The Grizzly Bear Recovery Project

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Effects of the Proposed Black Ram Project on Yaak Grizzly Bears

Comments on the Black Ram Environmental Assessment

The assessment of prospective effects of the Black Ram Project on grizzly bears in the Environmental Assessment (EA) is premised on several critical assumptions. First, status of the Cabinet-Yaak grizzly bear population is assumed to have improved since 2012. Second, and related, the EA assumes that some erosion of security for grizzly bears is therefore permissible, conditioned on a related assumption that security and road access standards employed by the Kootenai National Forest (NF) are sufficient for recovery of grizzly bears in this ecosystem. Third, the EA assumes that abundance of huckleberries are demographically limiting for grizzly bears in this region. And, fourth, the EA further assumes that Project treatments will substantially enhance abundance of huckleberries to an extent sufficient to offset any losses of habitat security.

All of these assumptions are unwarranted and, as a result, the conclusions reached in the EA about Project effects on grizzly bears are likewise unwarranted. More problematic yet, all of the conclusions of relevance to grizzly bears likely under-estimate negative impacts and over-estimate prospective benefits. Briefly:

- The weight of available evidence does not support concluding that population status has improved. For one, the methods used to estimate trend and current population size are beset with a host of problems. For another, the information able to be distilled from demographic data suggests that any improvement has stalled since 2014.

- Variations in population size and trajectory between 1999 and 2010 are more likely attributable to variations in abundance of natural foods—berries in particular—that affect exposure of bears to humans rather than to any increased mitigations. During years of scant berries, bears likely forage more widely and more often end up in conflict situations or exposed to malicious killing.

- The population of grizzly bears in the Yaak/Yahk is far smaller than even the smallest size posited to be viable by any researcher. Related, the population remains acutely vulnerable to even the smallest increases in bear mortality that are predictably more likely to occur with any increase in road access and associated human activity.

- Malicious and other unjustified killing by humans remains the dominant cause of death for grizzly bears in the Cabinet-Yaak Ecosystem. These kinds of killings are predictably associated with roads. As a result, levels of road access need to be substantially reduced and related levels of habitat security substantially increased rather than the opposite, as is being proposed for the Black Ram Project.

- Road density and habitat security standards used by the Kootenai NF are patently deficient, partly because they are based on research that conflates behavioral phenomena such as avoidance and displacement with demographic phenomena, notably survival. The scale is wrong as well, given that exposure to mortality hazards logically accrues over years as a consequence of cumulative annual movements of bears vis-à-vis hazardous environs. As a corollary, the fact that standards on the Kootenai NF are more lax than standards on the Flathead NF is self-evidently nonsensical given that grizzly bears in the Cabinet-Yaak Ecosystem remain in a much more precarious status compared to grizzly bears in the Northern Continental Divide Ecosystem.

- There is little or no evidence that food abundance is a significantly limiting factor for grizzly bears in the Cabinet-Yaak Ecosystem—especially as manifest in reproduction. On the other hand, there is ample evidence that human-caused mortality had governed and continues to govern the fate of this population, with food effects manifest primarily in the extent to which grizzly bears are exposed to human-related hazards during years when berries are in shorter supply.

- There is essentially no prospect of any increased berry production attributable to Project treatments offsetting increases in hazards associated with increased human access and activity. Moreover, enhancement of huckleberry production is more likely to occur as a result of wildfire rather than as a result of timber harvest. Given that the Davis fire recently burned substantial acreages in habitat conducive to huckleberry production, any need for ostensible berry-enhancing treatments has already been met naturally—and in an area that is likely to be more secure from humans.

- Compounding prospective problems with the Black Ram Project, proposed activities are concentrated in an area that is vital for facilitating movement of grizzly bears west-east between core habitats. Project activities will diminish rather than enhance security needed not only to facilitate transit of bears, but also increase odds that exposed bears will survive.

In short, the Black Ram Project promises to harm grizzly bears in the Cabinet-Yaak Ecosystem. In what follows, I elaborate on most of the points outlined above, and substantiate my conclusion.

A. Weight of Evidence Does Not Support Concluding that Status of the Cabinet-Yaak Population has Improved Since 2012

A.1. The 2.1% Per Annum Growth Rate for the Cabinet-Yaak Population is Not Justified or Applicable

Use of a 2.1% per annum growth rate to project total size of the Cabinet-Yaak population from the Kendall et al. (2016) 2012 point estimate, as was done by Kasworm et al (2018), is not defensible. Such use is, moreover, guaranteed to produce spurious results that cannot legitimately be used to reach conclusions of management relevance. There are several unambiguous reasons.

A.1.a. The growth rate is not representative of the total population

First, the estimated 2.1% per annum growth rate only applies to an unknown fraction of the total Cabinet-Yaak grizzly bear population. Vital rates used to estimate this growth rate were based solely on "native" or "natural" research-trapped bears, and expressly excluded bears captured because of conflicts or part of the augmentation program (Kasworm et al. 2018: 10). The growth rate, moreover, applies almost exclusively to the Yaak portion of the population given that 95% of the data used to estimate survival rates and 85% of the data used to estimate reproductive rates came from this subpopulation (ibid: 36)—protestations by the authors notwithstanding (ibid: 36). On top of this, the 2.1% per annum rate was estimated only for the female portion of this high-grade (ibid: 10), which is of consequence even though female survival is disproportionately important in determining growth rate, as such.

In other words, the 2.1% per annum growth rate can only be legitimately applied to females residing in the Yaak subpopulation that were not trapped and marked as a result of conflicts nor part of the augmentation program. Put another way, management-trapped bears, augmentation bears, and males would need to be represented in a modeling framework if any estimated population growth rate were to be prima facie representative of the total population. Moreover, if the fates of all such bears were to be considered, estimated population growth rate would almost certainly be lower given that survival rates of males, augmentation bears, and management bears are substantially less than survival rates of the females used to estimate the 2.1% per annum growth rate (ibid: 33-35).

If a growth rate were to be used to project a *total* population estimate, comparable to the Kendall et al. 2012 point estimate of 49 bears (95% CI = 44-62), then such a growth rate would need to represent birth and death rates of the *total* population, and apply specifically to the period of interest (e.g., 2012-2017) rather than a longer period of time that masks the relevant trajectory (see my point below).

A.1.b. The growth rate does not represent 2012-2017

The 2.1% per annum growth rate used by Kasworm et al to project 2017 population size was calculated using data that span 1983-2017 and so, therefore, axiomatically represent a generalized growth rate for Yaak females during this lengthy 35-year period. Put another way, the 2.1% per annum growth *is not* an estimate of growth for the period 2012-2017. For it to be so, the rate would have necessarily been estimated only using data from the approximate 2012-2017 period.

More to the point, estimates of growth for the Yaak female population are increasing back-weighted by inclusion of data that are, on average, increasingly old. Figure 1 (herein) shows the approximate average age of data used to calculate vital rates with the passage of time (from ibid: Table 17, 40-42). Notice that average age has increased from around 6-7 years in 1998 to nearer 15 years in 2017. In other words, with the progression of time estimates of population growth for the female segment of the Yaak population have become increasingly irrelevant to judging current population trajectory.

The Government retort to these contentions would probably be that the data from such a short period of time would be so sparse as to preclude a usefully accurate estimate. That is almost certainly the case, and a commentary in its own right on the profound limitations imposed by intrinsically small sample sizes. Nonetheless, this does not negate the point that the 2.1% per annum growth rate for 1983-2017 is spurious when applied to the 2012-2017 period. As Figure 11 clearly suggests (ibid: 37), population growth rate has almost certainly varied over time, albeit in largely indeterminate ways (see my following point).

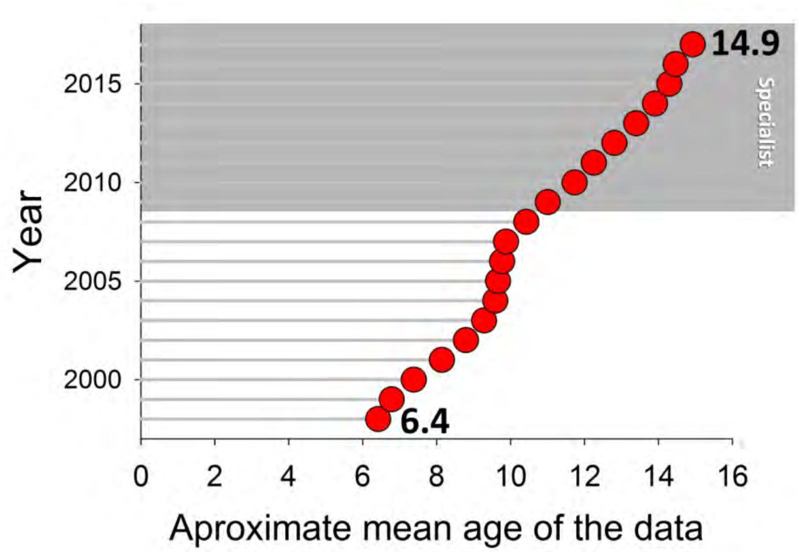

Figure 1. Trend in mean age of data used to calculate vital rates of Cabinet-Yaak grizzly bears with passage of years from 1998 to 2017. Mean age has more than doubled, with trend towards increased aging accelerating since deployment of a conflict management specialist in the ecosystem. Increasing age renders estimated vital rates increasingly irrelevant to current conditions.

A.1.c. Uncertainty of the growth rate as currently (or even ideally) calculated debars use

Small sample sizes impose very real constraints on the precision and accuracy of all demographic rates being used by Cabinet-Yaak researchers and managers. These constraints follow ineluctably from the small size of the Cabinet-Yaak grizzly bear population, which is a non-negotiable feature of this ecosystem.

As a practical upshot, all of the population growth rates calculated to date have uncertainty intervals (e.g., 95% confidence intervals) that not only substantially overlap zero (i.e., no growth) but also, over time, each other. More specifically, despite purporting to show trend in cumulative growth rate over time, the confidence intervals shown in Figure 10 (ibid: 37) all overlap—most almost completely (see also Figure 2A herein). Because of this, there is little or no basis for concluding that growth rate has varied with time. Likewise, taking a *precautionary* approach, there is little or no justifiable basis for concluding that growth rate is currently positive, despite statements in Kasworm el al. such as “The probability that the population was stable or increasing was 73%” (ibid: 36), especially in light of the fact that the point estimate of 2.1% per annum is a *cumulative rate* spanning 1983-2016 with little or no known relationship to *current rate* of population increase or decline.

Moreover, when the totality of point estimates and uncertainty is taken into consideration for the period 1998-2017, there is a cumulative 62% probability that the population was *declining* during these 19 years, consistent with the 2017 estimate of population size for Yaak females still being around 52% less than the estimate of population size for 1998 (Figure 2A and 2B herein).

The implications of uncertainty are thrown into relief by examining the specifics of projecting population size forward in time from 1983 to 2017 using the 1.021 (95% CI = 0.949-1.087) growth rate, noting up front that uncertainty in annual growth rate magnifies exponentially over time when manifest in population size. For example, after back-casting to obtain a plausible 1983 population starting point, deterministic projections of population size using the upper and lower confidence intervals of growth allow for a current population (2017) of anywhere between 3 and 256. Stochastic projections, e.g., using the software RISKMAN, generate a similar and not particularly useful range of 4 to 154 individuals.

The point here is that the raw cumulative uncertainty is huge, especially when dealing with a time period as long as 1983-2017. It is also important to note that this exercise takes the 1.021 estimate of lambda at face value, which, as per my previous points, is unwarranted.

Related to this last point, the current basis for modeling population growth rate using Booter (ibid: 10-11) is egregiously simplistic given the self-evident structural complexity of grizzly bear population demography in the Cabinet-Yaak Ecosystem. For any estimate of growth rate to be realistic, explanatory, relevant, and accurate, all of the main structure needs to be accommodated. More specifically, a relevant demographic model would ideally include source-sink structures accounting for management-trapped versus research-trapped bears, bears in the Yaak area versus the Cabinet Mountains, augmentation bears versus in situ bears—in addition to accounting for the male segment as well as inter-annual variation attributable to variation in key food resources (see later). The model described in Kasworm et al. does none of this.

Again, the probable retort would be that sample sizes are too small to support estimating the many rates required for such a model. But that is, indeed, the point. And no amount of hand-waving or protest will make it otherwise nor redeem the deficiencies in current estimates of demographic rates. The uncertainty is real and unavoidable, and should be acknowledged in management decision-making.

A.2. Even taking estimated growth rate at face value, current population status is problematic
Even taking the population growth rate estimated by Kasworm et al. at face value, the most defensible conclusions would be, first, that status of the population has worsened during 2014-2017 compared to 2006-2013, and, second, that numbers are still substantially less than the presumed peak reached around 1998. These conclusions are based on trend in *population growth rate* over time (as per ibid: 37), and trend in *population size* estimated by projections using year-specific cumulative population growth rates (e.g., projecting population size for 1998 using the 1983-1998 growth rate estimate, and then doing the same for each successive year, with 1983 the starting year throughout).

Figure 2 (herein) shows seminal results. In Figure 2A I've identified three periods typified by trends in *population growth*: rapid decline of 2% per annum during 1998-2006, coincident with the berry famine (see below); a nearly as rapid 1.1% rate of improvement during 2006-2014; followed by stalling in the rate of improvement to around 0.2% per annum since 2014—an 82% decline in rate of change—coincident with population growth rate finally reaching positive territory. Importantly, this refers to the *per annum rate of deterioration or improvement* in *population trajectory*, which is perhaps the most relevant information to be gleaned from the estimates of population growth rate presented by Kasworm et al.

Finally, Figure 2B (herein) shows trend in estimated *size* of the Yaak female population, both as a central tendency (dark green line) as well as bounding uncertainty (light green band, based on projections using the upper and lower confidence intervals for each cumulative estimate of growth rate). Parenthetically, I transformed the values to a natural log scale in Figure 2B to visually emphasize trends given that the bounds of uncertainty explode with projections increasingly farther forward in time. The take-away point is that, according to these values, population size peaked during 1998, reached a nadir during the height of the berry famine in 2006, increased through 2014, and then stalled during 2015-2017 at a size that was still around 52% less than peak numbers reached during 1998.

The key points here are that improvement in status of the female segment of the Yaak population *stalled* beginning in 2014 at numbers that were still approximately 52% less than the peak reached during 1998. Having said this, both of these conclusions remain severely compromised by the intrinsic uncertainties, lack of relevance, and bias of methods used by Kasworm et al.

A.3. Conclusion
The upshot of all this is that there is *no legitimate basis* for estimating current population size (e.g., 55-60) by applying a biased 1983-2017 growth rate—based on high-graded data representing only a fraction of the population—to a point population estimate made during 2012. Moreover, even taken at face value, the current cumulative population growth rate shows stalled improvement in population status and a population still substantially less than peak numbers reached during 1998.

The best that can be perhaps be invoked is a contrast between the presumed minimum estimate of 35 bears during 2014-2017 (ibid: 27) and the 2012 estimate of 49 (44-62) bears reported by Kendall et al.

(2016). The estimate of 35 for 2014-2016 is self-evidently less than the lower bound of the 2012 confidence interval, more consistent with a *static or even declining population* than with an increasing one. Of greater relevance to the draft EIS, this general conclusion also holds for comparisons specific to the Cabinet population (a current minimum of 13 bears compared to lower confidence intervals of around 20 reported by Kendall et al. for 2012).

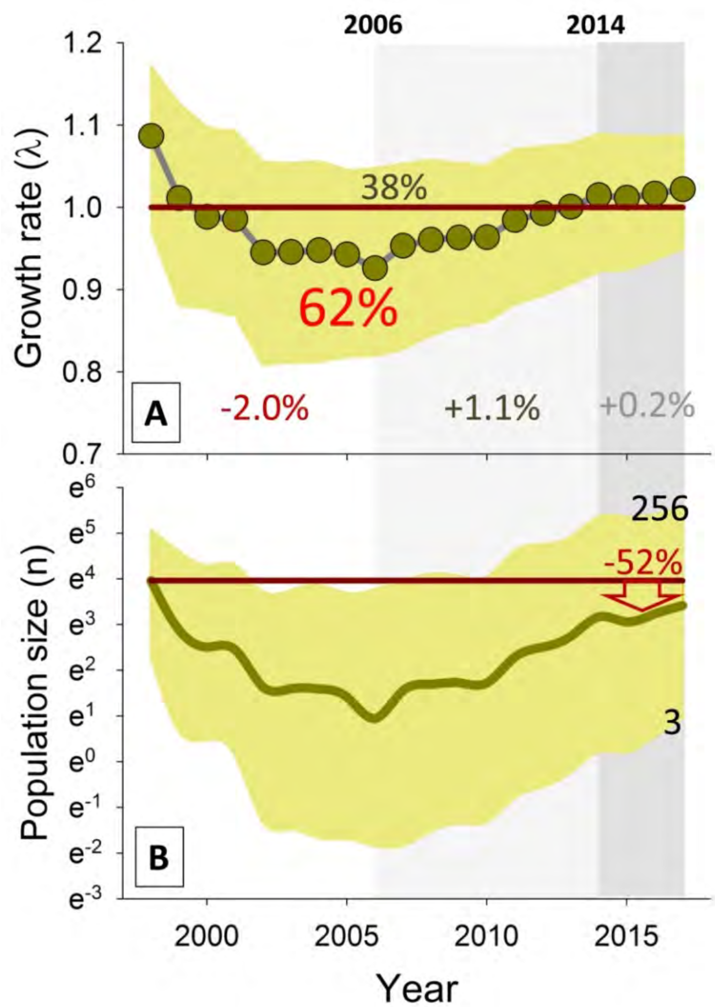

Figure 2. Trend in estimated population growth rate (A) and related estimated total population size (B) for Cabinet-Yaak grizzly bears, with the notable proviso that both sets of estimates are based almost wholly on data obtained from female grizzly bears in the Yaak population. Dark green dots or lines denote central tendencies, large green bands bounds of uncertainty. The horizontal dark red line in (A) denotes no growth, with any values above leading to increase and any values below leading to decline. The red line in (B) corresponds with estimated population size in 1998. In (A) I also show the cumulative weight of evidence for population declines versus increases for 1998-2017 along with average annual rates of change in lambda during three periods characterized by non-stationary shifts in dynamics. The numbers at right in (B) correspond to the range in estimated population size given uncertainties in growth rate (3-256), as well as the deviance in current estimated population size from the 1998 benchmark.

B. Comparison of Pooled Survival Rates in Kasworm et al. (2018) is Not Legitimate

As ancillary support for the proposition that size of the Cabinet-Yaak population has increased between 1999-2006 and 2007-2017, Kasworm et al state that “Grizzly bear survival of all sex and age classes decreased from 0.899 during 1983–1998 to 0.792 during 1999–2006 and then rose to 0.934” (ibid: 34), and then summarize these same numbers in Table 13 (ibid: 34).

Most of the problems and associated bias noted above applies to this comparison. Note, first, that the 95% confidence intervals reported in Table 13 for pooled estimates from all three time periods overlap, which precludes confidently concluding there is any difference in mean rates. Moreover, note the restriction to “native” bears, which excludes any consideration of conflict-trapped or augmentation bears, which were very much a component of the 2012 point estimate of population size.

The other problematic aspect of this comparison is that data from all bear sex and age-classes were pooled, without any apparent attempt to determine whether this collapse of data preserves representation of the population at large. Are males over- or under-represented?...likewise subadults versus adults? Some sort of weighting scheme reflective of current or even stable population structure could provide some remedy, but without compensating for other biases.

The other interesting aspect of this data-pooling is the extent to which it is at odds with other results and commentary in Kasworm et al. More specifically, this aggregation of data ignores the disproportionate importance of subadult females to population dynamics. This importance is evident in the near 85% variance in estimated population trend attributed to survival of subadult and adult female bears in Booter calculations (but with 60% attributable to subadult female survival, Table 15; ibid: 37), as well as the different contextual emphasis placed by the authors on female survival on Pages 32 (“…it is important to consider the rate of female mortality”) and 37.

The implication of all this is that the comparison of survival rates estimated from pooled data presented by Kasworm et al on Pages 33 and 34 does not mitigate the many fatal problems with their estimates of population growth rate.

C. Comparison of Annual Average Deaths in Kasworm et al. (2018) is Uninformative

Kasworm et al. (2018) present information on grizzly bear deaths in the Cabinet-Yaak Ecosystem in terms of numerous contrasts and adjustments presumably designed to be of relevance to various management deliberations. On pages 15-16 a running average of annual mortalities is related to recovery criteria; on pages 16-18 a full list of deaths with ancillary details is provided; and on pages 31-33 mortality is summarized in multiple ways presumably relative to different management considerations. Throughout, the parsing, categories, and nomenclature are confusing, obfuscated, and confounded. As a result, I needed to reconstruct much of the analysis of mortalities presented by

Kasworm from the raw data on pages 16-18. The contrast among time periods presented in Table 11 (ibid: 33) was a particular focus.

C.1. Table 11 in Kasworm et al. (2018) is a Tangled Mess

The totals in the column farthest right in Table 11 of Kasworm et al. (2018) include all mortalities—human-caused, natural, within 16-km of the Recovery Area boundary, in the US as well as Canada—plus the estimated unrecorded *human-caused* mortalities. For some inexplicable reason, and unlike in the NCDE and GYE, natural mortalities and mortalities of unknown cause were not accounted for in estimations of unrecorded mortalities.

The upshot is that the row totals in Table 11 represent a mishmash of natural, human-caused, and estimated unrecorded human-caused mortalities, without any straight-forward connection to judging overall population status. In fact, the inattention and even outright dismissal in this context of natural mortality as a factor in judging population status is mystifying given that a dead bear, for whatever reasons, matters in assessing the toll taken by mortality.

C.2. Comparison of 'rates' between 1999-2006 and 2007-2017 is Uninformative

By contrast, the comparison of annually-averaged human-caused mortality between 1999-2006 and 2007-2017 on Page 32 of Kasworm et al. only considers human-caused mortality, but without including any of the estimated unrecorded human-caused mortality included in Table 11—and without any cogent explanation. The confusion implicit to this inexplicable parsing is compounded by use of the term 'rate' in reference to an annual average, in context of 'rate' being used elsewhere in reference to survival and reproductive rates referenced to fates of individual bears. On top of this, a typo was made in reference to the 2007-2017 'rate,' which should be 2.2, not 2.1. This error amplified the potential for confusion arising from comparing '2.1' with '2.25' and calling the first value an increase over the second.

Reducing this chaos to something comprehensible: the annually averaged number of known and probable human-caused deaths during 1999-2006 was 2.13. Using all currently available data, for 2007-2018 the average was 2.08. When the estimate of unreported human-caused deaths is included, the average for 1999-2006 was 2.75 (95% CI 1.6-3.9). For 2007-2018 it was 3.2 (95% CI 2.2-4.2). Considering total known-probable mortality plus estimated unreported human-caused mortality—but without any correction for unreported natural deaths—the annual averages for 1999-2006 and 2007-2018 were virtually identical: 3.9 and 3.8.

The important point is, here again, that rote statistical uncertainty debars any conclusion about increase, stasis, or decrease in numbers of human-caused deaths. The confidence intervals of annual averages overlap substantially, which is not surprising given the small sample of years and dead bears. This statistical uncertainty is amplified by uncertainty attached to detecting any bear death other than that of an actively radio-monitored animal. Considering only human-caused deaths, this certainly holds for poached bears, deaths 'under investigation,' and deaths from unknown (but human-related) causes. A back-of-the-envelope calculation suggests that such deaths need to be increased by around 70 to 120% in year-end tallies.

In the face of such irrefutable uncertainty, Kasworm et al resort to focusing on and then emphasizing female mortality, which reduces the absolute values of calculated averages even further. When an estimate of unreported human-caused female mortalities is added to known mortalities (using the long-term proportion of F:M deaths=0.4), the result is an annual average of 1.75 (95% CI 0.83-2.67) female deaths for 1999-2006 and 0.80 (95% CI 0.34-1.54) female deaths for 2007-2018. All of the reported differences in mean values are so far within the range of statistical uncertainty as to render these comparisons a bit absurd.

C.3. Conclusion

Again, researchers and managers in this ecosystem might argue that small samples prevent any degree of certainty about conclusions, but this does not obviate the obligation to acknowledge uncertainty. Nor does it eliminate the practical consequences of small sample sizes and the compromising effects of chance processes—highlighted recently by a jump in recorded deaths from 1 in 2017 to 3 in 2018, a tripling in just one year. More certainly, it recommends humility and precaution in the face of such statistical ambiguities.

But all of this still leaves open the question of why natural mortalities as well as mortalities that cannot be definitively ascribed to human causes are not accounted for in assessing population status. This question is especially relevant given that Kasworm et al comment in several places on the extent to which variation in abundance of key natural foods likely drives population dynamics, often through the 'natural' death of dependent young (see below). Or, even, why, when considering only human-caused mortality, adjustments to account for unrecorded deaths were not included. This is all a bit mystifying as well as prima facie unjustified.

D. Status of the Cabinet-Yaak Population Remains Highly Precarious

The current vulnerability of the Cabinet-Yaak population can be illustrated through a simple exercise, even without accounting for spatial structure of the Cabinet and Yaak subpopulations. I input vital rates into a commonly-used risk management program named RISKMAN (currently being proposed for management of grizzly bear mortality in the NCDE). Using the stochastic function, I was able to reconstruct the c. 2.1% growth rate reported by Kasworm et al (2018) for 1983-2017. More specifically, the cumulative geometric mean growth rate (lambda) varied from a maximum of 1.035 to a minimum of 1.008. Accounting for variation in vital rates, the median ending population size at year 34 was 43, although the upper and lower 95% percentiles of simulated trajectories produced ending populations as small as 4 and as large as 154.

I then simulated what would have happened if just one additional female died each year. In this scenario, the geometric cumulative mean growth rate dropped from 0.952 (already much less than 1) to an astounding 0.202 at year 34 of the simulation (Figure 3 herein). Median total population size had reached 0 by year 23, with an upper 95th percentile of only 11 animals at the end of simulations. Results were not much improved when an additional 1 female was lost only once every 2 or 3 years. This is not presented as any definitive modeling result, but rather illustrative of how little the margin of error is, and how vulnerable this population is to even the smallest increased increments of mortality (e.g., Kendall et al. 2016). This point is especially germane given that *one adult female was killed by humans each of the last two years, during 2018 and 2019*. And this does not account for adult females that died and were not documented.

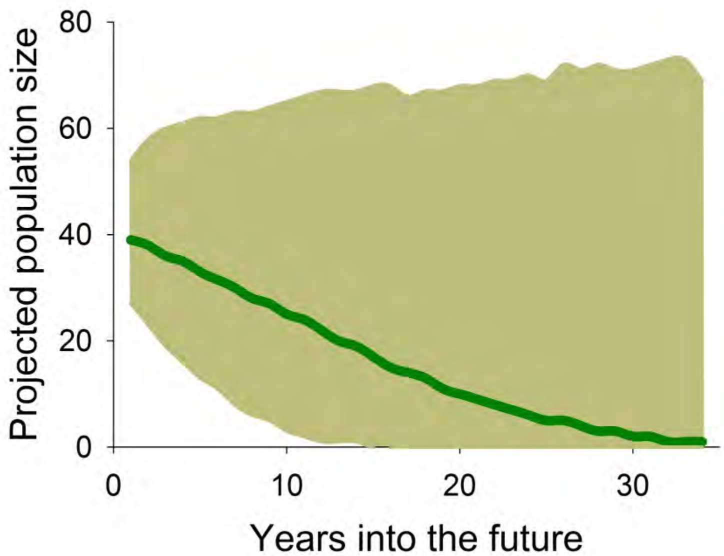

Figure 3. Results of RISKMAN projections for the Cabinet-Yaak population using vital rates reported by Kasworm et al. (2018), but introducing the death of an additional female grizzly bear once every 2 years. The thick green line represents the median trend of projections; the dusky green band above and below the variability of projections.

E. Weight of Available Evidence Emphasizes the Continued Importance of Malicious Killing

The extent to which poaching, malicious killing, or other suspect circumstances are associated with human-caused deaths is also instructive regarding the overall effectiveness of conflict mitigation efforts during 1999-2017 to offset the problematic effects of road-access and poaching. By its nature, malicious killing/poaching is a criminal act undertaken by criminals. Such behavior is rooted in attitudes and outlooks that are notoriously unresponsive to education and 'outreach'. The phenomenon is about willful malfeasance. As such, limitations on road access coupled with improved law enforcement and successful prosecutions are logically the most appropriate redress—not, for example, conflict mitigation by a specialist who is not tasked primarily with law enforcement.

Before pursuing this any farther, some clarification of obfuscations in the dead bear database is needed. During 1999-2017 a number of deaths were ascribed to 'Undetermined' human causes, 'Poaching' or listed as 'Under investigation'. The first and last categories are not explicit, but nonetheless strongly suggestive. Certainly, 'Under investigation' suggests that the death occurred under suspicious circumstances warranting investigation—with a strong likelihood of either poaching or other unwarranted lethal action by the involved people. Such suspicions are rarely definitively resolved. 'Undetermined' is also more suggestive of malfeasance rather than innocence on the part of the involved people. Given the alternatives, such deaths are more defensibly allocated to causes more resistant than not to mitigation.

With all of this as context, there were a total of 7 known-probable deaths during 1999-2006 attributed to either poaching or undetermined causes, representing 58% of total human-caused deaths. During 2007-2018 there were a total of 13 deaths either under investigation or ascribed to poaching, representing a nearly identical 59% of the total known-probable human-caused deaths. These are major fractions in their own right, but leave estimated numbers of unreported deaths unaccounted for. As Kasworm et al make clear (ibid: 33), their estimate of 'unreported' deaths did not apply to bears that were radio-collared or removed by managers, which leaves this unreported estimate levied almost entirely against malicious or otherwise suspect causes. When these unreported estimates are added to the known-probable toll taken by poaching, unknown causes, or suspicious circumstances, the percentage increases to around 70% during 1999-2006 and approximately 77% during 2007-2016.

Taken together, these figures support concluding that (1) malicious or otherwise suspect causes account for a large portion—if not majority—of grizzly bear deaths in the Cabinet-Yaak Ecosystem; (2) the fraction and even total numbers of deaths attributable to such causes did not decrease from 1999-2006 to 2007-2018; and (3) that aggressive limitations to road access by the USFS are needed, especially in areas with concentrations of productive habitat (Proctor et al. 2015, 2017).

F. Access Management is Critical to Limiting Malicious & Other Unjustified Killing

The consensus of relevant research is unambiguous about the link between road access and grizzly bear mortality. The more access, the more dead bears there are, with disproportionate concentrations near roads (Brannon et al. 1988; Benn & Herrero 2002; Nielsen et al. 2004; Wakkinen & Kasworm 2004; Boulanger & Stenhouse 2014; McLellan 2015; Proctor et al. 2017, 2018). Dead bears tend to be concentrated within 100 to 500 m of roads, averaging around 300 m (± 195 m) among studies where distance was noted.

Unfortunately, there is a common conflation of the extent to which radio-marked grizzly bears spatially avoid roads with the geospatial configuration of mortality risk and, even more important, decrements in survival and population growth. These parameters are not synonymous. Even though a bear might underuse habitats within a certain distance of roads, this does not translate into a 1:1 correlation with exposure to risk of human-related mortality during a bear's lifetime. Conflation of avoidance with mortality risk has led to the unstated assumption that the former can be used to set standards for the latter. Such is the case for road density and habitat security standards set by the Kootenai National Forest based on the results of Wakkinen & Kasworm (1997).

Taking 300 m as a ballpark figure, road densities of roughly 0.6 km/km^2 translate into areas remote from where human-caused mortality is concentrated that amount to only 84 ha (208 acres), which is trivially small for a grizzly bear. This sort of geospatial buffer still means that grizzly bears are frequently exposed to hazards of human-caused death to the predictable extent that they must and will move from one presumably secure area to another—even assuming that these bears exhibit "average" avoidance of human features such as roads. In other words, the level of buffering from human-caused mortality offered by road density and related security standards invoked in the Black Ram EA is guaranteed to be inadequate.

The inadequacy and inappropriateness of road density and security standards used by the Kootenai National Forest in application to the Black Ram Project are highlighted in contrast to standards applied in the Northern Continental Divide Ecosystem (NCDE), as well as in contrast to trajectories of populations in the NCDE and Greater Yellowstone Ecosystem. The populations of already relatively numerous grizzly bears in the NCDE and GYE have increased substantially since the early 1990s to 2000s, in contrast to in the Cabinet-Yaak where precariously few bears have fared poorly (see my Points A-D, herein). Tellingly, Wilderness Areas and Inventoried Roadless Areas where road access is not allowed comprise around 56% of the NCDE and GYE. In the Cabinet-Yaak Ecosystem this figure is less than half as much, nearer 21%. This difference alone can explain much of the corresponding difference in fates of grizzly bear populations.

Despite these telling differences in fates and trajectories of grizzly bear populations, the road density and habitat security standards applied by the Kootenai National Forest are more lax, not less, than those applied on the Flathead National Forest. On the Kootenai, areas allowed with >1 mile/mile2 of roads are 1.7-times greater; areas with >2 miles/mile2 of roads are 1.4-times greater; and extents of secure habitat nearly 20% less compared to what is ostensibly allowed on the Flathead NF. These disparities are perverse and not able to be explained on the basis of differences in the extent of movements by grizzly bears. If anything, bears range more widely in the Cabinet-Yaak Ecosystem compared to the NCDE (Kasworm et al. 2018).

As a bottom line, existing and proposed access management in the Black Ram Project Areas has jeopardized and will continue to jeopardize grizzly bears.

G. More Grizzly Bear Deaths Are Occurring On USFS Jurisdictions Now Compared to During 1999-2006

The argument for more aggressive management to prevent human-caused grizzly bear mortality on USFS jurisdictions is given greater weight by differences in locations of bear deaths between 1999-2006 and 2007-2018. Data from Kasworm et al. (2018) and Kasworm (2018)show an increase in the proportion of grizzly bear deaths on USFS lands from 25% (95% CI = 0.5-49.5%) during 1999-2006 to 56.5% (36.3-76.8%) during 2007-2018. Although sample sizes are small, confidence intervals large, and overlap of the intervals non-trivial (17%), these results *do not* support concluding that hazards for grizzly bears have remained constant or declined on USFS lands. Rather, by weight of evidence, the better supported conclusion is that hazards have *increased* and, because of that, imperatives to control mortality on public lands have likewise increased, including on lands part of the proposed Black Ram Project. As per my point F, above, the most efficacious means available to the USFS for addressing this imperative is through providing increased rather than diminished habitat security, axiomatically through *reducing* road access in the Project area.

H. The Proposed Black Ram Project is Inconsistent with the Best Available Science

Although I could invoke a large corpus of relevant science that has been neglected in the Black Ram EA, I focus here to one piece of research—Proctor et al. (2017). This paper is highly relevant to judging the trade-off between proposed forest treatments and habitat security for grizzlies, especially vis-à-vis any prospective increase in huckleberry production and hazards associated with road access. Some relevant recommendations from Proctor et al. (2017) include:

- That 25% of the landscape, particularly portions with important huckleberry fields, have no roads.
- That 60% of the landscape be >500 m from an open road in patches >5-10 km^2 and those secure habitat patches encompass higher quality habitats.
- That some portion (roughly 25%) of the landscape not have any roads.
- That areas be mapped where wildfires will not be extinguished unless conditions are too risky for private property.

Some of the more germane quotes from Proctor et al. (2017) include:

> "While managing the landscape for huckleberry forage is sometimes a medium to long term option, managing for mortality risk through road density has potential to be implemented over shorter time scales."
>
> "We found that 74% of huckleberry patches were not in cut blocks."
>
> "We also found that planted cut blocks were less likely to have huckleberry patches. Slash burning didn't increase the probability of a cut block yielding a huckleberry patch—it was reduced. Fires (not slash burns) were ~3x more likely to yield a huckleberry patch than a cut block."

The results of Proctor et al. (2017) are consistent with other research showing that naturally-occurring shrub-fields are positively selected by grizzly bears, presumably to consume berries (Waller & Mace 1997, McLellan & Hovey 2001, Apps et al. 2004), whereas recently harvested (<40-years) stands tend to be avoided, partly because of associated human activity (Apps et al. 2004). Parenthetically, descriptions of habitat use given by Kasworm et al. (2018: 48-50) are of limited applicability simply because the analysis did not consider use in relation to availability.

Given all of these considerations, the Davis Fire has almost certainly yielded as many benefits as might be needed for grizzly bears insofar as enhancements of habitats for huckleberry production. No further forest treatments are likely needed, even of the sort recommended by Proctor et al. (2017). More certainly, any (if any) gains in enhanced huckleberry habitats will almost certainly be negated by maintenance of existing road access, much less any of the local increases proposed under terms of the Black Ram Project.

I. Activities of the Black Ram Project Are Problematic in a Larger Geospatial Context

The Black Ram EA failed to evaluate the impacts of proposed activities on grizzly bears in a larger geospatial context. Mattson & Merrill (2004) and Proctor et al. (2015) are perhaps most relevant to such an evaluation. The former research mapped existing core habitat as well as higher-probability source habitats in the Cabinet-Yaak Ecosystem (shades of green; Fig. 4A), whereas the latter mapped core habitat (green) along with prospective corridors (yellow; Fig. 4B). Importantly, Proctor et al. (2015) modeled habitat at a finer grain that approximated daily movements, whereas Mattson & Merrill (2004) modeled habitat at the scale of grizzly bear life ranges, which is more relevant to judging overall odds of survival reckoned over a span of years.

As has been repeatedly emphasized by researchers in the Transboundary region, connectivity is essential to long-term prospects for recovery (Proctor et al. 2012, 2015, 2018; Kasworm et al. 2018). And, as also emphasized by these authors, connectivity is inextricably rooted in habitat security. However, Figure 4 highlights a fundamental problem for grizzly bears residing in the Yaak/Yahk region: core habitat to the west is comparatively isolated from core habitat to the east by a zone that is less secure. Ideally, aggressive restoration efforts would be undertaken in this problematic area to reestablish free exchange of bears likely to survive the transit. Logically, this restoration would emphasize road closure and permanent decommission (as per my points F and H). Instead, the Black Ram Project proposes to not only maintain an extensive road infrastructure in this potential west-east connector, but more egregiously ramp up human activities, with the prospect of both displacing bears and also exposing bears venturing into the Project Area to even greater risk of mortality.

The results of Mattson & Merrill (2004) and Proctor et al. (2015) allow for the identification of proposed Project Activities that are especially problematic and should be cancelled altogether if there is serious intent of meaningfully recovering grizzly bears in the Yaak/Yahk region. As a priority, regeneration and intermediate harvests planned for Units 77-83 should be eliminated from the Project, as should activities proposed for Blocks 1-17, 66-76, and 84. Ideally, activities in Blocks 18-44 would also be removed from the proposal to promote reestablishment of west-east connectivity in an area with broadly the greatest potential.

Adding weight to this recommendation, the under-construction Pacific Northwest Trail is located such that it will amplify the impacts of proposed activities in the Black Ram Project (Fig. 4). Parenthetically, this consideration highlights the additional fact that the Black Ram EA did not consider the prospective cumulative impacts of all on-going, proposed, and foreseeable human activities—including increases in human activity along the Pacific Northwest Trial and foreseeable activities associated with the proposed Rock Creek and Montanore Mines.

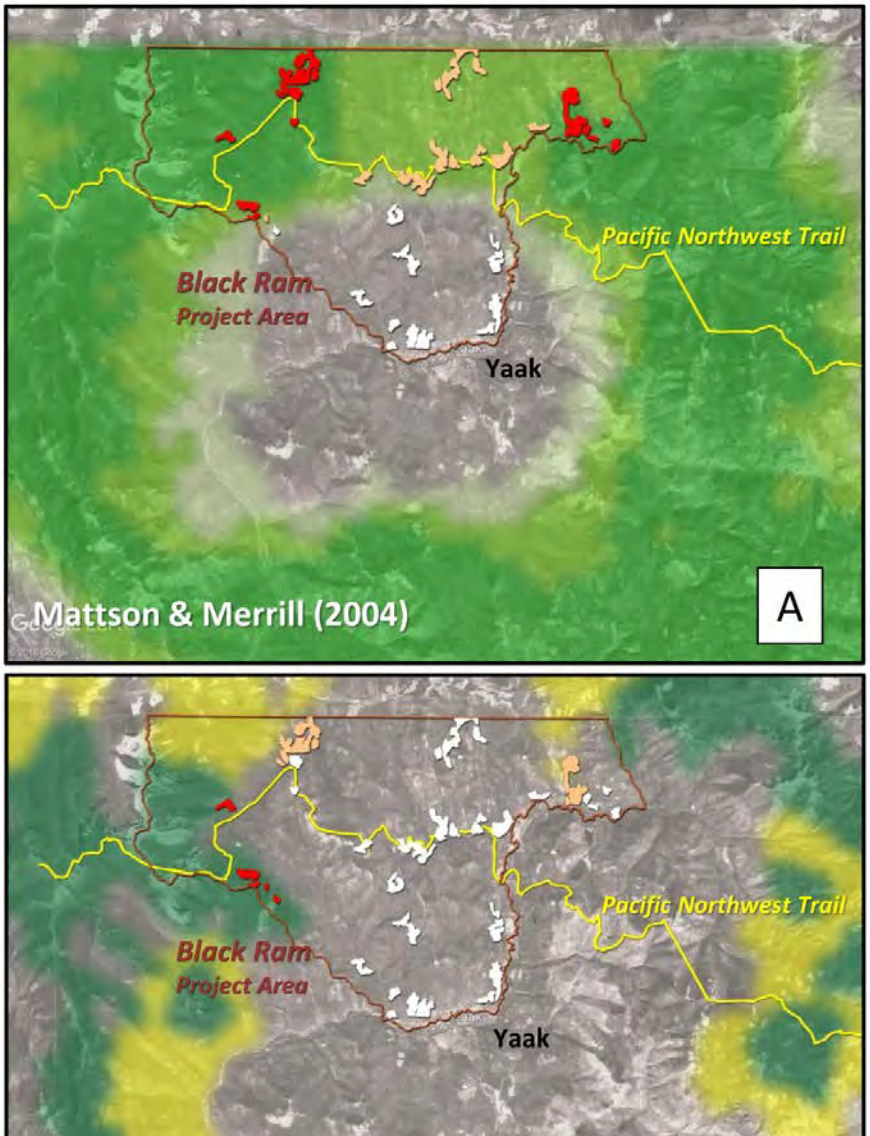

Figure 4. *Maps of (A) potential core and source habitat and (B) cores and corridors for grizzly bears (green and yellow, respectively) in the Yaak region. The boundary of the Black Ram Project Area is shown in burgundy juxtaposed with the Pacific Northwest Trail in yellow. Units proposed for regeneration and intermediate harvest in areas that promise maximum impacts to currently relative secure habitat are shown in red. Units that promise harm in habitats important to connectivity are shown in orange. The remainder of proposed harvest units are shown in white.*

J. Impacts of Black Ram will be Synergistically Amplified by the Pacific Northwest Trail

The Pacific Northwest Trail (PNT), currently under construction, will prospectively introduce an additional 100-150 people into the heart of previously secure habitat in the Yaak region each year, with almost all of that increase during July-September, the critical hyperphagic period for grizzly bears. Importantly, the PNT creates a linear disturbance that is contiguous with and extending beyond disturbances proposed during the Black Ram Project (Fig. 5). This attenuation of distrubances insures that there will be minimal areas free of human impacts anywhere in the Black Ram Project Area. An ample corpus of relevant research unambiguously shows that hikers using a trail such as the PNT will displace grizzly bears, with impacts predictably greatest on reproductive females (Mattson 2019a). Moreover, these impacts will almost certainly be greater in and near the lineated high-elevation habitats in the Yaak (Mattson 2019a), with displaced bears having few places to go that will not also be impacted by human activities associated with the Black Ram Project.

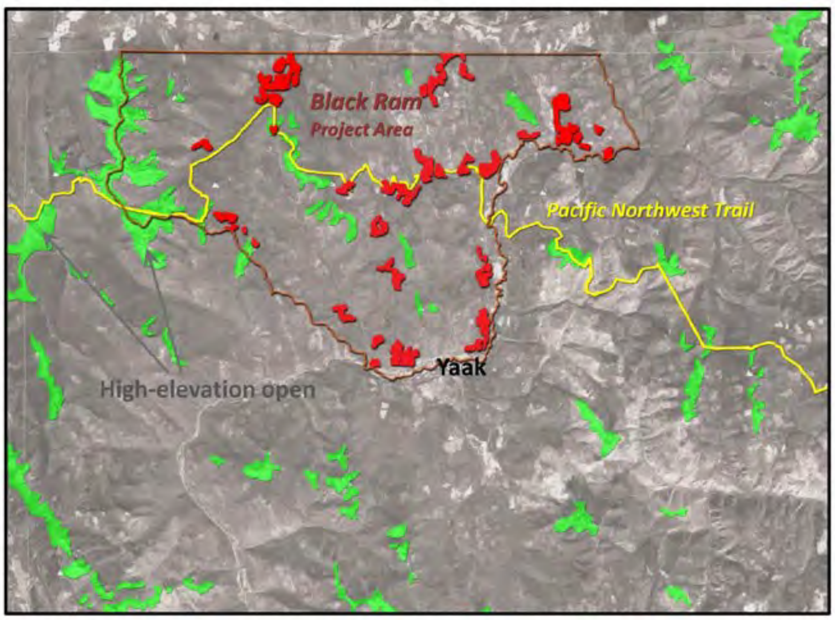

Figure 5. *Location of the Black Ram Project and associated intermediate and regeneration harvest treatments (in red) juxtaposed with the Pacific Northwest Trail (in yellow) and high-elevation open habitats (in green).*

Impacts from the PNT highlight the patent failure of the Black Ram EA to assess the cumulative effects of other current, foreseeable, and proposed human activities. Moreover, with the Cabinet-Yaak Recovery Area as a logical unit of analysis, any assessment of cumulative effects needs to account for other on-going and planned human activities associated with forest treatments and harvest in this Ecosystem, as well as foreseeable impacts associated with the proposed Rock Creek and Montanore Mines; as well as on-going and foreseeable impacts associated with the human transportation infrastructure (e.g., railways and associated highways that already fragment grizzly bear distribution in this Ecosystem, Mattson et al. [2019b]), all with the potential to amplify impacts arising from the Black Ram Project.

K. A Devil's Bargain Will Not Rescue This Small Population

K.1. The Yaak Population is Not Viable and Remains Acutely Vulnerable to Increased Mortality

The Cabinet-Yaak grizzly bear population is smaller than the smallest census population size ever posited as being viable. The Yaak/Yahk subpopulation has limited connectivity with grizzly bear populations elsewhere, and the Cabinet Mountains subpopulation is more isolated yet (Apps et al. 2016; Kendall et al. 2016; Proctor et al. 2012, 2015). Such isolation is well-known to magnify risk. The degree of this risk is evident in the fact that fates of populations as small of that of the Cabinet-Yaak grizzlies can be dictated solely by chance variation in birth and death rates, known as demographic variation. Yet demographic variation is a relatively minor stressor compared to environmental variation, catastrophes, negative deterministic trends, and loss of genetic diversity—all of which are documented or potential factors in the Cabinet-Yaak. The contemporary consensus of researchers is that populations of large mammals such as grizzly bears need to consist of thousands of animals to withstand all of these stochastic and deterministic threats over meaningful periods of time.

More to the point, as I emphasize in Point D, above, the Yaak and Cabinet grizzly bear populations remain acutely vulnerable to even small changes in levels of mortality. Under such circumstances, a precautionary approach to managing spatial hazards and habitat security is not only advisable, but mandatory. Unfortunately, there is no evidence of caution or even meaningful recognition of threats to the Yaak population in the Black Ram EA.

K.2. Variation in Population Trajectory Has Likely Been Driven by Exposure to Humans

As a hypothetical, it is worth taking claims regarding an improvement in status of the Cabinet-Yaak grizzly bear population between 1999-2006 and 2007-2018 at face value. Again, the emphasis here is on the hypothetical given all of the compromising or even fatal flaws in analyses and conclusions reported in Kasworm et al. More specifically, if an improvement did occur, what was (were) the likely driver(s)?

Causation is notoriously hard to establish with any reliability or confidence. Nonetheless, even taking comments in Kasworm et al (again) at face value, one can establish how these authors ascribed causation based on the balance of their comments. The relevant quotes include:

> "The increase in total known mortality beginning in 1999 may be linked to poor food production during 1998-2004 (Fig. 9). Huckleberry production during these years was about half the long term average…Poor nutrition may not allow females to produce cubs in the following year and cause females to travel further for food, exposing young to greater risk of mortality from conflicts with humans, predators, or accidental deaths." (emphasized in Figure 10; ibid: 32; see Fig. 6, herein).

> "Some of this decrease [in survival] in the 1999-2006 period could be attributed to an increase in natural mortality probably related to poor berry production during 1998-2004. Mortalities on private lands within the U.S. increased during this period, suggesting that bears were searching more widely for foods to replace the low berry crop." (ibid: 34).

In reference to a probable increase in size of the Cabinet Mountains subpopulation from around <15 (possibly 5-10) in 1988 to around 22-24 in 2012: "These data indicate the Cabinet Mountains population has increased 2-4 times since 1988, but this increase is largely a product of the augmentation effort with reproduction from that segment." (ibid: 36).

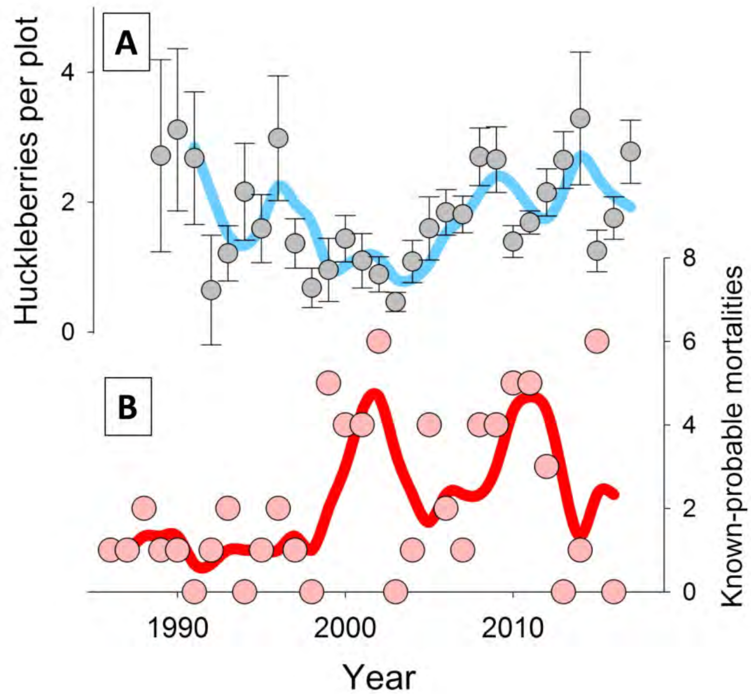

Figure 6. *Trends in size of huckleberry crops (A) and in numbers of known and probable grizzly bear deaths (B) in the Cabinet-Yaak Ecosystem. Annual data are shown as dots and 3-year moving averages as solid lines. The negative relations between berry crop size and numbers of deaths is clearly evident, largely driven by related variation in exposure to humans.*

On balance, Kasworm et al ascribe more weight to variation in natural foods and the augmentation program than to any mitigation measures as drivers of decline for the Cabinet-Yaak grizzly bear population during 1999-2006 and the subsequent presumed improvement in status during 2007-2017. This conclusion is consistent with that reached by McLellan (2015) from research in the nearby North Fork of the Flathead River and by McCall et al. (2013) from the Purcell Mountains showing a major influence of huckleberry fruit crops on demography of the local grizzly and black bear populations.

Importantly, though, Kasworm et al clearly associate the increase of deaths on during 1999-2006 and subsequent decline during 2007-2016 to bears foraging more widely—including into conflict situations—during times of dearth in late-season fruit crops. In other words, variation in population trajectory as a result of variation in size of huckleberry crops is not linked to changes in female reproductive success, but rather to effects on exposure of bears to humans.

K.3. Any Prospective Increase in Huckleberries Will Not Offset Impaired Habitat Security
The consensus of available research is unequivocal about survival of especially female grizzly bears being more important than any likely increase in reproductive rates, in this case prospectively from enhanced berry production (Eberhardt et al. 1994, Hovey & McLellan 1996, Garshelis et al. 2005, Harris et al. 2006, Mace et al. 2012), especially if huckleberry patches are not meaningfully secured from human access and protected from human harvest (Proctor et al. 2017; see also https://news.gov.bc.ca/releases/2019FLNR0186-001439). And, moreover, reiterating my points under H, above, harvest treatments as part of Black Ram will not likely provide substantial increases in huckleberry production, especially relative to the benefits already imparted by the Davis fire. Rather than planning more timber harvest, the Forest Service would provide more certain benefits for grizzly bears by prescribing a more widespread natural fire regime combined with aggressive closure and retirement of roads in portions of the Black Ram Project Area highlighted in Figure 4.

L. Conclusion

Reiterating my conclusion in the Introduction to these comments, the Black Ram Project as described in the EA promises to harm grizzly bears in the Cabinet-Yaak Ecosystem, with bears occupying the transboundary region of the Yaak bearing the brunt of harm. In contrast to the current proposed actions entailing widespread timber harvest and maintenance and even construction of supporting roads, the Forest Service could unequivocally benefit grizzly bears in this area by instead emphasizing the promotion of natural fire regimes and closure and retirement of roads.

Please contact me at davidjmattson@gmail.com or 406-222-4702 if you have any questions. My somewhat dated resume is attached, current only up through my retirement shortly after 2011.

David Mattson, PhD

M. References

Apps, C. D., McLellan, B. N., Woods, J. G., & Proctor, M. F. (2004). Estimating grizzly bear distribution and abundance relative to habitat and human influence. The Journal of Wildlife Management, 68(1), 138-152.

Apps, C. D., McLellan, B. N., Proctor, M. F., Stenhouse, G. B., & Servheen, C. (2016). Predicting spatial variation in grizzly bear abundance to inform conservation. The Journal of Wildlife Management, 80(3), 396-413.

Benn, B., & Herrero, S. (2002). Grizzly bear mortality and human access in Banff and Yoho National Parks, 1971-98. Ursus, 13, 213-221.

Boulanger, J., & Stenhouse, G. B. (2014). The impact of roads on the demography of grizzly bears in Alberta. PloS One, 9(12), e115535.

Brannon, R. D., Mace, R. D., & Dood, A. R. (1988). Grizzly bear mortality in the northern Continental Divide ecosystem, Montana. Wildlife Society Bulletin, 16(3), 262-269.

Eberhardt, L. L., Blanchard, B. M., & Knight, R. R. (1994). Population trend of the Yellowstone grizzly bear as estimated from reproductive and survival rates. Canadian Journal of Zoology, 72(2), 360-363.

Garshelis, D. L., Gibeau, M. L., & Herrero, S. (2005). Grizzly bear demographics in and around Banff National Park and Kananaskis country, Alberta. The Journal of Wildlife Management, 69(1), 277-297.

Harris, R. B., Schwartz, C. C., Haroldson, M. A., & White, G. C. (2006). Trajectory of the Yellowstone grizzly bear population under alternative survival rates. Wildlife Monographs, (161), 44-55.

Hovey, F. W., & McLellan, B. N. (1996). Estimating population growth of grizzly bears from the Flathead River drainage using computer simulations of reproduction and survival rates. Canadian Journal of Zoology, 74(8), 1409-1416.

Kasworm , W. F., Radant, T. G., Tesiberg, J. E., Welander, A., Proctor, M., & Cooley, H. (2018). Cabinet-Yaak Recovery Area 2017 research and monitoring progress report. US Fish & Wildlife Service, Missoula, Montana.

Kasworm, W. (2018). Selkirk/Cabinet-Yaak IGBC Subcommittee, meeting notes: 2018 research/monitoring update. http://igbconline.org/wp-content/uploads/2018/11/181108_SCYE_Mtg_Summary.pdf

Kendall, K. C., Macleod, A. C., Boyd, K. L., Boulanger, J., Royle, J. A., Kasworm, W. F., ... & Graves, T. A. (2016). Density, distribution, and genetic structure of grizzly bears in the Cabinet-Yaak Ecosystem. The Journal of Wildlife Management, 80(2), 314-331.

Ladle, A., Avgar, T., Wheatley, M., Stenhouse, G. B., Nielsen, S. E., & Boyce, M. S. (2018). Grizzly bear response to spatio-temporal variability in human recreational activity. Journal of Applied Ecology.

Mace, R. D., Carney, D. W., Chilton-Radandt, T., Courville, S. A., Haroldson, M. A., Harris, R. B., ... & Schwartz, C. C. (2012). Grizzly bear population vital rates and trend in the Northern Continental Divide Ecosystem, Montana. The Journal of Wildlife Management, 76(1), 119-128.

Mattson, D. J. (2019a). Effects of pedestrians on grizzly bears: An evaluation of the effects of hikers, hunters, photographers, campers, and watchers with reference to the proposed Pacific Northwest Trail. Grizzly Bear Recovery Project, Report GBRP-2019-3.

Mattson, D. J. (2019b). Effects of trains and railways on grizzly bears: An evaluation of the effects of increased train traffic on the Burlington Northern Santa Fe and Montana Rail-Link Railways, Montana-Idaho. Grizzly Bear Recovery Project, Report GBRP-2019-1.

Mattson, D. J., & Merrill, T. (2004). A model-based appraisal of habitat conditions for grizzly bears in the Cabinet–Yaak region of Montana and Idaho. Ursus, 15(1), 76-90.

McCall, B. S., Mitchell, M. S., Schwartz, M. K., Hayden, J., Cushman, S. A., Zager, P., & Kasworm, W. F. (2013). Combined use of mark-recapture and genetic analyses reveals response of a black bear population to changes in food productivity. The Journal of Wildlife Management, 77(8), 1572-1582.

McLellan, B. N., & Hovey, F. W. (2001). Habitats selected by grizzly bears in a multiple use landscape. The Journal of Wildlife Management, 65(1), 92-99.

McLellan, B. N. (2015). Some mechanisms underlying variation in vital rates of grizzly bears on a multiple use landscape. The Journal of Wildlife Management, 79(5), 749-765.

Proctor, M. F., Paetkau, D., Mclellan, B. N., Stenhouse, G. B., Kendall, K. C., Mace, R. D., ... & Wakkinen, W. L. (2012). Population fragmentation and inter-ecosystem movements of grizzly bears in western Canada and the northern United States. Wildlife Monographs, 180(1), 1-46.

Proctor, M. F., Nielsen, S. E., Kasworm, W. F., Servheen, C., Radandt, T. G., Machutchon, A. G., & Boyce, M. S. (2015). Grizzly bear connectivity mapping in the Canada–United States trans-border region. The Journal of Wildlife Management, 79(4), 544-558.

Proctor, M. F., Lamb, C. T., & MacHutchon, A. G. (2017). The grizzly dance between berries and bullets: relationships among bottom-up food resources and top-down mortality risk on grizzly bear populations in southeast British Columbia. Trans-border Grizzly Bear Project, Kaslo, British Columbia, Canada, http://transbordergrizzlybearproject.ca/research/publications.html.

Proctor, M. F., McLellan, B. N., Stenhouse, G. B., Mowat, G., Lamb, C. T., & Boyce, M. S. (2018). Resources roads and grizzly be4ars in British Columbia and Alberta, Canada. Trans-border Grizzly Bear Project, Kaslo, British Columbia, Canada, http://transbordergrizzlybearproject.ca/research/publications.html.

Wakkinen, W. L., & Kasworm, W. (1997). Grizzly bear and road density relationships in the Selkirk and Cabinet-Yaak recovery zones. US Fish and Wildlife Service, Kalispell, Montana.

Wakkinen, W. L., & Kasworm, W. F. (2004). Demographics and population trends of grizzly bears in the Cabinet–Yaak and Selkirk Ecosystems of British Columbia, Idaho, Montana, and Washington. Ursus, 15(1), 65-76.

Waller, J. S., & Mace, R. D. (1997). Grizzly bear habitat selection in the Swan Mountains, Montana. The Journal of Wildlife Management, 61(4), 1032-1039.