallatin #2	155	YNP	0	2	5	1	12	1	0
Gallatin #3	218	GNF	0	2	12	0	6	9	0
		YNP	0	0	0	0	0	0	0
Hellroaring/Bear #1	185	GNF	0	4	9	0	4	2	8[3]
		YNP	0	0	1	0	0	1	0
Hellroaring/Bear #2	229	GNF	0	0	0	0	0	0	0
		YNP	0	0	0	0	2	0	0

1 Area in parenthesis is the area of the subunit without large lakes > 1 square mile.
2 Mining claims with plans of operation are considered developed sites for this baseline. Not all sites currently have active projects.
3 Includes one materials mineral site with an outside contractor.

Bear Management Subunit	Area (mi^2)	Administrative Units	Summer Home Complexes	Developed Campgrounds	Trail Heads	Major Developed Sites and Lodges	Administrative or Maintenance Sites	Other Developed Sites	Plans of Operation for Minerals Activities
Henrys Lake #1	201 (191)	TNF	2	3	1	0	8	5	1 0
Henrys Lake #2	153 (140)	TNF GNF	0 4	0 3	1 5	0 0	1 0	1 1	1 0
Hilgard #1	202	BNF GNF	0 0	0 0	1 4	0 0	2 2	0 4	0 0
Hilgard #2	141	GNF YNP	0 0	0 0	7 3	0 0	0 0	1 0	0 0
Lamar #1	300	YNP GNF SNF CNF	0 0 0 0	1 2 0 0	5 8 0 0	0 0 0 0	3 3 0 0	2 7 0 0	0 6 0 1
Lamar #2	181	YNP	0	0	0	0	4	0	0
Madison #1	228	GNF YNP	0 0	2 0	11 0	0 0	1 0	10 0	0 0
Madison #2	157 (149)	GNF YNP	5 0	2 0	1 1	0 0	1 2	5 1	0 0
Pelican/Clear #1	108	YNP	0	0	2	0	0	0	0
Pelican/Clear #2	252	YNP	0	1	4	1	4	3	0
Plateau #1	286	TNF GNF YNP	1 0 0	0 0 0	0 1 0	0 0 0	1 0 1	0 0 0	0 0 0
Plateau #2	431 (420)	TNF YNP	0 0	2 0	1 0	0 0	2 4	1 0	0 0
Shoshone #1	122	SNF	1	2	0	0	0	3	0
Shoshone #2	132	SNF	0	0	1	1	0	0	0
Shoshone #3	141	SNF	2	0	2	1	0	0	0
Shoshone #4	189	SNF	3	3	3	6	0	7	0

Subunit	Area (mi^2)	Administrative Units	Summer Home Complexes	Developed Campgrounds	Trail Heads	Major Developed Sites and Lodges	Administrative or Maintenance Sites	Other Developed Sites	Plans of Operation for Minerals Activities
South Absaroka #1	163	SNF	0	0	0	0	0	0	0
South Absaroka #2	191	SNF	0	0	0	0	2	0	1[4]
South Absaroka #3	348	SNF	1	3	4	1	2	6	1[4]
Thorofare #1	273	BTNF YNP	0 0	0 0	0 0	0 0	2 4	0 0	0 0
Thorofare #2	180	BTNF YNP	0 0	0 0	0 0	0 0	0 0	0 0	0 0
Two Ocean/Lake #1	485 (372)	YNP BTNF GTNP	0 0 0	2 3 0	3 3 1	1 0 0	3 1 0	2 0 1	0 0 0
Two Ocean/Lake #2	143 (125)	YNP BTNF	0 0	0 0	0 0	0 0	2 1	0 0	0 0
Washburn #1	178	YNP	0	2	8	2	7	6	0
Washburn #2	144	YNP	0	1	6	0	1	4	0
Primary Conservation Area	**9210 (9036)**	**ALL**	**20**	**67**	**180**	**26**	**119**	**151**	**27**

4 One operating plan that encompasses all claims in both subunits

Livestock Grazing

There were 88 commercial livestock grazing allotments inside the PCA in 1998 and 17,279 permitted sheep AMs (Table 4). Allotments with less than 100 acres inside the PCA were not included. Where several allotments are managed as one, this was counted as a single allotment. Sheep animal months (AMs) are calculated by multiplying the permitted number of sheep times the months of permitted use. Actual use by sheep in many cases may have been less than the permitted numbers identified for 1998.

Allotments include both vacant and active commercial grazing allotments. Vacant allotments are those without an active permit but may be used periodically by other permittees at the discretion of the land management agency to resolve resource issues or other concerns. Reissuance of permits for vacant cattle allotments may result in an increase in the number of permitted cattle but the number of allotments would remain the same as the 1998 baseline. Combining or dividing existing allotments would be allowed as long as acreage in allotments does not increase. Any such use of vacant cattle allotments resulting in an increase in cattle numbers will only be done after an analysis to evaluate impacts on grizzly bears. Where chronic conflicts occur on cattle allotments inside the PCA, and an opportunity exists with a willing permittee, one alternative for resolving the conflict may be to phase out cattle grazing or to move the cattle to a currently vacant allotment where there is less likelihood of conflict.

It should be noted that the Caribou-Caribou-Targhee National Forest closed 2 sheep allotments and is keeping 2 additional sheep allotments vacant since 1998, resulting in a reduction of 7,889 sheep AMs over that shown for the Caribou-Targhee in Table 4.

Table 5. Number of commercial livestock grazing allotments and sheep animal months inside the Yellowstone Primary Conservation Area in 1998.

Adminstrative Unit	Cattle Allotments		Sheep Allotments		Sheep AMs
	Active	Vacant	Active	Vacant	
Beaverhead-Deerlodge NF	4	2	0	0	0
Custer NF	0	0	0	0	0
Bridger-Teton NF	4	0	0	0	0
Gallatin NF	21	0	2	0	1,800
Caribou-Targhee NF	8	0	9	2	15,479
Shoshone NF	33	0	0	2	0
Grand Teton NP	1	0	0	0	0
Total in PCA	71	2	11	4	17,279

Habitat Effectiveness

Habitat effectiveness outputs from the Grizzly Bear Cumulative Effects Model (CEM, Dixon 1997, Weaver *et al*. 1986, Mattson *et al.* in press) as of 1998 are presented in Table 5. Habitat effectiveness as displayed in Table 4 is a relative measure of that part of the energy potentially derived from the area that is available to bears given their response to humans (Mattson *et al*. in press). The higher the number the greater the habitat effectiveness. The highest values in the estrous period are associated with cutthroat trout spawning streams, high vales in early hyperphagia are a result of moth aggregation sites and high values in late hyperphagia are primarily due to white bark pine. HE is calculated using the ICE9 software (Bevins, 1997), which evaluates information, contained in several Geographic Information System and tabular databases. The databases include digital maps of vegetation, ungulate winter ranges, and point, linear and dispersed human activities, coefficient tables that categorize the relative values of vegetation and human activities and tables that identify the type, intensity and duration of the human activities.

Table 6. 1998 Habitat Effectiveness (HE) values by season from the Yellowstone Grizzly Bear Cumulative Effects Model (CEM) for each of the 40 Greater Yellowstone Area grizzly bear management subunits.[1]

Subunit	Spring (3/1-5/15) HE	Estrus (5/16-7/15) HE	Early Hyperphagia (7/16-8/31) HE	Late Hyperphagia (9/1-11/30) HE
Bechler/Teton#1	116	64	44	274
Boulder/Slough#1	105	105	119	853
Boulder/Slough#2	123	112	111	521
Buffalo/Spread Cr#1	79	86	78	267
Buffalo/Spread Cr#2	58	98	125	863
Crandall/Sunlight#1	53	94	78	800
Crandall/Sunlight#2	52	82	124	329
Crandall/Sunlight#3	53	50	156	208
Firehole/Hayden#1	96	189	162	244
Firehole/Hayden#2	45	843	66	342
Gallatin#1	139	144	198	635
Gallatin#2	104	97	105	585
Gallatin#3	78	69	89	599
Hellroaring/Bear#1	85	74	95	678
Hellroaring/Bear#2	117	99	98	628
Henrys Lake#1	41	39	32	178
Henrys Lake#2	41	41	33	225
Hilgard#1	99	68	91	614
Hilgard#2	81	97	132	902
Lamar#1	127	118	136	571
Lamar#2	132	167	180	795
Madison#1	53	115	227	390
Madison#2	41	60	147	63
Pelican/Clear#1	103	324	105	560
Pelican/Clear#2	105	2253	203	997
Plateau#1	26	49	36	109
Plateau#2	75	81	56	442
Shoshone#1	39	50	115	264
Shoshone#2	51	56	1424	387
Shoshone#3	65	57	583	484
Shoshone#4	57	78	327	392
South Absaroka#1	55	57	392	399
South Absaroka#2	41	45	339	250
South Absaroka#3	46	73	303	551
Thorofare #1	84	488	298	956
Thorofare #2	79	82	295	583
Two Ocean/Lake#1	115	1300	64	426
Two Ocean/Lake#2	117	2401	107	1079
Washburn#1	121	110	126	404
Washburn#2	99	86	85	272

[1] Bevins 1997, USDA Forest Service 1990. HE values are based on productivity coefficients depicting an average year (Mattson et al. in press). The higher the number the greater the habitat effectiveness.

Habitat Value of Secure Habitats

The decision to set the baseline for secure habitat at the levels existing in 1998 required an evaluation of the relative habitat quality of the existing secure habitats. Secure habitat should meet the seasonal habitat needs of the bear and should be representative of seasonal habitats in the entire analysis area (IGBC 1998). The Yellowstone Grizzly Bear Cumulative Effects Model (CEM) was used to compare the average habitat value of the secure habitat to that for the entire subunit for each of six categories (Table 7). These averages were not compared statistically as secure habitats in each subunit showed similar proportions of each category as compared to the subunit average. Habitat value was used rather than habitat effectiveness so that the presence or absence of human activities did not influence the comparison. Had existing secure habitats shown lower habitat value than the subunit average, it might have been necessary to recommend closing roads to create appropriate seasonal secure habitat.

It should be noted that habitat effectiveness, which is habitat value reduced by the presence of human activities, is much higher in secure habitats than non-secure habitats. Motorized access routes reduce habitat values more than any other human activity. Secure habitats do not contain motorized access routes.

Table 7. Percent of 6 habitat value categories in each of 40 Greater Yellowstone Area grizzly bear management subunits and percent of the 6 habitat value categories in secure habitat in each subunit.[1]

Subunit	mi^2	Habitat Value Category Percent of Subunit[2]						Secure Habitat mi^2(% of subunit)	Habitat Value Category Percent of Secure Habitat[2]					
		VL	L	LM	HM	H	VH		VL	L	LM	HM	H	VH
BECHLER/TETON #1	534	11	20	15	49	1	4	417(78)	14	20	13	48	1	4
BOULDER/SLOUGH_#1	282	12	1	40	45	2	0	272(97)	13	1	42	43	2	0
BOULDER/SLOUGH_#2	232	9	6	33	52	1	0	227(98)	9	6	34	50	1	0
BUFFALO/SPREAD_CR_#1	220	25	20	13	39	2	0	194(88)	25	20	14	39	2	0
BUFFALO/SPREAD_CR #2	508	14	12	21	51	3	0	412(81)	14	10	22	52	2	0
CRANDALL/SUNLIGHT_#1	130	10	34	43	11	2	0	105(81)	11	35	42	10	2	0
CRANDALL/SUNLIGHT_#2	316	5	30	34	30	1	0	260(82)	4	32	34	29	1	0
CRANDALL/SUNLIGHT_#3	222	2	43	42	13	1	0	178(80)	1	45	42	12	0	0
FIREHOLE/HAYDEN_#1	339	2	4	65	21	5	3	300(88)	1	2	69	21	4	3
FIREHOLE/HAYDEN_#2	177	3	7	68	7	1	14	152(88)	1	7	74	6	1	10
GALLATIN_#1	128	6	1	29	62	1	0	123(96)	6	1	29	62	1	0
GALLATIN_#2	155	2	8	27	63	1	0	140(90)	2	4	29	65	1	0
GALLATIN_#3	218	18	17	13	51	1	0	120(55)	21	12	12	55	1	0
HELLROARING/BEAR_#1	185	17	20	12	51	0	0	142(77)	17	15	11	57	0	0
HELLROARING/BEAR_#2	229	21	5	26	47	2	0	228(99)	21	5	26	46	2	0
HENRYS_LAKE_#1[3]	191	47	7	10	36	0	0	87(45)	31	9	11	50	0	0
HENRYS_LAKE_#2	140	7	19	26	46	2	1	64(46)	9	17	24	50	0	1
HILGARD_#1	201	19	12	18	51	1	0	140(70)	20	10	19	51	0	0
HILGARD_#2	141	13	8	17	61	1	0	100(72)	15	8	13	64	1	0
LAMAR_#1	300	4	2	26	68	1	0	268(89)	4	1	25	70	0	0

1 Habitat value or habitat productivity as measured by the Yellowstone grizzly bear cumulative effects model (CEM) (Mattson et al. in press). Large lakes >1 mi^2 were excluded from this analysis.
2 Six-part categories were determined from raw CEM habitat value outputs that provide relative comparisons across seasons. VL = Very Low (0-15), L = Low (16-42), LM = Low Moderate (43-122), HM = High Moderate (123-355), H = High (356-1032), VH = Very High (>1032). Percent rounded to the nearest whole number.
3 Includes Henry's Lake Flat (private land) where habitat type map data was not available and was counted as having no value in this analysis.

Table 7 (continued). Percent of 6 habitat value categories in each of 40 Greater Yellowstone Area grizzly bear management subunits and percent of the 6 habitat value categories in secure habitat in each subunit.[1]

Subunit	mi^2	Habitat Value Category Percent of Subunit[2]						Secure Habitat mi^2(% of subunit)	Habitat Value Category Percent of Secure Habitat[2]					
		VL	L	LM	HM	H	VH		VL	L	LM	HM	H	VH
LAMAR_#2	181	4	1	34	60	1	0	181(100)	4	1	34	60	1	0
MADISON_#1	227	4	12	52	21	10	2	163(72)	5	12	58	17	8	1
MADISON_#2	149	2	6	69	19	3	2	99(67)	0	4	79	14	2	1
PELICAN/CLEAR_#1	108	1	8	6	80	6	0	106(98)	1	7	7	79	6	0
PELICAN/CLEAR_#2	257	2	8	33	33	7	16	237(94)	2	8	34	33	7	17
PLATEAU_#1	286	2	29	58	11	0	0	197(69)	1	28	58	13	0	0
PLATEAU_#2	420	0	19	37	42	0	1	372(89)	0	20	36	42	0	1
SHOSHONE_#1	122	1	53	45	2	0	0	120(99)	1	53	45	2	0	0
SHOSHONE_#2	132	2	63	29	6	0	0	131(99)	2	63	29	6	0	0
SHOSHONE_#3	141	1	47	43	9	1	0	136(97)	1	48	43	7	1	0
SHOSHONE_#4	189	2	40	35	23	1	0	179(95)	1	41	34	23	1	0
SOUTH_ABSAROKA_#1	163	2	3	86	9	0	0	162(99)	2	3	86	9	0	0
SOUTH_ABSAROKA_#2	191	1	2	93	3	1	0	190(100)	1	2	93	3	1	0
SOUTH_ABSAROKA_#3	348	1	4	90	2	4	0	337(97)	1	4	90	2	3	0
THOROFARE_#1	274	5	2	82	3	5	3	273(100)	5	2	82	3	5	3
THOROFARE_#2	180	8	2	83	1	5	0	180(100)	8	2	83	1	5	0
TWO_OCEAN/LAKE_#1	372	1	2	74	3	9	12	358(96)	1	2	74	3	8	12
TWO_OCEAN/LAKE_#2	125	1	1	71	3	6	18	125(100)	1	1	71	3	6	18
WASHBURN_#1	178	6	8	18	68	1	0	148(83)	6	6	22	66	1	0
WASHBURN_#2	144	27	2	40	30	1	0	133(92)	27	1	41	30	1	0

1 Habitat value or habitat productivity as measured by the Yellowstone grizzly bear cumulative effects model (CEM) (Mattson et al. in press). Large lakes >1 mi^2 were excluded from this analysis.

2 Six-part categories were determined from raw CEM habitat value outputs that provide relative comparisons across seasons. VL = Very Low (0-15), L = Low (16-42), LM = Low Moderate (43-122), HM = High Moderate (123-355), H = High (356-1032), VH = Very High (>1032). Percent rounded to the nearest whole number.

Appendix G. Motorized Access Management Inside and Outside the Primary Conservation Area

Introduction

This conservation strategy identifies the standard for motorized access for subunits inside the Primary Conservation Area (PCA). No motorized access standards are identified for areas outside the PCA in this conservation strategy; however, access standards exist outside the PCA on some Forests as part of big game standards in Forest Plans. This appendix provides additional specifics on how the motorized access habitat standard will be applied inside the PCA and what the application of temporary changes really means in terms of area of secure habitat potentially affected. Information is also presented that identifies the potential for future increases in motorized access outside the PCA given current management prescriptions.

Application of the Motorized Access Habitat Standard in the PCA

The habitat standard for motorized human access in this document requires that secure habitat will be maintained or improved from the 1998 baseline, while maintaining options for management of project activities at approximately the same level as existed in 1998 (Appendix V). The grizzly bear achieved all recovery goals by 1998 concurrent with many ongoing forest management activities. Timber sales and other vegetation management activities occurred regularly. Roads were built and roads were decommissioned. Humans continued to recreate and share the landscape with the bears. Application of the standard to maintain the 1998 baseline, will allow forest management activities to continue at approximately the same level that was occurring in 1998.

The 1997 Targhee Forest Plan standards and guidelines, when fully implemented and adopted, meet the intent of the habitat standard for motorized access for this Conservation Strategy.

The following section restates the access management standard, provides definition of terms used and additional information on how this standard will be applied.

Motorized Access Standard

The following motorized access standard will be monitored and maintained on public lands within all subunits in the PCA as measured from the 1998 baseline.

The percent of secure habitat within each Bear Management Subunit must be maintained at or above levels existing in 1998 (Appendix F, Table 1). Temporary and permanent changes are allowed under specific conditions identified below. The definitions and rule set on the following page and in table 1 will be used in management and evaluation of projects and habitat management actions as appropriate under this Conservation Strategy.

Application Rules for Changes in Secure Habitat

Permanent Changes to Secure Habitat. A project may permanently change secure habitat provided that replacement secure habitat of equivalent habitat quality (as measured by CEM or equivalent technology) is provided in the same grizzly subunit. The replacement habitat must either be in place prior to project initiation or be provided concurrently as an integral part of the project plan.

Temporary Changes to Secure Habitat. Temporary reductions in secure habitat can occur to allow projects, provided that all of the following conditions are met:

- Only one project is active per grizzly subunit at any one time.
- Total acreage of active projects within a given bear management unit (BMU) will not exceed 1 percent of the acreage in the largest subunit within that BMU (Table 2). The acreage of a project that counts against the 1% limit is the acreage associated with the 500-meter buffer around any motorized access route that extends into secure habitat.
- Secure habitat is restored within one year after completion of the project.

Definitions and Rules

Definition of Secure Habitat. Secure habitat is defined as any area greater than 500 meters from an open or gated Motorized Access Route and greater than 10 acres in size.

Definition of Replacement Secure Habitat. Must be of equal or greater habitat value and maintained for a minimum of 10 years.

Time of Secure Habitat Application. Secure habitat application will apply from March 1 to November 30 each year for each of two seasons. There are no secure habitat standards in the winter season (12/1 – 2/28). Season 1 is 3/1 – 7/15 and season 2 is 7/16 – 11/30.

Definition of a Project. A project is defined as any activity requiring a new open motorized access route in presently secure habitat. Under this definition, projects only include activities in secure habitat that require construction of new roads, reconstructing or opening a restricted road or require recurring helicopter flight lines at low elevations.

Recurring Helicopter Flights. While helicopter access is generally preferred over construction of roads to minimize long term effects on secure habitat, recurring flight lines at low elevation (such as required for helicopter logging) have much the same displacement potential as roads during the duration of the activity.

Inclusions in Secure Habitat. Secure habitat can include roads restricted with permanent barriers (not gates), decommissioned or obliterated roads and/or non-motorized trails.

Activities that are allowed in secure habitat. Activities that do not require road construction, reconstruction, opening a restricted road, or recurring helicopter flight lines at low elevation do not detract from secure habitat. Examples of such activities include thinning, tree planting, prescribed fire, trail maintenance, and administrative studies/monitoring. However, these activities should be concentrated in time and space to the extent feasible to minimize disturbance. Land management agencies will also be sensitive to these activities occurring adjacent to active projects and will analyze the effects in the NEPA process for the project.

- Helicopter use to respond to emergencies such as fire suppression or search and rescue activities does not detract from secure habitat under this definition. Likewise, helicopter use for short-term activities such as prescribed fire ignition/ management, periodic administrative flights, and other similar activities does not constitute a project under this definition.
- High use non-motorized trails, winter snow machine trails and other motorized winter activities do not count against secure habitat for the following reasons:
    - A Biological Assessment on the Effects of Snowmobile Use on Grizzly Bears on the Gallatin, Beaverhead-Deerlodge, Custer, Bridger-Teton and Shoshone National Forests was completed by the Forest Service and submitted to the U. S. Fish and Wildlife Service in 2001. The U. S. Fish and Wildlife Service issued a Biological Opinion on this assessment, stating that current authorized snowmobile activity on the Custer, Shoshone, Gallatin, Beaverhead-Deerlodge, and Bridger-Teton National Forests is not likely to jeopardize the continued existence of the grizzly bear (U. S. Fish and Wildlife Service Biological Opinion dated May 30, 2002). Effects of snowmobile use on grizzly bears on the Caribou-Targhee National Forest was consulted on with the U. S. Fish and Wildlife Service as part of the 1997 Revised Forest Plan for the Targhee National Forest.
    - Research addressing grizzly interactions with high use, non motorized trails is very limited and has not identified impacts to grizzly bears, particularly when other management practices are employed to reduce conflicts i.e. food storage orders. Likewise, limited research is available regarding the effects of winter-motorized activity on denning grizzly bears (see Record of the Workshop on Snowmobile Effects on

Wildlife: Monitoring Protocols. 2001 National Park Service. University of Montana School of Forestry, Missoula, MT. 110 pp.).

- Further research is needed to address the potential impact of high use non-motorized trails and winter motorized activities on grizzly bears. As such research information becomes available, an adaptive management approach will be used as necessary to incorporate any new information.
- If conflicts develop with winter use activities, either during denning or after den emergence in the spring, the land management agencies can address these conflicts with localized area restrictions under current travel management rules.

Subunits with planned temporary secure habitat reduction. Secure habitat values for subunits Gallatin #3 and Hilgard #1 will temporarily decline below 1998 values due to the Gallatin Range Consolidation Act. This temporary decline is acceptable because upon completion of the land exchange and associated timber sales, secure habitat and motorized access route density levels in these subunits will be improved from the 1998 baseline (Table 1).

Subunits with potential for improvement. Henrys Lake #2, Gallatin #3 and Madison #2 on the Gallatin National Forest have the potential for improvement in the amount of secure habitat. The timing and amount of improvement will be determined through the Gallatin National Forest Travel Management Planning process. This process has been initiated.

Changes in OMARD and TMARD associated with Secure Habitat Changes

- The percentage of OMARD and TMARD may change slightly from the baseline with permanent changes in location of secure habitat under the standard. There is not a one to one relationship between OMARD, TMARD and secure habitat due to the nuances of the moving windows analysis used to estimate these motorized access parameters. Resulting slight changes in OMARD and TMARD will become the new baseline for monitoring changes in these parameters.
- Temporary increases in both OMARD and TMARD parameters may occur as secure habitat is temporarily decreased as identified in the habitat standard. Actual percentage increase will vary depending on how new open roads are situated on the landscape and their location relative to other existing roads. The resulting somewhat proportional increase in OMARD and/or TMARD will return to baseline levels as secure habitat is restored under the secure habitat standard.

Potential for Changes in Motorized Access Inside and Outside the PCA

All three state grizzly bear management plans recognize the importance of areas that provide security for bears both inside and outside the PCA. The Montana and Wyoming plans make recommendations for maintaining average road densities of one mile per square mile of habitat or less outside the PCA on federal lands. Both states have similar recommendations for elk habitat and note that this level of motorized access has been demonstrated to meet the needs of a variety of wildlife while maintaining reasonable public access. The Idaho State plan encourages land management agencies to monitor motorized access outside the PCA and to focus on those areas that provide security for bears (areas that have no motorized access routes or motorized access routes less than or equal to 1.0 mile per square mile). The area most likely to be occupied by grizzly bears outside the PCA in Idaho is on the Caribou-Targhee National Forest. The 1997 Targhee Forest Plan includes motorized access standards and prescriptions outside the PCA.

The land management activities that are most likely to influence existing secure habitat and motorized access route density on federal lands are oil and gas development and timber harvest. However, USDA Forest Service Interim Directive 1920-2001-1 that became effective December 14, 2001 currently regulates activities in 'inventoried roadless areas'. Under this directive little road building or timber harvest can be done in inventoried roadless areas until Forest Plans are

revised or amended to specifically address activities in roadless areas. The Chief of the Forest Service may approve a project in roadless areas under special circumstances. Given the time frames of adjusting Forest Plans and the speed with which court cases and other political processes move, it is likely that the Forest Service will be operating under the interim direction for the next 5 or more years. The Targhee National Forest is exempt from this interim directive because they are operating under a Revised Forest Plan, which addresses the management of roadless areas. Motorized access and other management activities are addressed by specific Management Prescription direction in the Revised Forest Plan. In general the Management Prescription direction provides that existing roadless areas remain roadless.

Current oil and gas leasing stipulations, timber management prescriptions and/or timber suitability designations and their relationship to inventoried roadless areas were evaluated within secure habitats inside the PCA (Table 3) and all habitats outside (Table 4) within proclaimed Forest Service Boundaries. It should also be noted that the USDA Forest Service initiated a new process in August 1999 to assist managers in balancing the benefits of access and the road-associated effects on other values and resources. Projects that may result in construction or reconstruction of roads require a roads analysis. Roads analysis is also integrated as part of watershed analysis, landscape assessments or other analysis. The roads analysis process (USDA 1999) is a science based ecological, social and economic approach to transportation planning, addressing both existing and future roads. It is not a decision document. The roads analysis will provide information to help land managers develop road systems that are safe and responsive to public needs, are affordable and efficiently managed, have minimal negative ecological effects on the land and are in balance with available funding for needed management actions. It will inform future management decisions on the benefits and risks of constructing new roads, relocating, reconstructing and decommissioning existing roads, managing traffic and road system maintenance.

Inside the PCA

Eightly-Six percent of the habitat inside the PCA is secure with 85 percent of the secure habitat occurring within National Parks or Forest Service Wilderness Areas (Table 2). Only 25 subunits of the 40 subunits in the PCA have portions that extend outside National Park or National Forest Wilderness Areas. Even in those 25 subunits, 81 percent of the habitat is secure with 75 percent of that secure habitat occuring within National Parks or National Forest Wilderness Areas where projects requiring new motorized access routes are prohibited or highly unlikely (Table 3). Therefore, only 15 percent of the secure habitat inside the PCA (25 percent of the secure habitat in the 25 subunits) has any potential for projects and a maximum of 25 projects (one per subunit) could occur within in the PCA at any one time under the rule set. Further, 62 percent of the 15 percent potentially available for projects is classified as roadless (Table3).

If all 25 possible projects were underway simultaneously, these projects could result in a maximum temporary reduction of only ½ of one percent of the secure habitat (39.7 square miles) in the PCA at any given time (Table 2). In reality, not every subunit is likely to have a project operating simultaneously nor is each project likely to use the maximum acreage, so the effect on secure habitat at any given time would likely be considerably less. Other factors such as Forest/Park Plan Standards and Guidelines and land use prescriptions, issues raised in the NEPA/public involvement process and other practical realities may further limit management activities and projects. For example, of the 1,201 square miles of secure habitat potentially available for projects within the 25 subunits, only 30 percent (363 square miles) includes suitable timber lands, land management prescriptions that emphasize timber management or oil and/or gas leasing stipulations that allow surface occupancy. Fourty-one percent of the 30 percent is within a roadless area where new roads are regulated by the Forest Service Interim Directive or regulated by the Targhee Forest Plan. The bottom line is that only about 213 square miles of the secure habitat on Forest Service Administered Lands in the PCA is identified for oil, gas or timber management outside roadless areas. An additional 49 square miles of secure habitat exists on

private and state inholdings within the PCA that have various levels of potential for increased motorized access.

Outside the PCA

There are a total of 13,306 square miles within proclaimed Forest Service Boundaries (including inholdings) outside the PCA in the GYA (Figure 3). While estimates are not available for secure habitat percentages outside the PCA, 22 percent of this area is wilderness and 49 percent is designated as a roadless area where new motorized access routes are regulated by the Forest Service Interim Directive or management prescription direction in the Targhee Forest Plan (Table 4.) It should be noted that some of the inventoried roadless areas contain existing open motorized access routes. However, new roads in these areas require the same scrutiny as roadless areas without roads.

Thrity-seven percent of the total area within proclaimed Forest Service Boudaries outside the PCA either allows surface occupancy for oil and gas development, has suitable timber or a management prescription that promotes timber harvest, however 51 percent of this area is within a roadless area (Table 4). Thus only about 2,400 square miles of Forest Service Administered Lands, 18 percent of the lands inside proclaimed Forest Service boundaries outside the PCA, are available for oil, gas or timber resource management outside roadless areas. Only a small portion of this total would actually be managed at any give time. Forest Plan Standards and Guidelines to protect other resources will reduce the actual area affected by these activities. An additional 437 square miles of private and state inholdings have various levels of potential for increased motorized access. Many of the inholdings are already roaded as are some of the Forest Service Administered lands identified as allowing timber harvest or oil and gas development.

Availability of lands for oil and gas development or timber harvest are not likely to increase significantly over time or under new Forest Plans. Current public processes have already identified lands that have the potential to be managed for timber, oil or gas and restrictions necessary to protect other resources. The exceptions are the Deerlodge, Gallatin and the Pryor Mountain portion of the Custer National Forests that have not yet completed an analysis of oil and gas leasing potential. Public lands on these forests are currently suspended from leasing until an Environmental Impact Statement is completed.

Table 1. The rule set for secure habitat management in the Yellowstone PCA.

Criteria	Definition
Software, Database and Calculation Parameters	ARC INFO using the moving window GIS technique (Mace et al. 1996). 30-meter pixel size, square mile window size and density measured as miles/square mile. Motorized access features from the CEM GIS database
Motorized Access Routes in Database	All routes having motorized use or the potential for motorized use (restricted roads) including motorized trails, highways, and forest roads. Private roads and state and county highways counted.
Season Definitions	Season 1 – 1 March to 15 July. Season 2 – 16 July to 30 November. There are no access standards in the winter season (1 December to 28 February)
Habitat Considerations	Habitat quality not part of the standards but 1) Replacement secure habitat requires equal or greater habitat value 2) Road closures should consider seasonal habitat needs.
Secure Habitat	More than 500 meters from an open or gated motorized access route or reoccurring helicopter flight line. Must be greater than or equal to 10 acres in size. Replacement secure habitat created to mitigate for loss of existing secure habitat must be of equal or greater habitat value and remain in place for a minimum of 10 years. Large lakes not included in calculations.
Project	An activity requiring construction of new roads, reconstructing or opening a restricted road or recurring helicopter flights at low elevations.
Activities Allowed in Secure Habitat	Activities that do not require road construction, reconstruction or opening a restricted road or reoccurring helicopter flights. Over the snow use allowed until further research identifies a concern.
Inclusions in Secure Habitat	Roads restricted with permanent barriers (not gates), decommissioned or obliterated roads and/or non-motorized trails.
Temporary Reduction in Secure Habitat	One project per subunit is permitted that may temporarily reduce secure habitat. Total acreage of active projects in the BMU will not exceed 1 percent of the acreage in the largest subunit within the BMU. The acreage that counts against the 1% is the 500-meter buffer around open motorized access routes extending into secure habitat. Secure habitat is restored within one year after completion of the project
Permanent Changes to Secure Habitat	A project may permanently change secure habitat provided that replacement secure habitat of equivalent habitat quality (as measured by CEM or equivalent technology) is provided in the same grizzly subunit. The replacement habitat must either be in place prior to project initiation or be provided as an integral part of the project plan.
Subunits with Planned Temporary Secure Habitat Reduction	Secure habitat for subunits Gallatin #3 and Hilgard #1 will temporarily decline below 1998 values due to the Gallatin Range Consolidation Act. Upon completion of the land exchange and associated timber sales, secure habitat in these subunits will be improved from the 1998 baseline.
Subunits with Potential for Improvement	Access values for Henrys Lake #2, Gallatin #3 and Madison # 2 have the potential for improvement. The quantity and timing of the improvement will be determined by the Gallatin National Forest Travel Management plan.
Proactive Improvement in Secure Habitat	A proactive increase in secure habitat may be used at a future date to mitigate for impacts of proposed projects of that administrative unit within that subunit.
Exceptions for Caribou-Targhee NF	When fully implemented and adopted, the Standards and Guidelines in the 1997 Revised Targhee Forest Plan meet the intent of maintaining secure habitat levels.

Table 2. Square miles of secure habitat in various management categories for each of the 40 Bear Management subunits in the Yellowstone Primary Conservation Area. (Figures include inholdings within proclaimed Forest Service Bourndaries. Large lakes greater than 1 mile per square mile were excluded.)

SUBUNIT NAME	Subunit Area (mi^2)	Total Secure Habitat (mi^2)	Wilderness or Park Secure Habitat (mi^2)	Non Wilderness or Non Park Secure Habitat (mi^2)	Total Area of Non Wilderness or Non Park In Subunit (mi^2)	Maximum area of Secure Habitat Available for Projects Under Rule Set at any one time[1] (mi^2)
Bechler/Teton #1	534	417	322	95	197	5.3
Boulder/Slough #1	282	272	269	3	11	2.8
Boulder/Slough #2	232	227	227	0	0	
Buffalo/Spread Creek #1	220	194	187	7	9	5.1
Buffalo/Spread Creek #2	508	412	300	112	208	
Crandall/Sunlight #1	130	105	57	48	72	3.2
Crandall/Sunlight #2	316	260	175	85	139	
Crandall/Sunlight #3	222	178	97	81	123	
Firehole/Hayden #1	339	300	300	0	0	0 (3.4)[2]
Firehole/Hayden #2	177	152	152	0	0	
Gallatin #1	128	123	123	0	0	2.2
Gallatin #2	155	140	140	0	0	
Gallatin #3	218	120	8	112	209	
Hellroaring/Bear #1	185	142	101	41	81	2.3
Hellroaring/Bear #2	229	228	228	0	0	
Henrys Lake #1	191	87	0	87	191	1.9
Henrys Lake #2	140	64	0 0	64	140	
Hilgard #1	201	140	107	33	90	2.0
Hilgard #2	141	100	63	37	72	
Lamar #1	300	268	256	12	29	3.0
Lamar #2	181	181	181	0	0	
Madison #1	227	163	108	55	114	2.3
Madison #2	149	99	94	5	48	
Pelican/Clear #1	108	106	106	0	0	0 (2.6) [2]
Pelican/Clear #2	257	237	237	0	0	
Plateau #1	286	197	124	73	161	4.2
Plateau #2	420	372	298	74	119	
Shoshone #1	122	120	100	21	22	1.9
Shoshone #2	132	131	115	16	18	
Shoshone #3	141	136	131	5	9	
Shoshone #4	189	179	163	16	25	
South Absaroka #1	163	162	130	32	33	3.5
South Absaroka #2	191	190	174	16	16	
South Absaroka #3	348	337	266	71	82	
Thorofare #1	274	273	273	0	0	0 (2.7) [2]
Thorofare #2	180	180	180	0	0	
Two Ocean/Lake #1	372	358	358	0	0	0 (3.7) [2]
Two Ocean/Lake #2	125	125	125	0	0	
Washburn #1	178	148	148	0	0	0 (1.8) [2]
Washburn #2	144	133	133	0	0	
Total	**9035**	**7756**	**6556**	**1201**	**2218**	**39.7 (53.9)[2]**

1 See table 1 in this appendix for a summary of the secure habitat management rule set.

2 Area identified in parenthesis is potentially available for projects however, because the entire BMU is within a National Park or Wilderness Area, new road construction is highly unlikely. The total in parenthesis includes this potentially available area.

Table 3. Project Potential for Secure Habitats in the 25 Subunits in the PCA with non-wilderness and non-National Park secure habitat.[1]

Bear Management Subunit	Subunit Area (mi^2)	Secure Habitat (mi^2) [2]	Non-Wilderness or Non-Park Secure Habitat		Non-Wilderness or Non-Park Secure Habitat where • Suitable Timber has been identified or • Management Area Allows Timber Harvest as part of the ASQ and/or • Oil and Gas Surface Occupancy Allowed (mi^2)[2, 3]	
			Total mi^2	Roadless mi2 (%)	Total mi2	Roadless mi2 (%)
Bechler/Teton #1	534	417	95	20 (21)	54	13(24)
Boulder/Slough #1	282	272	3	1 (33)	2	1 (50)
Buffalo/Spread Creek # 1	220	194	7	7 (100)	3	3 (100)
Buffalo/Spread Creek # 2	508	412	112	66 (59)	32	16 (50)
Crandall/Sunlight # 1	130	105	48	41 (85)	14	1 (7)
Crandall/Sunlight # 2	316	260	85	74 (87)	21	16 (76)
Crandall/Sunlight # 3	222	178	81	72 (89)	28	26 (93)
Gallatin # 3	218	120	112	95 (85)	11	10 (91)
Hellroaring/Bear #1	185	142	41	29 (71)	4	3 (75)
Henrys Lake #1	191	87	87	26 (30)	24	8 (33)
Henrys Lake #2	140	64	64	26 (41)	9	3 (33)
Hilgard # 1	201	140	33	28 (85)	4	2 (50)
Hilgard # 2	141	100	37	35 (95)	4	2 (50)
Lamar #1	300	268	12	11 (92)	7	5 (71)
Madison #1	228	163	55	52 (95)	9	8 (89)
Madison #2	149	99	5	0 (0)	5	0 (0)
Plateau #1	286	197	73	2 (3)	44	1 (2)
Plateau #2	420	372	74	0 (0)	51	0 (0)
Shoshone #1	122	120	21	21 (100)	7	7 (100)
Shoshone #2	132	131	16	16 (100)	6	6 (100)
Shoshone #3	141	136	5	5 (100)	0	0 (0)
Shoshone #4	189	179	16	14 (88)	0	0 (0)
South Absaroka #1	163	162	32	27 (84)	8	8 (100)
South Absaroka #2	191	190	16	16 (100)	4	4 (100)
South Absaroka #3	348	337	71	64 (90)	12	7 (58)
Total (%)	**5957**	**4845 (81)**	**1201**	**748 (62)**	**363**	**150 (41)**

1 Lakes > 1 mi^2 are excluded from calculations. Subunit and secure area totals include inholdings within proclaimed Forest Service boundaries but were not included in timber harvest and oil or gas development totals. There are 49 mi^2 of secure habitat within the inholdings in the PCA.
2 Area roadless totals were calculated using inventoried roadless areas maps except for the portions of Henry's Lake #1 and #2, Plateau #1 and #2, and Bechler/Teton #1 that are on the Targhee National Forest that were calculated from the roadless management prescriptions identified in the 1997 Revised Targhee Forest Plan.
3 The area overlapping a map of the suitable timber base was used in these calculations for the Shoshone, Gallatin and Bridger-Teton National Forests. Management Area Prescriptions that allow timber harvest as part of the allowable sale quantity (ASQ) or suitable timber base were used for the Beaverhead-Deerlodge, Custer and Caribou-Targhee National Forest. The suitable timber maps probably more accurately portray the actual areas where timber harvests could occur. Management area prescriptions are a more general designation of areas that allow timber harvest.

Table 4. Potential for land management activities within proclaimed Forest Service Boundaries in State Bear Management Areas outside the Primary Conservation Area in the GYA.

Administrative Unit	Area in GYA outside PCA (mi^2)[1]	Wilderness mi^2 (%)	Roadless mi^2 (%)	or Management Area Allows Timber Harvest as part of ASQ [1]		Oil and Gas Stips Allow Surface Occupancy		• Suitable Timber or • Management Area Allows Timber Harvest as part of ASQ and/or • Oil and Gas Surface Occupancy Allowed[1]	
				Total mi^2	Roadless mi^2 (%)	Total mi^2	Roadless mi^2 (%)	Total mi^2	Roadless mi^2 (%)
Bridger-Teton National Forest	4269	1113 (26)	2302 (54)	476	288 (61)	1519	981 (65)	1605	1053 (66)
Gallatin National Forest	1920	479 (25)	739 (38)	320	127 (40)	0	0 (0)	320	127 (40)
Shoshone National Forest	1928	739 (38)	691 (36)	92	17 (18)	707	346 (49)	716	349 (49)
Custer National Forest	764	354 (46)	144 (19)	349	112 (32)	50	13 (26)	350	113 (32)
Beaverhead/Deerlodge National Forest	3241	86 (3)	2181 (67)	534	218 (41)	1082	693 (64)	1441	848 (59)
Caribou-Targhee National Forest [2]	1185	108 (9)	512(43)	447	23 (5)	206	0 (0)	488	23 (5)
Total	**13,307**	**2879 (22)**	**6569 (49)**	**2218**	**785 (35)**	**3564**	**2033 (57)**	**4920**	**2513 (51)**
% of total area in GYA outside PCA	**100**	**22**	**49**	**17**	**6**	**27**	**15**	**37**	**19**

1 The area overlapping the suitable timber base was used in these calculations for the Shoshone, Gallatin and Bridger-Teton National Forests. Management Area Prescriptions that allow timber harvest as part of the allowable sale quantity (ASQ) or suitable timber base were used for the Beaverhead-Deerlodge, Custer and Caribou-Targhee National Forest. The suitable timber maps probably more accurately portray the actual areas where timber harvests could occur. Management area prescriptions are a more general designation of areas that allow timber harvest. Lakes > 1 mi^2 are excluded from totals. Inholdings were included in all totals where applicable except totals for timber, oil or gas prescriptions. There are about 437 square miles of inholdings outside the PCA with varying potential for increases in motorized access.
2 Roadless figures derived from the area managed as roadless under the Management Prescriptions in the 1997 Revised Targhee Forest Plan. Inventoried roadless areas were used as the basis on other administrative units.

Appendix H. Annual Cost Estimates by Agency for Implementing this Conservation Strategy

Task	YNP	USGS BRD - IGBST	Wyoming	Montana	Idaho	USFS	FWS LE	GTNP	TOTAL
Annual GIS layer updates[5]	5,000	-	-	-	-	95,000	-	2,000	102,000
GIS run for Habitat Effectiveness calculation (CEM)[5]	5,000	-	-	-	-	23,000	-	2,000	30,000
GIS run for secure habitat/subunit[5] ; TMARD >1 mi/sq mi/subunit[5]; OMARD > 2 mi/sq mi/subunit[5]	12,000	-	-	-	-	23,000	-	8,000	43,000
Monitor developed sites and livestock grazing						2,000			
Monitor hunter numbers			1,000	1,000	1,000				
Cutthroat trout spawners (Kokanee – Idaho)	7,000	14,590	-	-	1,250	_	-	-	22,840
Spring carcass surveys	12,000	29,210	-	2,000	-	4,000	-	-	47,210
Whitebark cone transects	2,500	48,703	1,000	1,500	-	2,000	-	-	55,703
Moth presence	-	43,830	3,500	3,500	-	-	-	4,000	54,830
Private land status	-	-	1,500	10,000	1,100	-	-	-	12,600
Monitoring unduplicated females w/cubs	15,000	87,584	29,150	35,000	4,000	-	-	4,000	174,734
Mortality	1,000	38,858	11,000	20,000	700	-	20,000	2,000	93,558
Distribution of family groups	7,500	35,792	7,500	10,000	5,000	-	-	-	65,792
Maintaining 25 adult females w/radios	10,000	187,859	80,750	60,000	5,500	-	-	-	344,109
Human/bear conflict mgt.	600,000[1]	-	593,000	229,000[4]	14,700	650,000[6]	-	100,000[1]	2,186,700
Outreach and education	20,000	-	10,000	30,000	68,500	60,000	10,000	8,000	206,500
Monitor genetic variation[2]	-	8,811	-	10,000	2,500	-	-	-	21,311
Miscellaneous		10,000			18,500				28,500
Total per agency per year	697,000	505,237[3]	738,000[1]	412,000	124,700	859,000	30,000	130,000	-
TOTAL COST PER YEAR									**3,496,337**
Total new cost per year	151,800	10,000	-	188,000	10,000	600,000	-		

[1] This cost is a projected ongoing cost and is not specific to the Conservation Strategy.
[2] This cost will be covered through USFWS monitoring of possible changes in genetic variation in all lower 48 grizzly populations.
[3] Much of this cost is in current IGBST operations.
[4] $35,000 is currently being spent.
[5] These are new costs to manage habitat but are already required as per the Recovery Plan.
[6] 30% currently funded; 70% currently needed but unfunded and are currently necessary to minimize bear-human conflicts as per the Recovery Plan.

Appendix I. Lead Agencies for Actions under this Conservation Strategy

AGENCY LEADS AND PARTICIPANT AGENCIES HABITAT AND POPULATION MONITORING				
TASK	**LEAD AGENCY**	**PARTICIPANT GENCIES**	**TASK LEADER**	**ANNUAL REPORT LEADER**
Habitat Effectiveness (GIS run and database updates)	USFS	YNP,GTNP	USFS	USFS
Secure Habitat/OMARD and TMARD (GIS runs and database updates)	USFS	YNP,GTNP	USFS	USFS
Cutthroat trout spawners	YNP	IGBST	YNP	YNP
Spring carcass surveys	YNP	IGBST	YNP	YNP
Whitebark cone transects	IGBST	YNP,USFS	IGBST	IGBST
Moth presence	WY	YNP, GTNP, IGBST	IGBST/WY	IGBST/WY
Mortality reduction	WY, MT, ID, NPS, USFS, FWS/LE	WY, MT, ID, NPS, USFS, FWS/LE	Cooperative	Cooperative
Developed Sites and Livestock Grazing	USFS	NPS	USFS	IGBST
Hunter Numbers	WY,ID, MT	WY,ID, MT	WY	IGBST

AGENCY LEADS AND PARTICIPANT AGENCIES HABITAT AND POPULATION MONITORING				
TASK	**LEAD AGENCY**	**PARTICIPANT AGENCIES**	**TASK LEADER**	**ANNUAL REPORT LEADER**
Private land status	Private conservation groups in cooperation with states	WY,ID, MT	To be selected	To be selected
Unduplicated females w/cubs	IGBST	WY,YNP,MT,ID,GTNP	IGBST	IGBST
Mortality	IGBST	MT,WY,ID,YNP,GTNP, FWS/LE	IGBST	IGBST
Distribution	IGBST	WY,YNP,MT,ID,GTNP	IGBST	IGBST
Maintaining 25 adult females with collars	IGBST	WY,YNP,MT,ID,GTNP	IGBST	IGBST
Monitoring genetic diversity	IGBST	IGBST and USFWS	IGBST	IGBST
Control action and conflict reporting	YNP	WY,YNP,MT,ID,GTNP	YNP	YNP/IGBST
Public outreach and information	All	WY,YNP,MT,ID,GTNP,USFS, FWS/LE	To be selected	To be selected

Appendix J. The Relationship Between the Five Factors in Section 4(a)(1) of the ESA and the Existing Laws and Authorities

The relationship between the five factors in Section 4(a)(1) of the Endangered Species Act and the existing State and Federal laws and regulations is important to assure that the existing laws and authorities can address all the factors necessary to assure recovery under the Endangered Species Act. This table presents the State and Federal laws and authorities and which of the five factors are addressed by that law or authority.

Sec. 4. (A) General. - (1) The Secretary shall by regulation promulgated in accordance with subsection (b) determine whether any species is an endangered species or a threatened species because of any of the following factors:

A. the present or threatened destruction, modification, or curtailment of its habitat or range;

B. overutilization for commercial, recreational, scientific, or educational purposes;

C. disease or predation;

D. the inadequacy of existing regulatory mechanisms;

e) other natural or manmade factors affecting its continued existence.

FEDERAL AND STATE LAWS AND REGULATIONS	Five Factors				
	A	**B**	**C**	**D**	**E**
The Act of Congress March 1, 1872 - Set Yellowstone National Park as a Public Park	X	X		X	X
National Park Service Organic Act of 1916, 16 U.S.C. 1, 39 Stat. 535	X	X		X	X
Lacey Act of 1900, as amended, 16 U.S.C. 701, 702; 31 Stat. 187, 32 Stat. 285; Criminal Code Provisions, as amended, 18 U.S.C. 42-44, 62 Stat. 87				X	
Fish & Wildlife Coordination Act of 1934, as amended, 16 U.S.C. 661-666c; 48 Stat.401	X	X		X	X
The Act of Congress September 14, 1950 - Expansion of Grand Teton National Park to include Jackson Hole National Monument	X			X	
Sikes Act, 1960, as amended, 16U.S.C. 670a-670o; 74 Stat. 1052, Pub. L. 86-797	X	X			X
Multiple-Use Sustained-Yield Act of 1960, 16 U.S.C. 528-531, 74 Stat. 215, P.L. 86-517	X	X			X
National Environmental Policy Act of 1969, as amended, 42 U.S.C. 4321, 83 Stat. 852, Pub. L. 91-190	X	X			X
The Act of Congress August 25, 1972 - Establish John D. Rockefeller, Jr. Memorial Parkway	X	X			
Endangered Species Act of 1973, as amended, 16 U.S.C. 1531-1543; 87 Stat. 884	X	X	X	X	X
Forest and Rangeland Renewable Resources Planning Act, 1974, Pub. L. 93-378	X	X		X	X
National Forest Management Act of 1976, U.S.C. 1600 et. seq., Pub. L. 94-588	X	X			X

FEDERAL AND STATE LAWS AND REGULATIONS	Five Factors				
	A	**B**	**C**	**D**	**E**
Federal Land Policy and Management Act of 1976, as amended,43 U.S.C. 1701 et. seq., Pub. L. 94-579, 90 Stat. 2744		X			X
Fish & Wildlife Improvement Act of 1978, 16 U.S.C. 742 l, 92 Stat. 3110				X	
Fish and Wildlife Conservation Act of 1980, 16 U.S.C. 2901-2904; 2905-2911; 94 Stat. 1322, Pub. L. 96-366	X	X		X	X
36 CFR 1.5 (a)(1)		X		X	
36 CFR 1.7(b) and 2.10(d)				X	X
36 CFR 1.7(b) and 7.13 (l)		X		X	X
36 CFR 2.2		X		X	X
36 CFR 2.10				X	X
36 CFR 219		X			X
36 CFR 219.19	X			X	
36 CFR 219.27 (a)(6)	X			X	X
36 CFR 261.50 (a), (b) and (c)				X	X
36 CFR 261.53 (a) and (e)				X	X
36 CFR 261.58 (e), (s) and (cc)				X	X
WYOMING STATE STATUTES					
23-1-101 (a)(xii)				X	
23-1-103		X		X	
23-1-302 (a)(ii)		X		X	
23-1-901					X
23-2-101 (e)				X	X
23-2-303 (d)				X	X
23-3-102 (b)		X		X	
23-3-103 (a) & (b)		X		X	X
23-3-106				X	X
23-3-107		X		X	X
23-3-109		X		X	X
23-3-112		X		X	X
23-3-301				X	X

FEDERAL AND STATE LAWS AND REGULATIONS	Five Factors				
	A	**B**	**C**	**D**	**E**
WYOMING GAME AND FISH COMMISSION REGULATIONS					
Chapter XLIII		X		X	X
Chapter XXVIII		X		X	X
Chapter III		X		X	X
IDAHO STATE STATUTES					
36-103 (a)		X		X	X
36-103 (b)				X	X
36-201				X	X
36-716		X		X	X
IDAHO FISH AND GAME COMMISSION REGULATIONS					
IDAPA 13 G 1.9		X		X	X
IDAPA 13 G 2.2				X	
MONTANA STATE STATUTES					
Section 87-1-301	X	X		X	X
Section 87-5-301	X	X		X	X
Section 87-5-302		X		X	
Section 87-2-101		X		X	X
ADMINISTRATIVE RULES OF MONTANA					
MCA 12.9.103 Grizzly Bear Policy (1)	X	X	X	X	X
MONTANA DEPARTMENT OF STATE LANDS					
Title 75, Chapter 1 MCA - Montana Environmental Policy Act	X				
Title 76, Chapter 14, MCA - Montana Rangeland Resource Act	X				
Title 77, Chapter 1 MCA - Administration of State Lands	X				X
Title 87, Chapter 5, MCA - Nongame and Endangered Species Conservation Act	X			X	X
Montana Constitution. Article IX - Environment and Natural Resources. Section 1 - Protection and Improvement	X				
Montana Constitution. Article X - Education and Public Lands. Section 4 - Board of Land Commissioners.	X				

FEDERAL AND STATE LAWS AND REGULATIONS	Five Factors				
	A	**B**	**C**	**D**	**E**
FEDERAL PLANS AND GUIDELINES - NATIONAL PARK SERVICE					
NPS-77, Natural Resource Management Guidelines, May 16, 1991		X			X
Final Environmental Impact Statement, Grizzly Bear Management Program, Yellowstone National Park, July, 1983	X	X	X	X	X
Yellowstone National Park Annual Bear Management Plan		X			X
Grand Teton National Park Human/bear Management Plan, 1989	X	X	X	X	X
U.S. FOREST SERVICE (Regions 1,2, and 4)				X	
Beaverhead NF Plan (1986) Deerlodge NF Plan (1987)	X			X	X
Bridger-Teton NF Land and Resource Management Plan (1989)	X		X	X	X
Custer NF and Grasslands Land Resource Management Plan (1987)	X		X		
Gallatin NF Plan (1987)	X		X	X	X
Shoshone NF Land and Resource Management Plan (1986)	X		X	X	
1997 Revised Forest Plan - Targhee National Forest	X		X	X	X
OTHER GUIDANCE					
Grizzly Bear Compendium. National Wildlife Federation, Washington, D.C. 1987					X
Interagency Grizzly Bear Committee Taskforce Report, Grizzly Bear/Motorized Access Management. 1994. Revised 1998.				X	
Yellowstone Grizzly Bear Investigations				X	X
Public Information and Involvement Strategy for IGBC.				X	X