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Comments on Proposed Rule and Conservation Strategy Regarding Delisting Yellowstone Grizzly Bears

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Lana Ciarniello, Aklak Wildlife Consulting, aklak@telus.net
Co-chair Human-bear Conflicts Expert Team, IUCN Bear Specialist Group

David Garshelis, Minnesota Department of Natural Resources, dave.garshelis@state.mn.us
Co-chair IUCN Bear Specialist Group

Bruce N. McLellan, B.C. Ministry of Forests, Lands, and Natural Resource Operations, bruce.mclellan@gov.bc.ca
Red List Authority IUCN Bear Specialist Group; Past Co-chair Bear Specialist Group; Past President IBA

Karyn Rode, U.S. Geological Survey, Alaska Science Center, krode@usgs.gov
Polar Bear Specialist Group

We were asked by the International Association for Bear Research and Management (IBA) to provide a science-based review of the US Fish and Wildlife Service's proposed rule (Federal Register 81 [48], March 11, 2016) (henceforth 'rule') to delist grizzly bears (*Ursus arctos horribilis*) in the Greater Yellowstone Ecosystem (GYE) as a distinct population segment (DPS) under the U.S. Endangered Species Act, and the Draft 2016 Conservation Strategy (CS) that outlines what would follow after such delisting. The objective of this review is to critically evaluate the science, the interpretation, and the proposed monitoring and management practices as they relate to the security and sustainability of the grizzly bear population in the Yellowstone ecosystem. We recognize that a sustainable grizzly bear population can occur under federal (listed) or state (delisted) management, and our intention is not to endorse or oppose delisting, but rather to ensure decisions are based on sound science. As a group of scientists with considerable experience in bear ecology, nutrition, demography, and management, we respectfully submit these comments as a critical evaluation of the scientific findings upon which the rule and CS are

based. None of us have any direct relationship with the politics, science, or activities related to the delisting, and so we hope that our comments are taken as objective, with the caveat that we may not understand all of the intricacies of the data. We would like to mention, in that regard, that the data are extensive, and so took considerable effort to understand, while much of the writing in the rule and CS is quite verbose, repetitive, and not organized in such a way as to lend to easy interpretation. We thus relied to a large degree on published papers and our own experiences in the field.

For delisting, the USFWS has established habitat-based recovery criteria and 3 demographic recovery criteria. These have been updated and revised multiple times, based on new understanding of this population and ecosystem. The habitat-based recovery criteria have the goal of maintaining or improving habitat conditions at levels that existed in 1998 when conditions allowed the population to expand. These criteria are based on measures of motorized access levels, areas of secure habitat, number of developed sites, and livestock allotments. All of these measures have been met. However, future management will need to involve careful monitoring of continued changes in food resources. This is particularly important in light of the vast uncertainty that surrounds the effects of climate change relative to the distribution and abundance of bear foods as well as the response by bears to the changing landscape. Therefore, while the quantity of developed sites and the amount of secure habitat may remain at levels similar to 1998 it is unlikely that the habitat remains in the same condition as it was in 1998. An example of this change may be best exemplified with whitebark pine (*Pinus albicuaulis*), which not only continues to die due to white pine blister rust fungus (*Cronartium ribicola*) and the mountain pine beetle (*Dendroctonus ponderosae*) epidemics, but is also being successionally replaced by other tree species, particularly subalpine fir (*Abies lasiocarpa*)(Keane and Arno 1993).

The 3 demographic recovery criteria relate to population size, distribution, and stability. Specifically these are: 1) maintain a population size of at least 500 bears and at least 48 females with cubs in the demographic monitoring area (DMA), calculated by the most updated protocol; 2) ensure a broad geographic distribution of reproducing female bears by confirming occupancy of females with young in at least 16 of the 18 bear management units (with no 2 adjacent units unoccupied over a 6-year period); and 3) maintain the population around the 2002-2014 average of 674 bears (95% CI 600 – 757) through control of mortality within specified limits.

We agree that, for more than a decade, all of these habitat and demographic criteria have been met. We commend the enormous effort over the past 40 years that the various US agencies and US citizens have put into recovering grizzly bears in the GYE; yours has been one of the most successful programs for bears world-wide. Having a population increase from perhaps 230-300 bears in the early 1970s to over 700 (2–3% per year) is a substantial achievement and, in comparison to 40 years ago, must seem to local researchers and managers like there are a lot of grizzly bears. In truth, however, ≈700 bears in an isolated region is not a large population. The IUCN rates populations threatened with extinction as vulnerable, endangered, or critically endangered, using explicit criteria related to population size, growth, and degree of isolation. Recently, the IUCN Bear Specialist Group categorized the level of risk for all isolated brown bear populations worldwide (44 populations in North America, Europe, and Asia) using the IUCN Redlist Criteria. The GYE was categorized as Vulnerable (using criteria D1: 250–1000 mature individuals), meaning that it is threatened of becoming endangered; if the population drops to

under 250 mature individuals (533 total population for GYE grizzly bears given present estimated composition), this population would be categorized as endangered by IUCN. Although we trust that the states of Montana, Wyoming, and Idaho, and the Federal agencies that will remain responsible for managing bears and their habitat will continue to do an excellent job of management, we want to stress that the grizzly bears in the GYE still form a small, isolated population, and must continue to be managed with great care.

The Fish and Wildlife Service (Federal Register 70 [221], Sept. 17, 2005) argued convincingly that the grizzly bears in the GYE were a DPS in part because of their discreteness (isolation and genetic differences), but also because they are ecologically and genetically "significant" or have a significant evolutionary legacy. Under the DPS policy, if delisted from the ESA, they should be as if they were their own species or, in this case, the last Yellowstone grizzly bears on earth.

Below we outline issues that pertain to managing such a small and isolated population. We underscore a number of continued risks moving forward that generally fall under the broad issue of uncertainty. We believe that a number of looming uncertainties exist in terms of changing food supplies, methods of population size and trend monitoring, and employing the associated mortality limits. We list these under subheadings of specific issues, and provide some recommendations.

Issue: Uncertain implications of measured declines of important grizzly bear foods

Cutthroat trout (*Onchohynchus clarki*), whitebark pine seeds, elk (*Cervus elaphus*), and army cutworm moths (*Euxoa auxiliaris*) have been identified as high quality forage for grizzly bears in the GYE. Cutthroat trout, whitebark pine, and elk are in decline due to various factors including competition with or predation by other species, disease and insect outbreaks, and/or climate change.

Although high-quality grizzly bear foods have declined, data on bear body condition and diet suggest bears have a flexible diet and have used alternative food sources to meet energetic requirements. Schwartz et al. (2014) and IGBST (2013) found no difference in the body condition of male or female bears during poor vs. good pine seed production years. The IGBC (2013:35) concluded that the decline in whitebark pine seeds "has had no profound negative effects on grizzly bears at the individual or population level." Bears that previously consumed cutthroat trout appear to now eat elk calves (Felicetti et al. 2003, Fortin et al. 2012, Ebinger et al. 2016). During years with poor whitebark pine seed production, bears appear to consume more meat (Felicetti et al. 2003, Schwartz et al. 2014) and false truffles (*Rhizopogon* spp.)(Fortin et al. 2012). van Manen et al. (2016) suggested that the population trend has recently stabilized due to density-dependent processes, including probable intraspecific killing.

Grizzly bears in Yellowstone National Park continue to adjust to changing food resources (Fortin et al. 2012) and available data have limitations in assessing effects. For example, some elk herds that use the GYE have declined as grizzly bear, wolf and human predation increased (IGBST 2013, Fortin et al. 2012). Elk appear to be the primary food replacing declining cutthroat trout (Fortin et al. 2012) and whitebark pine (Schwartz et al. 2014). Schwartz et al. (2014) documented no declines in female body mass but a decline in percent body fat that reached levels known to cause reproductive failure (i.e., <20%; Robbins et al. 2012). With additional years of data, IGBST (2013) found no decline in percent body fat, but

acknowledged important limitations with sample size and seasonal variability of data collection. Also, they did not consider the reproductive state of females in their analysis. Moreover, we observed that their data (IGBST 2013: Fig. 4) show female bears with exceedingly low levels of body fat in fall only since 2008, when loss of whitebark pine has been most severe.

Separating density-independent from density-dependent effects on population growth is acknowledged in the rule as being challenging. van Manen et al. (2016) concluded that the decline in population growth in recent years was associated more with increasing grizzly bear density than whitebark pine decline. This finding, though, cannot discount the potential role of whitebark pine seed production in demographics of grizzly bears in this ecosystem. The direct relationship between whitebark pine production and survival rates of independent bears is well documented (Blanchard and Knight 1995, Mattson 1998, Schwartz et al. 2006). This relationship is driven mainly by increased human-bear conflicts during years of poor whitebark pine production (Schwartz et al. 2006). This effect continues to date (IGBST 2013). It is also possible that social interactions change in complex ways with changing food conditions, with male bears sequestering some key food resources, and putting younger bears (especially dependent offspring) at increased risk (Garshelis and Pelton 1981, Ditmer et al. 2016); this may not be detectable from an analysis of body condition or the type of modelling conducted by van Manen et al. (2016). The challenges of quantifying mechanisms that influenced changes in population growth rate are significant. Based on the cumulative body of literature on nutrition and demography of bears in North America (e.g., Elowe and Dodge 1989, Costello et al. 2003, Obbard and Howe 2008, McLellan 2015), we believe that declining food resources must have played some role in recent reductions in population growth and is an important factor warranting consideration in management and monitoring in the future.

We recommend that the minimum size of core security areas be increased to allow for dietary flexibility required to respond to continually changing food resources. The minimum size of core security areas is typically set as approximately 10-km^2 (1,000 ha), because that size takes into account the size of average daily movement for an adult female grizzly bear (Gibeau et al. 2001). It should be at least 500 m from high human use defined as >100 human visits/month, not just an “open or gated motorized access route”.

Issue: Uncertain implications of climate change

The impact of climate change on grizzly bears is highly uncertain because bears are opportunistic omnivores with a diverse diet (Roberts et al. 2014). Whitebark pine is a high elevation species currently being negatively affected by climate change. In years of low availability of whitebark pine seeds, bears seek alternate foods usually at lower elevations where they are more likely to encounter humans. Grizzly bear deaths depend on type and frequency of contact between bears and humans (Mattson et al. 1996) and habitats that attract bears to high-risk areas can result in local population sinks (Delibes et al. 2001, Naves et al. 2003, Schwartz et al. 2010). For grizzly bears inhabiting the GYE, survival was lowest in the fall corresponding with the highest incidence of human-bear conflicts (Schwartz et al. 2006). The ability of grizzly bears to adapt to climate change via use of alternative foods can be misleading if those habitats also hold a higher risk of human-caused mortality.

In light of recently observed declines in some food resources and uncertain effects of climate change, we recommend continued monitoring of the relationship between the availability and use of key food resources and the vital rates and body condition of grizzly bears (such monitoring does not appear to be part of the CS). We also recommend a more thorough examination of the uncertainty surrounding the effects of climate change on the loss, redistribution, and potential change in grizzly bear foods (including army cutworm moths) and associated effects on bears attempting to access high quality food sources.

Issue: Limitations of methods for estimating population size and trend

The IGBST developed an index of bear density, based on counts of unique females with cubs of the year (FCOY), to monitor the recovery of the Yellowstone grizzly bear population. The premise was that FCOY were spread out enough that FCOY individuals could be distinguished using a combination of litter size, distance between sightings (and intervening barriers to movements), and days between sightings. The process for doing so was formalized by Knight et al. (1995), and has come to be known as the Knight rule set. (Different rule sets have been proposed for counts of FCOY brown bears in Europe; Ordiz et al. 2007).

Application of the Knight rule set provides an estimate of the minimum number of FCOY per year and estimates of frequencies of sightings of each known FCOY. This is converted to an estimate of total FCOY (i.e., including those not observed) using the Chao2 estimator (Chao 1989, Cherry et al. 2007).

Schwartz et al. (2008) investigated the important assumption that changes in density do not affect the ability to distinguish FCOY. They found that at higher densities, with more overlap among FCOY, it became increasingly difficult to distinguish individuals based on the Knight rule set. Hence, the Chao2 estimator becomes increasingly negatively-biased with increasing density. As such, Chao2 estimates could level off while the population continues to increase, giving a false sense of the population reaching carrying capacity (K). Likewise, once the population has exceeded the density threshold of FCOY that precludes further differentiation of distinct individuals, a decline also would not be detectable until dropping below this threshold.

If the Choa2 estimates are increasingly biased low, then mortality limits would also be biased low, allowing for the population to increase. Hence, the bias in the Chao2 is considered to be “conservative”. It is conservative, though, only when the population is indeed increasing. If vital rates or mortalities are mis-estimated, the population could also decline, undetected. Thus, once a density threshold of population density has been exceeded, it is difficult to accurately assess population size and trend using the Chao2 estimator.

An additional important potential limitation of the Chao2 estimator is that it might become even more unreliable with increased mortality of cubs, as has apparently occurred (van Manen et al. 2016). This higher mortality could lead to more confusion in distinguishing FCOY (because litter sizes are changing and because more FCOY have similar litter sizes) and more errors in tabulating sighting frequencies used in the Chao2 estimator.

A series of workshops conducted by the IGBST including a contracted biometrician tried to rectify this problem (IGBST 2012) and concluded that it is not possible to rely on FCOY observations as a trend indicator at the current high densities. Thus, continued reliance of FCOY estimates for trend monitoring and estimation of total population size and composition (through matrix modelling of vital rates) is problematic and has important implications on the ability to accurately set sustainable mortality limits.

Problems with the Chao2 estimator spurred the IGBST to seek an alternative method for monitoring this population (IGBST 2012). The most promising seems to be a mark-resight method that uses observations made during search flights for FCOYs, and checking (after sighting them) for whether they are marked (collared) using a telemetry receiver. An advantage of this method is that unlike the present Chao2, it is unbiased with respect to bear density. Also, data are available back to 1997 (with 2 search flights per year), so provide an indication of trend since then. Unfortunately, these data are rather sparse, so the precision is low (Higgs et al. 2013). That issue could be addressed, though, with larger samples of collared bears and/or more search flights. Whereas the IGBST clearly sees the value in pursuing and enhancing this alternative method (Haroldson and van Manen 2014), the sole reliance on Chao2 for monitoring this population has an associated risk of bias and inaccuracy that is not acknowledged in the Rule or Conservation Strategy (CS).

In sum, the Chao2 estimator is likely no longer the "best available science" for continued monitoring of this population. We agree that this method has served well to demonstrate the strong recovery of this population, but it is likely not the best method for continued monitoring to (a) discern whether the population is truly at K, and (b) to derive limits of allowable mortality. We recommend an eventual conversion to mark–resight methodology, with a management system adaptable enough to accommodate the inevitable changes in population estimates that this will entail.

Issue: Biological and management challenges of maintaining a population at a specified size

A stated goal of the rule is to try to maintain a total population of 674 grizzly bears in the demographic monitoring area (based on the Chao2 estimate of FCOY, extrapolated to the full population using data on vital rates). We believe this is unrealistic for a population in an ecosystem in which food resources vary year to year and for which estimates of population size and trend are inexact (and still being modified — see below). The confidence intervals of Chao2-based population estimates are large. For example, the 2014 Chao2 estimate had a 95%CI that ranged from 588–721 (Haroldson and van Manen 2014), making it quite uncertain whether the true population was above or below the target of 674. Additionally, there are many sources of variation not included in the CIs (IGBST 2012: Table 2.1), so the actual CIs are even larger.

Second, the population may already be above 674. Telemetry data indicated that survival rates have changed, and when these new rates are applied (to convert Chao2 estimates of FCOYs to estimate the total population size), the "updated" estimates are much larger. The updated estimates for the most recent years show a much higher number of males and slightly lower number of females than estimates calculated with the older survival rates (Haroldson and van Manen 2014). This not only begs the question: which is right, but also demonstrates the considerable uncertainty in the monitoring system.

One of the most ironic outcomes is that the "updated" estimates put the population into the highest category of allowable mortality (10% for females, for a population exceeding 747), whereas the non-updated estimates for the same year put the population in the lowest allowable mortality category (<7.6% for females) — that is, although the estimated number of females was *lower* in the updated estimates, the allowable mortality for females was higher because the population had more males, which does not make biological sense.

Setting a goal of maintaining a population at a certain numerical size is not only impractical, but seems counter to other stated conservation strategies of allowing the population to fill all suitable habitat, and even expand to connect to the NCDE and possibly the Bitterroots, the latter of which is a significant and important component of protecting grizzly bears as a species. There seems to be evidence that the population is still expanding geographically, and may still be increasing numerically: vital rates suggest that the current growth rate is 0–2% (based on 2 different assumptions [IGBST 2012]; the true CI is much wider; see Harris et al. 2007, IGBST 2012: Table 2.1). If the population is indeed still increasing, this further challenges the ability to maintain it at its present level: in other words, *is the goal to prevent a further increase even if it is increasing now, or is the goal to maintain the current numbers because the population is believed to be relatively stable?* The rule and CS do not make this distinction clear. We do not believe that either strategy is in the best interest of conserving this population (further discussion below).

Issue: Sustainable mortality rates may be over-estimated

The rule suggests that a total mortality rate (natural mortalities, plus known and unknown human-caused mortalities [HCM]) of 7.6% for independent female bears would maintain a long-term average population of 674 bears (the average population during 2002–2014). This rate was estimated by: 1) field methods and survival rate estimation following Schwartz et al. (2006), 2) reproductive rate estimates based on transition probabilities (ground-based capture in the GYE has led to biased capture with respect to female reproductive states plus short-term monitoring of individuals) described by Schwartz and White (2008), and 3) adjustments in reproductive rates to account for mortality of cubs before they were first observed, following Mace et al. (2012). These complex results were used in a matrix projection model and independent female survival was varied until population growth was stable.

We have two concerns over the use of the 7.6% total female mortality limits (and even higher limits for higher populations). First is the lack of confidence limits on the estimate. Given the large CIs on the survival estimates of each sex-age group and non-age-specific reproductive rate (IGBST 2012), the estimated mortality limit must have wide uncertainty.

Second are potential problems with how these vital rates were combined into the projection model. We used the data from the IGBST report (2012: Table 4.1) in a projection matrix and obtained the same 7.6% sustainable female mortality rate, provided all females aged 4–30 years maintained an average reproductive rate (0.336). Although the estimation of stable states may have been based on transitions of immature animals and old bears in reproductive senescence, we believe using this static reproductive rate for females of all of these ages (i.e. 4 to 30) may not only add considerable uncertainty, but may

have caused the sustainable mortality to be over-estimated. Schwartz et al. (2006, p. 19) found that during 1983–2001, when the population was below carrying capacity and increasing rapidly, the mean age of first reproduction was 5.8 years (95%CI: 5.6–6.3) and age was an important predictor of reproductive output (Schwartz and White 2008). Prime reproductive age is 10–15 years old, followed by declining reproduction through 25 years of age when reproduction usually stops in these southern grizzly bear populations (Schwartz et al. 2003, Schwartz and White 2008, McLellan 2015).

If average reproduction spans ages 6–26 years (Schwartz et al. 2006), then independent females could only sustain a 5.4% mortality rate. However, the average age of first litter may have increased as the population supposedly approached carrying capacity so this rate may be even lower. For example, if the mean age of first litter increased to 7 years, then the sustainable rate would be only 4.7%. If cub and yearling survival have declined to the extent presented by van Manen et al. (2016, Fig. 3)(i.e., 0.4 and 0.3 instead of the 0.553 and 0.539 used in the IGBST [2012] report), then female survival would have to be 100% through senescence to maintain a stable population.

We recommend a re-consideration of these mortality limits in light of the above concerns combined with known declines (and further projected declines) in survival of dependent young, and probable instability of the age structure caused by institution of a hunt (thus likely violating a key assumption of the matrix modelling upon which the allowable mortality rates are calculated).

Issue: Uncertainty in the estimates of annual mortality

There is considerable uncertainty in the number of bears that die of natural causes and that are killed by people but not reported. In 2013 and 2014, for example, estimated unreported losses formed a significant contribution to the number of deaths in the total mortality. With data from 1983 to 2011, Schwartz et al. (2006) did not document any independent radio-collared females dying of natural causes although 3 died of undetermined causes. Natural deaths are uncommon, but other studies towards the southern edge of the grizzly bear distribution have found natural mortalities of females (Garshelis et al. 2005, Mace et al. 2012, McLellan 2015) and Schwartz et al. (2006) did document 5 cases of males dying of natural causes. Bears do sometimes kill one another, they die in avalanches, rock slides, collapsed dens, and some starve. Recently, some natural deaths of females in the GYE have been documented (van Manen 2016). Without telemetry, most of these deaths go unrecorded and must be estimated using rates of collared bears, which adds another layer of uncertainty.

The total number of human-caused deaths (reported and unreported) was estimated using a Bayesian approach. This method resulted in broad credible intervals that depend on different priors that in turn, depend on how the unexplained and unresolved cases are dealt with (Cherry et al. 2002). For example, depending on which of the 3 priors are chosen for years 1997-1999 (typical years), the 95% posterior probability intervals could vary from 16 to 78 bears killed by people when “possible” cases are included. When possible cases are excluded, the number could vary from 12 to 64. In IGBST annual reports, a point estimate is used and the credible intervals are not shown. Natural deaths are not included in this estimation method (Cherry et al. 2002).

Unknown mortalities are an inherent part of any management system, but we saw little recognition in the rule or CS of the effects of 3 intersecting uncertainties that should be considered in establishing quotas on discretionary killing: (1) the unknown number of bears that died of other causes, (2) the true sustainable mortality rate, given the uncertainty in the vital rates, and (3) the true population size.

Issue: High mortality rates above targeted population size are risky

Allowable female mortality rates of 9% for total populations 675–747 and 10% for populations >747 would not be sustainable, especially with increasing density dependent forces (van Manen et al. 2016). We interpret these high allowable rates as a purposeful attempt to reduce the population to achieve the target of 674 bears if it is found to exceed that number. However, this is never explicitly stated, and the language in the rule and CS seem not to be consistent as to whether the aim is to reduce the population if it exceeds 674.

Given the uncertainties in the estimation of sustainable mortality, uncertainties in estimation of population size and trend, and lack of understanding of the density dependent factors at play as well as effects of changing food conditions (discussed below), we believe it is risky to manage for a purposeful reduction in this population.

Issue: Managing with uncertainty calls for conservatism

An important question that grizzly bear managers must answer is how confident they want to be that the number of bears being killed by people is sustainable. Using "best estimates" for population size, vital rates used in the matrix model, unknown HCM, and natural deaths, as appears to be the case in this proposed rule, would mean that there would be a 50% chance that the mortality limits would be unsustainable. Given the breadth of all confidence limits, there is a reasonable chance that the mortality limits might be far from sustainable and may cause the population to decline. To manage grizzly bear populations more conservatively (preferring to err on the side of an increase rather than a decline), other management agencies do not set mortality limits at "best estimates", but rather nearer the lower confidence limit. For example, British Columbia (BC) has > 20 times as many grizzly bears over an area 16 times as extensive as the GYE, and many BC bears are in rich ecosystems with many fruit species, ungulates, and salmon. BC's bears are also fully connected with bears in Alaska, the Yukon, Alberta, and Northwest Territories. In hunted populations, BC has a total HCM annual limit (recorded including hunting, human-bear conflicts, road or train kills, and estimated unrecorded kills) of 4-6% and only 30% of these can be females. In threatened BC populations, any human-caused mortality is opposed to the population recovery objective so there is no legal hunting. In areas where there is higher uncertainty around population estimates, BC managers typically choose total HCM limits lower than maximums. The limit used in BC, although the population is much larger and more secure, is far more conservative than the 7.6% and 15% for female and male bears suggested in the proposed rule for managing GYE.

We recommend that a more conservative approach to harvesting, analogous to that used in BC, be used in the GYE.

Issue: Population target seems counter to the goal of population expansion and reconnection

Large population size and connection with neighboring populations should be a prime strategy for long-term population persistence of grizzly bears, particularly during a time of unpredictable ecological conditions stemming from climatic change. However, the overall thrust of the rule appears to purposefully limit both population growth and dispersion. The rationale for this decision is not clear, and seems counter to the overall objective of ensuring the persistence of Yellowstone grizzly bears post-delisting. Even without the documented density-dependent decline in cub and yearling survival and reproductive output (van Manen et al. 2016), revised population estimates (Haroldson and van Manen 2014) will result in high and unsustainable mortality rates, which will reduce population size and likely preclude expansion. We believe that continuing to encourage population grown, expansion, and ultimately reoccupying the Bitterroot Ecosystem (BE) and connecting with the Northern Continental Divide Ecosystem (NCDE) should be the long-term goal.

The rule (p 13192) suggests that future connectivity is desirable and will be actively managed for, but tying increasing mortality limits to increasing population size does just the opposite. We suggest a re-examination of the apparent discord between the high, unsustainable mortality limits when the population exceeds the target of 674, versus encouraging the population to expand westward (towards the BE) and northward (towards the NCDE). Having spatially-explicit targets for this population would be better, we believe, than exact numerical targets. For example, mortality limits could be higher in the southern and eastern portions of the GYE than to the west and north. Moreover, habitat to the north and west of the DMA should be managed in such a way as to encourage population expansion. We acknowledge that such expansion would occur within the constraints of human-bear conflicts (i.e., social carrying capacity), but in the face of a changing biological carrying capacity, it would seem prudent to view the social carrying capacity as also adaptable in the long term.

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