MISCONCEPTIONS, IRONIES, AND UNCERTAINTIES REGARDING TRENDS IN BEAR POPULATIONS

INVITED PAPER

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Abstract: Despite our rapidly increasing knowledge of bears, there are few places in the world where we really know how bear populations are faring. I argue that bear conservation would benefit by highlighting rather than hiding this uncertainty. Assessments of bear populations often are based on records of dead animals and trends in habitat availability. These data produce dubious indications of population trend. Case studies relating to the trade in bear parts, sport harvests, and nuisance kills indicate that records of human-killed bears may not be accurate and may not necessarily reflect changes in population size. Increasing bear populations may continue to rise with increased levels of human exploitation (as long as it is below the maximum sustainable take), whereas declining populations may continue to plummet despite reduced exploitation. Similarly, whereas loss of habitat (forest area) probably engenders a decline (of unknown magnitude) in bear populations, unchanging or increasing forested area may not necessarily result in stable or increasing bear numbers. Ironically, bear populations that have been managed for sustained harvests have generally fared better than populations in which hunting has been prohibited, mainly because the former better controls illicit hunting than the latter. Long-term conservation of bears requires better information on population trends, but better techniques are unlikely to be developed if faults and inadequacies of current data are not clearly recognized.

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In most human societies, knowledge empowers, whereas uncertainty signifies fallibility, timidity, and weakness. Scientists are presumed to be knowledgeable, and thus able to produce accurate facts, explanations, and predictions; those that do so with certainty tend to be held in high esteem by the public. The soothsayers of the past were probably wrong more often than are modern forecasters of environmental and astronomical events, but even today's complicated computer models are prone to error because we lack a full understanding of most natural systems.

A major concern in today's world is the threat of species extinctions due to the activities of humans. There is a strong relationship between human population size and threat of extinction of native fauna (McKinney 2001). Although we recognize the basic causes of extinction (Diamond 1989) and we have been able to identify taxa, ecosystems, and geographic areas that are most susceptible to extinctions (Cole et al. 1994, Mace and Balmford 2000), ecologists and conservation biologists have been struggling to understand how to relieve species from extinction threats. Seemingly basic questions, such as "What is the minimum viable population size and what level of human exploitation is sustainable? What habitats does the species require and how much area should be protected within reserves?" are routinely debated, because empirical data are lacking. Unfortunately, the science of ecology is by nature inexact and laden with uncertainty.

Shrader-Frechette and McCoy (1993:123–124) contend that due to the inherent complexity of ecology, there are few governing principles, so case studies are the best means for achieving understanding. The method of case studies involves scrutinizing the details of particular situations in an attempt to "make sense" of them. Accumulating and comparing results from a series of related case studies advances the science.

For large mammals such as bears, experimentation is rarely employed as a part of the case study. Instead, bear biologists tend to reach conclusions based on patterns in the data, logic, insight, and knowledge of other studies. Case studies generally enter the body of science through a process of peer-review, although much information is contained in less formal reports and even raw data.

At periodic junctures it is worthwhile to review the basis of conclusions and direction of thinking. In experimental sciences, predictions that are not upheld empirically are ultimately discarded. In sciences based on case studies, apparent anomalies may represent truly unique situations, making it difficult to tease out erroneous information. Nevertheless, occasional re-examinations may prove to be fruitful — if not to correct the past, to guide the future — especially in terms of species conservation.

In this paper I draw attention to several misconceptions related to the monitoring and conservation of bear populations. I rely heavily on case studies to illustrate my points. These are used mainly as counter-examples to prevailing views or to exemplify common problems.

A principal purpose of this critique is to highlight the uncertainty, and hence fallibility, of our understanding of bear populations. There are few places in the world where biologists would admit to not knowing whether a bear population was increasing, decreasing, or stable, yet the reality is that there are few places where we really *do* know for sure how bears are faring. Seemingly contrary to my opening remarks, I believe that ultimately we, as bear biologists benefit — because bears benefit — by critically examining the basis of our knowledge and admitting to our foibles and uncertainties.

MISCONCEPTIONS REGARDING POPULATION TREND

Trend Ascertained from Numbers of Dead Bears

In a population of unknown size, a large death toll is obviously unnerving. Because most bear populations are of unknown size, a record of increasing known deaths is often taken as *prima facie* evidence of a population decline. Moreover, even poor records with no clear trend but occasional documentation of a surge of deaths may be cause to fear a population decline.

Records of bear parts (principally gall bladders) traded among Asian countries are a salient example of tallies of dead bears being used to interpret population trends. Several good investigative reports exposed the broad geographic scope of this trade (Mills and Servheen 1991; Mills 1995; Mills et al. 1995, 1997), although it was not possible to accurately quantify it. Some evidence suggested increases or decreases in bear kills in certain countries, based on documented or estimated numbers of exported or imported parts. However, population trend assessments based on trends in the trade in bear parts, and hence numbers of bears killed, have been inconsistent. Consider the cases of 3 countries that have been heavily involved in this trade.

China.—In China, the killing of bears (other than giant pandas [*Ailuropoda melanoleuca*]) for their parts was legal until 1989. In the decade preceding this restriction (1979–88), several thousand bear gall bladders were exported from China to Japan (Servheen 1990). Additional, but smaller numbers of gall bladders were exported to South Korea (Mills et al. 1995). However, trends and quantities of bears killed for the trade in gall bladders are nearly impossible to discern from bile export data, due to many confounding issues, including trade in fake bear bile (gall from animals other than bears that are claimed as bears) and farmed bile (bile drained from live, captive bears)(Box 1).

Farming bears for their bile began in China in 1984. During 1985–89 hundreds or thousands of bears were removed from the wild to stock captive populations (Fan and Song 1997). However, since 1989, all of the species of bears in China (brown [*Ursus arctos*], Asiatic black [*U. thibetanus*], and sun [*Helarctos malayanus*]) have been protected, inasmuch as killing or capturing is illegal without a special permit, and selling of parts of wild bears is also prohibited (Mills and Servheen 1991, Fan and Song 1997). Has this supposed change in exploitation of bears enabled bear populations to increase? The answer is unclear.

Santiapillai and Santiapillai (1997:23) indicated that "throughout China, bear populations are in decline." They cite an estimate of 15,000–20,000 Asiatic black bears in China, which matches the range reported by Ma and Li (1999), based on "1994 statistics". Ma and Li (1999) believed that over-hunting for bear parts was causing this species to decline, although their chief evidence for recent declines were diminishing numbers of purchased bear skins during 1986–1991. Cheng (1999:123), referring to these same data, concluded that "In recent years, ... the number of bears [both black and brown, in one province] has dropped significantly..." Li et al. (1996; citing Ma et al. [1994]), presented higher population estimates (20,000–32,000 Asiatic black bears and 12,000–14,000 brown bears), but also suggested that populations were shrinking. Fan and Song (1997:11) called these estimates "an emotional guess" and presented their own estimates of 46,500 Asiatic black bears, 14,800 brown bears, and 400 sun bears, based on field surveys and interviews with local people. They claimed that after bears were protected in 1989, populations increased. Ma et al. (2001) conducted a more recent survey, also based on field sign and local interviews, and concluded that Asiatic black bears numbered <20,000 and were still declining numerically and geographically. Differences in these opinions appear to be just that — beliefs lacking much factual basis.

Russia.—Exportation of bear gall bladders increased dramatically in Russia in the early 1990s for various political and economic reasons (Chestin 1998). Chestin (1998) believed that because of increased economic incentives, legal harvests of brown bears, generally totaling 4,000–4,500 nationwide, might have been matched by an equal number of illegally taken (poached) bears. Imports of bile to South Korea from Russia showed a sharp increase in the 1990s, but still represented a small number of bears killed/year (Box 1). Prior to this rise in poaching, the total number of Russian brown bears appears to have increased, from an estimated population of 80,000 in 1981 to 125,000 in 1990, and the geographic range expanded concomitantly (Chestin 1999). Annual sustainable harvest quotas were established so as not to exceed 10% of the population, but in reality appeared to be far below that. Thus, even if poaching was as high as posited by Chestin (1998), the overall rate of human exploitation may have been sustainable. Most killing for gall bladders has occurred in the Russian Far East (Kamchatka), where some reports suggested an annual take of 1,500–2,000 brown bears, possibly 20% of the population (Nikolaeno

Box 1. Records of bile imports or exports have been used to estimate the number of bears killed to support that trade. Tabulated below are the supposed numbers of bears killed/year to account for imports of bear bile recorded by the Korean Customs Administration (Mills 1995, Mills et al. 1995) for 4 countries of origin discussed in the text.

	Calculated number of bears killed/year		
Country of origin	1970s	1980s	1990s
China	3	3	490
Russia	0	0	15
Japan	7100	26	7
Indonesia	690	3	0

Although these numbers seem to indicate clear trends in bears killed over time, the data are too confounded to draw such conclusions. Several major difficulties exist in converting bile to bears.

Variation in Gall Bladder Mass.—The amount of bile in gall bladders varies by species, geographic area, and time of year, so any conversion of bile mass to dead bear equivalents is subject to appreciable error. Values tabulated are based on 30 grams/whole, dried bear gall bladder (Lay 2001). Mills (1995) suggested an average of 60 grams/gall bladder, but did not present supporting documentation. Further uncertainty involves whether the Korean customs records relate to grams of bile, grams of whole gall bladders, or a combination of both.

Changes in Regulations, Enforcement, and Recording of Imports.—Mills (1995) and Mills et al. (1995) reported Korean bile import data by decade, covering 24 years, 1970–93. The Republic of Korea joined the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in July 1993, and in 1996 accepted the Appendix II listing of bears whose populations were not considered threatened; this listing requires documentation to ensure legal import. Korea also concomitantly improved surveillance and enforcement. These actions resulted in better recording of bear imports and more seizures, so the total amount of bile rose by nearly an order of magnitude from the 1990–93 period shown in the table to 1994–99 (Mills et al. 1997, Yoon 1997, Sohn 2001).

Counterfeit Bile.—Several investigative reports (Lau et al. 1994, Chang et al. 1995, Gaski 1997) indicated that a very high proportion of the presumed bear gall bladders on the Asian market (94–98%) are from animals other than bears. This would severely inflate the estimate of dead bears based on bile imports. Trade in non-bear gall bladders likely explains the unreasonably large quantity of bile from Indonesia and Japan in the 1970s. It would be impossible to remove >7,000 bears annually for 10 years from a Japanese population of 10,000–15,000 black bears (Hazumi 1999) and 2,000–3,000 brown bears (Moll 2001). Moreover, Japan also has an internal market for bear bile, and exports to countries other than South Korea, so the amount of bile obtained in Japan is far more than indicated on Korean customs records.

Farmed Bile.—Bile obtained from catheterized, captive (farmed) bears probably explains the sharp increase in imports from China in the 1990s. Lau et al. (1994) indicated that virtually all the bile imported from China (into Hong Kong) in the early 1990s was from captive bears, not dead bears. The Korean import data do not discriminate between powdered bile (most likely from farmed bears) and whole gall bladders (dead animals, most of which are not bears).

1993, cited in Chestin 1999). In this area it is assumed that numbers declined, although population estimates from aerial surveys showed an equivocal trend (Revenko 1998).

Commercially-motivated poaching of Asiatic black bears in the Russian Far East (the only area of Russia inhabited by this species) also has increased, but estimates of population size and presumed rates of decline have been highly variable and contradictory (Yudin 1993). Moreover, references to population declines in this species generally refer to the distant past. Chestin and Yudin (1999) suggested that Russian Asiatic black bears numbered 25,000–35,000 at the beginning of the 1800s, only 6,000–8,000 in 1970, and 4,000–5,000 by 1985, which is thought to be about the same remaining at present. Until 1983, Russians legally harvested 300–400 Asiatic black bears/year. Since then, black bear hunting has been illegal. It is uncertain whether the previous legal harvest of 300–400 was sustainable (it would seem so if the population was 4,000–5,000), and if so, whether illegal harvests now exceed that. Sustainability of the harvest relates only to the number of bears killed, not whether they were legally or illegally taken. Of course the former is more readily adjusted to remain sustainable, but the later is not by definition unsustainable.

Japan.—Japan is an importer and exporter of bear bile, as well as a user of products obtained from native bears. Both import and export of bile appeared to decline dramatically from the 1970s to the early 1990s (Mills et al. 1995), although these data are difficult to interpret (Box 1). Harvesting of brown bears (on Hokkaido) and black bears (on Honshu) is legal, but rather loosely regulated in part because there has been a long-term, purposeful effort to reduce numbers of bears. Hunters can legally sell all parts of bears they harvest, and there are no government-imposed restrictions on the number they can take during the hunting season. It is believed that the opportunity to sell bear parts is largely what sustains interest in hunting (Moll 2001).

Some hunting restrictions were imposed during the 1980s and 1990s (e.g., elimination of the brown bear season during spring when hunters could snow-track bears to or from their dens) (Mano 1998, Moll 2001); this reduced the kill, but not in all areas (Kaji and Mano 1996). Mano and Moll (1999:129) thought that brown bear harvests still exceeded sustainable limits in some places, such as the Oshima peninsula, threatening the "long-term persistence of that subpopulation." In another report, however, Mano (1998:179) indicated that the Oshima brown bear population "persists in high numbers," but suggested that bears in more lightly hunted areas were declining. Aoi (1991:135) described the overall Hokkaido brown bear population as "declining rapidly," whereas Kaji (1992:413) thought that "Further studies are needed to analyze the population trend..." It seems clear from the conflicting reports that Kaji's call for more study is warranted.

Approximately 2,000 Asiatic black bears have been taken annually on Honshu, half by hunting and half explicitly for pest control (Hazumi 1994). Based on density estimates produced from springtime snow-tracking, capture–recapture, and habitat assessment across the island, the total population size has been estimated at 10,000–15,000. The veracity of this estimate is difficult to assess, and even if it is assumed to be accurate, the span is wide enough to preclude judgment as to whether present levels of exploitation are sustainable. Hazumi (1999:209) considered Japanese black bears to be "facing a crisis," due to the combined effects of habitat degradation and uncontrolled harvesting, but he had no real evidence of a population decline. Some prefectural government studies have attempted to assess local population trends, but flaws in their methodology undermined the credibility of their results (Huygens and Hayashi 2001).

Generalities.—The 3 countries highlighted above were selected not because they exemplified situations with inadequate data on bear population trends, but rather because, compared to other Asian countries impacted by the gall bladder trade, they had considerably *more* data on their bear populations. Additionally, unlike most of the other Asian countries, some records of the gall bladder trade exist for these 3, and each of the 3 exhibited an apparent temporal trend in the volume of this trade (Box 1). Despite these data, bear population trends in these 3 countries are equivocal, even controversial. The status of bears in other Asian countries is even more uncertain.

I am not suggesting that the gall bladder trade is not cause for grave concern — certainly it is. But this concern should arise from the uncertainty, not the certainty, of the impacts. We cannot discount the possibility that in many areas, the exploitation of bears for parts is *sustainable*. That is, we cannot reject the null hypothesis of *no detrimental effect*. However, employing statistical terminology, we have insufficient power (due to a paucity of data) to reject this hypothesis. Normally, we are concerned mainly with type-I errors: we attempt to avoid erroneously rejecting a true null hypothesis. However, in cases involving harm, to people or the environment, it may be ethically more responsible to err on the side of caution by trying to avert effects that may be nonexistent (i.e., putting more effort toward avoiding type-II errors; Mapstone 1995). Shrader-Frechette and McCoy (1993:153) put it this way: "in cases of *uncertainty* [my emphasis], ecologists ought to adopt an ethical (rather than purely scientific) account of ecological rationality." Thus, for rare species, the burden of proof should switch from proving that a population decline has occurred, to proving that it has not (Taylor and Gerrodette 1993).

A problem with emphasizing the avoidance of type-II errors in cases of potential harm, especially irreversible harm such as extirpation, is losing track of the underlying uncertainty. It can become all too easy, once accepting that a detrimental effect may exist, to begin to prophesize the magnitude of the effect. Without real data, this can become a game of emotional guesstimation. In cases such as the gall bladder trade, where to most Westerners the practice is culturally alien and repugnant, claims of effects often become exaggerated, especially if they are thought to help instigate remedial action. Hence, assertions of Asian bear populations being "devastated," "decimated", or "depleted" (Knights 1996) tend to be widely accepted, or at least not questioned. It is doubtful that such unsubstantiated claims serve the best interest of bear conservation. I believe they do not, mainly because they falsely reflect the certainty of our knowledge. Hence, they create more opportunity for further misinformation, especially related to population level effects of highly visible mortality.

Increases or decreases in levels of human exploitation may not necessarily result in attendant changes in population size. An increasing population may continue to increase in the face of heightened exploitation, whereas a declining population may continue to plummet despite reduced exploitation. The discovery of a massive shipment of gall bladders or a large number of dead bears should not, in itself, be construed to represent a population decline, and neither should the absence of these be cause for complacency.

The examples so far concerned Asian bears and the gall bladder trade. Because this exploitation is largely unregulated, it is presumed to be unsustainable. In contrast, recreational (sport) harvests are overseen by management agencies whose responsibility is to ensure that they are sustainable. Nevertheless, unusually large sport harvests often raise concerns, if not by the management agency, by others interested in bears. I offer 2 examples dealing with American black bears (*U. americanus*).

Tennessee.—The legal harvest of black bears in Tennessee in 1997 was at least twice that of previous years, due to a natural food failure that prompted many bears to leave the sanctuary of the Great Smoky Mountains National Park. Pelton (1998:26) reported that in reaction to this high harvest, some biologists, bear advocacy groups, and alarmists in the general public claimed that the population was being "slaughtered" and "driven to extirpation." Long-term research (Pelton and van Manen 1996), however, showed that the population had been increasing for many years and continued to increase afterwards. Unfortunately, the body count was obvious, whereas the biological data either were not appreciated or did not constitute as appealing a story.

Minnesota.—Hunting of black bears in Minnesota has, since 1982, been regulated by restrictions (quotas) on the numbers of licenses available. This system was implemented to reduce the rate of harvest on what was thought to be a declining population. After a few years of sharply curtailed harvests, there was ample evidence that the population was growing. However, a food failure in 1985 disrupted normal feeding activities, which resulted in an unusually large number of bears being killed as nuisances. This large killing attracted considerable attention by the news media. Moreover, one bear biologist, who had been monitoring a few radiocollared bears at the time, suggested, in a memo to the management agency, that the food failure caused "severe malnutrition," possibly leading to reduced reproduction and starvation of cubs. He also warned that 2 age classes of young bears might have been "virtually eliminated," thus compounding the high kill (L. Rogers, 1986, unpublished report). Based on this report, an environmental group concluded that "it would be surprising if the black bear population has not already been nearly eliminated ..." (Sierra Club, North Star Chapter, Minneapolis, Minnesota, 1986, unpublished report). Hindsight showed these forecasts to be wrong. Collections of bear teeth from subsequent harvests, used for age determination, showed no indication of weak cohorts. Furthermore, population modeling interfaced with 2 statewide, mark–recapture population estimates (Garshelis and Visser 1997) indicated that the population grew steadily at ~5% annually (D. Garshelis, unpublished data). Fifteen years after the 1985 "high kill" the population had tripled, and despite steadily increasing harvests, the agency's goal of stabilizing population size had not been achieved. As in the other examples above, these data demonstrate that population trend cannot reliably be ascertained from numbers of dead bears.

Trend Ascertained from Area of Habitat

It seems almost tautological that bear populations decline as a result of habitat loss. However, the explanation for this relationship is not as simple as it may at first appear. If humans did not exploit bears, bear populations would likely exist at or near the carrying capacity (K) of the habitat over the long-term. Any loss of habitat in this case would diminish K, eventually resulting in a population decline from increased natural mortality, diminished reproduction, or both. In the modern world, however, very few bear populations exist at K. Conceivably then, habitat loss would not necessarily cause a population to decline. As an example: if, due to human exploitation, a bear population existed at 1/3 K, and the area of habitat was reduced by 1/3, this reduced area could still easily support the existing population, which — other things being equal — would now be at 1/2 K (Fig. 1).

This seeming paradox is resolved by considering further ramifications of the loss of habitat. If the level of human exploitation remained constant, the above situation might indeed occur; habitat could be lost without affecting bear numbers until the point that the remaining population, confined to a smaller area, exceeded K. In reality though, bear mortality would likely increase inside the smaller patch of habitat because of heightened human exploitation (Fig. 1). Exploitation levels would tend to increase for several reasons. (1) The reduced area would increase the proportion of bears living at the edge, and these edge animals would be more vulnerable to hunters and also more likely to wander into adjoining crop fields and be killed as pests. This explanation seemed to account for dramatic declines in orangutans (*Pongo pygmaeus*) following logging (van Schaik et al. 2001). (2) The diminished size of the patch would make the interior area more accessible to hunters that kill bears either intentionally or inadvertently when seeking other species (e.g., by snaring); in essence, the reduced area would lessen the chance for some part of the region to function as a bear sanctuary. (3) Because bears are known to travel widely, especially during years of natural food failure, they would be more likely to leave the bounds of the smaller patch of habitat and thus be exposed to greater human contact. Recent studies have shown that although protected areas (e.g., national parks) are reasonably effective in maintaining habitat (vegetation) for animals (Bruner et al. 2001), the persistence of wide-ranging animals (including bears), are strongly related to edge effects (Woodroffe and Ginsberg 1998, Revilla et al. 2001) and surrounding human density (Woodroffe 2000). Among the carnivores, it is ironic that the more opportunistic-natured bears, which can often adapt to altered habitats, are thus more prone to encountering humans and associated risks of mortality.

There are also many additional synergistic interactions

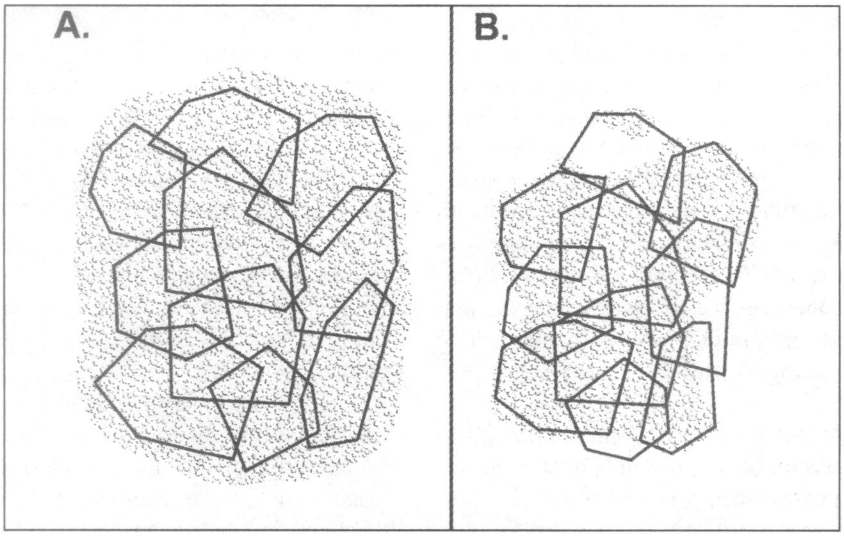

Fig. 1. Hypothetical representation of the effects of habitat loss on bears. In panel A, 10 bears, whose home ranges are indicated by convex polygons, are below carrying capacity because of human exploitation. In panel B, these same 10 bears are forced into a smaller patch of habitat, the fringes of which have been converted to agriculture. This remaining patch of habitat might still suffice to support the 10 bears. However, the smaller size and more irregular shape of the patch makes bears more vulnerable to human exploitation because bears at the edge may be more prone to venture out into the agricultural fields, and people can more easily reach once-secluded areas in the middle.

between habitat loss and other factors that might impact bear populations. Small patches of habitat are more prone to catastrophic fires or food failures (Cochrane 2001) and have less capacity to regenerate fruit-bearing plants because frugivorous seed-dispersers are less likely to visit there (Cordeiro and Howe 2001). Shrinking, isolated patches of habitat also may be less likely to attract immigrant bears, so whereas local overharvest in contiguous habitat can be overcome through source–sink dynamics (what Brown and Kodric-Brown [1977] called the "rescue effect"), small, insular patches of habitat are more prone to extirpation (Peres 2001). Finally, and perhaps most importantly, decreased habitat limits the potential for a population to increase; even if habitat loss does not directly cause a population decline, it may preclude recovery.

For these reasons, habitat loss should be foretelling of reduced bear numbers and population viability. However, the actual relationship between habitat loss and population decline is far from clear. Moreover, sustained or increased habitat is not necessarily indicative of a stable or increasing bear population. These points are illustrated by examples from Asia.

Giant Pandas in China.—Two range-wide surveys of giant pandas have been conducted (and a third is nearly completed). These surveys accomplished 2 things: (1) they estimated panda numbers, and (2) they estimated the area of remaining habitat. In the first survey, conducted during the mid-1970s, some 3,000 people scoured the panda's range, recording panda sightings and scats. A "rough" population estimate of 1,050–1,100 was obtained (Schaller et al. 1985:15–16). This narrow range belies the inherent inaccuracies of the method employed and variability among survey participants (Schaller 1993). A decade later, a smaller team of 35 biologists repeated the survey using more rigorous sampling procedures to measure density of sign, including both scats and bedsites. The resulting estimate of about 900–1,400 pandas provided no indication of population change.

A major finding from these surveys, though, was that panda habitat was being lost at a rapid rate. Large tracts of agricultural land bisected the range into small, fragmented populations. Moreover, low elevation areas that once likely provided optimal habitat were no longer available to pandas (Reid and Gong 1999). In response, many more protected areas have been established (total >30) to prevent further loss of habitat. However, it has become increasingly clear that this alone is insufficient to ensure viability of panda populations because these protected areas are small and disconnected by expanses of unsuitable habitat (Loucks et al. 2001); furthermore, habitat quality, even within some of the protected areas, is deteriorating. A case in point is Wolong Nature Reserve, one of the original and presently largest of the Nature Reserves established explicitly for the protection of pandas. Wolong is also an International Biosphere Reserve and the site of both a panda breeding facility and the first intensive study of radiocollared pandas (Schaller et al. 1985). In 1975, the size of this reserve was expanded 10-fold (to 200,000 ha) to improve protection of panda habitat. Since then, the human population *within* the reserve (mainly minority ethnic groups, who are exempt from China's restrictions on family size) has grown by nearly 70% and the number of households has more than doubled (Liu et al. 1999). Number of households is significant because it is related to timber and fuelwood consumption, which has increased dramatically (An et al. 2001). Accordingly, suitability of the habitat for pandas in Wolong has steadily diminished (Liu et al. 2001*a*). There is some debate as to whether Wolong is atypical (Baragona 2001, Brooks et al. 2001) or just the worst-case of a growing problem (Liu et al. 2001*b*), but either way it exemplifies the point that habitat quality can deteriorate from the bear's perspective while outwardly seeming intact from the human perspective.

A good deal of effort is presently being expended to map as well as assess remaining panda habitat using sophisticated procedures for estimating density of their staple food, bamboo, from satellite imagery (Linderman et al. 2000, Loucks and Wang 2002). This is a promising approach, although the knowledge to define suitable habitat for this species is still lacking (e.g., species and density of bamboo, overstory trees, den trees, hill slope; Reid and Hu 1991, Reid and Gong 1999). Thus, quantifying changes in density of bamboo, although better than simple habitat mapping, might still not accurately depict population trend (Reid 1994).

Sun and Sloth Bears in Southern Asia.—During 1994–96, J.L.D. Smith and I attempted to initiate a field study of sun bears in Thailand. Our greatest difficulty was in locating an area with sufficient bear density. The Khoa Ang Rue Nai Wildlife Sanctuary in southeastern Thailand was recommended to us because it had a new research facility and satellite maps showed it to have a dense forest. The southern border of the reserve abuts other densely-forested protected areas. Stewart-Cox (1995:107) characterized this area as "the largest tract of lowland evergreen forest in Thailand." A few roads and trails penetrated the forest, which facilitated access for trapping and radiotracking. The main entrance was guarded and gated, and there were several guard stations inside. From these indications we expected this to be an ideal study site.

We set out traps and baits and conducted sign surveys. Although we found some old sign, we soon concluded that there were few bears in this reserve; in fact, there was little sign of any medium–large mammals, even at places where they would typically congregate, such as fig trees (*Ficus* spp.) laden with fruit, salt licks, and water holes. We heard numerous reports of poaching, saw signs of poaching encampments, and heard gunshots. One night a binturong (*Arctictis binturong*) was poached near our camp. We noticed that during both day and night, motorcyclists rode freely around the closed gates and past the guards. We learned that one of the roads through the reserve was a main thoroughfare connecting two parallel highways. This sanctuary was certainly not the "secluded world" that Stewart-Cox (1995:107) had described.

Despite suitable habitat, this area exemplified what Redford (1992:412) called an "empty forest." "Often trees remain in a forest that human activities have emptied of many of its large animals...We must not let a forest full of trees fool us into believing all is well."

We encountered a similar situation with sloth bears (*Melursus ursinus*) in Nepal. We surveyed their entire range, a narrow strip of lowland forest and scattered grasslands called the terai. Sloth bears were abundant in Chitwan National Park, in the center of this range, but were absent at the eastern and western extremities of the range, despite suitable habitat. These areas had good forest cover and abundant termites (a staple food for sloth bears)(Garshelis et al. 1999*a*), but sloth bears had apparently been poached out during the previous 2 decades (Garshelis et al. 1999*b*), creating vacant bear habitat.

Sun Bears in Borneo.—Meijaard (2001) reported just the opposite situation for sun bears in Kalimantan (Indonesian Borneo). Here, disappearing forests seemed to be filled with bears, despite supposed periods of heavy poaching. During the 1970s poaching of sun bears appeared to be rampant in Indonesia, as evidenced by the amount of bile illegally exported. During that decade, Meijaard (1999) estimated that gall bladders from about 7,000 Indonesian sun bears were sent to South Korea; additional shipments of gall went to other countries. I previously showed that quantities of traded bile cannot be converted to reliable estimates of numbers of dead bears, or even used to construe trends in levels of bear mortality (Box 1). Nevertheless, it appears from the presently low amount of bile exported, low in-country demand, and according to information from local people, few bears killed for their parts, that during the past 2 decades, the trade in parts has not resulted in large numbers of bears killed (Meijaard 1999).

Interviews with local people across Kalimantan in the mid-1990s indicated that sun bears were still "relatively abundant" in most forested areas (Meijaard 2001). The forests, however, were rapidly being cut, which presumably would escalate human-related mortality and thus reduce numbers of bears (Fig. 1). It is difficult, though, to accept Meijaard's (2001) estimate that habitat loss caused 10,000 sun bears to die in Kalimantan during the 1980s, given his evidence that human-caused sun bear deaths appeared to be relatively low during that decade. Also, while habitat loss is obviously troubling, equally troubling is Meijaard's (2001) tenuous prediction that within another decade, 14,000–28,000 more bears will die.

An irony in presenting such alarming numbers is that one could use them to back-calculate an estimate of present numbers of sun bears. Meijaard (2001) converted habitat loss to numbers of dead bears using a "very crude" density estimate of 1 bear/4 km^2 presented by Davies and Payne (1981). This estimate of density was derived from only 2 bear sightings and 9 observations of sign. Extrapolating this density to all of Kalimantan would yield >90,000 sun bears. Extending this density to forested areas of Malaysian Borneo (Sabah and Sarawak) and Sumatra would increase the total to about 190,000 bears (forest areas from Mayaux et al. 1998). Even if sun bear densities in mainland southeast Asia are much lower, the total world population would still well exceed 200,000, which would make this species numerically equivalent to brown bears on a global scale, and second only to American black bears.

The reality is that sun bears are listed by the IUCN as "data deficient," because reliable estimates of population size and trend are unavailable (Baillie and Groombridge 1996). Creating unsubstantiated estimates in the hope of rousing more conservation interest may, as illustrated here, contravene the intended result. Without far better information on the relationship between bear density and habitat, attempts to quantify bear numbers and trends from forest cover data are likely to be misguided.

IRONIES REGARDING HUNTING AND POPULATION TREND

A particularly noteworthy irony regarding bear populations is that most legally-protected populations seem to be declining, whereas most hunted populations are increasing. One explanation is that protected populations tend to be small, and thus more prone to decline as a simple consequence of low numbers (Caughley 1994). Another explanation is that many of these legally-protected populations are really heavily exploited. Oftentimes, the level of human exploitation may be less under a system of managed hunting than supposed total protection. The reasons for this seeming contradiction have a lot to do with the people, finances, energies, and ideologies entailed in a managed harvest, resulting in an infrastructure of managers, scientists, bureaucrats, and hunters, with non-hunters and anti-hunters as overseers. This complex structure is often lacking in the management of protected areas. However, it is also true that countries with managed bear hunting tend to have stronger economies, which can support bear management activities (e.g., research, enforcement) better than countries where hunting is banned. These points are illustrated first by contrasting the management of American and Asiatic black bears, followed by an example regarding polar bears (*U. maritimus*).

American versus Asiatic Black Bears.—The 2 species of black bears are similar in terms of their life histories, and seem similar in terms of reproductive potential, although reproductive data on wild Asiatic black bears is presently insufficient to enable a true quantitative comparison (Garshelis 2002). However, the 2 species are managed very differently. Legal hunting is the main source of mortality for American black bears in most parts of their range, whereas hunting for Asiatic black bears is legal only in Japan. Most American black bear populations appear to be increasing (Williamson 2002), whereas Asiatic black bears are thought to be declining in most areas. The difference is that human exploitation is monitored and controlled in the former case, surreptitious in the latter.

A reviewer of this paper asserted that the cause and effect thesis posed here is reversed. That is, legalized hunting did not result in numerically abundant bear populations; rather, hunting was legalized because bears were numerically abundant. I disagree with this. American black bears were severely over-exploited through the early-mid 1900s. Although regulated exploitation of other North American species, such as white-tailed deer (*Odocoileus virginianus*), dates to the 1600s (Gilbert and Dodds 1987), black bears were much less valued as food so did not inspire efforts to limit the take. Moreover, bears did not generate much interest among recreational hunters on whose behalf game laws were made (Schullery 1983). In fact, mainly during the 1800s and early 1900s, federal, state, and local governments supported programs to destroy both black bears and grizzly bears because they were considered detrimental to raising livestock and crops as well as potentially dangerous to people (Spencer 1955, Cardoza 1976, Brown 1996). An evolution in ideology, beginning in some U.S. states in the early 1900s, eventually led to the designation of black bears as a big game species, with the objective of a sustained harvest (Miller 1990). These laws were passed *because* bear populations had noticeably diminished. Minnesota was one of the last states to classify black bears as big game (1971). In one Minnesota county where bears had been considered "very nearly extinct" prior to their big game listing (Cahn 1921:70, Special Committee on the Conservation of Wildlife Resources 1940), a long-term telemetry study revealed a high density of bears following 20 years of legal hunting (D. Garshelis, unpublished data).

There are many factors — economical, political, historical, cultural, and spiritual — that make it difficult to transfer to Asia the Western traditions of sustained-yield hunting. Proponents of sustainable use in developing countries argue that people are more apt to conserve resources when they have a vested interest in a return from these resources (Gadgil 1992, Kothari et al. 1995, Saberwal 1996). Others, though, have observed that high human densities, abject poverty, class systems, and corrupt governments create a situation where it is nearly impossible to regulate harvests (Bennett and Robinson 2000, Madhusudan and Karanth 2000, Meijaard 2001).

A high market value for bears in Asia makes the regulation of harvest an even more daunting problem. In Korea, for example, Asiatic black bears were subjected to the same sort of government-supported removal efforts as American black bears during the early-mid 1900s (Won 2001). Unlike the situation in North America, however, Korean bear populations continued to plummet from overexploitation into recent times because they were sought commercially. A lesson learned during the evolution of the North American system was that market hunting was detrimental to wildlife populations and was therefore incompatible with recreational and subsistence hunting (Geist 1988, 1994). In fact, recreational hunting enthusiasts were largely responsible for legislation that eventually prohibited market hunting for wildlife in North America (Reiger 1978).

Harvesting animals for profit, though, is not uniformly detrimental to wildlife populations. In North America, many species of furbearing mammals are trapped specifically for sale of their pelts, so the kill fluctuates with fur prices; nevertheless, their populations have been carefully managed by government agencies (Novak et al. 1987). In several European countries, hunters routinely sell their game for personal profit or for income for the landowner or hunting club; in some cases, hunters can only retain a portion of their take. This system has worked for centuries (Bolen and Robinson 1995). In Japan, Moll (2001) suggested that a prohibition against the sale of bear parts might lead to diminished interest in legitimate bear hunting and higher prices for bear gall, which together could result in reduced stewardship of the resource and hence increased danger of bears being over-exploited by poachers.

To be clear, my purpose here was to point out the seemingly paradoxical effects of legal hunting, not to suggest that sport hunting should be promoted where it does not now occur. Simply instituting a legal harvest is obviously not the solution to declining bear numbers. Historically though, in both North America and Europe, managed hunting has been an effective system for protecting bear populations. It has worked because it has enlisted a clientele interested in ensuring continued abundance of the resource. It also has worked because, for species such as bears that can be a nuisance and a threat, it transfers the killing of animals from the general public to a smaller group of people (i.e., the hunters). Both these issues have been instrumental in shaping bear management and conservation in North America, Europe (Klenzendorf and Vaughan 1999, Zedrosser et al. 2001), and Japan (Huygens et al. 2001, Moll 2001). Linnell et al. (2001:348) commented "There is no doubt that the concept of hunting large carnivores as game species is far older in Europe than in North America and has contributed greatly to their persistence." Ironically, in places such as India and Nepal, where bear hunting is now prohibited, preserves that were set aside explicitly for hunting (by both local and European aristocrats) during the 1800s formed the basis of a system of parks and wildlife sanctuaries that now constitute virtually the only remaining areas of intact habitat with viable populations of large mammals, including bears (Israel and Sinclair 1987, Mishra and Jefferies 1991, Rangarajan 2001; negative consequences of these royal hunts and exclusionary policies notwithstanding [Saberwal et al. 2001]).

Polar Bears.—During the 1960s it became evident that polar bears were being over hunted. In 1973 an historic conservation agreement was signed among all 5 nations with populations of polar bears (U.S., Canada, Norway [for Svalbard], Denmark [for Greenland], and the former U.S.S.R.). Interestingly, the International Agreement on the Conservation of Polar Bears (International Legal Materials 13:13–18), which took effect in 1976, did not prohibit hunting, but rather limited it to native people using traditional methods (IUCN/SSC Polar Bear Specialist Group 1999). Within this restriction, the member nations went in different directions. Canadian jurisdictions imposed hunting quotas in most areas, whereas the U.S. could not, under the constraint of the Marine Mammal Protection Act of 1972 (16 U.S. Code 1361–1407). However, non-mandatory harvest guidelines have been developed for native communities in Alaska. In Greenland, there are no quotas on polar bears but the harvest is limited to native people who hunt or fish full time. In Svalbard, hunting of polar bears was forbidden after the Agreement. In Russia, a prohibition on the hunting of polar bears predated (1956) the Agreement. Russia thus appears the most restrictive for the longest time, yet in reality, the strongest concerns about poaching polar bears exist in Russia (IUCN/SSC Polar Bear Specialist Group 1999). What may superficially seem ironic but pertinent to this discussion is a recent agreement to permit native Russian people to hunt polar bears in the population shared with Alaska. The presumption is that a legal hunt, with the self-serving interest to remain within sustainable limits, would be more effective at conserving this population than striving (probably unsuccessfully) for total protection. Management for harvest tends to be more successful because it broadens the number and scope of people with a stake in maintaining a healthy population.

UNCERTAINTY AND CONSERVATION

The only real certainty in bear conservation is that human intrusion, via both direct over-exploitation and habitat destruction, is the main factor threatening bears worldwide. The degree of threat, however, is very uncertain. The best information exists for North American and European bear populations. Geographic ranges are generally well-delineated, and population estimates and growth rates, though often inexact, are usually based on some research data (Table 1). For Andean bears (*Tremarctos ornatus*) of South America, there are good distribution maps but no data-based estimates of abundance or trend (Peyton et al. 1998). Good range maps and an estimate of population size exist for giant pandas, but there is no good information on population trend (Reid and Gong 1999). Very generalized range maps, poor population estimates, and weak evidence of population trend are available for the other Asian species (Table 1). All these species, though, are perceived to be in trouble.

To aid in the conservation of these species, many believe it is necessary to provide population numbers and extinction scenarios. Population viability analyses are certainly productive exercises that may be especially important in illuminating sensitive population parameters and weaknesses in the data (Sæther et al. 1998, Wiegand et al. 1998); however, these should not be confused with actual population projections (Mills et al. 1996, White 2000). We have rarely been able to track population trends in the present, and because we lack vital biological information for many of the species (Garshelis 2002), it seems improbable that forecasting the future would be very accurate. Referring to large whales, but describing a situation applicable to bears, Gerber et al. (2000:318) observed: "our limited knowledge... makes it extremely difficult to quantify the degree to which a population may go extinct in a specific period of time... Unfortunately, the public and the press have not been entirely aware of these difficulties. Worse, advocacy groups on both sides of the environmental continuum and even some scientists have filled this void with inaccuracies."

Some believe that admitting to uncertainty would muddy the message, and thereby detract from conservation initiatives. That view holds that firm, bold, and clear assertions, even if not entirely backed by factual information, yield better results in terms of protecting environmental welfare than does revealing uncertainties. Those opposing this approach warn of blurring the distinction between science and advocacy, which can be especially tempting when both are harbored in the same individual (Bowen and Karl 1999). Schrader-Frechette and McCoy (1999) argue that occasionally compromising science in favor of advocacy will ultimately create the perception that science was abandoned. If we do not universally adhere to all the principles of science, then we must be prepared to wade into ethical battles, where scientific viewpoints no longer have ascendancy.

There is an obvious counter-argument to this reasoning: brandishing uncertainty may not be a powerful means of swaying policies toward better conservation of bears. Acknowledgment of uncertainty in the scientific arena is one thing; highlighting it in the political arena is quite another. There is certainly some wisdom in this, but I offer several reasons why there is usually greater merit in making the uncertainties clear to the public and the politicians. (1) If new data do not support previous suppositions (e.g., about a population decline), and if the uncertainties inherent in the original suppositions were not made clear, scientific credibility will be damaged and future conservation efforts based on scientific information may be compromised. (2) Optimism generally provides more motivation for conservation action than pessimism (Beever 2000), and, in many cases, uncertainty provides a greater array of optimistic scenarios. Uncertainty in this context should not be confused with ignorance, which is always detrimental (Garshelis 1997). (3) Incognizance of uncertainties may detract from efforts to gather more data and improve methodologies. False con-

Table 1. Relative degree of certainty regarding geographic range, population numbers, and population trends of the 8 species of bears. Symbols (++ reasonably good, + fair, 0 poor or nonexistent) represent subjective ratings by the author[a] for comparisons within and among columns.

		Informational quality[a]		
Species	Geographic area	Range	Numbers	Trend
American black	North America	++	+	+
Brown	North America	++	+	+
	Europe	++	+	+
	Asia	+	0	0
Polar	Arctic	++	+	+
Andean	South America	++	0	0
Giant panda	Asia	++	+	0
Asiatic black	Asia	+	0	0
Sloth	Asia	+	0	0
Sun	Asia	+	0	0

[a] Based mainly on Servheen (1990), Servheen et al. (1999), Williamson (1999), Sathyakumar (2001), and Zedrosser et al. (2001), plus accumulated knowledge and personal experience.

fidence in presumptions about population declines thus may inhibit discoveries that could aid in detecting population change. This is an extension of Gibbs et al.'s (1998:940) view: "The primary consequence of failing to improve methodologies for identifying population change in ecology will be a chronic failure to detect population change. Unfortunately, these statistical errors will frequently be misconstrued as reflecting population 'stability,' lack of treatment effect, or ineffectiveness of management." Hence, if the uncertainties are not eventually remedied, even effective conservation programs may yield no measure of success because it will not be possible to detect a population increase.

I contend that in the interests of both science and conservation, biologists should *emphasize* the uncertainties of population assessments and thus the necessity for more rigorous research. This may seem counter-intuitive in terms of conservation, but the logic is this: in the presence of uncertainty efforts should be directed toward ensuring no irreparable harm. The wide range of uncertainty about bear populations should be reason enough for claiming a wide berth in erring on the side of caution.

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