POPULATION MANAGEMENT OF BEARS IN NORTH AMERICA[1]

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Abstract: Population management for black bears (*Ursus americanus*), brown-grizzly bears (*U. arctos*) and polar bears (*U. maritimus*) in North America is reviewed. In different areas bear populations are managed to achieve goals of population control, conservation, or sustained yield. Most North American bears are managed for sustained yields and this topic is emphasized. The consequence of error in population management is high as bears reproduce slowly and reduced populations will require many years to recover. Simulation results where reproductive rates were generous, natural mortality rates were low, and harvests were 75% of maximum sustainable rates indicated that populations reduced by half will require >40 years to recover for brown (grizzly) bears and >17 years for black bears. Under optimal conditions for reproduction, natural mortality, and with males twice as vulnerable as females, maximal sustainable hunting mortality was estimated as 5.7% of total population for grizzly bears and 14.2% for black bears. In recent decades, all 3 species have obtained the status of game animals in most jurisdictions and management for control objectives is increasingly uncommon. Management for conservation requires primary emphasis on habitat protection and on minimizing mortalities from any source. Managers of hunted bear populations use information from hunters, from sex and age composition of killed bears, from research programs, and from computer simulation studies. Non-critical uses of data from any of these sources may lead to management error. Data on age-at-harvest is especially prone to misinterpretation. Techniques used to limit harvests by managers of hunted bear populations are reviewed. The primary constraints facing bear population management derive from inadequate habitat protection, political pressures, technological limitations of available population management techniques, and inadequate financial support for management.

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Population management efforts designed to enhance or stabilize bear numbers are recent in the United States and Canada. In the last century and early portion of the present century, black and grizzly bears were widely regarded as impediments to desired development and human safety. Bounties for killing bears were offered in many jurisdictions. This attitude, combined with habitat destruction, led to the elimination of grizzly bears throughout most of the United States except Alaska and the reduction of black bears especially in the southern and southeastern United States (Cowan 1972, Jonkel 1987). By the mid-19th century, polar bear populations were also greatly reduced by market hunting for their hides (Anon. 1965, Stirling 1986).

Attitudes towards bears began to change in the 20th century. Instead of being classified as "predators" or "vermin" that could be killed indiscriminately, bears were classified as fur animals subject to regulated commercial harvests. By the 1920's, bears were elevated to the status of "game" animals in most areas (Table 1). Typically, limitations on sale of hides, meat or other bear products came along with game animal status as well as significant limitations on hunting opportunities (seasons, bag limits, techniques, etc.). In some areas further limitations resulted when bear populations were greatly depleted. At this point populations were classified as "threatened", the status of the grizzly in the lower 48 states, or "endangered", such as the black bears in Texas since 1987 (Wallace 1987). There is nothing inevitable about a downward trend in bear numbers to a threatened or endangered status. For black bears, at least, populations currently are stable in much of the United States and Canada. Also, in some regions, with formerly depleted populations of all 3 species, bears have recovered to a secure status.

The techniques used in modern bear management are the subject of this paper. These techniques are applied to 3 general goals for population management listed by Caughley (1977:168): *control* (treatment of a population that is too dense to stabilize or reduce its density), *conservation* (treatment of a small or declining population in such a way as to raise its density), and *sustained yield* (exploitation to take from a population a long-term sustained yield of surplus animals without causing a population decline). Although all 3 of these goals are discussed, primary emphasis in this paper is on sustained yield management.

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Table 1. Year in which bears were declared to be game animals in different portions of North America. Dates refer to black bears except where indicated by "G" for grizzly bear or "P" for polar bear.

Location	Year classified as game animal	Source
Alaska	1902(G)[a]	57th Cong. Sess. I Chap. 1037
New York	1903	Clark (1978)
Pennsylvania	1905[a]	Alt and Lindsey (1980)
British Columbia	1909[a]	Anon. 1980
Montana	1923	Dood et al. (1986)
Montana	1923(G)	Dood et al. (1986)
Oregon	1925, 1970[b]	Anon. (1987*a*)
Texas	1925	Winkler (1975)
Michigan	1925	Harger (1980)
Quebec	1926[c]	Caron (1980)
Yukon Territory	1928[a]	MacHutchon and Smith (1988)
Alberta	1929(G)	Nagy and Gunson (1988)
Arkansas	1927[d]	Conley (1977)
Arizona	1927[a]	LeCount (1977)
South Carolina	1927	Stokes (1977)
Arizona	1929(G)	Brown (1985:154)
Wisconsin	1930	Kohn (1982)
Maine	1931[e]	McLaughlin (1986)
Washington	1933, 1969[f]	Poelker and Hartwell (1973)
Alaska	1939	Code Fed. Reg. Title 50(91.1)
Vermont	1941	Wiley (1978)
Colorado	1941	Beck (1979)
Manitoba	1942	Shoesmith (1977)
Idaho	1943	Beecham (1986)
California	1948[g]	Anon. (1987*b*)
Alaska	1948(P)[a]	Anon. (1965:51)
Northwest Territories	1949(P)	Urquhart and Schweinsburg (1984)
Maryland	1949	Taylor (1984)
Yukon	1950(P)[h]	Stirling and Calvert (1985)
Oklahoma	1951	Vohs (1977)
Massachusetts	1953[g]	Cardoza (1978)
Newfoundland	1961	Russell and Forsey (1978)
Ontario	1961[i]	Clarke (1961)
New Brunswick	1961	Cartwright (1978)
Newfoundland	1962	Mahoney (1984)
Saskatchewan	1963	R. Seguin (Sask. Parks, Recreation and Culture, Meadow Lake, pers. commun.)
Nova Scotia	1966	Patton (1978)
Utah	1967	Burruss (1979)
West Virginia	1969	Rieffenberger and Allen (1978)
Quebec	1969(P)[h]	Stirling and Calvert (1985)
Ontario	1970(P)[h]	Stirling and Calvert (1985)
Manitoba	1970(P)[h]	Stirling and Calvert (1985)
Newfoundland	1971(P)	Stirling and Calvert (1985)
Minnesota	1971	Hugie et al. (1978)
New Hampshire	1983[j]	Orff (1987)

[a] Date of first bag limit or season restriction.
[b] First declared game in 1928, replaced, then redeclared in 1970.
[c] Vallee (1977) gives 1970 as date game status was assigned in Quebec.
[d] Date of total season closure.
[e] Hermes and Hugie (1977) note black bears were bountied until 1957 and game status was being recommended in 1977.
[f] Date of first season, game animal status repealed in 1951 in some areas, reinstituted in 1969.
[g] Burton (1977) gives 1957 as date game status was assigned in California.
[h] Date of legal basis for current management.
[i] Polar bears in Ontario have been treated, for management purposes, as a furbearer since 1971 (G. Kolenosky, Ont. Ministry of Nat. Resour., Maple, pers. commun.).
[j] $20 bounty removed from bears in 1955, first season in 1961.

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CONSEQUENCE OF ERROR

For all 3 management objectives the consequence of error in managing bear populations is high. Bear populations that are inadvertently reduced to lower levels than desired will require many years to recover. This is because all 3 species of North American bears have long lifespans (>20 years), low reproductive rates (an average of 2 cubs produced by adult females every 2-6 years), delayed reproductive maturity (first breeding at 3-7 years), high survivorship of adults, variable survivorship of young, which is frequently dependent on environmental conditions (Rogers 1983), and typically little fluctuation in number of adults from year to year (Jonkel 1987, Kolenosky and Strathern 1987, Kolenosky 1987).

The period required for recovery of reduced populations of black and grizzly bears was simulated using a simple deterministic model (Miller and Miller 1988) in a scenario involving overharvests by hunters. In these simulations, maximally productive populations of black and grizzly bears that were stabilized by hunting were suddenly overharvested by doubling the exploitation rate. When the population declined to half its original size, hunting was restricted and the time required for the population to recover to its initial size was noted. When no hunting occurred during the recovery period, the black bear population recovered in 6 years compared to 10 years for grizzly bears (Table 2). When hunting during the recovery period occurred at 75% of the maximum sustainable hunting rate, it took almost 3 times longer for black bears to recover and 4 times longer for grizzly bears (Table 2). These results are minimal values as the reproductive and natural mortality rates used were set at the most optimistic values that have been reported (Miller 1989).

Table 2. Simulation results for estimating period required to recover from overhunting that caused a 50% reduction in maximally productive grizzly bear and black bear populations. During recovery period population was subject to hunting rates of 0, 50, and 75% of the initial rates at which populations were stable.

	Grizzly bear	Black bear
Years required to recover from reduction when hunting is held at following fractions of initial hunting rate:		
(No hunting)	10	6
50%	19	9
75%	40	17

MANAGEMENT FOR CONTROL OF BEAR NUMBERS

Until the current century, reduction of bear numbers was the most common objective for bear population management. In some parts of North America, bears are still sufficiently abundant or troublesome to humans that management efforts involve reducing bear densities (e.g., Poelker and Hartwell 1973, Jorgensen et al. 1978, Ambrose and Sanders 1978, Poelker and Parsons 1980, Will 1980, Miller 1990*a*, Gasaway in press). Such areas have become increasingly rare and geographically restricted in recent decades. They will likely become even rarer.

However, where human and bear populations coexist, managers will have to deal with some problems. These problems can result in bear mortalities that are large enough to be significant from a population management standpoint. For Yellowstone grizzlies, control killings of only a few additional females may mean the difference between continued population decline and recovery (Knight and Eberhardt 1984, 1985). In such cases, human populations, not bear populations, must make the needed accommodations for coexistence.

Many states and provinces compile data on the number of human-bear conflicts reported. It is sometimes implied that an increase in the number of nuisance bear complaints reflects an increase in bear numbers. More often, however, increased complaints reflect a change in human use of bear habitat. Increased human-bear conflicts more commonly correspond to a decline in bear populations, not an increase.

Research in Alaska and other northern regions has demonstrated that predation from bears and wolves (*Canis lupus*) can inhibit recovery of depleted moose (*Alces alces*) and caribou (*Rangifer tarandus*) populations (Ballard et al. 1980, Franzmann et al. 1980, Gasaway et al. 1983, Ballard and Larsen 1987, Ballard and Miller 1987, Boertje et al. 1988). These findings have resulted in pressure from sportsmen and subsistence hunters to reduce numbers of predators to permit faster growth and higher harvests of prey populations. In response to such pressures, grizzly bear seasons have been liberalized and harvests have increased in many portions of interior Alaska (Miller 1989). These changes represent a geographically widespread shift from conservative grizzly bear management strategies to more aggressive ones in which the likelihood of management error in these areas is increased. In at least 2 areas, increased harvests have resulted in declines in grizzly bear populations (Reynolds and Hechtel 1988, Miller 1990*a*). Elsewhere, results are inconclusive (Gasaway 1988) or no field studies designed to evaluate trends in bear numbers are ongoing.

Responsible management of bear populations under circumstances such as these is especially challenging because the techniques available to document changes in bear populations are imprecise. This makes it difficult to establish realistic criteria by which to judge when bear population reduction goals have been met. Also, there sometimes is inadequate recognition that management prescriptions for predator reduction programs that involve bears (e.g., Gasaway in press) need to be different than for predators like wolves that have much higher reproductive rates.

MANAGEMENT FOR CONSERVATION

Conservation is the general objective for grizzly bear management efforts in the lower 48 states, for black bear management in southern and southeastern states, and for management of all species in national parks. In some large Alaskan parks where grizzly bears are abundant, population management is less important than people management and habitat protection. Most national parks in the U.S. and Canada are not so fortunate and require active bear management to assure perpetuation of bear populations (Knight and Eberhardt 1987, Knight et al. 1988, Horejsi 1989).

Even in parks and other protected areas, baseline data on population density and composition may be critically important in evaluating impacts of environmental accidents or changing patterns of human use of areas occupied by bears. The absence of a systematically obtained baseline estimate of bear density in Katmai National Park made it difficult to evaluate whether the bear population had declined as a result of the 1989 oil spill from the *Exxon Valdez*. Similarly, in Glacier National Park, the lack of a systematically collected historical record of bear numbers made it difficult to isolate human use patterns, which may have caused a reduction in bear numbers (Hayward 1989, Keating 1989).

In parts of North America small bear populations survive only in small pockets of habitat isolated from each other. This fragmentation exposes these populations to higher probabilities of extinction because of chance events and environmental variation. Managers of these populations must determine how large these populations and reserves should be to insure persistence in the face of natural catastrophes, and random environmental, demographic, and genetic events (Schaffer and Sampson 1985). For Yellowstone grizzlies the minimum viable population that gave a 95% probability of surviving for 100 years was estimated at 50-90 bears (Schaffer and Sampson 1985). Using the same general approach but with different data on mortality rates, the estimated minimum viable population was estimated at 125 bears (Suchy et al. 1985).

The Interagency Grizzly Bear Committee formed in 1983 is an example of the kind of coordination that is essential if remnant populations of grizzly bears are to survive in the lower 48 states (Salwasser et al. 1987). This committee includes representatives of 5 U.S. agencies that manage portions of grizzly habitat plus 6 state or provincial wildlife agencies, and 2 Indian tribes. Working together, these agencies have developed a coordinated set of objectives and strategies to direct conservation efforts. The International Agreement on the Conservation of Polar Bears and their Habitat is another example of the kind of cooperation needed to perpetuate healthy bear populations (Stirling 1986, 1988*a*).

Techniques for estimating population size and trend, discussed below, are especially needed in management of reduced populations for conservation objectives. Unfortunately, some of the techniques that provide the most accurate population estimates may frequently be inappropriate for very small remnant populations of bears because these techniques are usually imprecise when applied to small populations. In addition, subjecting such populations to the additional stress and mortality associated with marking studies may be unwise. In managing greatly reduced remnant populations of bears, managers may find it more productive to concentrate on habitat protection issues rather than on efforts to document bear numbers or mortality rates with marking studies that may produce only uncertain results.

SUSTAINED YIELD

Sustained yield management of bear populations is the management goal in most areas of North America inhabited by bears. Most commonly the yields are taken by hunters. Perhaps because sustained yield management is not usually conducted in a crisis atmosphere where bear populations are threatened or where bears are seen as damaging to humans' economic interests, sustained yield management has not received as much attention as it deserves. More concern is merited because correct management of populations that have sustained yield goals may prevent crisis situations from developing. Also, population management techniques are especially important in managing for sustained yields. For these reasons this topic is given primary emphasis in this review.

The principle behind sustained yield management is that populations produce a surplus of animals that can be removed or harvested without causing population declines. Under sustained yield management, harvesting takes the place of mortalities that would occur from old age and other causes. In very dense populations, reproductive rates of bears are suppressed by density-dependent mechanisms that act to prevent the population from overshooting carrying capacity. Because population growth is suppressed by these mechanisms, populations at carrying capacity can support little harvest. If such dense populations are harvested and bear density declines, reproductive rates should increase and natural mortality rates should decline,which produces a surplus that can be taken annually without causing declines in bear numbers. Maximum sustainable yield (MSY) is the point where population size and productivity balance to produce the maximum size of harvest without causing a population decline. At populations lower than MSY, productivity and sustainable harvest rates remain high but fewer total bears can be harvested without causing a population decline.

For bear managers the MSY population size is more useful theoretically than practically since the "optimum" population size will be unknown. This is because reproductive and mortality rates can vary from year to year in a density independent fashion based on fluctuations in food supply (Jonkel and Cowan 1971; Rogers 1976, 1983, 1987). Managers striving for sustained yields from exploited bear populations try to maintain populations that have good average reproductive rates and small average natural mortality rates in the expectation that such populations will be producing harvestable surpluses at high levels.

The challenge facing managers managing bear populations for sustained high harvests is to identify correctly what harvest levels are sustainable and when sustainable levels are exceeded. To assist in making these determinations the population manager may have information available from hunters, from the animals that are harvested, from field investigations, and from simulation studies.

Information Provided by Hunters

Hunters can provide valuable information to managers of exploited bear populations. Information from hunters is most useful as a flag that alerts managers to potential problems or helps to form hypotheses about population status. These hypotheses can then be evaluated using other lines of evidence.

Number Killed.—Perhaps the single most basic and useful piece of information that can be provided by hunters is the number of bears killed. Increasing numbers of bears killed should alert managers that populations could be declining. Of course, population trend is not necessarily correlated with number killed; increasing harvests could occur without population decline as long as sustainable harvest levels were not exceeded.

Probably the best way to use data on harvest number requires calculation of sustainable harvest rate. With information on reproductive and mortality rates derived from research, this rate can be estimated using simulation models, discussed below. The calculated sustainable rate can be compared with actual harvest rate obtained by dividing number killed by estimated population size. This approach resulted in a recommendation against an increase in polar bear hunting quotas in the Northwest Territories (Stirling et al. 1985).

In a few instances, efforts have been made to use kill numbers to derive population size by assuming the kill represents some percentage of the total population, usually the calculated sustainable harvest rate, and back-calculating from this rate to derive a total population estimate. This is a reasonable procedure only if managers have independent evidence that the population is stable.

Unreported sport or nuisance kills and wounding losses can represent significant sources of mortality that managers should consider. In rural northwestern Alaska, less than half the grizzly bear sport and subsistence harvest is reported as required (W. Ballard, Alas. Dep. of Fish and Game, Nome, pers. commun.). On the heavily hunted Kenai Peninsula in Alaska, where reporting is thought to be fairly complete, wounding loss of black bears was estimated to be 13-16% of reported kill based on mortalities of radio-marked bears (Schwartz and Franzmann in prep.). "Control kills" of nuisance black bears accounted for 36% of known human-caused mortalities and unreported control kills were estimated to equal or exceed reported ones in the Yukon (MacHutchon and Smith 1988). Poaching accounted for 9% of deaths of marked black bears in Maine (Hughie 1982). A third of the known grizzly bear mortality was illegal harvest in Alberta (Peek et al. 1987). The mortality rate of marked grizzlies in Montana was estimated at 0.47, all from illegal, unreported kills (Knick and Kasworm 1989). In 6 studies of marked grizzly bears, 26% of mortalities were caused by illegal harvests compared to 42% by legal hunting (McLellan 1990). Managers need to incorporate estimates of all significant mortality sources into their bear management efforts.

Hunter Effort.—Number of bears killed is best interpreted along with information on level of hunting effort. Increases in number of bears killed under conditions where effort is constant may lead managers to suspect an increasing bear population. The same increase in harvest number where effort is also increasing may suggest an increased exploitation rate and a declining bear population. This indicator was used in Alberta, where managers noted that harvests of grizzly bears increased 100% during a period when effort increased 350% (Nagy and Gunson 1988). In a heavily hunted area in south-central Alaska where grizzly bear density was reduced by about half as a consequence of liberalized hunting regulations, successful hunters reported spending more time before shooting a bear than before the density was reduced (Miller 1990*a*). Typically, hunter effort data are highly variable and statistical tests seldom reveal significant differences. This does not, however, invalidate the cautious use of such effort data to assist managers in forming hypotheses about population trends.

Hunter Success.—Hunter success rates are influenced by improved access or hunter technology, motivation, and number of bears. This means that effort indices should be used with caution. Variability in effort unrelated to population status is apparent in Alaska where non-resident grizzly bear hunters are required to hunt with guides and pay high fees. Resident hunters have no such restriction and need only buy a $25 tag. These differences in cost of hunt affects motivation and is reflected in success rates. The statewide success rate for non-residents is much higher (52% in 1987) than for residents (8.4%) (Alaska Dep. of Fish and Game [ADF&G] unpublished file data). However, in the Game Management Unit that includes Kodiak Island, where highly prized brown bear hunting permits are allocated by lottery, both types of hunters had higher success rates: 19% for residents and 74% for non-residents (ADF&G unpublished file data for 1986). In contrast, average harvest success rate for grizzly bear hunters in Alberta was 3% for residents and 12% for non-residents during 1971-1987 (Nagy and Gunson 1988). Even where hunting is limited by permits, hunter success can be low; Arkansas black bear permittees had 0.4-2.2% success in different years (Pharris and Clark 1987). This variability underscores the need to look for trends in success rates within groups that are as homogeneous as possible with respect to residency, transportation type, motive, and area hunted.

Kill density.—The geographic location of hunter kills is also important. Harvest number with geographic location permits managers to estimate kill per unit area or kill density. Excluding effects of immigration, sustainable kill density can either be calculated (like sustainable harvest numbers) or estimated based on areas where both population trend and kill density are known. Kill density divided by population density was used to approximate grizzly bear harvest rate in a heavily hunted portion of Alaska (Miller 1988, 1990*a*). Kill density estimates were used to illustrate that dangers of overkill of a grizzly bear population was higher in the Canadian portion of the Northern Continental Divide Ecosystem where a legal hunting season was in place than on the U.S. side (Horejsi 1989). Kill density also can be used to establish quantifiable management objectives in management plans.

Integrated Approaches.—In Minnesota, black bear population managers use a hunter survey to collect data on hunting success and bears killed per hunter-day. Data are adjusted to correct for annual variation in food abundance and are used to select the most conservative growth curve that fits this trend from a series of model-generated curves of population growth. Managers then use the selected curve and some subjective criteria to develop estimates of population size and to set harvest quotas (D. Garshelis, Minn. Dep. of Nat. Resour., Grand Rapids, pers. commun.). This approach appears to be a worthwhile effort toward integrating information obtained from hunters with that from other sources into a standardized management framework useful in making objective management decisions.

Information Provided by Harvest Composition

Detection of bear population trend from the sex and/or age structure of harvested bears is more often attempted than achieved (Caughley 1974; Wiley 1980; Gilbert et al. 1978; Bunnell and Tait 1980, 1981; Miller and Miller 1988). Procedures that are appropriate for more productive ungulate populations (e.g., Fraser 1976, Fryxell et al. 1988) are difficult to apply to bears because they are a long-lived and low density species that can sustain only low harvest rates (Harris and Metzgar 1987*a*). Low harvest rates provide a small sample of harvested animals from which to make inferences about the population and a delay in the time required for harvest to perturb the population's sex and age structure. Sex ratio of harvest is more sensitive as an indicator of population status than age structure (Harris 1984), perhaps because all the harvest is distributed between only 2 sexes compared to 20 or more age classes. It is popular to try to use data on age composition of harvest because hunters can be required to submit teeth from their kills. These can be sectioned and age estimated by counting cementum annuli (Stoneberg and Jonkel 1966). The age of harvest results in tables of supposedly "hard" data, the utility of which is more frequently assumed than demonstrated.

Differences in the sex and age composition of bear populations subjected to different levels of hunting have been documented (Jonkel and Cowan 1971; Beecham 1980; Kolenosky 1986; Reynolds and Hechtel 1988;

Miller 1988, 1990*a*). As yet, these differences have not been clearly related to differences in the age composition of bears harvested from these populations. Increases in number of black bear females harvested has been correlated with increased harvest rate in Ontario (Kolenosky 1986).

The sample of bears shot by hunters will seldom directly reflect the population composition. Hunters are selective and bears have differential vulnerability based on sex, age, or reproductive status (Bunnell and Tait 1980). A further problem is that most interpretations of harvest composition data assume a stable age distribution, which is usually inappropriate. Relaxation of stable age distribution assumptions may be possible if independent information on rate of change in population is available (Eberhardt 1985, 1988).

Commonly, age data on sex or age composition of bear harvests are used to infer that populations are stable because mean (or median) age or sex ratio of harvested animals is constant. Similarly, some managers look for decreasing mean (or median) age of harvest (especially of males) or increasing proportions of females in the kill as indicators of overharvest. Such interpretations can lead managers into unwarranted complacency about population status. When birth and death rates are constant, the sex and age composition of the population will stabilize regardless of population trend. This has been recognized since the 1907 paper by Lotka (Caughley 1977) but remains a source of confusion. When birth and death rates are not constant or vulnerability by sex or age class is changing, harvest composition may change in response. This change, however, is not necessarily related to a change in population status.

Managers should be cautious in setting planning objectives based on age or sex ratio in harvest statistics. Benchmarks such as "no fewer than 60% males in the total harvest" may be inadequate to prevent overexploitation. The sex ratio of harvest at sustainable harvest levels is not a constant. Instead, this value is a function of a number of factors including the relative vulnerability of each sex to human-caused mortality, sex and age-specific natural mortality rates, proportion of total mortality that is represented by harvest, and sex ratio at age of first vulnerability to hunters. Failure to meet an objective of at least 60% males could, for example, be "remedied" by adding an early spring season when males have high vulnerability (O'Pezio et al. 1983, Miller 1990*b*, Van Daele et al. 1990) rather than by decreasing kill of females. It is preferable to set exploitation guidelines in terms of the total adult female harvest as has been done for polar bears (Taylor et al. 1987*b*).

A promising approach to interpretation of sex and age composition of black bear harvest data was suggested by Fraser et al. (1982). This approach exploits the higher harvest vulnerability of males compared to females (Bunnell and Tait 1980), which results in a progressive decline in the proportion of males in older age classes. At some age, the higher vulnerability of males will be offset by the larger number of surviving females and the harvest at that age and older will favor females. In lightly exploited populations the age at which females predominate in the harvest will be older than in heavily exploited populations. A regression of percent males in harvest on age class will have a steeper negative slope in heavily hunted populations (Fraser et al. 1982).

Simulation studies have indicated that for bears this model is sensitive to a number of likely violations of underlying assumptions (Harris and Metzgar 1987*a*). Even if it lacks robustness, however, this approach may be useful as a tool to examine conflicting interpretations of available data. In a portion of south-central Alaska, the Fraser et al. (1982) approach was successfully used to document that current grizzly bear exploitation rate was higher than formerly (Miller 1988). Even though harvest rate could not be directly estimated because of violations of the model's assumptions, this analysis was useful in discrediting the hypothesis that the bear population was unaffected by increased harvests. Also, the most likely bias in the use of the Fraser et al. (1982) approach in Alaska would have resulted in an underestimation of harvest rate. This was because vulnerability of females declined in the adult age classes when females were periodically protected by being accompanied by offspring (it is illegal to shoot grizzly bears accompanied by cubs or yearling offspring). Because the estimated harvest rate was an overestimate but was still higher than the calculated sustainable rate, it was useful in demonstrating a clear need for reduced harvests.

A more complex approach for interpreting sex and age composition of harvest data was developed by Tait (1983). Using sex and age composition of harvest data, Tait's approach uses non-linear optimization procedures to develop maximum likelihood estimates for historic population size, hunting rate, recruitment rate, and other parameters. Unfortunately, Tait's model has yet to be adequately tested with real harvest data or evaluated to see how robust it is when underlying assumptions are violated. Alaska is currently making an effort to conduct such tests.

The limitations of sex and age composition of harvest data should not discourage managers from collecting these data and continuing to investigate meaningful ways of using them. Compared to field studies as a way of evaluating population status, harvest data are much less expensive to collect. In using these data, managers must be aware of the limitations, however, as common misinterpretations could lead managers into misclassifying declining populations as stable. With existing technology, it is clear that the limitations on use of composition of bear harvest data are such that hunting remnant populations of bears cannot be justified on the basis that such data would be helpful in evaluating population status.

Information Obtained from Research

Research is an important component of sustained yield management for bears. Research is necessary because bear population management has few generally accepted techniques that can be widely applied to evaluate population size or trend (Harris 1986). Research is not needed for each exploited population. Frequently, adequate results can be obtained by cautious extrapolation from research done elsewhere. However, it should be recognized that responsible sustained yield management of a bear population will be expensive and may require field studies to estimate population size, population density, movements, or critical reproductive and mortality rates.

Population Size and Trend.—Research programs most commonly address estimation of population size. Frequently, population size is estimated using some variation of capture-mark-recapture procedures such as the Seber-Jolly technique (DeMaster et al. 1980, Beecham 1983, Amstrup et al. 1986, Kolenosky 1986). This technique requires an estimate of survival rate in addition to the other standard assumptions of capture-recapture procedures (Seber 1982). Where survival estimates are not available, black bear population estimates have been obtained using more traditional Lincoln Index procedures (Jonkel and Cowan 1971, LeCount 1982, Young and Ruff 1982, Miller and Ballard 1982, Beecham 1983, Aune and Brannon 1987). Frequently it is difficult to convert population estimates obtained using such techniques to density estimates because of uncertainty about size of the area occupied by the estimated population.

In Alaska, intensive capture-recapture techniques using radio-telemetry to correct for lack of population closure have been used to derive black and grizzly bear density estimates in small (<2,000 km^2) areas (Miller et al. 1987). With this approach the area occupied by the estimated population does not have to be estimated. In 1 area this technique was used to document statistically significant declines in bear numbers caused by hunting (Miller 1990*a*). Elsewhere, these estimates serve as baselines for documenting potential changes in density caused by hunting, development, or habitat deterioration (Schoen and Beier 1989, Miller and Sellers 1989, Ballard et al. 1990). Such density estimates were made for 9 grizzly bear and 3 black bear populations in Alaska in a variety of habitats and over a range of bear densities from 6.7-380 bears/1,000 km^2 (Miller et al. 1987, Barnes et al. 1988, Schoen and Beier 1989, Miller and Sellers 1989, Miller 1990*a*, Ballard et al. 1990, Schwartz and Franzmann in prep.). Not all of the problems associated with using these techniques have been resolved. The best methods for dealing with capture bias, small sample sizes, and extrapolation of results to larger areas need additional study. A correction factor for small sample bias in such estimates was developed by Eberhardt (in press).

Other approaches to estimating bear density are based on movements of radio-marked bears (Rogers 1977, Hughie 1982, Reynolds et al. 1987, Schwartz and Franzmann in prep.). Typically, these techniques involve plotting home ranges of individual bears over a study area and calculating the proportions of each home range overlapping the study area. These proportions are summed to derive a population estimate and divided by the size of the study area to obtain a density estimate. Such estimates are usually identified as minimum values because of the possibility that not all bears in the study area were radio-marked. These estimates usually do not include a variance estimate and may contain subjective elements that make them difficult to replicate by different observers. However, they may provide more accurate density estimates than capture-recapture procedures used when the size of the area occupied by the estimated population is uncertain (Hughie 1982).

In the small but politically significant populations of grizzly bears in Glacier and Yellowstone National Parks, grizzly bear population size and trend were estimated from direct observations of bears (Martinka 1971, 1974; Craighead et al. 1974; Knight and Eberhardt 1984, 1985; Keating 1986; McDonald et al. 1988; Hayward 1989). In a Montana study area, number of grizzlies was estimated by adding marked bears known present with unmarked bears seen (Aune and Brannon 1987). Systematic application of direct observation techniques may be preferable for deriving such estimates for critically small populations of grizzly bears such as in the Yellowstone area (Harris 1986). However, these approaches are too labor-intensive to be useful to managers of exploited bear populations. They also lack variance estimates, which makes it difficult to evaluate the significance of reported changes in population numbers.

Another promising approach towards estimating bear density without marking animals was described by Dean (1987). This method employs intensive aerial surveys and a sightability correction factor to estimate number of animals missed during aerial searches.

Research aimed at developing indicators of population trend have not yet produced consistently reliable procedures. Different techniques have been used to detect changes in bear numbers (see review in Harris 1986 and discussions in Pelton et al. 1978, Phelps 1979, LeFranc et al. 1987). There are ongoing efforts to develop trend indices based on use of bait stations (D. Garshelis, Minn. Dep. of Nat. Resour., Grand Rapids, pers. commun.), scent stations (Lindzey et al. 1977, J. Beecham, Id. Dep. of Fish and Game, Boise, pers. commun.), and on track counts in Florida (J. Wooding, Fla. Game and Fresh Water Fish Comm., Wildl. Res. Lab., Gainesville, pers. commun.). In some parts of Alaska, annual aerial counts of bears are conducted at food concentration sites such as along salmon streams. Correct interpretation of such data from any 1 year requires many replicate counts (Erickson and Siniff 1963). Also, the utility of counts at food concentration areas to detect population trend is questionable. Numerous bears would likely be observed in such areas long after the number of bears in less preferred habitats had declined significantly. Away from food concentration areas, high direct annual counts from aircraft may provide a useful index of trend where bear populations are dense and visibility is high. Such counts in alpine habitats are conducted in southeastern Alaska (Schoen and Beier 1989) and on Kodiak Island (R. Smith, Alas. Dep. of Fish and Game, Kodiak, pers. commun.).

Vital Rates.—Rates of birth, death, and recruitment for bear populations can only be established by research programs or by extrapolation from research. For long-lived species with low reproductive and adult mortality rates like bears, estimation of these parameters requires many years of study of >10 radio-marked females (Miller 1989). Estimates of survivorship rates based on regular locations of radio-marked animals can be calculated using procedures developed by Heisley and Fuller (1985) and Pollock et al. (1989). These procedures have been applied on populations of grizzly and black bears (Knick and Kasworm 1989, Schwartz and Franzmann in prep.). Other approaches using kill rates of tagged black bears were described by LeCount (1982), Kolenosky (1986), and Miller (1987). For polar bears and grizzly bears, mortality rate of adult females was shown to be the most critical factor in correctly estimating population growth rate or sustainable mortality rates (Knight and Eberhardt 1985, Taylor et al. 1987*b*).

In cases where research objectives require capture or handling of bears, the studies themselves will result in some mortalities or other stresses on bear populations. These stresses may not be significant to healthy bear populations (Ramsay and Stirling 1987), but they may make such studies inappropriate for depleted populations. Whether conducted on depleted populations or not, all proposed studies requiring handling of bears should receive adequate peer review to assure that poorly designed or implemented projects are not authorized.

Information Obtained from Simulation Studies

Information obtained from hunters, from harvested bears, and from field studies needs to be integrated into a conceptual framework or model where it can be used to make management decisions. Managers are increasingly finding mathematical models of populations to be useful tools for organizing and making decisions from such information. Computers are useful tools for examining such models as they permit managers to quickly make the lengthy and repetitive calculations needed to estimate parameters like sustainable harvest levels.

Deterministic models used to estimate sustainable harvest levels have only 1 result per set of inputs. These models are relatively simple to make. Useful deterministic models can be made by persons without programming talents using conventional spreadsheet software. Such models may introduce systematic error in species, like bears, with multi-year periods of maternal care (Taylor et al. 1987*c*).

Stochastic models, where life history events are assigned probabilities instead of fixed rates, are useful in examining the range of possible outcomes per set of inputs. Software useful in constructing stochastic models for species with any kind of life history has been developed by Harris et al. (1986). This software was used to evaluate sensitivity of harvest data (Harris and Metzgar 1987*b*) and is useful in predicting, for example, probability of survival of small populations of bears.

Deterministic models based on ANURSUS (Taylor et al. 1987*a*) with optional stochastic features have been developed specifically for each of the 3 North American bear species. ANURSUS attempts to mimic the dynamics of bear populations and, as a result, requires a daunting number of input parameters. The 3 species models based on ANURSUS are currently being linked and documented (M. Taylor, Northwest Territ. Dep. of Renewable Resour., Yellowknife, pers. commun.). When this is accomplished, ANURSUS can be more widely tested and used to establish management objectives based on sustainable yield.

ANURSUS was used to estimate sustainable harvest levels for adult female polar bears. Less than 1.6% of the total population of all bears could be harvested as adult females (Taylor et al. 1987*b*). Based on this finding sustainable harvest number can be approximated (M. Taylor, Northwest Territ. Dep. of Renewable Resour., Yellowknife, pers. commun.) as:

$$H = (N)(0.015/F),$$

where H is number of bears that can be harvested, N is total population number, F is the proportion of adult females in the harvest, and the 0.015 constant is derived from the simulation result that <1.6% population can be harvested as adult females (Taylor et al. 1987*b*). It follows that if the whole harvest is adult females, then harvests of 1.5% of the population can be sustained. If a proportion of the harvest is male then a larger percent harvest can be sustained. The maximum sustainable harvest rate for polar bears was estimated at 4.5%; this occurred when 33% of the harvest is female (Stirling 1988*b*). In the Yukon Territory, ANURSUS was used to make preliminary estimates of maximum sustainable harvest levels for male (6%) and female (2.5%) segments of regional grizzly bear populations (B. Smith, Yukon Dep. of Renewable Resour., Fish and Wildl. Branch, Whitehorse, pers. commun.).

Deterministic models were used by Bunnell and Tait (1980) to estimate sustainable mortality from all causes. When natural mortalities are subtracted separately, such models estimate sustainable harvest levels. The consequence of error simulation discussed earlier illustrates this application. With the generous estimates of reproductive rates and survivorship from natural mortality used to estimate population recovery period (Table 1), maximum sustainable harvests were estimated at 7.8% for grizzly bears older than 2.0 and at 15.9% for black bears >1.0 (Table 3). These were converted to estimates of sustainable harvest of the whole population using typical values for mortality of cub and yearling grizzly bears and cub black bears (Bunnell and Tait 1985). Under these conditions and assumptions, maximal sustainable annual hunting mortality was 5.7% for grizzly bears and 14.2% for black bears (Table 3). Elsewhere this approach was used to estimate that sustainable harvests for Yukon grizzlies was 2-3% of the population (Sidorowicz and Gilbert 1981). McCullough (1981, 1986) estimated higher sustainable harvest levels than other models by incorporating density-dependent effects on recruitment. There is both direct and indirect evidence for such relationships (Rogers 1983; Kemp 1972; Stringham 1980, 1983; Young and Ruff 1982; Schwartz and Franzmann in prep.). In my view, however, these relationships are as yet too poorly understood to be safely incorporated into estimates of sustainable harvest levels for hunted populations.

Estimates of sustainable harvest rates derived from models may be compared with calculated harvest rates derived from kill numbers and population estimates. In cases where different sexes have different vulnerability to hunters, population harvest rate can be estimated directly from information on the sex and age composition of the population (Bunnell and Tait 1985) or harvest (Fraser et al. 1982). Such comparisons should be viewed skeptically especially when age distributions are not stable (Caughley 1974, 1977; Harris 1984).

Harvest Controls

Managers have numerous regulatory tools for influencing the number or composition of bears harvested (Phelps 1979, Harger 1978). The effectiveness of any particular tool will vary among areas depending on hunting conditions and the type and motives of the hunter.

Seasons and Bag Limits.—Number of bears taken by hunters can usually be reduced by shortening seasons and increased by lengthening seasons. However, season length works to reduce kill only to a point; in Pennsylvania, 736 black bears were taken in a 1979 open hunting season only 1 day long (Lindzey et al. 1983). A similar number are currently being taken with a 3-day season (G. Alt, Pa. Game Comm., Moscow, pers. commun.). Seasons can be held periodically instead of shortened. On the Alaska Peninsula in southwestern Alaska, grizzly bear hunting has been allowed only on alternate years in an effort to reduce harvest and maintain open hunting (Sellers and McNay 1984).

Shorter seasons may give managers just as many bears killed by hunters in a shorter time, hunting under more crowded conditions. When this occurs, managers may choose to limit the number of hunters by issuing permits. Hunting by limited permit can augment the quality of the

Table 3. Estimated sustainable yield from maximally productive populations of grizzly and black bears (input parameters reported in Miller [1989]).

Annual hunting rate for initial stabilized population of grizzly bears (>2.0-years-old) and black bears (>1.0-years-old)	7.8%	15.9%
Equivalent hunting rate for total population (all ages)	5.7[a]	14.2%[b]

[a] Total population estimated from number of 2-year-olds by assuming yearling and cub mortality rates of 0.20 and 0.35, respectively, for each sex.
[b] Total population estimated from number of yearlings by assuming cub mortality rates of 0.22 for each sex.

hunting experience and, depending on number issued, may serve to maintain trophy bears in the population.

In some parts of Alaska, special permits and seasons are also used to minimize damage by and danger from grizzly bears that enter rural villages. By providing for these bears to be taken legally, managers achieve more accurate records on kill rates and allow the public to effect control actions that would otherwise have to be accomplished at public expense.

Seasons can also be adjusted to influence the sex of bears taken. This is particularly true during spring seasons because male bears tend to leave dens earlier than females, move greater distances, are not accompanied by cubs, and spring hunters may be more selective for large (male) bears (O'Pezio et al. 1983, Schoen et al. 1987, Miller 1990*b*, Van Daele et al. 1990). During spring grizzly bear seasons in Alaska (1984-1988), 74% of grizzly bears taken (n = 2,563) were male compared to 55% of bears taken in fall seasons (n = 2,963) (ADF&G unpublished data). A similar pattern was evident for black bear where 75% of spring bears harvested during 1984-1988 were male (n = 4,691 bears killed) compared to 64% in fall harvests (n = 2,887). In some areas, polar bears killed in seasons open during the den entrance and emergence period are more likely to be females because dens are typically on land, which may be near villages of native hunters, and only pregnant females use dens (Stirling 1986, Kolenosky 1987). In parts of Canada pregnant females are protected from hunting by delaying opening of hunting until 1 December, after the den entrance period (Stirling and Calvert 1985). Black bear tracking studies in Maine revealed that black bears are likely to be distant from their breeding ranges during early fall seasons. The geographic distribution of kill at such times would not, as a result, accurately depict the origins of harvested bears (Hughie 1982).

A chronology of sex ratio in kill of grizzly bears harvested in a portion of southcentral Alaska was given by Miller (1990*b*). There was little change in sex ratio of kill over time during fall seasons, but there was an increase in the proportion of females killed as the spring season progressed. This suggested that the last part of spring seasons should be eliminated if hunter kills need to be reduced. However, 2-4 times as many females are killed during each of the first 2 weeks of September than during any week of the spring seasons. Also, the percentage of females in the kill is higher in the early fall than at any other time of the year (Miller 1990*b*). In this area, more females would be protected from hunters if the first 2 weeks of the fall season were closed than could be accomplished by closing the whole spring season.

Bag limits can also be adjusted to influence the number of bears taken. In most of Alaska, grizzly bear bag limits are 1 per 4 years. The multi-year bag limit serves to make hunters more selective as by taking a bear they forego the opportunity to take a better bear in subsequent years. In Alaska's Game Management Unit 13, grizzly bear bag limits were changed from 1 per 4 years to 1 per year during fall seasons in 1982-1986 and harvests averaged 81 bears per season (range 59-96). When bag limits were 1 per 4 years, fall harvests were lower averaging 60 bears (range 40-73) in the 5 years before the change and 53 (range 48-58) bears per season in the 2 years after the change (ADF&G unpublished data).

Increase in reported kill for certain areas may result from misreporting by hunters. This may occur when areas with multiple-year and annual-year bag limits are mixed. Differences in bag limits and seasons give hunters incentives to report location of their kills incorrectly. In Alaska, an investigation by Fish and Wildlife Protection Officers resulted in the prosecution of a guide who had misreported the location of at least 25 grizzly bears killed by himself, his relatives, and his clients during 1 season. Half of these were wrongly reported as having been taken in areas with a 1 per year bag limit when they had, in fact, been taken in an area with a 1 per 4 year bag. Such misreporting can result in serious management error in circumstances where managers rely on accuracy in these statistics.

Another way to affect bear harvests is to time bear seasons to occur at the same time as hunts for other species. In some areas harvests will be increased if hunters can take bears incidental to hunts designed primarily to take ungulates. This approach has been used in a number of different areas to influence taking black bears (Burk 1977) as well as grizzly bears in Alaska.

Closed Areas.—Areas closed to hunting are also a potentially useful tool for managers. Closed or lightly hunted reservoir areas can be sources of surplus animals that immigrate to open or more accessible areas where they can be hunted (Beecham 1986). To be effective such areas must be large.

Methods, Means, and Legal Bear Definitions.—Besides season adjustments and limited entry systems, managers use restrictions on methods and means of hunting to influence harvest. These include restrictions on weapon type, transportation methods, use of attractants like bait, and use of dogs. In Michigan, the age of black bears taken by hunters using dogs was older than for hunters not using dogs (Harger 1978). However, regulations that permit baiting or hunting with dogs could result in adverse population impacts if hunters select for females, which are more likely to "tree" when chased by dogs.

One of the most effective ways to maximize sustainable harvest of bears with minimal influence on the reproductive capability of the population is to direct hunter harvest away from adult females by prohibiting shooting females accompanied by offspring. Where female bears produce new litters every 2 or 3 years, adult females are vulnerable only 1/2 to 1/3 as frequently as males. Experiments are ongoing in the Yukon Territory to direct the harvests of outfitters away from female grizzly bears by giving them incentives to harvest males (Smith 1990). Clearly, it is difficult to differentiate between sexes of bears but it will be done more often if hunters have incentives to do so. In Ontario it was shown that not all females contribute equally to reproduction (Kolenosky 1990). Protection of maternal black bear females, which are producing the bulk of recruits, is especially important when this is the case (Kolenosky 1990).

Commercialization and Restrictions by Class of Hunter.—Limitations on commercial use of bear parts is a useful tool in preventing excessive harvests. Commercial exploitation of wildlife has the potential to reduce and eliminate wildlife populations and species quickly (Geist 1988). The Lacy Act of 1900 in the United States was a largely successful effort to stop the trend of commercial overexploitation of many wildlife species and populations (Trefethen 1961). In many areas the sale of black or grizzly bear hides or parts, such as gall bladders, is illegal as is the sale of polar bear hides. These restrictions reduce harvests over what would occur if commercial sales of bear parts were allowed (Geist 1988). Commercial sales of bear parts could be allowed in some states without creating local management problems, but this may exacerbate problems elsewhere by giving lawbreakers the ability to claim the parts came from somewhere sales were legal.

Regulations designed to benefit or restrict special groups of hunters such as resident, non-resident, native, sport, trophy, or subsistence hunters can be used to constrain harvests. As an alternative to using limited entry permits, such regulations allow only certain classes of hunters to participate. This is the system in effect for polar bear in both Canada and the United States where only indigenous people have hunting rights.

DISCUSSION

In most areas, bear population management has evolved from efforts to reduce bear numbers to objectives based on maintenance or augmentation of population numbers. The ability of managers to maintain or increase bear population numbers successfully is limited by 4 major constraints.

The first constraint is adequate protection of bear habitat. This topic is treated elsewhere (McLellan and Shackleton 1988, McLellan 1990, Mattson 1990, Schoen 1990).

The second constraint on bear population management is political. Bear population managers are pressured by many special interest groups, frequently with diametrically opposed objectives. In Alaska, for example, subsistence and sport hunters frequently pressure managers to reduce populations of bears and other predators. On the other extreme are groups that agitate to reduce or eliminate hunting. One such group managed to eliminate the black bear hunting season in California in 1989 (D. Koch, Calif. Dep. of Fish and Game, Sacramento, pers. commun.). Wildlife managers must spend an increasing amount of their personnel and financial resources dealing with the demands and proposals of special interest groups. These expenditures represent resources that are diverted from habitat and population management programs. Some groups, commonly those opposed to hunting, even target resource management agencies as the problem. These activities reduce public confidence and support for management efforts. An important challenge facing wildlife managers is to direct the activities of these groups into activities that increase support for soundly based management. This is easier to say than to do. Clearly, however, in the North American political system, the concerns of such groups cannot be ignored without ultimate counter-productive consequences. Although it may be frustrating at the time, it will help if these special-interest groups are involved in the development of management plans. This provides a forum where their concerns can be heard by managers and managers' concerns can be heard by them.

There appears to be more political will to protect remnant populations of bears than there is to reestablish bears in areas where they have been eliminated. Although grizzlies have been eliminated from 99% of their former range south of Canada (Servheen 1987 cited by Jonkel 1987), there is little interest in reestablishing them in places like California, Colorado, or Arizona (Brown 1985). In Texas much of the public is opposed to management actions that would result in a significant increase in black bear population numbers or distribution (C. Winkler, Tex. Parks and Wildl. Dep., Austin, pers. commun.). Reintroduction of bears into an area where they have been eliminated is a positive action that will provoke some opposition. In North American political systems, it appears to be more difficult to take action than to do nothing. Thus, it is important to assure that the *status quo* includes bears.

The third constraint is the technological tool kit available for use by managers. Many of the tools used by managers to assess the success or failure of bear management strategies lack precision, estimates of variability, or produce potentially biased results. It is easier to detect potential sources of bias and imprecision in analyses of bear populations than it is to develop approaches without these flaws. Unfortunately, the decisions managers have to make do not disappear just because the information available has uncertain accuracy or precision. In making these decisions, however, managers should incorporate the limitations of the data into their management strategies. Usually this will mean setting management objectives and guidelines on the conservative side of what might be estimated to be optimal. The costs associated with unintended population declines and the difficulties of detecting such declines until they are far advanced mandate a conservative approach to bear population management.

The fourth major constraint to population management is financial. Some of the technological constraints of existing bear population management techniques can be overcome if adequate funds were available. Where the commitment to spend the necessary money is lacking, bear population managers have little choice but to implement conservative management strategies.

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