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Confronting uncertainty in wildlife management: performance of grizzly bear management in British Columbia, Canada

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Abstract

Scientific management of wildlife requires confronting the complexities of natural and social systems. Uncertainty poses a central problem. We examined outcome uncertainty and components of biological uncertainty in hunt management of grizzly bears (*Ursus arctos horribilis*) in British Columbia, Canada (2001–2011). We found that unaddressed uncertainty compromised management performance. Outcome uncertainty alone — discrepancy between expected and actual mortality levels — led to excess mortality in 19% of cases (population-years) examined. Accounting for uncertainty around estimated biological parameters revealed excess mortality might have occurred in up to 70% of cases. We offer a method to identify targets that incorporate uncertainty and maintain the probability of exceeding mortality limits below specified thresholds. At thresholds of 25% and 5%, this would require average target mortality reductions of 47% and 81%, respectively. Application of our transparent and generalizable framework to this or other systems could improve management performance in the face of uncertainty.

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Introduction

Confronting uncertainty poses a central problem in wildlife management. Decisions made without proper consideration of uncertainty can have undesirable consequences, and have been implicated, for example, in widespread overfishing (Harwood & Stokes 2003). Although often poorly accounted for or ignored, uncertainty exists about the "true" value of estimated parameters (Reckhow 1994, Hilborn & Mangel 1997, Regan et al. 2005, Armitage et al. 2009). Parameter uncertainty propagates to uncertainty in important management estimates, including the magnitude of mortality a population can withstand without experiencing long-term declines or other deleterious effects (hereafter "mortality limit uncertainty"; Caddy & McGarvey 1996, Prager et al. 2003). Management performance can also be compromised by outcome uncertainty, defined as the difference between targeted and realized mortality levels (Holt & Peterman 2006). Remarkably, scholarly and independent retrospective examination of wildlife or fisheries management performance — in the face of uncertainty, or, in general — are rarely conducted (but see Ross et al. 1996, Holt & Peterman 2006, Linnell et al. 2010).

Several methods can account for and incorporate uncertainty into decision-making, estimating *a priori* the probability that specific scenarios will lead to over-exploitation (Reckhow 1994, Harwood & Stokes 2003). Key to implementing these approaches, however, is differentiating targets (mortality levels management aims to achieve) from limits (mortality levels management should never exceed). Given that the probability of exceeding a target is non-zero, dictated by a system's outcome uncertainty, conservative management should avoid setting targets as high as, or conflating them with, limits (Caddy & McGarvey 1996, Prager et al.

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2003). Incorporating uncertainty into setting such targets could better ensure that populations are managed within specified limits.

Grizzly bears (*Ursus arctos horribilis*) provide an ideal model species for assessing uncertainty in wildlife management. Management of most populations occurs with limited demographic information (Miller 1990, Mattson et al. 1996, McLoughlin 2003). Moreover, grizzly bear life-history characteristics — including long lifespans, low reproductive rates, delayed reproductive maturity, and slow population growth rates (Miller 1990)— make them highly vulnerable to population declines (Reynolds 2003). Finally, as with many large carnivores, mortality is primarily human-caused (Peek et al. 1987, Miller 1990, Treves 2009). As such, management decisions can have a considerable influence on population viability (McLoughlin 2003, McLoughlin & Messier 2004).

Grizzly bear mortality management in British Columbia (BC) provides a particularly useful case study for examining effects of uncertainty on management performance. Most populations are managed for sustained yield whereby, in theory, a maximum number of bears ("mortality limit") can be killed (mostly by hunting; Figure 1) annually by humans without causing population declines (Hamilton & Austin 2002, Lunn & Ethier 2007, British Columbia Ministry of Environment 2010). However, uncertainty in mortality limits is only partially addressed. Estimated biological parameters and calculated mortality limits are treated as point estimates, with uncertainty adjustments dictated by professional judgement (Hamilton & Austin 2004), not probabilistic assessments. As such, "true" mortality limits might be lower than point estimates used (Mattson et al. 1996, McLoughlin 2003). Additionally, outcome uncertainty is

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not incorporated; mortality limits are used to allocate mortality (Austin et al. 2004, Lunn & Ethier 2007), thereby conflating targets with limits.

Debate about grizzly bear management in BC is highly contentious. Independent scientists have recommended more conservative management (McLoughlin 2003, Peek et al. 2003). Societal and political debate led to a single-season moratorium on hunting in 2001 (Peek et al. 2003, British Columbia Ministry of Environment, 2010). Citing sustainability concerns, the European Union has banned the import of BC grizzly bear parts since 2002 (Peek et al. 2003, Knapp 2006). More recently, a consortium of First Nations governments issued a declaration banning grizzly bear hunting across most of coastal BC (Hume 2012). Despite concerns, and concurrent with an increasing number of populations gaining threatened status, hunting mortality increased across the province from 2001–2011 (Figure 1; Ministry of Forests, Lands and Natural Resource Operations 2012, unpublished data).

Clearly, this case study of prominent debate requires empirical insight. We used data on reported mortality and mortality limits (including parameter values used to derive these limits) to explore potential effects of outcome and mortality limit uncertainty on grizzly bear management in BC from 2001–2011. Others have highlighted the need to quantitatively confront management uncertainty (McLoughlin 2003, Peek et al. 2003, McLoughlin & Messier 2004); we additionally provide empirical insight by retrospectively assessing historic management. Specifically, we tested management performance and characterized outcome uncertainty by comparing reported human-caused mortality to allowable levels. We then used simulation modeling to estimate mortality limit uncertainty and to assess how this uncertainty

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might affect overmortality probabilities. Finally, we incorporated outcome and mortality limit uncertainty into a generalizable and transparent method for identifying mortality targets that maintain the probability of overmortality below pre-determined thresholds.

Methods

We conducted our analyses at the Grizzly Bear Population Unit (GBPU) spatial scale, thought to reflect ecologically and demographically relevant sub-populations (British Columbia Ministry of Environment 2010). We divided our study period into the same multi-year allocation periods (2001–2003, 2004–2006, and 2007–2011) used by the British Columbia Ministry of Environment (hereafter "government"). We followed government procedures for setting mortality limits based on estimates of population size, annual allowable mortality (AAM; proportion of a population that can theoretically be removed without causing population declines), and unreported mortality. The government sets mortality limits for the entire population (Eq 1) and for females separately (Eq 2) to account for the sensitivity of populations to female mortality (Harris 1986, Hamilton & Austin 2002, British Columbia Ministry of Environment 2010). The government subtracts predicted non-hunt mortality (e.g. road kill, animal control kills, and illegal hunting) estimates from mortality limits and allocates the remaining mortality to hunting. We additionally calculated female mortality as a percentage of total mortality. Details on mortality limit calculations, and on how limits differed among periods, are provided in Appendices S1 and S2, respectively.

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Eq 1: total mortality limit = population estimate × uncertainty factor × (annual allowable mortality – estimated unreported mortality) × period length – previous period total overmortalities

Eq 2: female mortality limit = 0.3 × population estimate × uncertainty factor × (annual allowable mortality – estimated unreported mortality) × period length – previous period female overmortalities

Management performance and mortality patterns

We assessed management performance across GBPUs and across study periods by calculating the difference between reported mortality and allowable mortality:

Eq 3:

$$\text{percent difference} = 100 \times \frac{\text{reported mortality-allowable mortality}}{\text{allowable mortality}}$$

We further explored patterns of mortality types associated with overmortality events.

We characterized management performance as a function of targeted mortality. Using maximum likelihood estimation, we fit Michaelis-Menton curves to model reported mortality ($U_{reported}$) as a function of allowable mortality (U_{target}), for each period, and for total and female mortality:

Eq 4:

$$U_{actual_i} = a + \frac{b(U_{target_i})}{(c + U_{target_i})} + \varepsilon_i, \qquad \varepsilon_i \sim \text{Negbin}(0, k)$$

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where *i* represents a GBPU-period; a, b, c, are estimated parameters of the curve; ε_i represents residual error; and k is the estimated size parameter of a negative binomial error distribution with a mean of 0. We used this error distribution because targets must be positive integer values. We fit the models using *optim* in R with the Nelder-Mead method and with estimated parameters in log space.

Mortality limit uncertainty and probability of overmortality

We propagated parameter uncertainty to estimate cumulative uncertainty around mortality limits using simulation modeling (Milner-Gulland et al. 2001, Peterman 2004), and assessed how this uncertainty might affect the probability of overmortality. We focused on three key parameters currently treated as point estimates. Because empirically-derived uncertainty estimates are lacking for most BC populations, we derived parameter uncertainty estimates from a literature review (Appendix S3). For each parameter, we took random draws from a continuous uniform distribution centered on existing point estimates. The distributions were bounded by: population estimates: +/- 40% of point estimate; AAM: +/- 2% of population estimate (because AAM is a percentage of population estimate); and unreported mortality: from half to double the point estimate (Appendix S3). We calculated simulated female and total mortality limits by substituting randomly drawn parameter values into Eq 1 and Eq 2. We repeated these simulations 1000 times in each GBPU and period to construct a distribution of realistic mortality limits (the simulated breadth of mortality limit uncertainty). We used the percentage of simulations in which simulated mortality limits fell below actual mortalities as a proxy for overmortality probability.

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Identifying targets that incorporate outcome and mortality limit uncertainty

We used derived distributions of outcome and mortality limit uncertainty to calculate targets for each GBPU that maintained the probability of overmortality below 5% (low risk-tolerant, conservation-prioritizing scenario) or 25% (higher risk-tolerant, hunter-prioritizing scenario), using data from 2007–2011. For a given target, we used a "plug-in" approach (Bolker 2011) to estimate outcome uncertainty from the stochastic component (the negative binomial error) of Eq 4, assuming that the deterministic component (the Michaelis-Menten curve) was fixed at the maximum likelihood estimate. We calculated the intersection of the resultant outcome and mortality limit distributions. For each GBPU, we iteratively repeated this technique with varying target values until we found the maximum value for which the resultant outcome uncertainty distribution intersected with less than the maximum area (the given threshold, 5% or 25%) of the mortality limit distribution (keeping mortality limit distributions fixed). The government reviewed these methods and results and did not identify any methodological or data concerns. We performed all analyses with R 2.14.1 (R Core Team 2012, R Foundation for Statistical Computing).

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Results

Management performance and mortality patterns

Management performance varied across GBPUs and periods, with discrepancies between allowable and realized mortality more pronounced for female than total comparisons (Figure 2, S1, and S2). Mortality was mostly below allowable levels, though overmortalities occurred in at least one period in 26 of the approximately 50 GBPUs open for hunting, and most frequently in

southern and eastern BC (Figure 3). Overmortalities (18 total cases and 33 female cases from 2001-2011) occurred more frequently in GBPUs with smaller allowable mortalities (Figure 2, S1, S2, and S3). In seven GBPUs, overmortality events occurred in two periods, whereas in three GBPUs they occurred in all three (Figure 3). Overmortality events ranged from one bear to 24; overmortality limits were also frequently approached but not exceeded (Figure S3).

The most common factor associated with total overmortalities was unpredicted non-hunting mortality. However, most of the total overmortalities from 2001–2011 (17 of 18, or 94%) could have been avoided with reduced hunting mortality (Figure S3). The most common factor associated with female overmortalities was hunting mortality. Most female overmortalities (25 of 33, or 76%) could have been avoided with reduced hunting mortality (Figure S3).

The female component exceeded 30% of total mortality in 55% of all cases and in 94% of all female overmortality events (Figure 4 A and B, respectively). The female component exceeded 30% of total hunting in 50% of all cases and in 82% of all female overmortality cases (Figure 4 C and D, respectively).

A detailed summary of mortality types associated with overmortality events in each GBPU and period is available by request.

Mortality limit uncertainty and probability of overmortality

Accounting for uncertainty revealed overmortalities might have occurred in 90 of 127 (71%) examined female cases and 89 of 127 (70%) examined total cases. This comprised an additional 45% of female cases and 56% of total cases relative to overmortality assessments that did not consider uncertainty (Figure 5 A and B, S4 A and B, and S5 A and B). Even in the face of

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uncertainty, reducing hunting by half would have reduced the probability of overmortality by an average of 85% for total and 75% for female overmortality cases (Figure 5C, S4C, and S5C), whereas completely eliminating hunting would have reduced the probability of overmortality by an average of 96% for total and 89% for female overmortality cases (Figure 5D, S4D, and S5D).

Identifying targets that incorporate outcome uncertainty and mortality limit uncertainty

To maintain the probability of overmortality below a 5% threshold, mortality targets would need to be reduced by an average of 81% across all GBPUs, and by 100% in 15 (Figure 6 A, B and E). For a 25% threshold, mortality targets would need to be reduced by an average of 47%, and by 100% in four GBPUs (Figure 6 C, D, and F).

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Discussion

This case study illustrates the importance of assessing management performance and considering uncertainty in management. We found that grizzly bear overmortalities were spatially widespread, occurred repeatedly, and were more frequent in females. Considering uncertainty around mortality limits revealed that many additional populations might have experienced overmortalities. A target-setting framework that incorporated outcome and mortality limit uncertainty identified that considerable reductions in targeted mortality would be required to improve management performance.

Considerations

Our approach was descriptive and illustrative, not prescriptive. Estimated probabilities of overmortality and reductions in targeted mortalities would change if empirically-derived and area-specific ranges of uncertainty were known for each GBPU. Similarly, given that the outcome uncertainty was estimated from management performance over a short time period, our derived distributions likely underestimated the true range. Additionally, because we used parameter uncertainty ranges considerably narrower than those found in the literature, overmortality risks and target reductions identified might have been underestimated. Finally, our analyses did not address government assumptions in setting specific parameter point estimates for each area, or in adjusting estimates among periods, which could have affected our ability to detect overmortalities (Appendix S2). Given these considerations, our results could be interpreted as suggestive of minimal requirements for improving performance; should management adapt our approach, we additionally recommend obtaining more accurate data, with management re-evaluated, updated, and refined iteratively.

Additional sources of uncertainty

Our analyses addressed only a subset of uncertainty in wildlife management. For example, there is additional uncertainty about the appropriateness of models used in setting limits ("model selection error"; Harwood & Stokes 2003); genetic, phenotypic, or social effects of exploitation (e.g. Swenson et al. 1997, Wielgus et al. 2001, Darimont et al. 2009); time required for population recovery (Reynolds 2003); effects of declining food availability (Levi et al. 2012); and the cumulative effect of other anthropogenic disturbances such as logging, mineral extraction, roads, and development (Mattson et al. 1996, Weaver et al. 1996, Peek et al. 2003).

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Management performance and outcome uncertainty

We explored multiple processes that contribute to outcome uncertainty. First, hunting mortality, especially in females, was often higher than expected. Guidelines that encourage hunters to avoid targeting females seem ineffective given that female mortality consistently exceeded the 30% threshold dictated by government models (e.g. Harris 1986, Hamilton & Austin 2002, British Columbia Ministry of Environment 2010). Similar barriers to limiting female mortality might also apply to other wildlife species in which sexes are not particularly dimorphic, with similar concerns about population dynamics (e.g. caribou, Festa-Bianchet et al. 2011). Importantly, most total and female overmortality events could have been prevented through hunting reductions. However, mortality sources beyond management control — including road kill, animal control kills, and illegal hunting — were also important, highlighting that measures beyond hunt reductions are likely required to safeguard populations. Finally, that overmortality occurred repeatedly in many areas highlights a consequence of not explicitly accounting for outcome uncertainty.

Mortality limit uncertainty

In addition to outcome uncertainty, uncertainty not explicitly accounted for in estimating biological parameters, such as mortality limits, can also lead to excessive mortality. By accounting for mortality limit uncertainty, our simulations revealed that overmortality events might have occurred in many cases in which mortality did not exceed mortality limit point estimates. We found that the probability of overmortality would have decreased considerably had hunting been reduced or eliminated, as expected for any system in which hunting

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comprises most mortality. This provides management a direct and easily controlled route to reducing the probability of over-exploitation.

Identifying targets that incorporate uncertainty

Our framework for transparently incorporating uncertainty identified targets that reduce the probability of over-exploitation. Of particular relevance to managers, the public, and management alike is how mortality management might change if this approach were implemented. Our simulations revealed the need for considerable reductions, consistent with the conservative bet-hedging recommended for cautious management (Reckhow 1994, Mattson et al. 1996). Importantly, given that female mortality seems difficult to control independently of total mortality, a given GBPU's total target mortality would need to be reduced sufficiently to maintain both total and female overmortality probabilities below thresholds. Although the acceptable probabilities of overmortality used in our approach (5% or 25%) were arbitrary, they might represent a low risk-tolerant, conservation prioritizing scenario and a higher risk-tolerant, hunter-prioritizing scenario respectively. This approach provides the first steps to a full decision analysis framework, a quantitative approach for weighing various management options that might be appropriate in future management deliberations (Reckhow 1994, Peterman 2004)

Importance of incorporating best-practices from other disciplines

This study illustrates the merit of incorporating approaches from other disciplines and taxa into wildlife management. Whereas BC grizzly bear management incorporates data and management techniques from grizzly bear management in other jurisdictions, it does not

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incorporate some promising methods from other disciplines. For example, our approach, which relies on the principle that targets should be set sufficiently low (and lower than most of the estimated range of mortality limits; Reckhow 1994, Caddy & McGarvey 1996, Prager et al. 2003) is used in fisheries but far less commonly in wildlife management, highlighting the need for better integration of best practices across taxa and disciplines.

Conclusion

Science can provide valuable insight into management issues often mired in heated debate. Management often occurs within contentious social environments, with interest groups advocating strongly for different scenarios, informed by varying ethical perspectives and philosophies (Lackey 1998, Linnell et al. 2010, Paquet & Darimont 2010, Darimont & Paquet 2012, Phillis et al. 2012). Science can inform such debate by assessing the ability of management to achieve objectives and by transparently communicating risks associated with various scenarios. We suggest that other management systems might benefit from retrospective examinations. These could be conducted as a part of the management process or, as in this study, by third parties. Results and predictions from such examinations in any system could help to communicate likely outcomes while simultaneously improving future management performance.

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References

Armitage, D.R., Plummer, R., Berkes, F., Arthur, R.I., Charles, A.T., Davidson-Hunt, I.J., Diduck, A.P., Doubleday, N.C., Johnson, D.S., Marschke, M., McConney, P., Pinkerton, E.W. & Wollenberg, E.K. (2009). Adaptive co-management for social-ecological complexity. *Front Ecol Environ*, 7, 95–102.

Austin, M.A., Heard, D.C. & Hamilton, A.N. (2004). Grizzly bear (Ursus arctos) harvest management in British Columbia. *British Columbia Ministry of Water, Land and Air Protection. Victoria, Canada*.

Bolker, B.M. (2011). *Ecological models and data in R*. Princeton University Press.

British Columbia Ministry of Environment, Fish, Wildlife and Habitat Branch. (2010). Grizzly bear hunting: frequently asked questions.

Caddy, J.F. & McGarvey, R. (1996). Targets or limits for management of fisheries? *N Am J Fish Manage*, 16, 479–487.

Darimont, C.T., Carlson, S.M., Kinnison, M.T., Paquet, P.C., Reimchen, T.E. & Wilmers, C.C. (2009). Human predators outpace other agents of trait change in the wild. *Proc Natl Acad Sci U S A*, 106, 952–954.

For Peer Review

Darimont, C.T. & Paquet, P.C. (2012). Lessons from the Lorax. a scientists' lament and a plea for ethics. In: *Faltering Light* (ed. Wright, A.). Cold Coast Press.

Festa-Bianchet, M., Ray, J.C., Boutin, S., Côté, S.D. & Gunn, A. (2011). Conservation of caribou (Rangifer tarandus) in Canada: an uncertain future. *Can J Zool*, 89, 419–434.

Hamilton, A.N. & Austin, M. (2004). *Estimating grizzly bear (Ursus arctos) population size in British Columbia using an expert-based approach*. British Columbia Ministry of Water, Land and Air Protection.

Hamilton, A.N. & Austin, M.A. (2002). Grizzly bear harvest management in British Columbia: background report. *British Columbia Ministry of Water, Land, and Air Protection, Biodiversity Branch, Victoria, British Columbia, Canada*.

Harris, R.B. (1986). Modeling sustainable harvest rates for grizzly bears. *Unpublished manuscript*.

Harwood, J. & Stokes, K. (2003). Coping with uncertainty in ecological advice: lessons from fisheries. *Trends Ecol Evol*, 18, 617–622.

Hilborn, R. & Mangel, M. (1997). *The ecological detective: confronting models with data*. Princeton University Press.

Holt, C.A. & Peterman, R.M. (2006). Missing the target: uncertainties in achieving management goals in fisheries on Fraser River, British Columbia, sockeye salmon (Oncorhynchus nerka). *Can J Fish Aquat Sci*, 63, 2722–2733.

Hume, M. (2012). B.C. first nations ban trophy bear hunting. *The Globe and Mail*.

Knapp, A. (2007). *A review of the European Union's import policies for hunting trophies*. A TRAFFIC Europe Report for the European Commission. Brussels, Belgium.

For Peer Review

Lackey, R.T. (1998). Seven pillars of ecosystem management. *Landscape Urban Plan*, 40, 21–30.

Levi, T., Darimont, C.T., MacDuffee, M., Mangel, M., Paquet, P. & Wilmers, C.C. (2012). Using Grizzly Bears to Assess Harvest-Ecosystem Tradeoffs in Salmon Fisheries. *PLoS Biol*, 10, e1001303.

Linnell, J.D.C., Broseth, H., Odden, J. & Nilsen, E.B. (2010). Sustainably harvesting a large carnivore? Development of Eurasian Lynx populations in Norway during 160 years of shifting policy. *Environ Manage*, 45, 1142–1154.

Lunn, K. & Ethier, T. (2007). *Grizzly bear harvest management*. British Columbia Ministry of Environment, Fish and Wildlife Branch.

Mattson, D.J., Herrero, S., Wright, R.G. & Pease, C.M. (1996). Science and management of Rocky Mountain grizzly bears. *Conserv Biol*, 10, 1013–1025.

McLoughlin, P.D. (2003). Managing risks of decline for hunted populations of grizzly bears given uncertainty in population parameters. University of Alberta.

McLoughlin, P.D. & Messier, F. (2004). Relative contributions of sampling error in initial population size and vital rates to outcomes of population viability analysis. *Conserv Biol*, 18, 1665–1669.

Miller, S. (1990). Population management of bears in North America. *Int C Bear*, 8.

Milner-Gulland, E.J., Shea, K., Possingham, H., Coulson, T. & Wilcox, C. (2001). Competing harvesting strategies in a simulated population under uncertainty. *Anim Conserv*, 4, 157–167.

Ministry of Forests, Lands and Natural Resource Operations. (2012). British Columbia grizzly bear population estimate for 2012.

For Peer Review

Paquet, P.C. & Darimont, C.T. (2010). Wildlife conservation and animal welfare: two sides of the same coin. *Anim Welfare*, 19, 177–190.

Peek, J., Beecham, J., Garshelis, D., Messier, F., Miller, S. & Strickland, D. (2003). *Management of grizzly bears in British Columbia - a review by an independent scientific panel*. Submitted to: Minister of Water, Land and Air Protection Government of British Columbia Victoria, B.C.

Peek, J.M., Pelton, M.R., Picton, H.D., Schoen, J.W. & Zager, P. (1987). Grizzly bear conservation and management: a review. *Wildlife Soc B*, 15, 160–169.

Peterman, R.M. (2004). Possible solutions to some challenges facing fisheries scientists and managers. *ICES J Mar Sci*, 61, 1331–1343.

Phillis, C.C., O'Regan, S.M., Green, S.J., Bruce, J.E.B., Anderson, S.C., Linton, J.N., Earth2Ocean Research Derby & Favaro, B. (2012). Multiple pathways to conservation success. *Conserv Lett*, 1–9.

Prager, M.H., Porch, C.E., Shertzer, K.W. & Caddy, J.F. (2003). Targets and limits for management of fisheries: a simple probability-based approach. *N Am J Fish Manage*, 23, 349–361.

Reckhow, K. (1994). Importance of scientific uncertainty in decision making. *Environ Manage*, 18, 161–166.

Regan, H.M., Ben-Haim, Y., Langford, B., Wilson, W.G., Lundberg, P., Andelman, S.J. & Burgman, M.A. (2005). Robust decision-making under severe uncertainty for conservation management. *Ecol Appl*, 15, 1471–1477.

For Peer Review

Reynolds, J.D. (2003). Life histories and extinction risk. In: *Macroecology* (eds. Blackburn, T.M. & Gaston, K.J.). Blackwell Publishing, Oxford, UK, pp. 195–217.

Ross, P.I., Jalkotzy, M.G. & Gunson, J.R. (1996). The quota system of cougar harvest management in Alberta. *Wildlife Soc B*, 24, 490–494.

Swenson, J.E., Sandegren, F., Söderberg, A., Bjärvall, A., Franzén, R. & Wabakken, P. (1997). Infanticide caused by hunting of male bears. *Nature*, 386, 450–451.

Treves, A. (2009). Hunting for large carnivore conservation. *J Appl Ecol*, 46, 1350–1356.

Wielgus, R.B., Sarrazin, F., Ferriere, R. & Clobert, J. (2001). Estimating effects of adult male mortality on grizzly bear population growth and persistence using matrix models. *Biol Conserv*, 98, 293–303.

Figure 1. Total grizzly bear (*U. arctos horribilis*) mortality from hunting (solid-black line) and non-hunting sources (dashed line) in British Columbia, Canada, from 2001–2011. A province-wide moratorium on the trophy hunt during one of two hunting seasons caused lower hunting mortality in 2001.

Figure 2. Management performance for A) total and B) female mortality in Grizzly Bear (*U. arctos horribilis*) Population Units (GBPUs) in British Columbia, Canada, 2007–2011 (see SI for additional periods). Black curve is a Michaelis-Menten curve fitted by maximum likelihood, assuming a negative binomial error distribution. Red dashed line indicates a 1:1 relationship; solid red dots above this line signal GBPU-level overmortality events. Grey-shaded region encompasses the 80% prediction interval (smoothed for visual purposes). Inset histograms

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show the distribution of GBPU-level percent difference between observed and allowable mortalities; red bars to the right of red dashed lines indicate overmortality events.

Figure 3. Number of allocation periods (2001–2003, 2004–2006, or 2007–2011) in which female or total overmortality occurred in Grizzly Bear (*U. arctos horribilis*) Population Units (GBPUs) of British Columbia, Canada. Shown are 2009 GBPU boundaries. Hunting is not allowed in areas denoted as “threatened”, “extirpated”, or “closed to hunting”. One additional GBPU (Blackwater-West Chilcotin) has been reclassified as threatened as of 2012.

Figure 4. Female mortality as percent of total mortality across Grizzly Bear (*U. arctos horriblis)* Population Units (GBPUs) in British Columbia, Canada, and allocation periods (2001–2003, 2004–2006, and 2007–2011). (A) female mortality as percent of all mortality, (B) female mortality as percent of all mortality in female overmortality events, (C) female hunting mortality as percent of all hunting mortality, and (D) female hunting mortality as a percent of all hunting mortality in female overmortality events. Vertical red lines indicate 30%, the threshold below which female mortality must remain for total mortality limits to be theoretically sustainable according to harvest policy.

Figure 5. A) Total and female overmortalities detected in Grizzly Bear (*U. arctos horribilis*) Population Units (GBPUs) of British Columbia, Canada, from 2007–2011 (see SI for additional periods), given actual hunting mortality levels and without consideration of mortality limit uncertainty. Blue indicates GBPUs with detected overmortality whereas white indicates GBPUs without. B–D) Simulated probability of total or female overmortality, incorporating uncertainty around mortality limits. Panel B shows simulated probability of overmortality given actual

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reported mortality rates; panels C and D show what the probability of overmortality would be had hunting mortality been reduced by 50% or 100%, respectively, assuming other sources of mortality remained unchanged. Increasingly dark red indicates an increasing probability of overmortality in a given period. Grizzly bears have been extirpated from dark-grey areas. Light-grey areas indicate GBPUs in which populations are either threatened or were closed to hunting during the study period.

Figure 6. A–D) Illustration of our method for setting female (A and C) and total (B and D) mortality targets, with maximum probability of overmortality of 5% (A and B) or 25% (C and D) by integrating outcome uncertainty (grey distribution) and mortality limit uncertainty (red distribution), using the Stewart Grizzly Bear (*U. arctos horribilis*) Population Unit (GBPU) as an example. Targets (dashed black lines) from this approach are set so that the resulting outcome uncertainty distribution (grey distribution) overlaps with a maximum of 5% or 25% of the mortality limit uncertainty distribution (red distribution). Red vertical lines represent mortality limits (conflated with targets in previous periods under harvest management policy) set by the government in 2007–2011. Magnitudes of recommended target reductions are shown by black double-headed arrows. E-F) Reduction in mortality targets, relative to 2007–2011 limits (conflated with targets at the time) required to maintain the probability of both female and total overmortality below E) 5% or F) 25%. Increasingly dark red indicates increasing target reductions identified.

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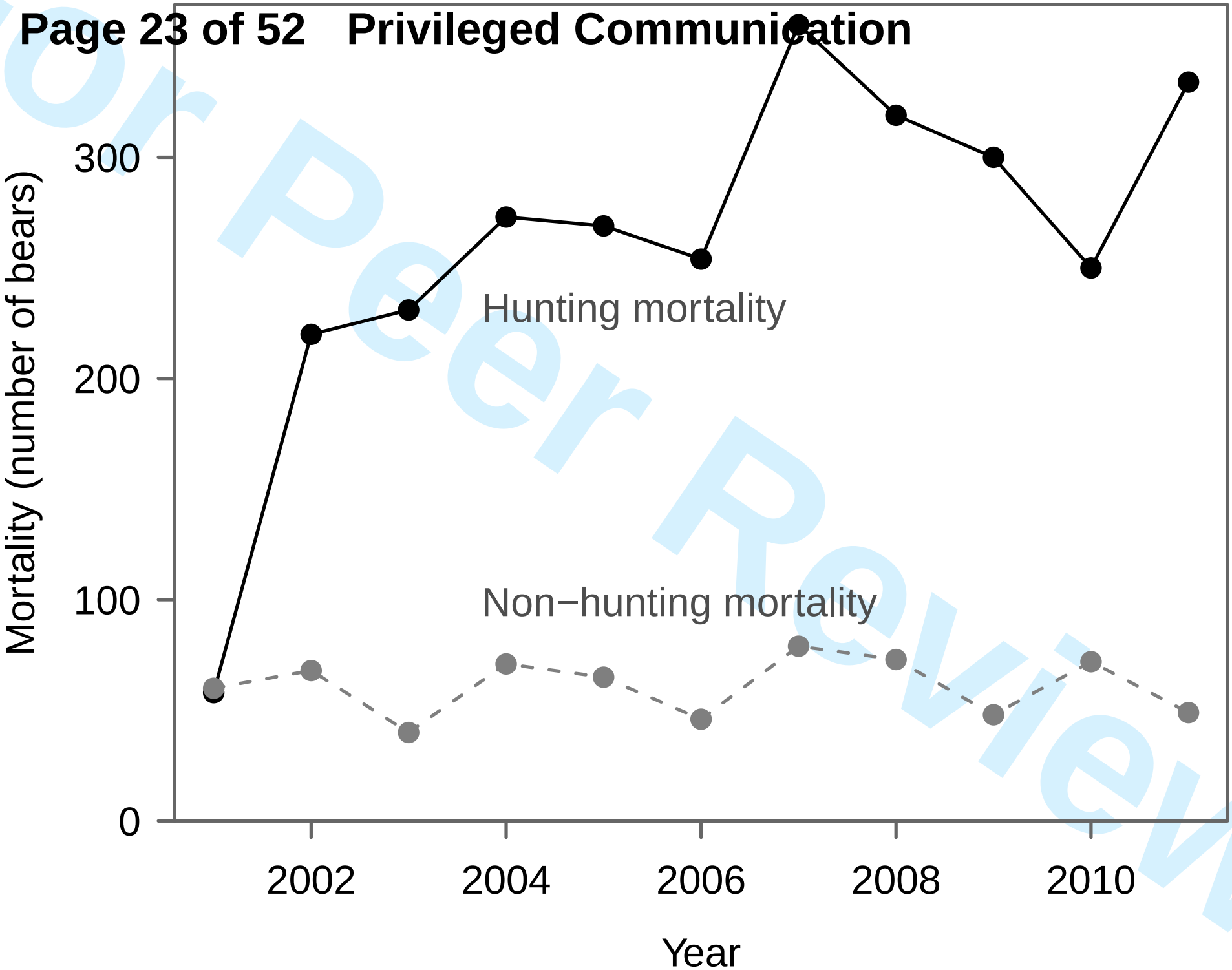

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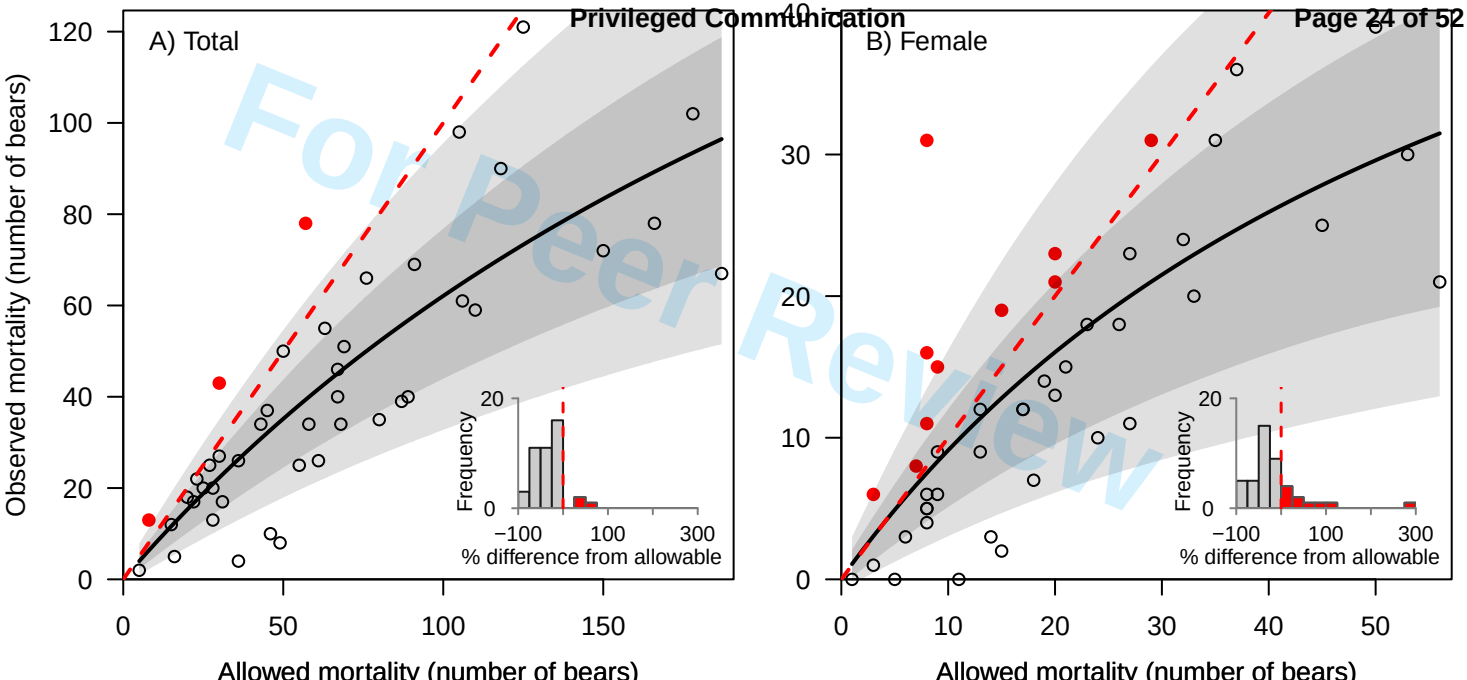

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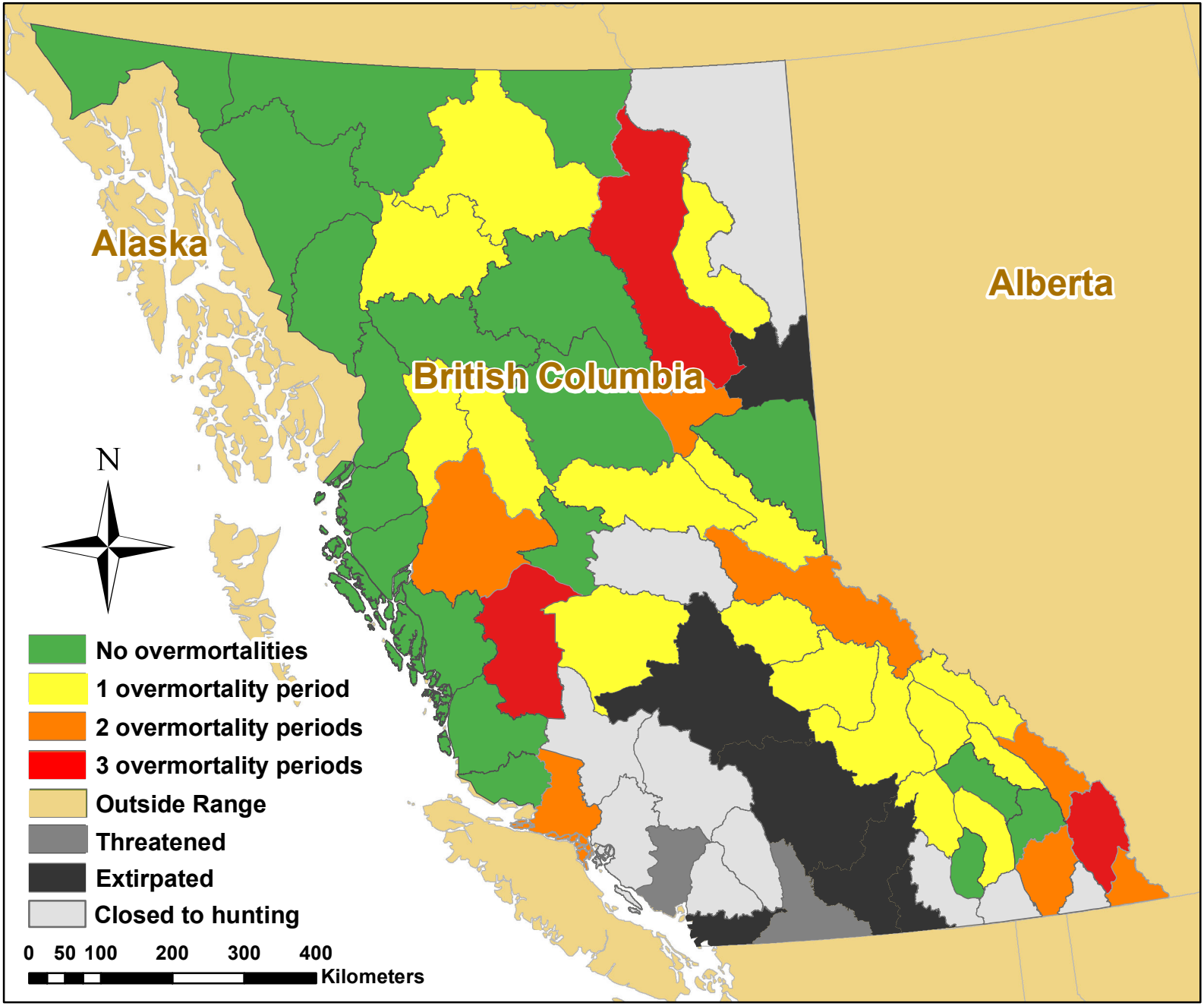
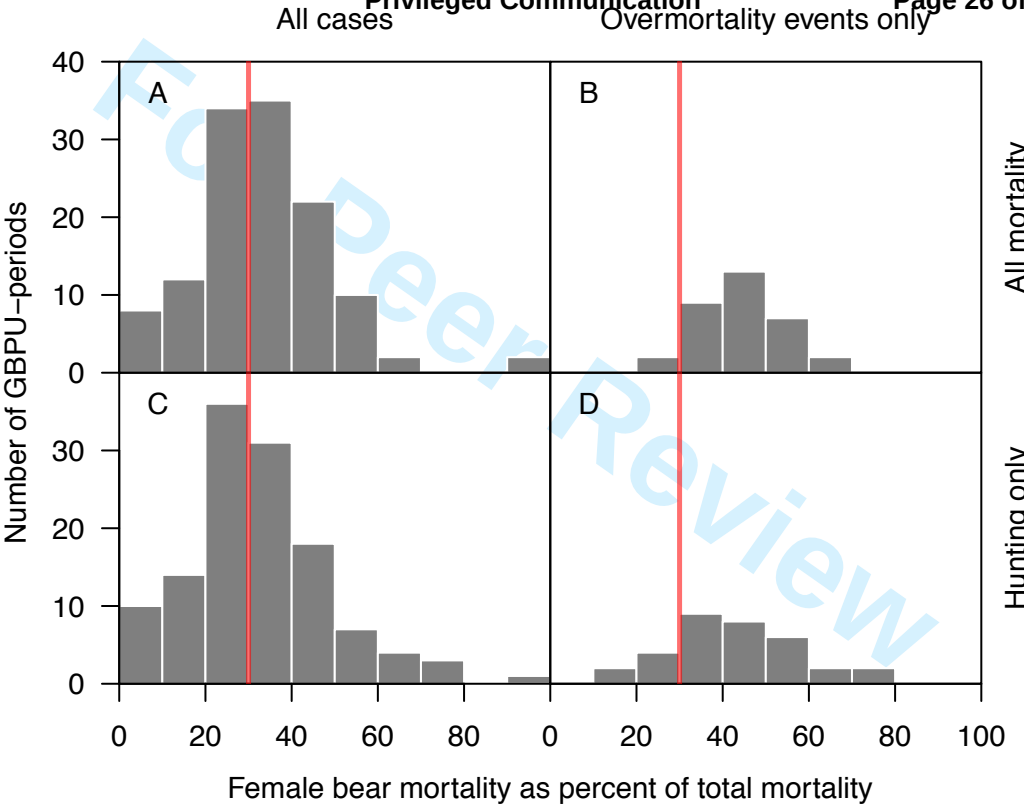

Alaska
Alberta
British Columbia
N
No overmortalities
1 overmortality period
2 overmortality periods
3 overmortality periods
Outside Range
Threatened
Extirpated
Closed to hunting
0 50 100 200 300 400
Kilometers



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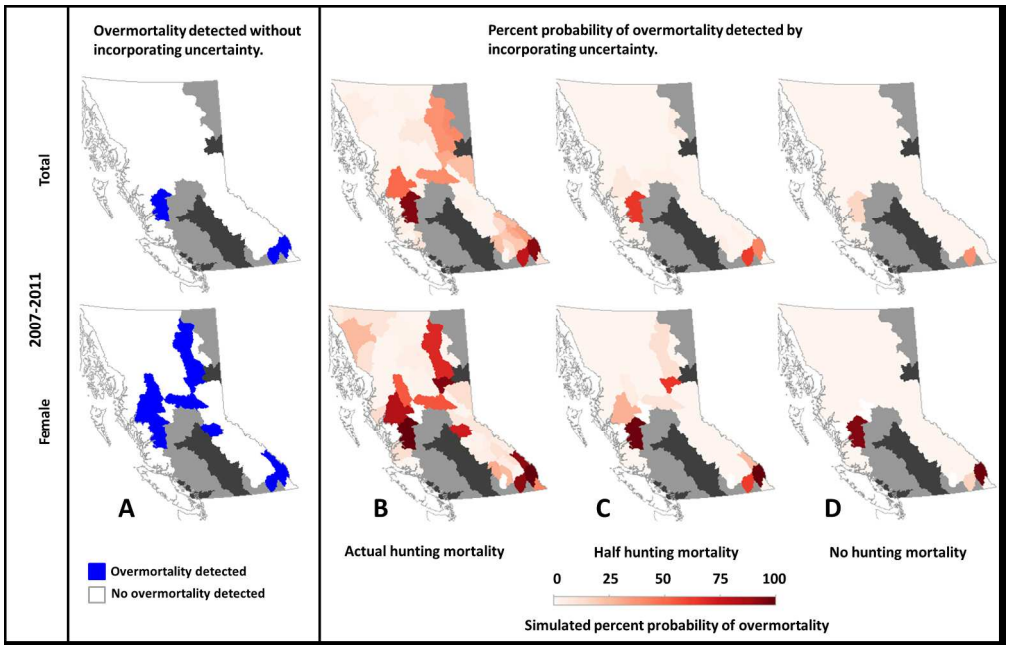

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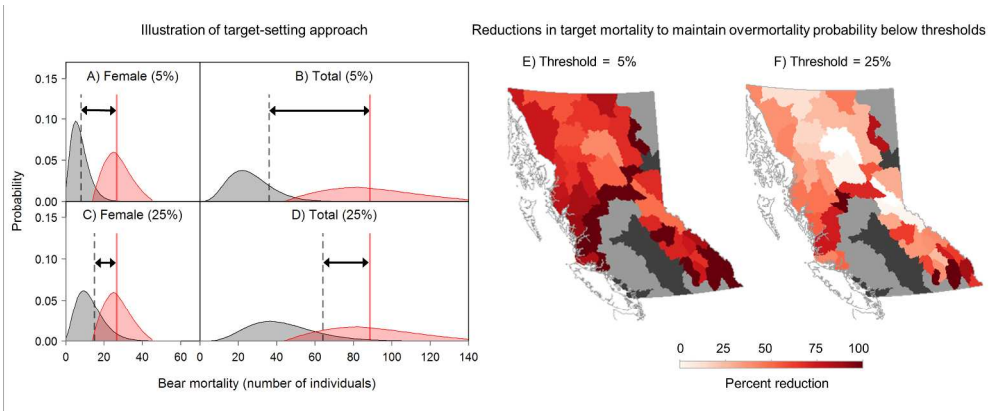

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Supporting Information

Confronting uncertainty in wildlife management: performance of grizzly bear management in British Columbia, Canada

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This document includes:

Appendix S1 - Parameters for setting mortality limits
Appendix S2 - Factors affecting detectability of overmortality
- Appendix S2 Figure 1
- Appendix S2 Figure 2
- Appendix S2 Figure 3
- Appendix S2 Figure 4
- Appendix S2 Figure 5
- Appendix S2 Figure 6

Appendix S3 - Parameter uncertainty ranges
- Appendix S3 Figure 1

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Figure S3
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Appendix S1 Parameters for setting mortality limits

The government estimates parameters at both the Grizzly Bear Population Unit (GBPU) and Management Unit (MU) scales. We performed all analyses first at the finer MU scale, then scaled up to the GBPU scale which is thought to reflect ecologically and demographically relevant sub-populations (British Columbia Ministry of Environment 2010).

1.1 Population estimates

The government sets population estimates using a variety of methods including: (1) DNA capture-recapture surveys (Hamilton & Austin 2004; approximately 14% of GBPUs as of 2010 (Hamilton, A.N. 2008, Apps 2010); (2) model estimates based on habitat suitability regression analyses (56% of GBPUs as of 2010); and (3) expert opinion (30% of GBPUs as of 2010).

1.2 Uncertainty factor

Uncertainty factors are multipliers used by the government that reduce the population estimate in some GBPUs and periods (0% of GBPUs in 2001-2003, 98% in 2004-2006, and 14% in 2007-2011) to account for uncertainty. The magnitude of corrections has been reduced in the most recent period (Appendix S2). The calculations used in setting individual factors are not publicly available.

1.3 Annual allowable mortality (AAM)

Annual allowable mortality is the annual proportion of the population for which human-caused mortality is assumed sustainable by the government, informed by Harris's (1986) work in Montana that estimated maximum AAM (rate with a 10% or less probability of causing chronic declines) in highly productive populations to be somewhere between 6.35% and 6.6%. AAM values were highest in 2007-2011 (Appendix S2).

1.4 Period length

Period length is three years for 2001–2003 and 2004–2006, and five years for 2007–2011.

1.5 Estimated unreported mortality

Estimated unreported mortality is an estimated proportion of the population killed annually by humans through sources including poaching, illegal kills, and unreported road accidents.

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1.6 Previous period overmortalities

Previous period (total or female) overmortalities account for the number of bears killed in excess of mortality limits in the previous period. Note that the government did not carry overmortalities forward from the 2001–2003 to 2004–2006 period.

Appendix S2 Factors affecting detectability of overmortality

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The government tests for overmortality events by comparing calculated allowable mortality limits to recorded (*i.e.,* actual) mortality. As such, detected frequency of overmortalities can be affected by changes to recorded mortality rates or, importantly, by changes to how mortality limits are calculated (Eq 1 and Eq 2). Additionally, the temporal scale at which these comparisons are made can affect results. Specifically, longer periods might conceal overmortality events that occur over shorter periods. Accordingly, we examined two aspects that could influence detectability of overmortality events.

First, we examined the distribution of AAM estimates, unreported mortality estimates, uncertainty corrections, and total allowable mortalities across allocation periods. Second, we tested the effect of allocation period length on detectability of overmortality events. We compared period overmortality to average yearly overmortality. **Period overmortality** was the number of overmortalities in a given period. **Yearly overmortality** was assessed by comparing *annual* mortality limits (period limit divided by period length) to *annual* reported mortality for GBPU. For each period, we also compared counts of how many GBPUs experienced a period overmortality to counts of how many GBPUs experienced at least one yearly overmortality.

Detection of overmortality events was likely influenced by harvest management procedures that differed among periods. We found that: i) AAM estimates increased across periods (Appendix S2 Figure 1, ii) estimates of unreported mortality decreased between 2001-2003 and subsequent periods (Appendix S2 Figure 2) and, iii) uncertainty corrections, introduced in period 2, were considerably reduced in period 3 (Appendix S2 Figure 3). Accordingly, resultant mortality limits also increased across periods (Appendix S2 Figure 4). Whereas these changes may have reflected the inclusion of more

accurate data, we note that apparent reductions in overmortality frequencies may have been due to adjusted parameters, not reduced mortality.

We also found evidence that assessing overmortality across entire periods had the effect of concealing inter-annual variation in mortality rate (Appendix S2 Figure 5); years in which mortality was unusually high were balanced by years in which it was unusually low. This concealment increased with the longer (*i.e.* five year) duration introduced in period 3, resulting in an apparent reduction of overmortality frequency compared to previous two periods, each of which were three years in length. This effect was most pronounced with total overmortality measures; whereas counts of whole period and yearly female overmortalities followed a similar pattern across periods, counts of whole period total overmortalities decreased considerably in period 3 whereas counts of yearly overmortalities did not (Appendix S2 Figure 6).

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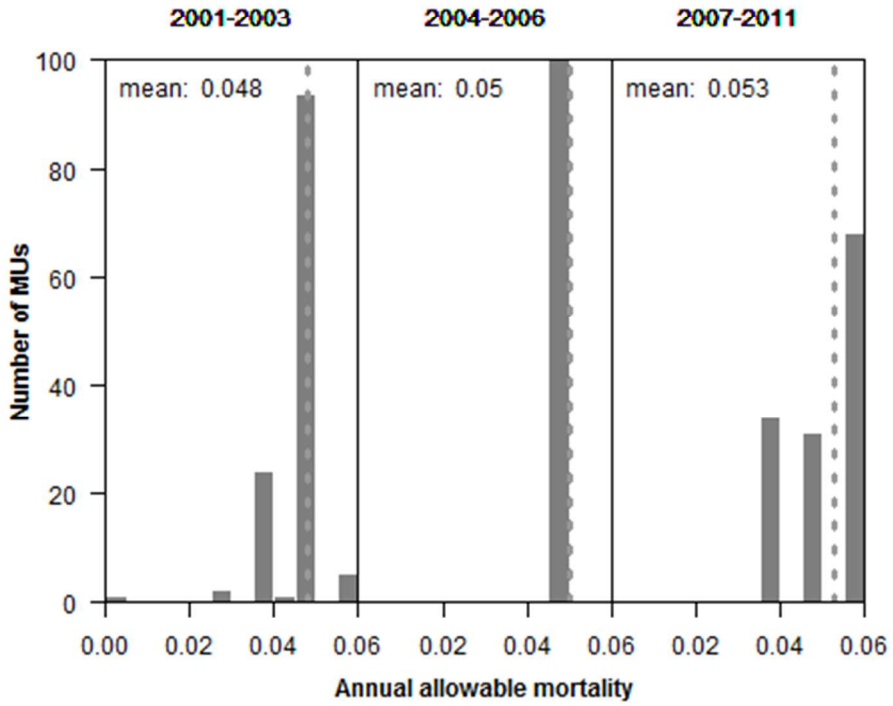
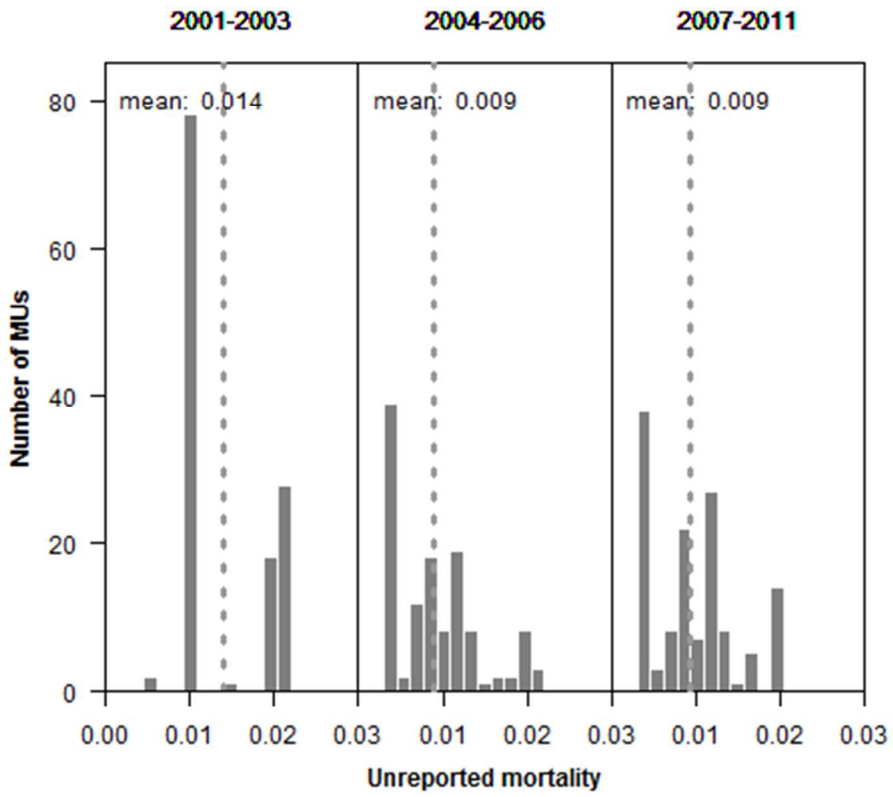


Appendix S2 Figure 1 Management unit annual allowable mortality estimates used in grizzly bear (U. arctos horribilis) management, British Columbia, Canada, 2001-2003, 2004-2006, and 2007-2011. Dashed line represents the period mean value.



Appendix S2 Figure 2 Management unit unreported mortality estimates used in grizzly bear *(U. arctos horribilis)* management, British Columbia, Canada, 2001-2003, 2004-2006, and 2007-2011. Dashed line represents the period mean value.

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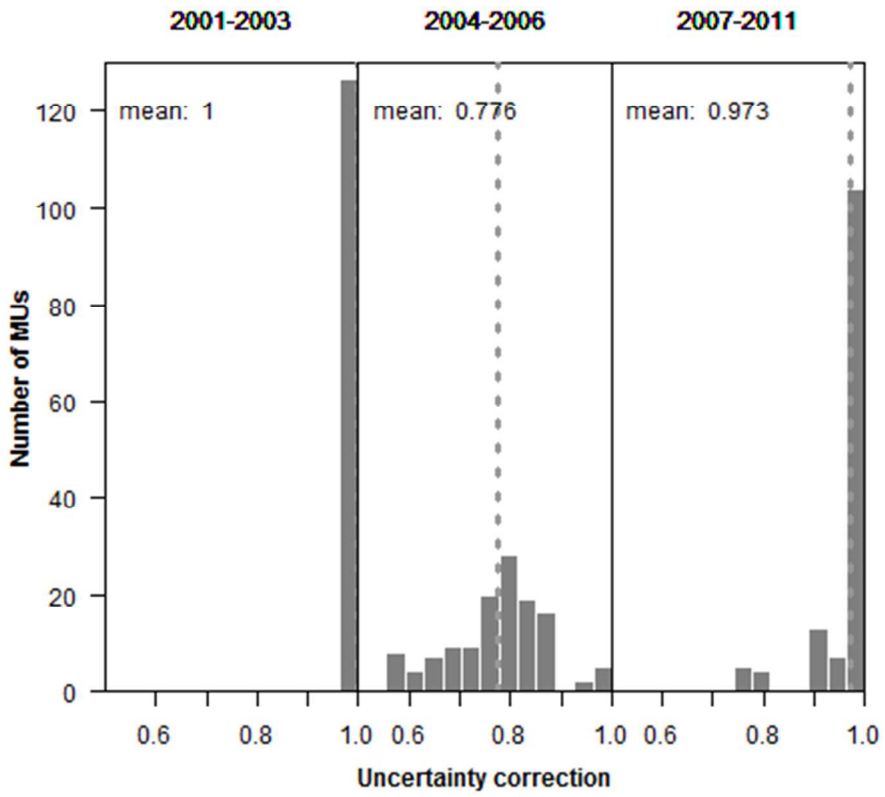
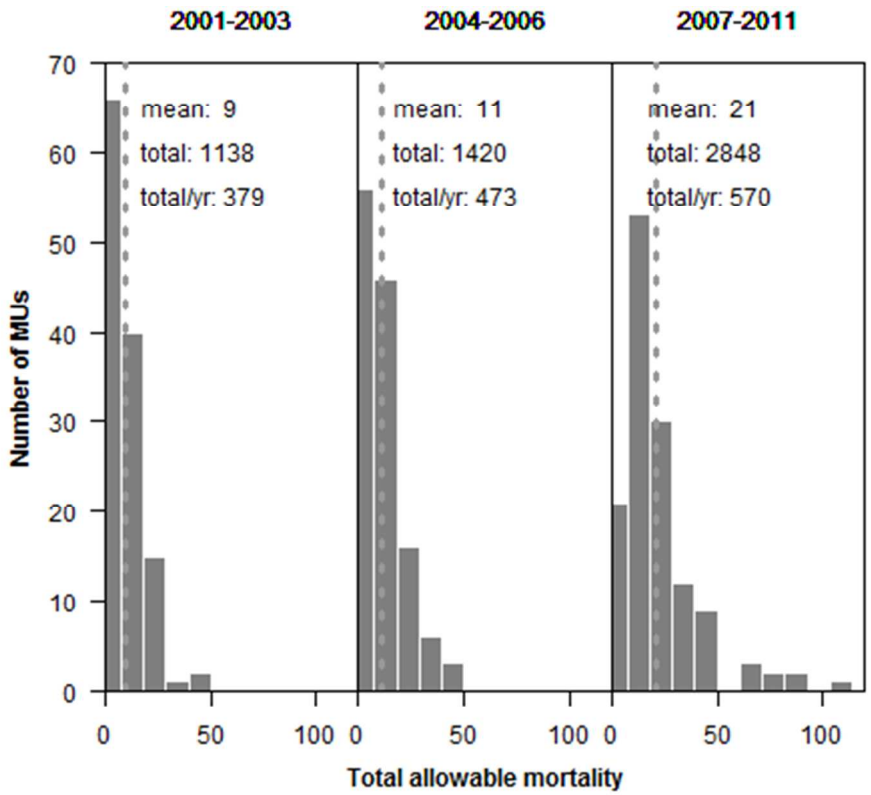


Appendix S2 Figure 3 Management unit uncertainty corrections used in grizzly bear *(U. arctos horribilis)* management, British Columbia, Canada, 2001-2003, 2004-2006, and 2007-2011. Dashed line represents the period mean value.

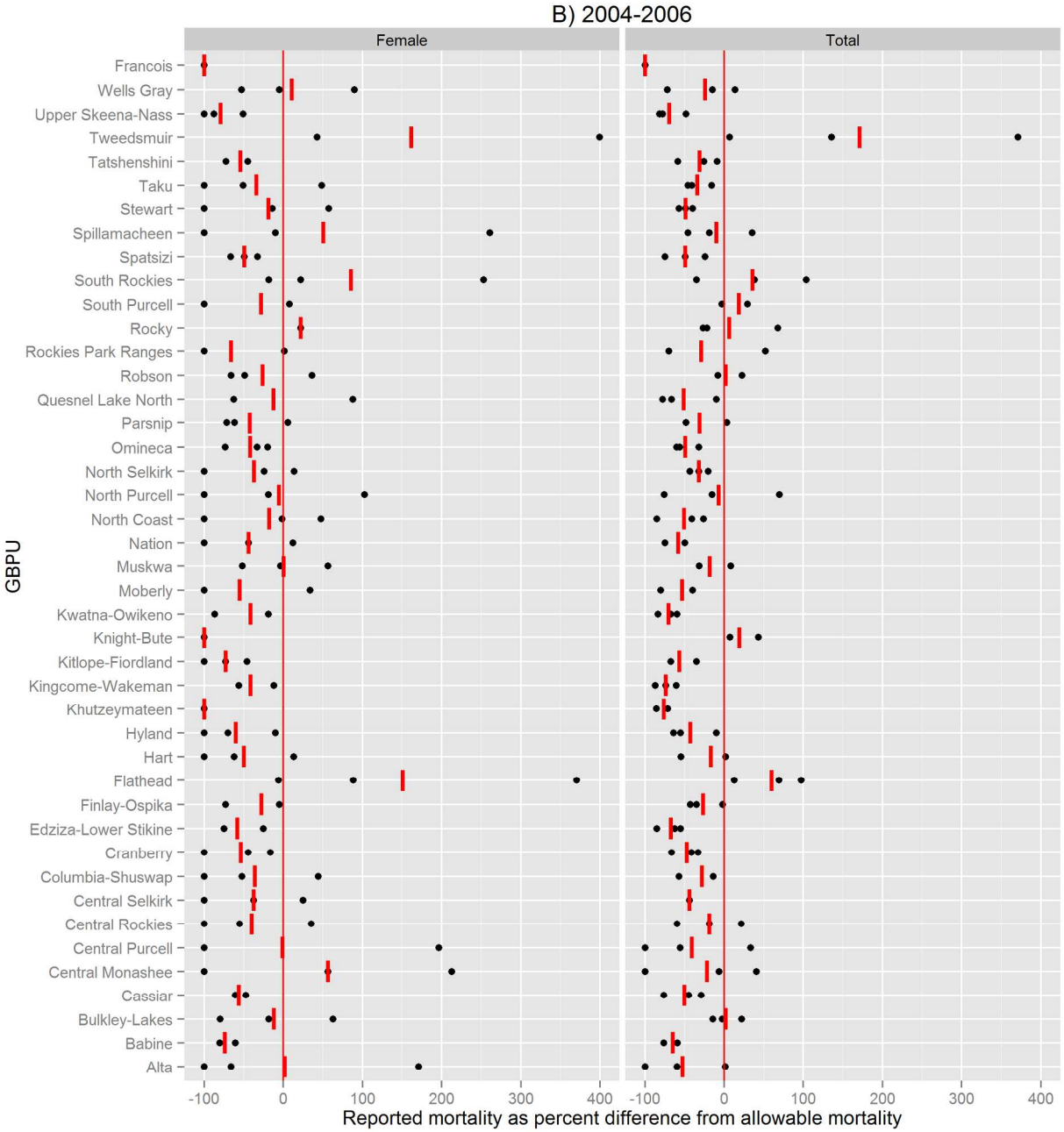
Review



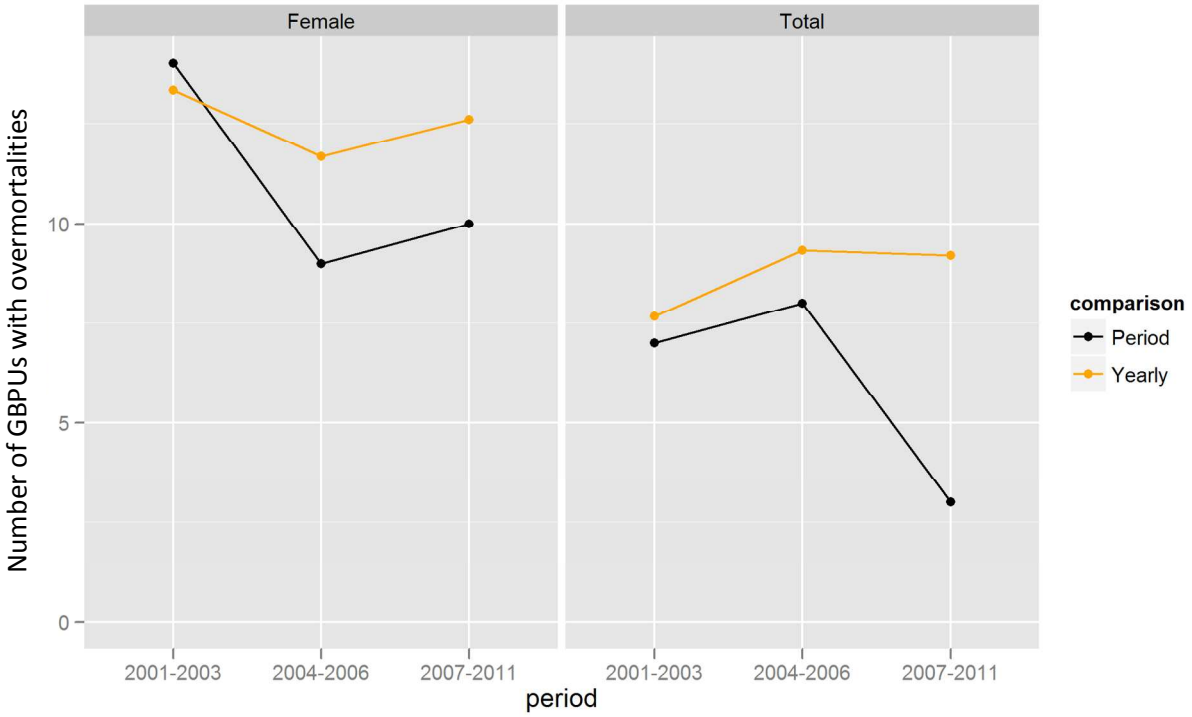
Appendix S2 Figure 4 Management unit total allowable mortality estimates used in grizzly bear *(U. arctos horribilis)* management, British Columbia, Canada, 2001-2003, 2004-2006, and 2007-2011. Dashed line represents the period mean value.

Review

A) 2001-2003
Female
Total
GBPU
Wells Gray
Upper Skeena-Nass
Tweedsmuir
Tatshenshini
Taku
Tagish
Stewart
Spillamacheen
Spatsizi
South Rockies
South Purcell
Rocky
Rockies Park Ranges
Robson
Quesnel Lake North
Parsnip
Omineca
North Selkirk
North Purcell
North Monashee
North Coast
Muskwa
Moberly
Kwatna-Owikeno
Knight-Bute
Klinaklini-Homathko
Kitlope-Fiordland
Kingcome-Wakeman
Khutzeymateen
Hyland
Hart
Francois
Flathead
Finlay-Ospika
Edziza-Lower Stikine
Cranberry
Columbia-Shuswap
Central Selkirk
Central Rockies
Central Purcell
Cassiar
Bulkley-Lakes
Bowron
Blackwater-West Chilcotin
Babine
Alta
-100
0
100
200
300
Reported mortality as percent difference from allowable mortality


B) 2004-2006
Female
Total
GBPU
Francois
Wells Gray
Upper Skeena-Nass
Tweedsmuir
Tatshenshini
Taku
Stewart
Spillamacheen
Spatsizi
South Rockies
South Purcell
Rocky
Rockies Park Ranges
Robson
Quesnel Lake North
Parsnip
Omineca
North Selkirk
North Purcell
North Coast
Nation
Muskwa
Moberly
Kwatna-Owikeno
Knight-Bute
Kitlope-Fiordland
Kingcome-Wakeman
Khutzeymateen
Hyland
Hart
Flathead
Finlay-Ospika
Edziza-Lower Stikine
Cranberry
Columbia-Shuswap
Central Selkirk
Central Rockies
Central Purcell
Central Monashee
Cassiar
Bulkley-Lakes
Babine
Alta
-100
0
100
200
300
400
Reported mortality as percent difference from allowable mortality

Appendix S2 Figure 5 Comparison of percent difference between actual and allowable mortality in Grizzly Bear *(U. arctos horribilis)* Population Units (GBPUs) of British Columbia, Canada, 2001-2011, at the yearly scale (black dots) and the period scale (red dashes). Vertical red line indicates a percent difference of 0 (reported mortality was equal to allowable). Periods examined are A) 2001-2003, B) 2004-2006, and C) 2007-2011



Appendix S2 Figure 6 Comparison of detected Grizzly Bear (*U. arctos horribilis)* Population Unit (GBPU) female and total overmortalities in British Columbia, Canada, from 2001-2003, 2004-2006, and 2007-2011. Black lines are number of GBPUs with detected period overmortalities (reported mortality for period exceeding allowable mortality for period). Orange lines are number of GBPUs with detected yearly overmortalities (reported yearly mortality exceeded period mortality/period length at least once in given period).

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Appendix S3 Parameter uncertainty ranges

3.1 Population estimate

In our simulation model scenarios, we randomly increased or decreased population estimates by up to 40%. Overall, the average change in government GBPU population estimates between 2004 and 2012 was approximately -10%, though adjustments ranged from -89% to 130% (Appendix S3 Figure 1A; Hamilton et al. 2004, Hamilton 2008, and Ministry of Forests, Lands and Natural Resource Operations 2012). These adjusted values reflected modifications to population estimates, such as inclusion of new capture-recapture data or model refinement, not changes in the population (Ministry of Forests, Lands and Natural Resource Operations 2012). DNA inventories (used in the estimation of approximately 14% of GBPUs as of 2010), the most rigorous population estimates used by the government, still contain

considerable uncertainty. The 95% confidence interval widths around estimates range from 33% to 152%, with an average of 70%, of the magnitude of the point estimate (Appendix S3 Figure 1B; Hamilton et al. 2004, Proctor et al. 2007, and Mowat & Fear 2004). Moreover, the magnitude of uncertainty for population estimates derived from models and/or from expert opinions is untested and unknown. In summary, the ±40 % range used in our simulations is considerably narrower than that suggested by the wide confidence intervals in DNA capture-recapture estimates, large fluctuations in the government's GBPU population estimates throughout the study period, and the heavy reliance on model-derived and/or expert-based estimates.

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3.2 Annual Allowable Mortality

Annual allowable mortality (AAM) values are not empirically derived for each GBPU. Instead, estimates are based on a sliding scale dictated by government estimates of a given GBPU's habitat quality and estimates of population vital rates (Hamilton & Austin 2002, Peek et al. 2003), with a maximum possible value of 6% (above; Harris 1986). Importantly, this rate assumes that female mortality comprises 30% or less of the total mortality. However, published estimates of maximum AAM values vary widely. Data from Bunnel & Tait (1980) modeling suggested maximum AAM for "maximally productive populations" to be 5.7%, though others considered their estimates of reproductive rates and survivorship were "very generous" (Miller 1990). Sidorowicz & Gilbert (1981) estimated AAM to be 2-3% in a Yukon population, whereas simulation by McLoughlin (2003) estimated maximum AAM values of 4.9% for good habitat, 2.8% for moderate, and 0% for poor habitat within BC. In the Northern Continental Divide area, Mace et al. (2012) calculated vital rates suggesting an AAM of approximately 3%. In Yellowstone, Schwartz et al. (2006) estimated up to 9% of female mortality could occur without population decline; however, this estimate included natural mortality, whereas the BC government AAM parameter represents only human-caused mortality. Hovey & McLellan (1996) estimated maximum finite rates of increase approximately 8.5% (95% CI of 3.2-13.6%), though, similar to Schwartz et al.'s (2006) this estimate is only relevant in terms of all mortality. Similarly, McCullough's (1981, 1986) simulation modeling found a total mortality (human and natural) of 10 bears per year to be sustainable in an initial population of 100 bears modelled after the Yellowstone populations, but this estimate depended on strong density-dependence for which there is mixed evidence (Miller 1990). Pease & Mattson (1999) found that survival in Yellowstone grizzly populations was food-dependent, with finite rates of increase of up to 7% in years with abundant food (white-bark pine mast years), and -5% in non-mast years. Overall,

population growth was likely positive, though given uncertainty in estimates the authors recommended cautious management. In our simulation model scenarios, we increased or decreased annual allowable mortality (AAM) values by up to 2% of the population estimate, resulting in maximum provincial AAMs ranging from 4% to 8%, centered on the maximum AAM point estimate of 6% currently used, a range considerably narrower than that suggested by the literature.

3.3 Unreported Mortality

We increased or decreased unreported mortality estimates, ranging from half to twice the current point estimate for each GBPU. Currently, unreported mortality estimates used by the province range from 0.3% to 2.2% of GBPU populations (unpublished data). Even when explicitly studied, however, unreported mortality is notoriously difficult to estimate (Peek et al. 1987, Cherry et al. 2002). Data from BC and beyond suggest that in many cases it could be considerably higher than estimates currently used. For example, in northwestern Alaska, less than half of grizzly bear hunting pre-1990 was thought to be reported (Miller 1990), whereas wounding loss of black bears was found to account for 9% of human-caused kills on Alaska's Kenai Peninsula from 1982-1987 (Schwartz & Franzmann 1991), losses that might have occurred unknown even to the hunters. In a study of radio-collared grizzly bears in the Selkirk mountains that straddle southeastern BC, northeastern Washington, and northwestern Montana, Knick & Kasworm (1989) found that the only source of death (27% of 11 bears collared) from 1983 to 1987 was illegal mortality. A meta-analysis by McLellan et al. (1999) of grizzly mortality in the Canadian and American Rockies, 1975-1997, found that 26% of kills were illegal. These data supported the estimate reported by Peek et al. (1987) that illegal kill of grizzlies comprised 25% of all human-caused mortality from 1970-1977. Our simulated uncertainty range, from half to twice the value of currently used estimates of unreported mortality, is therefore considerably narrower than the range suggested by the literature.

We excluded one GBPU (South Rockies) from these analyses in 2004–2006 because the allowable mortality for this period was set manually by the government, without record of relevant parameters. No mortalities were detected for Bowron, North Monashee, or Tagish in 2001-2003, or Francois in 2004-2006, so these were also excluded from simulated overmortality assessments.

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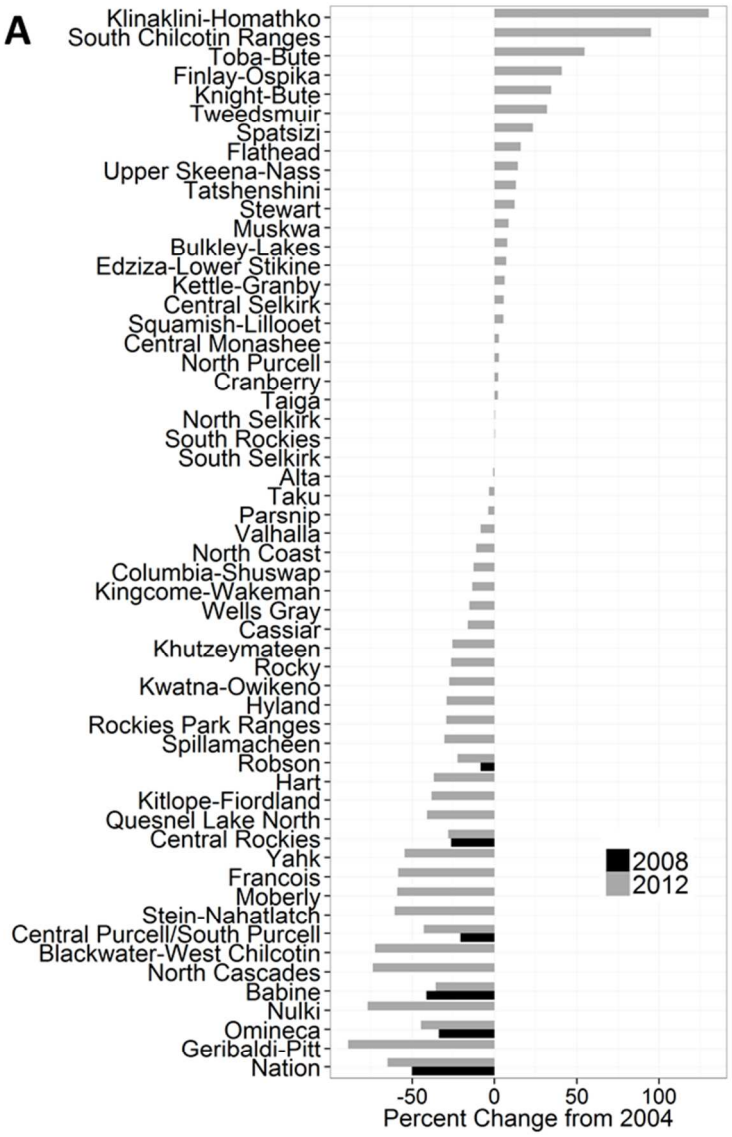
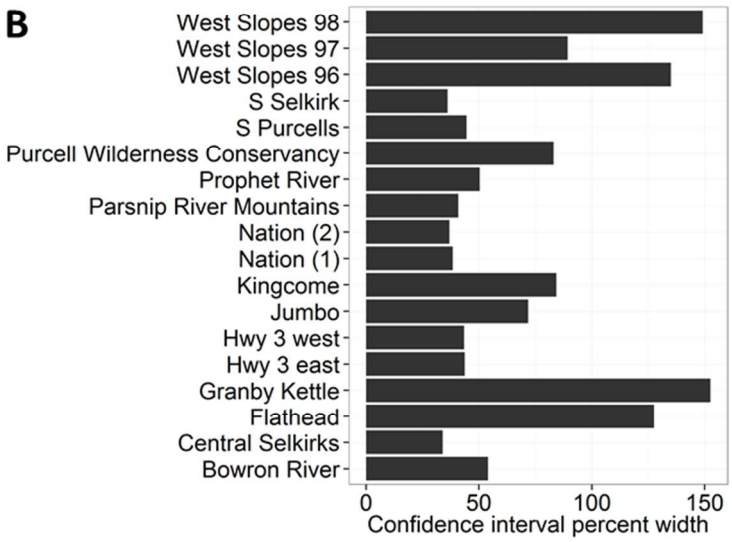

A
Klinaklini-Homathko
South Chilcotin Ranges
Toba-Bute
Finlay-Ospika
Knight-Bute
Tweedsmuir
Spatsizi
Flathead
Upper Skeena-Nass
Tatshenshini
Stewart
Muskwa
Bulkley-Lakes
Edziza-Lower Stikine
Kettle-Granby
Central Selkirk
Squamish-Lillooet
Central Monashee
North Purcell
Cranberry
Taiga
North Selkirk
South Rockies
South Selkirk
Alta
Taku
Parsnip
Valhalla
North Coast
Columbia-Shuswap
Kingcome-Wakeman
Wells Gray
Cassiar
Khutzeymateen
Rocky
Kwatna-Owikeno
Hyland
Rockies Park Ranges
Spillamacheen
Robson
Hart
Kitlope-Fiordland
Quesnel Lake North
Central Rockies
Yahk
Francois
Moberly
Stein-Nahatlatch
Central Purcell/South Purcell
Blackwater-West Chilcotin
North Cascades
Babine
Nulki
Omineca
Geribaldi-Pitt
Nation
2008
2012
-50
0
50
100
Percent Change from 2004


B
West Slopes 98
West Slopes 97
West Slopes 96
S Selkirk
S Purcells
Purcell Wilderness Conservancy
Prophet River
Parsnip River Mountains
Nation (2)
Nation (1)
Kingcome
Jumbo
Hwy 3 west
Hwy 3 east
Granby Kettle
Flathead
Central Selkirks
Bowron River
0
50
100
150
Confidence interval percent width

Appendix S3 Figure 1 A) Magnitude of confidence intervals as percent of point estimate for DNA capture-recapture inventories of Grizzly Bears (*U. arctos horribilis*) in British Columbia, Canada. Data from Hamilton et al. (2004), Proctor et al. (2007), and Mowat, G. & Fear, D. (2004). B) Percent change in estimated population size of Grizzly Bear (*Ursus arctos)* Population Units (GBPUs) in British Columbia, Canada, from 2004 to 2008 (black bars) and 2004 to 2012 (grey bars) Data from Hamilton et al. (2004), Hamilton (2008), and Ministry of Forests, Lands and Natural Resource Operations (2012).

Additional Supporting Figures

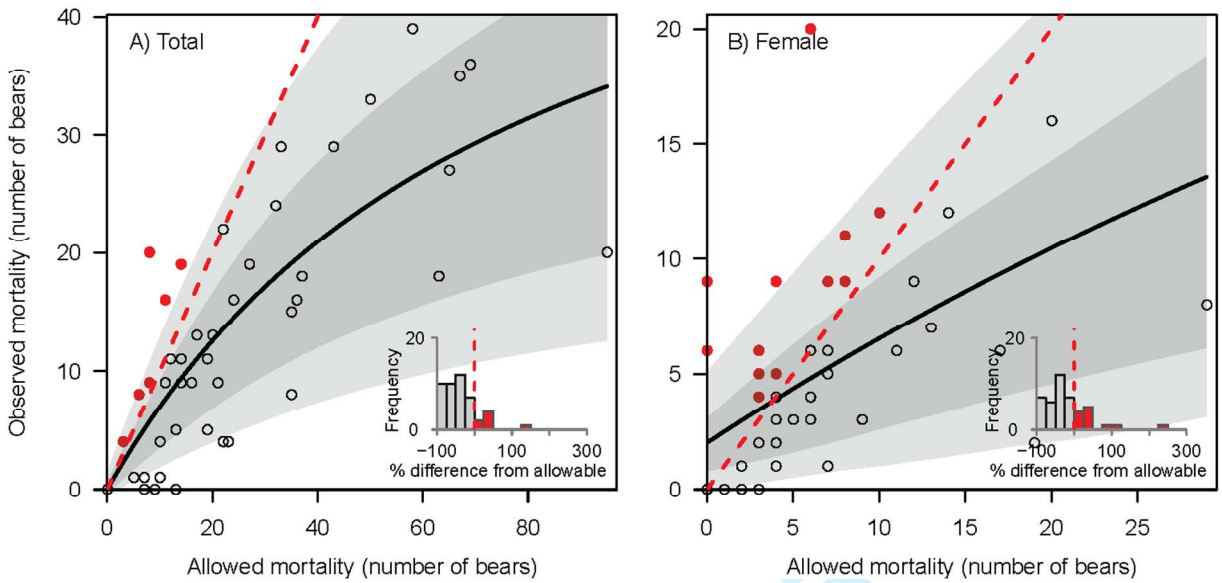


Figure S1 Management performance for A) total and B) female mortality in Grizzly Bear (U. arctos horribilis) Population Units (GBPUs) in British Columbia, Canada, 2001-2003. Black curve is a Michaelis-Menten curve fitted by maximum likelihood, assuming a negative binomial error distribution. Red dashed line indicates a 1:1 relationship; solid red dots above this line signal GBPU-level overmortality events. Grey-shaded region encompasses the 80% prediction interval (smoothed for visual purposes). Inset histograms show the distribution of GBPU-level percent difference between observed and allowable mortalities; red bars to the right of red dashed lines indicate overmortality events.

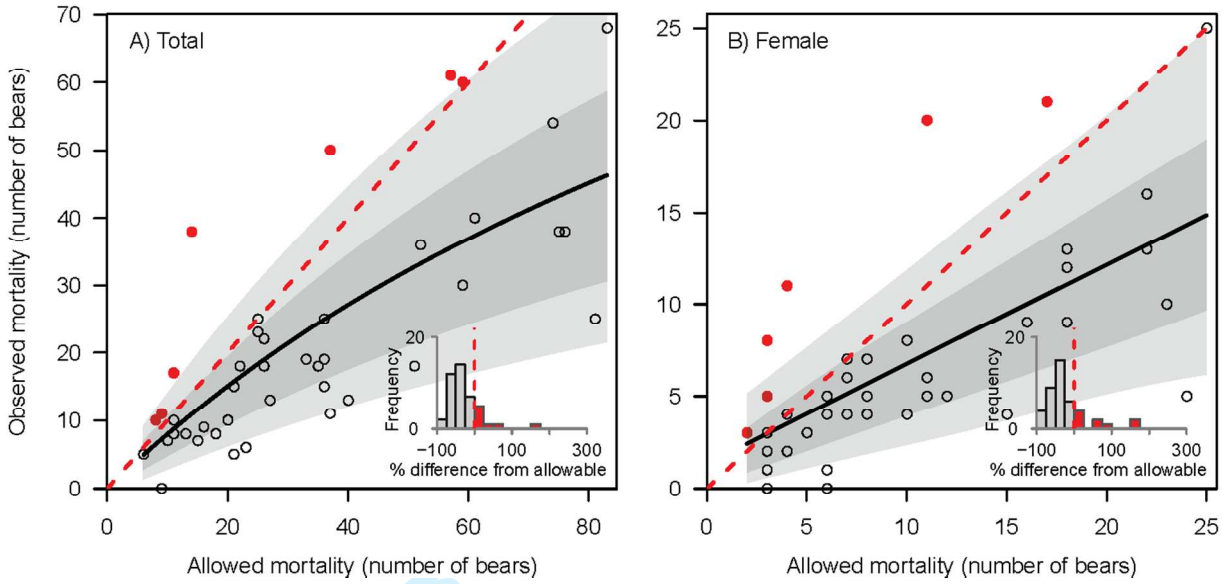


Figure S2 Management performance for A) total and B) female mortality in Grizzly Bear (U. arctos horribilis) Population Units (GBPUs) in British Columbia, Canada, 2004-2006. Black curve is a Michaelis-Menten curve fitted by maximum likelihood, assuming a negative binomial error distribution. Red dashed line indicates a 1:1 relationship; solid red dots above this line signal GBPU-level overmortality events. Grey-shaded region encompasses the 80% prediction interval (smoothed for visual purposes). Inset histograms show the distribution of GBPU-level percent difference between observed and allowable mortalities; red bars to the right of red dashed lines indicate overmortality events.

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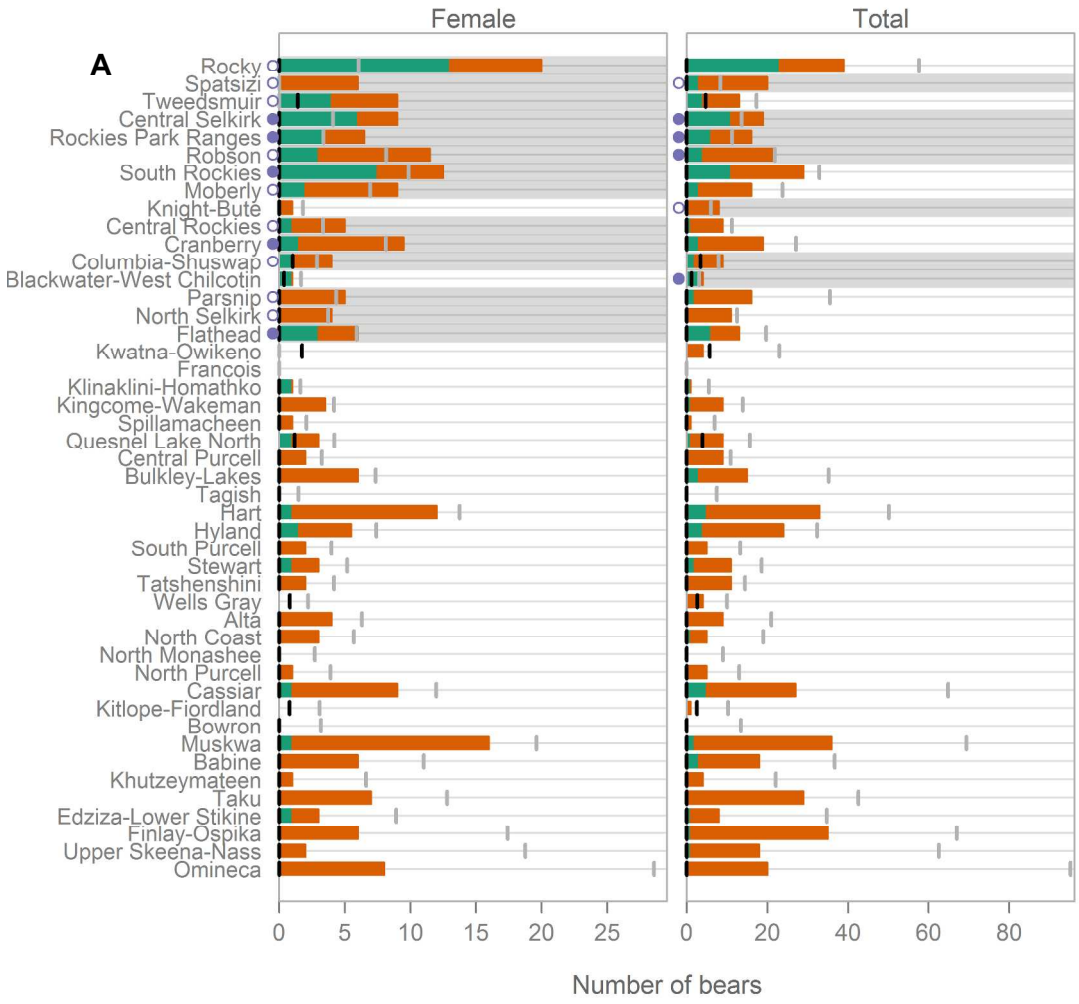
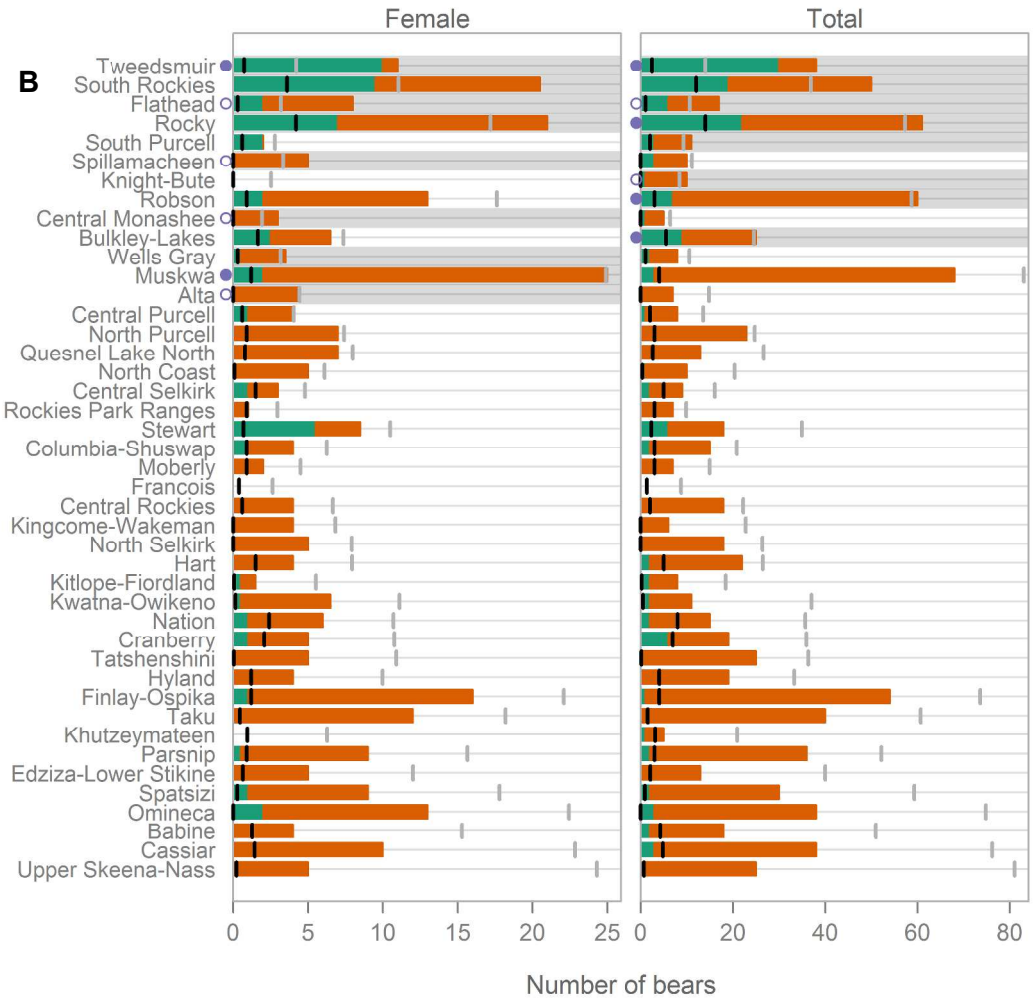

A
Female
Total
Rocky
Spatsizi
Tweedsmuir
Central Selkirk
Rockies Park Ranges
Robson
South Rockies
Moberly
Knight-Bute
Central Rockies
Cranberry
Columbia-Shuswap
Blackwater-West Chilcotin
Parsnip
North Selkirk
Flathead
Kwatna-Owikeno
Francois
Klinaklini-Homathko
Kingcome-Wakeman
Spillamacheen
Quesnel Lake North
Central Purcell
Bulkley-Lakes
Tagish
Hart
Hyland
South Purcell
Stewart
Tatshenshini
Wells Gray
Alta
North Coast
North Monashee
North Purcell
Cassiar
Kitlope-Fiordland
Bowron
Muskwa
Babine
Khutzeymateen
Taku
Edziza-Lower Stikine
Finlay-Ospika
Upper Skeena-Nass
Omineca
0 5 10 15 20 25
0 20 40 60 80
Number of bears

view


B
Female
Total
Tweedsmuir
South Rockies
Flathead
Rocky
South Purcell
Spillamacheen
Knight-Bute
Robson
Central Monashee
Bulkley-Lakes
Wells Gray
Muskwa
Alta
Central Purcell
North Purcell
Quesnel Lake North
North Coast
Central Selkirk
Rockies Park Ranges
Stewart
Columbia-Shuswap
Moberly
Francois
Central Rockies
Kingcome-Wakeman
North Selkirk
Hart
Kitlope-Fiordland
Kwatna-Owikeno
Nation
Cranberry
Tatshenshini
Hyland
Finlay-Ospika
Taku
Khutzeymateen
Parsnip
Edziza-Lower Stikine
Spatsizi
Omineca
Babine
Cassiar
Upper Skeena-Nass
0
5
10
15
20
25
0
20
40
60
80
Number of bears

view

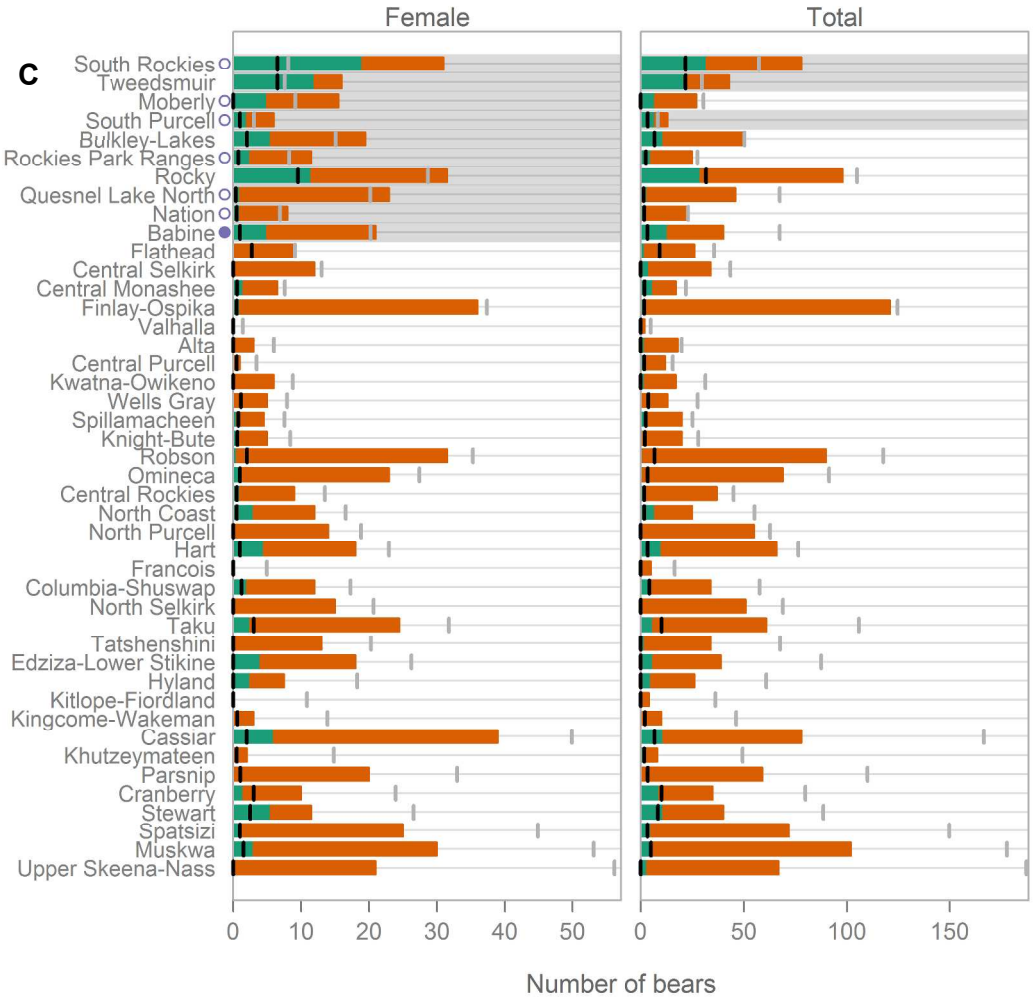


Figure S3 Allowable and reported mortalities for each Grizzly Bear (*U. arctos horribilis*) Population Unit (GBPU) in British Columbia, Canada, during A) 2001-2003, B) 2004-2004, and C) 2007-2011 allocation periods. Green and orange bars represent number of bears killed by non-hunting and hunting sources, respectively. Vertical grey lines denote allowable mortality and vertical black lines denote predicted non-hunt mortality for each period. GBPU rows in which reported mortality exceeded mortality limits ('overmortality') are shown with grey highlighting. Open blue circles denote GBPUs in which hunting mortality alone exceeded the mortality limits for all sources combined; filled blue circles denote areas in which the unpredicted non-hunting mortality (difference between actual and predicted non-hunting mortality) exceeded the excess mortality.

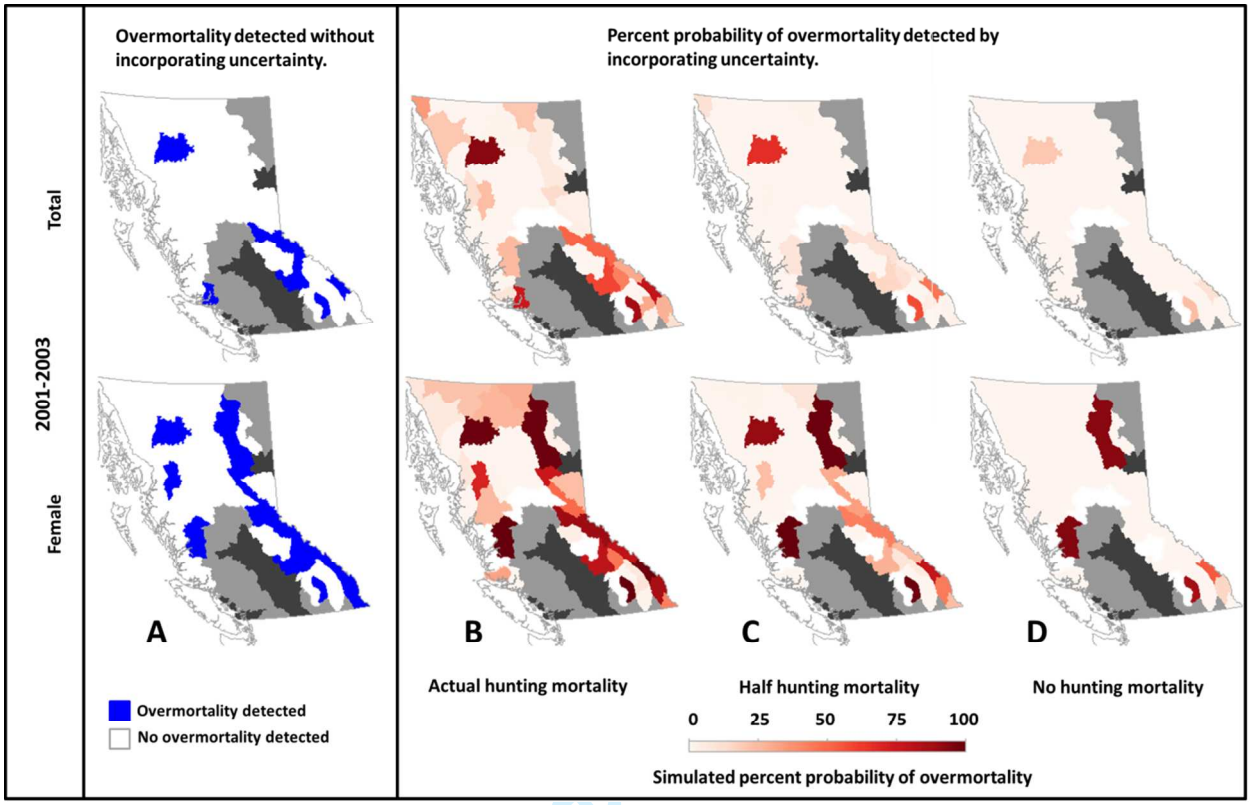


Figure S4 A) Total and female overmortalities detected in Grizzly Bear (U. arctos horribilis) Population Units (GBPUs) of British Columbia, Canada, from 2001-2003, given actual hunting mortality levels and without consideration of mortality limit uncertainty. Blue indicates GBPUs with detected overmortality whereas white indicates GBPUs without. B–D) Simulated probability of total or female overmortality, incorporating uncertainty around mortality limits. Panel B shows simulated probability of overmortality given actual reported mortality rates; panels C and D show what the probability of overmortality would be had hunting mortality been reduced by 50% or 100%, respectively, assuming other sources of mortality remained unchanged. Increasingly dark red indicates an increasing probability of overmortality in a given period. Grizzly bears have been extirpated from dark-grey areas. Light-grey areas indicate GBPUs in which populations are either threatened or were closed to hunting during the study period.

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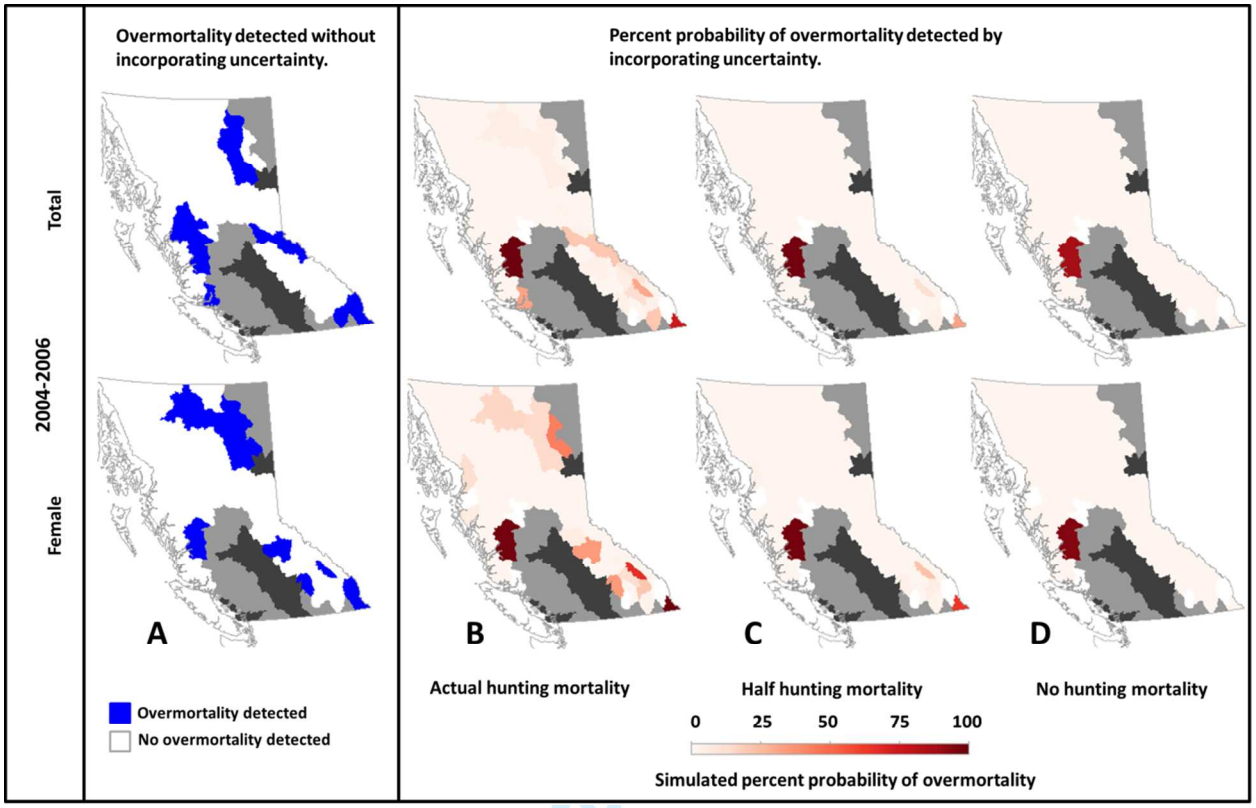

Figure S5 A) Total and female overmortalities detected in Grizzly Bear (U. arctos horribilis) Population Units (GBPUs) of British Columbia, Canada, from 2004-2006, given actual hunting mortality levels and without consideration of mortality limit uncertainty. Blue indicates GBPUs with detected overmortality whereas white indicates GBPUs without. B¬–D) Simulated probability of total or female overmortality, incorporating uncertainty around mortality limits. Panel B shows simulated probability of overmortality given actual reported mortality rates; panels C and D show what the probability of overmortality would be had hunting mortality been reduced by 50% or 100%, respectively, assuming other sources of mortality remained unchanged. Increasingly dark red indicates an increasing probability of overmortality in a given period. Grizzly bears have been extirpated from dark-grey areas. Light-grey areas indicate GBPUs in which populations are either threatened or were closed to hunting during the study period.

Supporting References

Apps, C. (2010). *The grizzly bear population inventory & monitoring strategy for British Columbia*. British Columbia Ministry of Environment and Habitat Conservation Trust Foundation.

British Columbia Ministry of Environment, Fish, Wildlife and Habitat Branch. (2010). Grizzly bear hunting: frequently asked questions.

Bunnell, F.L. & Tait, D.E.N. (1980). Bears in Models and in Reality: Implications to Management. *Int C Bear*, 4, 15–23.

Cherry, S., Haroldson, M.A., Robison-Cox, J. & Schwartz, C.C. (2002). Estimating Total Human-Caused Mortality from Reported Mortality Using Data from Radio-Instrumented Grizzly Bears. *Ursus*, 13, 175–184.

Hamilton, A.N. & Austin, M. (2004). *Estimating grizzly bear (Ursus arctos) population size in British Columbia using an expert-based approach*. British Columbia Ministry of Water, Land and Air Protection.

Hamilton, A.N. & Austin, M.A. (2002). Grizzly bear harvest management in British Columbia: background report. *British Columbia Ministry of Water, Land, and Air Protection, Biodiversity Branch, Victoria, British Columbia, Canada*.

Hamilton, A.N., Heard, D.C. & Austin, M.A. (2004). British Columbia Grizzly Bear (Ursus arctos) Population Estimate 2004.

Hamilton, A.N. (2008). 2008 Grizzly Bear Population Estimate for British Columbia.

Harris, R.B. (1986). Modeling sustainable harvest rates for grizzly bears. *Unpublished manuscript*.

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. *Can J Zool*, 74, 1409–1416.

Knick, S.T. & Kasworm, W. (1989). Shooting mortality in small populations of grizzly bears. *Wildlife Soc B*, 11–15.

Mace, R.D., Carney, D.W., Chilton-Radandt, T., Courville, S.A., Haroldson, M.A., Harris, R.B., Jonkel, J., Mclellan, B., Madel, M. & Manley, T.L. (2012). Grizzly bear population vital rates and trend in the Northern Continental Divide Ecosystem, Montana. *J Wildlife Manage*.

For Peer Review

McCullough, D.R. (1981). Population dynamics of the Yellowstone grizzly bear. In: *Dynamics of large mammal populations* (eds. Fowler, C.W. & Smith, T.D.). John Wiley and Sons, New York, New York, pp. 173–196.

McCullough, D.R. (1986). The Craigheads' data on Yellowstone grizzly bear populations: relevance to current research and management. *Int C Bear*, 21–32.

McLellan, B.N., Hovey, F.W., Mace, R.D., Woods, J.G., Carney, D.W., Gibeau, M.L., Wakkinen, W.L. & Kasworm, W.F. (1999). Rates and causes of grizzly bear mortality in the interior mountains of British Columbia, Alberta, Montana, Washington, and Idaho. *The Journal of wildlife management*, 911–920.

McLoughlin, P.D. (2003). Managing risks of decline for hunted populations of grizzly bears given uncertainty in population parameters. University of Alberta.

Miller, S. (1990). Population management of bears in North America. *Int C Bear*, 8.

Ministry of Forests, Lands and Natural Resource Operations. (2012). British Columbia grizzly bear population estimate for 2012.

Mowat,, G. & Fear, D. (2004). *Grizzly bear density in the Nation River area of British Columbia*. Submitted to: Slocan Forest Products and B.C. Ministry of Water, Land, and Air Protection.

Pease, C.M. & Mattson, D.J. (1999). Demography of the Yellowstone grizzly bears. *Ecology*, 80, 957–975.

Peek, J., Beecham, J., Garshelis, D., Messier, F., Miller, S. & Strickland, D. (2003). *Management of grizzly bears in British Columbia - a review by an independent scientific panel*. Submitted to: Minister of Water, Land and Air Protection Government of British Columbia Victoria, B.C.

Peek, J.M., Pelton, M.R., Picton, H.D., Schoen, J.W. & Zager, P. (1987). Grizzly bear conservation and management: a review. *Wildlife Soc B*, 15, 160–169.

Proctor, M., Boulanger, J., Nielsen, S., Servheen, C., Kasworm, W., Radandt, T. & Paetkau, D. (2007). Abundance and density of Central Purcell, South Purcell, Yahk, and South Selkirk Grizzly Bear

For Peer Review

Population Units in southeast British Columbia. *Report for BC Ministry of Environment, Nelson and Victoria, BC*.

Schwartz, C.C. & Franzmann, A.W. (1991). Interrelationship of Black Bears to Moose and Forest Succession in the Northern Coniferous Forest. *Wildlife Monogr*, 3–58.

Schwartz, C.C., Haroldson, M.A., White, G.C., Harris, R.B., Cherry, S., Keating, K.A., Moody, D. & Servheen, C. (2006). Temporal, spatial, and environmental influences on the demographics of grizzly bears in the Greater Yellowstone Ecosystem. *Wildlife Monogr*, 161, 1–8.

Sidorowicz, G.A. & Gilbert, F.F. (1981). The Management of Grizzly Bears in the Yukon, Canada. *Wildlife Soc B*, 9, 125–135.

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