

COMPARING EXTINCTION RISK AND ECONOMIC COST IN WILDLIFE CONSERVATION PLANNING¹

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Abstract. Planning regulations pursuant to the National Forest Management Act of 1976 require the USDA Forest Service to produce cost-effective, multiple-use forest plans that ensure the viability of native wildlife populations within the planning area. In accordance with these regulations, this paper presents a method for determining cost-effective conservation plans for sensitive wildlife species. The method is a decision framework for determining what forest areas should be managed as habitat to meet a population viability constraint and what areas should be used for timber production to maximize the present value of revenue from timber yields. The viability constraint is a minimum probability of meeting a standard for the risk of population extinction. This viability constraint focuses regulatory decisions on two key parameters: the standard for extinction risk and the probability of attaining the standard. The decision model is used to estimate the economic costs of these parameters. Examples for single- and multi-patch conservation problems show that the cost of habitat preservation increases as the standard for extinction risk becomes more stringent and as the required probability of attainment increases. Results from the decision model are useful for evaluating research and monitoring activities and determining the economically efficient risk standard.

Key words: cost of habitat preservation; decision model; economic efficiency; extinction risk; forest-dependent wildlife; forest management; habitat conservation planning; optimal harvesting; population models; population viability analysis; risk and uncertainty; stochastic simulation model.

INTRODUCTION

Regulations pursuant to the National Forest Management Act of 1976 (NFMA; 16 U.S.C. §§1600-1614)² provide an explicit charge to conserve the plant and animal diversity of national forests (Wilcove 1993). The regulations require the USDA (United States Department of Agriculture) Forest Service "to maintain viable populations of existing native and desired non-native vertebrate species in the planning area" (36 C.F.R. §219.19, 1992).³ The regulations define a viable population as "one which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is well distributed in the planning area" (36 C.F.R. §219.19, 1992). Although the regulations do not specify standards for species survival, the regulations clearly require the Forest Service to choose forest management activities that maintain native vertebrates within their existing ranges (Salwasser et al. 1984). This biodiversity goal can be achieved by reviewing species lists, undertaking viability analyses for sensitive species, and developing conservation plans (Salwasser et al. 1984, Wilcove 1993).

A species conservation plan may incur considerable economic costs, and, according to NFMA regulations, economic costs must be considered in the process of developing forest management plans (Teeguarden 1987). The regulations require the Forest Service to develop alternative forest plans that "represent to the extent practicable the most cost efficient combination of management practices examined that can meet the objectives established in the alternative" (36 C.F.R. §219.12(f)(8), 1992). Further, to inform the decision maker about economic impacts, "alternatives shall be formulated to facilitate evaluation of the effects on present net value, benefits, and costs of achieving various outputs and values that are not assigned monetary value but that are provided at specified levels" (36 C.F.R. §219.12(f)(3), 1992). In accordance with these regulations, a conservation plan should provide a cost-effective means of attaining species viability. Further, because the regulations do not codify viability standards, a set of plans should be developed showing the economic impacts of alternative standards. The trade-off between stricter standards and economic costs should facilitate the development and selection of a socially optimal conservation plan (Teeguarden 1987).

This paper presents a method for developing cost-effective conservation plans. The method is a decision framework for allocating land between competing uses: habitat for a forest-dependent wildlife population and timber production. The framework is based on a definition of population viability that depends on popu-

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² U.S.C. = United States Code, the official compilation of all federal law.

³ C.F.R. = Code of Federal Regulations, the official compilation of agency regulations that implement U.S. (federal) law.

lation size at the end of a predefined planning period. A wildlife population is viable if, at the end of the planning period, the risk of its extinction in the management area in future years is less than a predefined standard for extinction risk. The decision model is formulated to determine what forest areas should be preserved as habitat to meet the standard for extinction risk and what areas should be used for timber production to maximize the present value of revenue from timber yields.

A complication in the decision model is uncertainty in wildlife population dynamics. Sources of uncertainty include demographic, genetic, and environmental processes that interact to cause random variation in the basic population parameters of birth, survival, and dispersal (Shaffer 1981, Simberloff 1988). In studies of population persistence, the effects of these stochastic processes are estimated using stochastic simulation models (see Boyce [1992] for review). Here, population size is estimated using a discrete time and space model (e.g., Hastings 1992) in which the size of each patch population is predicted with a logistic growth equation. The growth rate for each patch population is a random variable representing the effects of unpredictable environmental events. The model is simple, which allows rapid determination of cost-effective land allocation, and it retains the metapopulation structure that is useful for modeling wildlife dynamics in fragmented habitat (e.g., Hanski and Gilpin 1991).

Because of uncertainty in population dynamics, population size at the end of the planning period and thus the risk of extinction in future periods are random variables. Consequently, the viability constraint is probabilistic and includes two policy parameters: the risk standard and a margin of safety. The risk standard is a predefined regulatory target representing the risk of population extinction after the end of the planning period. The margin of safety is the required probability of attaining the standard. Because both parameters are value judgements rather than scientific rules, it is important to know the economic impacts of alternative parameter values. Results from the decision model are used to estimate the relationship between economic cost, expressed here as foregone revenue from timber harvesting, and alternative risk standards and margins of safety. This relationship is the foundation for economic analysis of alternative conservation plans.

Note that the terms "risk" and "uncertainty" are used differently than in traditional economics. *Risk* is the probability of population extinction after the end of the planning period. Because population dynamics during the planning period are uncertain, the level of risk attained is a random variable with mean and variance. The magnitude of the variance is *uncertainty*.

This framework for comparing extinction risk and economic cost is one of the first attempts (see also Montgomery et al. 1994) to incorporate uncertainty in an economic analysis of wildlife conservation plan-

ning, and has several strengths. Defining population viability in terms of a risk standard and a margin of safety focuses attention on the separate economic effects of these two policy parameters. Requiring that the population meet a risk standard recognizes the relationship between population size at the end of the planning period and the long-term risk of extinction. Requiring compliance with the risk standard within a margin of safety recognizes the uncertainties in population prediction during the planning period and corresponds to a disaster-avoidance approach to protection. Similar methods have been used in other environmental regulatory settings (e.g., Lichtenberg et al. 1989). Finally, the approach meets the intent of NFMA regulations that require protection of wildlife populations at minimum cost.

The methodology is demonstrated using two hypothetical conservation problems. The first problem is to determine the cost-effective allocation of a single forest patch between wildlife and timber uses. The second problem is to determine how to manage a corridor connecting two habitat reserves.

METHODS

The methods involve determining the land-use strategy that maximizes the present value of timber harvest revenue while maintaining the viability of a forest-dependent wildlife population. The formulation assumes that the forest is divided into k habitat patches each of which may be depleted through harvesting. Each forest patch supports a subpopulation of the wildlife species. The growth of each patch population depends on patch size and dispersal depends on the location of neighboring patches. The maximization problem is solved with alternative risk standards and margins of safety providing estimates of the costs of these parameters in terms of foregone timber revenue.

Forest and wildlife dynamics

For simplicity, the formulation assumes that patch attributes (other than area) are constant over time and that the patches are non-renewable. For example, the patches could represent large areas (e.g., >400 ha) of forest in a matrix of agricultural land. Let $x_i(t)$, $i = 1, \dots, k$, be the area of forest patch i at the beginning of period t , and let $h_i(t)$, $i = 1, \dots, k$, be the percentage area harvested in patch i . The forest dynamics are

$$x_i(t+1) = x_i(t)[1 - h_i(t)] \\ i = 1, \dots, k. \quad (1)$$

Obvious extensions include age and species dynamics that model changes in patch attributes over time and activities such as planting and thinning that allow renewal of patch attributes.

Each forest patch supports a subpopulation of a wildlife species. Let $n_i(t)$, $i = 1, \dots, k$, be the size of the population in patch i at the beginning of period t . Each

patch population grows during a finite time interval and disperses at the end of the period.

The growth of each patch population is modeled with a logistic equation, which has two non-negative parameters: r_i is the finite rate of population increase at low population sizes, and c_i is carrying capacity of the patch. Letting $m_i(t)$ be the population size at the end of period t , the pre-dispersal dynamics are

$$m_i(t) = \begin{cases} n_i(t)r_i(t)^{[(c_i(t)-n_i(t))/c_i(t)]} & \text{if } r_i(t) > 1 \\ n_i(t)r_i(t) & \text{if } r_i(t) \leq 1. \end{cases} \quad (2)$$

The link between forest and wildlife dynamics is made through the carrying capacity of the patch. The carrying capacity $c_i(t)$ is the patch area after harvest divided by the average size of an individual's home range α_i :

$$c_i(t) = \frac{x_i(t)[1 - h_i(t)]}{\alpha_i} \quad (3)$$

Note that the growth rate $r_i(t)$ is time dependent and is affected by carrying capacity only when $r_i(t) > 1$. $r_i(t)$ is a lognormally distributed random variable representing random environmental effects on the finite population growth rate during period t . When $r_i(t) > 1$, the quotient $[c_i(t) - n_i(t)]/c_i(t)$ adjusts $r_i(t)$ depending on population size. When $n_i(t) \ll c_i(t)$, the population grows at rate $r_i(t)$. When $n_i(t) = c_i(t)$, the growth rate is 1 and population size does not change. When $n_i(t) > c_i(t)$, the growth rate is < 1 and the population declines. If $r_i(t) < 1$, the population declines independent of size.

Following population growth, individuals disperse between patches. The emigration rate from each patch increases linearly with population size so that the number of dispersers from patch i is proportional to the squared population size:

$$d_i(t) = \beta_i m_i^2(t). \quad (4)$$

The fraction of dispersers from patch j that are in patch i after dispersal is a constant γ_{ji} , where $\sum_i \gamma_{ji} = 1$. Between-patch dispersal rates may depend on the location and quality of patches. For example, the fraction of dispersers from a given patch that arrive at a distant patch may be less than the fraction of dispersers that arrive at a nearby patch. Equations for dispersal link subpopulations and complete the periodic growth cycle:

$$n_i(t + 1) = m_i(t) - d_i(t) + \sum_{j=1}^k \gamma_{ji} d_j(t), \quad i = 1, \dots, k. \quad (5)$$

Population viability constraint

The population viability constraint is a minimum probability that the population meet a predefined standard for extinction risk. The risk standard is the maximum acceptable risk of extinction for the population

present in the final planning period T . The risk standard represents long-term risk, recognizing that management decisions are planned over a relatively short horizon. The degree of risk attained is a random variable because of uncertainty surrounding population dynamics prior to period T . Consequently, the viability constraint is defined in probabilistic terms.

The first viability constraint is based on a minimum target N_0 for population size N in period T , where

$$N = \sum_{i=1}^k n_i(T). \quad (6)$$

The population size target N_0 is a predefined standard below which the population is subject to intolerable risks. For example, N_0 could represent the population size that has 1% chance of extinction over the next century. Because the wildlife dynamics are stochastic, N is a random variable with a distribution that can be used to estimate $\text{Pr}[N \geq N_0]$. The viability constraint is obtained by putting a lower bound P (e.g., 95%) on the probability that N meets the target N_0 :

$$\text{PR}[N \geq N_0] \geq P. \quad (7)$$

Eq. 7 (the constraint) is called a safety rule (Lichtenberg and Zilberman 1988) because it limits to some small amount $1 - P$ the probability of violating the risk standard N_0 . The lower bound P is a margin of safety and represents the decision maker's aversion to uncertainty about attainment of the risk standard. A higher margin of safety implies greater aversion to uncertainty. The margin of safety can be interpreted as a confidence level for a hypothesis test (Lichtenberg and Zilberman 1988). Similar to confidence levels used for scientific reliability, the margin of safety is either 95% or 99%.

It is useful to compare the decision-making behavior associated with Eq. 7, to that associated with a constraint on expected population size:

$$E[N] \geq N_0. \quad (8)$$

With the Eq. 8 constraint, actions are favored that increase the mean of the distribution of terminal population sizes without regard to its variance, implying a risk-neutral decision maker. With Eq. 7, increasing the margin of safety P places more emphasis on the tail of the distribution of N . Actions are favored that reduce the variance as well as increase the mean of the distribution, implying a risk-averse decision maker.

The meaning of the population size target N_0 is different from targets used in other analyses of population risk (e.g., Shaffer 1983, Montgomery et al. 1994). In those studies, targets are defined as extinction thresholds: a population that is smaller than the size threshold at the end of the planning period will assuredly go extinct. Population viability is the probability that population size exceeds the extinction threshold at the end of the planning period. The only policy parameter is

the minimum acceptable probability of exceeding the extinction threshold.

Rather than defining the population target in terms of an extinction threshold, I define the target in terms of the maximum acceptable risk of extinction after the planning period. Population viability is the probability that the population attains this risk standard. The risk standard is a policy parameter in addition to the probability of attaining it. Using two policy parameters allows the decision maker to distinguish between long-term risk of extinction (the risk standard) and the probability of attaining the risk standard over a finite planning period. Furthermore, making the risk standard a policy parameter allows the decision maker to analyze the economic impacts of alternative levels of long-term extinction risk. Using an extinction threshold as the population target as in previous definitions of population viability is equivalent to setting the risk standard $N_0 = 0$: the decision maker is unwilling to acknowledge any risk of extinction after the planning period.

The second viability constraint is based on an explicit risk standard R_0 for the maximum allowable population risk at the end of the planning period. Population risk is an explicit function of population size N in period T . Risk decreases as N gets large because threats from inbreeding depression or genetic drift, chance birth or death events, or environmental catastrophes are reduced (Boyce 1992). Further, it seems reasonable that the effect of an additional individual on extinction risk decreases as the population gets large. Therefore, extinction risk is modeled with a negative exponential function:

$$R = \exp(-\theta N), \quad (9)$$

where θ governs the shape of the function. Because N is a random variable, R is random and a constraint is placed on the probability that R is less than the maximum allowable risk R_0 ,

$$\Pr[R \leq R_0] \geq P. \quad (10)$$

Constraint 10 says that the risk of extinction must be less than the risk target at least P percent of the time.

Note the distinction between the Eq. 7 and Eq. 10 constraints. Eq. 10 is based on an explicit relationship between risk of extinction and population size. Although Eq. 7 carries an implicit assumption that the risk of extinction decreases linearly with increasing population size, the actual shape of the relationship is unknown. In this case, decision makers may be more comfortable with examining the economic impacts of alternative population size standards because they are more concrete than risk standards.

Objective function

The objective is to maximize the present value of timber harvest revenue over the planning period while meeting a wildlife viability constraint. Timber harvesting may take place in any patch over time. Letting

ρ_i be the revenue per unit area for timber in patch i and δ be the discount factor, the maximization problem is

$$\max_{\{h_i(t), t=0, \dots, T-1\}} \sum_{t=0}^{T-1} \delta^t \sum_{i=1}^k \rho_i x_i(t) h_i(t), \quad (11)$$

subject to any one of the viability constraints.

Cost function

A cost function $C(R_0, P)$ is estimated by repeatedly solving the optimization problem for alternative levels of R_0 and P . Each solution is the cost-efficient conservation plan for a particular risk standard and margin of safety. The economic cost is the timber revenue foregone to protect the population. For simplicity, this definition of economic cost is narrow. Economic cost could be broadly defined as the net change in social welfare including economic benefits of joint products associated with habitat protection (e.g., Montgomery et al. 1994).

The cost function is the foundation for economic evaluation of conservation plans. A curve showing cost vs. allowable risk for a given margin of safety can be used to identify inefficient plans (i.e., plans that attain a given risk standard with higher cost). Further, in a multi-species setting in which the objective is to maximize some aggregate measure of viability across species populations, cost functions can be used to determine an efficient set of risk standards subject to an overall cost constraint.

The marginal costs of reducing risk and uncertainty are estimated using the slopes of the cost function. If one imputes a benefit-cost rationale to the choice of a risk standard, the marginal cost of risk reduction implicitly gives an estimate of social willingness to pay for risk reduction with a given margin of safety (Lichtenberg et al. 1989). The marginal cost of uncertainty reduction (i.e., increasing the margin of safety) can be used to evaluate the economic efficiency of research and monitoring activities that reduce uncertainty in population size predictions (Lichtenberg and Zilberman 1988).

Solution method

The maximization problem (Eq. 11) is stochastic because of the viability constraints. The aim of the solution method is to transform the stochastic problem into a deterministic approximation using scenarios (e.g., Ruppert et al. 1984, Ermoliev and Wets 1988, Valsta 1992). A scenario is one realization over the planning period of the stochastic processes. In this case, growth rates for patch populations are independent, lognormal random variables. A scenario contains sequences of growth rates for the patch populations. The growth rates in each sequence are drawn independently from the appropriate lognormal distribution. One thousand scenarios are constructed, and each has equal probability of occurrence. Because the number of sce-

narios is finite, the problem is a deterministic approximation to the stochastic problem and can be solved using nonlinear programming techniques.

Each scenario is used to compute population size, which in turn is used to test the population viability constraint. Let $S = \{1, 2, \dots, S\}$ be a set of scenarios where each scenario $s \in S$ is a realization of the stochastic processes. Associated with each scenario is a probability p_s where

$$\sum_{s=1}^S p_s = 1.$$

Under scenario s , the population dynamics (Eqs. 2–5) are used to compute the patch population sizes $n_{st}(T)$, $i = 1, \dots, k$ at the end of the planning period and the size of the whole population:

$$N_s = \sum_{i=1}^k n_{st}(T). \quad (12)$$

An approximation of the Eq. 7 constraint is obtained by defining an indicator variable v_s for the violation of the target:

$$v_s = \begin{cases} 1 & \text{if } N_s \geq N_0 \\ 0 & \text{if } N_s < N_0. \end{cases} \quad (13)$$

Then Eq. 7 is approximated by

$$\sum_{s=1}^S p_s v_s \geq P. \quad (14)$$

The Eq. 10 constraint is approximated in the same way. The probability-weighted sum of population size is an approximation of $E[N]$ in Eq. 8, producing

$$\sum_{s=1}^S p_s N_s \geq N_0. \quad (15)$$

The constrained optimization problem is solved using a coordinate-search method coupled with a penalty-function method for handling the constraint. These methods are described in most operations research texts (e.g., Bazaraa and Shetty 1979) and have been used to solve other forest management problems (Roise 1986, Haight et al. 1992, Valsta 1992).

The principal drawback of the optimization method is its convergence to locally optimal solutions because of nonconvexities in the response surface defined by the wildlife model. As the number of decision variables increases, the variability of local solutions increases. Evaluation of several local optima found with different starting points for the harvest controls is required before estimating the global optimum.

A SINGLE-PATCH PROBLEM

Parameters

The first hypothetical conservation problem is to determine the cost-effective allocation of a single forest patch between wildlife and timber uses, i.e., the optimal

TABLE 1. Parameters for the stochastic simulation model of a wildlife population occupying a single forest patch.

Parameter	Description	Value
Forest		
$x_i(0)$	Initial forest patch size	500 ha
ρ_i	Economic value (revenue from timber sales)	\$4000/ha
Wildlife		
$n_i(0)$	Initial population size	1000 animals
α_i	Territory (average size of individual animal's home range)	0.5 ha
r_i	Annual population growth rate	
	Mean	1.2
	Standard deviation	0.4
β_i	Density-dependent migration	...
γ_{ji}	Dispersal matrix	...
T	Time horizon	100 yr

size of a single forest patch subject to a viability constraint for a resident wildlife population. The initial patch size is 500 ha (Table 1). The patch may be harvested in the first period for a return of \$4000/ha or preserved as wildlife habitat in perpetuity. Initially, the wildlife population fully occupies the patch. Population growth is projected in 1-yr intervals over a 100-yr horizon. The mean annual growth rate is 1.2 with a standard deviation of 0.4. The viability constraint is based on the population size in year 100.

The problem is solved using Eqs. 7 and 8 constraints. The first constraint states that the probability that the population size in year 100 exceeds a target must be greater than a margin of safety. Targets range from 20 to 100 animals in increments of 20. The margins of safety are 95% and 99%. The second constraint states that the expected population size in year 100 must be greater than the target. Without any constraints, the entire patch is harvested for a return of \$2 000 000. The cost of a viability constraint is the foregone revenue.

Results

The trade-offs between the target population size and foregone timber revenue are shown in Fig. 1. For each margin of safety, cost increases as the target increases. Further, for a given target, cost increases as the margin of safety increases. The constraint on the mean population size is equivalent to a safety rule with a 50–60% margin of safety and thus has lower cost.

The additional cost of meeting a target at a higher margin of safety is the premium paid for reducing uncertainty and is analogous to a risk premium in the economics literature. This "uncertainty premium" increases as both the target and the margin of safety increase. For example, for a target population size of 100 animals, the costs of a 1% increase in the margin of safety are \$12 000 and \$19 000 for 95% and 99% margins of safety, respectively. If there is a choice between reducing uncertainty by leaving more habitat or conducting research and monitoring activities that

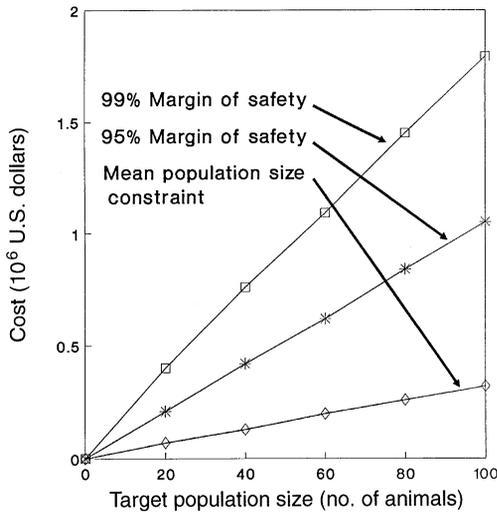


FIG. 1. Trade-offs between target population size and foregone timber revenue for the single-patch model.

reduce the variability in the estimated population growth, the uncertainty premium suggests how much can be efficiently spent on research and monitoring.

The cost of increasing the target by one individual for a given margin of safety is the marginal cost of reducing long-term risk (assuming that the long-term risk of extinction decreases as population size increases). The marginal cost of risk reduction is roughly constant for each margin of safety and increases as the margin of safety increases. For example, the marginal costs of risk reduction for 95% and 99% margins of safety are \$1200 and \$1500, respectively. These marginal costs can be compared with estimates of the marginal benefits associated with increasing population size and reduced extinction risk to determine an economically efficient population target.

Suppose the decision maker wants to attain a pop-

ulation target of 100 animals in 100 yr with 95% certainty. The patch size associated with this level of viability is 262.5 ha at a cost of \$1 050 000 (Fig. 1). The carrying capacity is 525 animals. Based on results from the stochastic simulation model, the population drops from 1000 animals to an expected population size that stabilizes at 330 animals in 100 yr.

A MULTI-PATCH PROBLEM

Parameters

The second hypothetical conservation problem seeks to determine how to manage a corridor connecting two habitat reserves. Assume there are three forest patches, each 375 ha in size (Table 2). Patches 1 and 3 are habitat reserves for a population of forest-dwelling animals; however, the animal population initially occupies patch 1 exclusively. Patch 2 is a corridor of forest connecting patches 1 and 3 and is open for timber harvesting, which creates unsuitable habitat. The goal is to establish a viable population in patch 3 using dispersers that travel from patches 1 and 2. The problem is to determine how much corridor in patch 2 can be harvested while meeting a constraint on the viability of the population that becomes established in patch 3.

Demographic parameters of each patch population are given in Table 2. The growth of each patch population is projected in 1-yr intervals over a 30-yr horizon. Mean annual growth rates of populations in patches 1 and 2 are 1.1; the growth rate for the population in patch 3 is 1.2, reflecting higher quality habitat. The standard deviation of growth in patch 1 is 0.1; the growth variability in patches 2 and 3 is higher, reflecting greater environmental uncertainty. Dispersal functions vary by patch. Relatively few animals move out of the habitat reserves (patches 1 and 3); the number of dispersers from the corridor is much higher. The dispersal matrix specifies that dispersers move in one direction: patch 1 to patch 2 and patch 2 to patch 3.

TABLE 2. Parameters for the stochastic simulation model of a wildlife population occupying three forest patches.

Parameter	Description	Numerical values		
		Patch 1	Patch 2	Patch 3
Forest				
$x_i(0)$	Initial forest patch size (ha)	375	375	375
ρ_i	Economic value (revenue from timber sales, \$/ha)	0	4000	0
Wildlife				
$n_i(0)$	Initial population size (no. of animals)	100	0	0
α_i	Territory (average size of individual animal's home range, ha)	2.5	2.5	2.5
r_i	Annual population growth rate			
	Mean	1.1	1.1	1.2
	Standard deviation	0.1	0.3	0.3
β_i	Density-dependent migration	0.0005	0.0050	0.0005
γ_{ji}	Dispersal matrix (the fraction of dispersers from forest patch j that are in patch i after dispersal)	0	0	0
		1	0	0
		0	1	0
θ	Density-dependent extinction	0.06
T	Time horizon (yr)	30	30	30

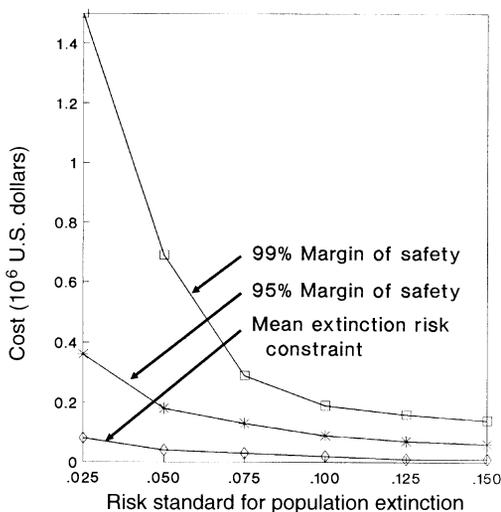


FIG. 2. Trade-offs between the risk standard and foregone timber revenue for the three-patch model.

Dispersers from patch 3 are lost. The viability constraint for the population in patch 3 is formulated in terms of the long-term risk of extinction associated with the population size in year 30. For populations with up to 40 individuals, the probability of extinction drops rapidly from 100% to 10%. For larger populations, extinction risk slowly approaches 0.

Because the size of the population in patch 3 is a random variable, extinction risk is a random variable and the viability constraint states that the probability that the long-term risk of extinction is less than a risk standard must be greater than a margin of safety (i.e., Eq. 10 constraint). Risk standards range from 0.025 to 0.150 in increments of 0.025. The margins of safety are 95% and 99%. The viability constraint is also formulated to constrain expected risk.

Harvest decisions are limited to patch 2. Harvesting may take place in 10-yr intervals spanning the 30-yr horizon. Without a viability constraint, patch 2 is harvested immediately with a return of \$1 500 000. The cost of the viability constraint is foregone harvest revenue.

Results

The trade-offs between the standard for long-term risk of population extinction in patch 3 and the cost of foregone timber revenue are shown in Fig. 2. For each margin of safety, cost increases as the risk standard becomes more stringent (smaller). More stringent risk standards are found by moving to the left on the x axis in Fig. 2. For a given risk standard, the cost increases as the margin of safety increases. The constraint on the mean risk is equivalent to a safety rule with a 65–75% margin of safety and thus has lower cost.

A cost function, $C(R_0, P)$, is estimated using the results in Fig. 2 (R_0 = maximum allowable risk, P =

lower bound on the probability). The functional form is based on the goodness of fit (adjusted $R^2 = 0.962$):

$$C(R_0, P) = \exp(-2.211 \ln P \ln R_0 - 19.673 R_0). \quad (16)$$

All coefficients are significant at the 0.05 probability level. Cost is measured in millions of U.S. dollars.

The cost function is used to estimate the uncertainty premium and the marginal cost of risk reduction. The uncertainty premium, $\partial C/\partial P$, is positive for the range of risk standards and margins of safety shown in Fig. 2. The uncertainty premium increases as the margin of safety increases and decreases as the risk standard increases. For example, for a risk standard of 0.05, the costs of adding an additional percentage point to the margin of safety are \$19 000 and \$24 000 for 95% and 99% margins of safety, respectively.

The marginal cost of risk reduction, $-\partial C/\partial R_0$, is positive and increasing with more stringent risk standards and higher margins of safety. Marginal costs of risk reduction increase with more stringent risk standards because of the shape of the relationship between population size and long-term risk of extinction: the increase in population size (and corridor habitat) required to attain a unit decrease in extinction risk increases as the extinction risk becomes small. With a 95% margin of safety, the cost of reducing risk 0.01 increases from \$20 000 for a risk standard of 0.10 to \$46 000 with a risk standard of 0.05. With a 99% margin of safety, the corresponding marginal costs of risk reduction increase from \$26 000 to \$67 000.

The effect of the margin of safety on the cost-minimizing corridor area is shown in Fig. 3. Assume that the risk standard is 0.05. Attaining the risk standard with either 95% or 99% margins of safety requires less than half of the area of patch 2 for habitat corridor. For each margin of safety, most of the timber harvesting takes place immediately because the discount rate reduces the value of future harvests. Some harvesting takes place in later years because, once a population is established in patch 3, fewer migrants and less corridor area are needed to support the population. As the margin of safety decreases, less corridor area is needed. The area of corridor required to attain a mean extinction risk of 0.05 is less than one tenth of the area required to attain a 0.05 risk standard with a 99% margin of safety.

The effect of the margin of safety on expected population size in patch 3 is shown in Fig. 4. In all cases, the population becomes established within 5 yr. Because of the larger corridor area and immigration into patch 3, the expected population size associated with a 99% margin of safety grows faster and attains higher levels than with smaller margins of safety. Although not shown on Fig. 4, the variability in expected population size decreases with higher margins of safety. For example, the standard deviation of the population size in year 30 with a 99% margin of safety is 19% of mean population size. For the constraint on mean risk

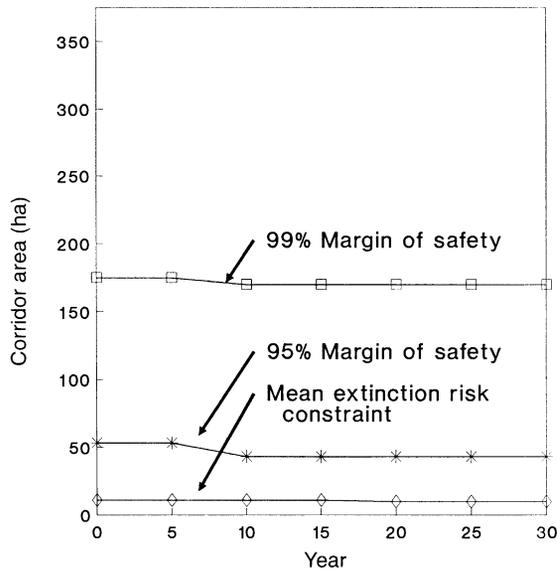


FIG. 3. Corridor area over time for cost-effective harvest policies that attain an extinction risk of 0.05 in year 30 with 95% and 99% margins of safety. The corridor area for a mean risk constraint of 0.05 is shown for comparison (lowest curve).

of extinction, the standard deviation is 38% of mean population size. Thus, although extinction risk associated with the expected population size in year 30 is <0.05 , the greater variability in population size makes the solution that attains a mean risk of extinction less attractive to risk-averse decision makers.

DISCUSSION

When timber harvesting conflicts with wildlife habitat requirements, conservation plans can be developed that attempt to maintain the viability of sensitive species populations. The safety rule for population viability presented here focuses regulatory decisions on two key parameters: the risk standard and the margin of safety. Because both parameters are value judgements, the parameter values are subject to political debate. The framework presented here is an approach to estimating economic costs of the parameters.

The economic impact of the risk standard has an important functional role in the decision-making process. The economic impact is the basis for quantifying distributional effects such as local and regional income and employment, which is required by National Forest Management Act regulations (Teeguarden 1987). Estimates of distributional effects provide a benchmark for evaluating alternative impact assessments by interest groups that have stakes in the outcome of the forest planning process. Finally, when economic benefits associated with alternative risk standards are reasonably well defined, the marginal cost of risk reduction in conjunction with estimates of marginal benefits can be used to determine an economically efficient risk standard.

As previously pointed out (Lichtenberg and Zilberman 1988), the margin of safety should be relatively easy to specify because it is analogous to a confidence level for a hypothesis test. The generally accepted minimum for scientific reliability is 95%. However, as demonstrated in the cases above, the marginal cost of uncertainty reduction can be very high. Thus, a process for considering trade-offs between the economic cost and the margin of safety needs to be established to facilitate the determination of a socially acceptable level of uncertainty.

The marginal cost of uncertainty reduction can also be used as a benchmark for the economic efficiency of research that may reduce uncertainty in population predictions. If the cost of reducing uncertainty by reserving more habitat is high, then the alternative of reducing uncertainty through research may be more efficient. Trade-offs between investments in habitat preservation and research can be formally addressed using a decision model in which the policy variables may affect both the means and variances of the stochastic parameters in the risk generation process (Lichtenberg and Zilberman 1988). For example, research and application of habitat improvement techniques may improve the mean survival rate for members of the population. Population monitoring may reduce the variances of estimated demographic parameters. The decision maker may want to find the minimum-cost set of activities that satisfies a safety rule for population viability. Alternatively, the decision maker may want to determine the research and monitoring activities that minimize the probability of attaining a predefined risk standard subject to a budget constraint. In either case, the efficient mix of investments will likely combine actions

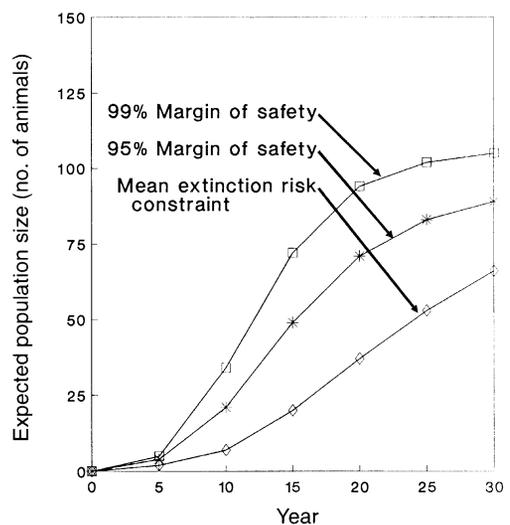


FIG. 4. Expected size trajectories for the population in patch 3 using a viability constraint with a risk standard of 0.05 and 95% and 99% margins of safety. The population trajectory for a mean risk constraint of 0.05 is shown for comparison.

aimed at mean risk reduction with actions aimed at uncertainty reduction.

Results from the decision model demonstrate the disadvantage of defining the population viability requirement with a constraint on a mean population parameter. Using population size as an example, the mean size does not measure the likelihood that the population size will dip below some threshold. Further, a mean-preserving increase in the variability of population size would increase the likelihood of extreme events without affecting the ability of conservation plans to meet the viability constraint. Accounting for extreme events is crucial when they represent probable extinction. Defining the population viability requirement as a minimum probability of meeting a predefined risk standard explicitly accounts for extreme events. A conservation plan that satisfies such a safety rule constitutes a disaster-avoidance strategy.

Regulations for land-use planning on national forests require the USDA Forest Service to protect wildlife populations and to determine the economic impacts of alternative levels of protection. The approach presented here is a first attempt at linking models for forest and wildlife dynamics in a decision model that estimates the economic cost protection. This framework is important because models for forest and wildlife dynamics are increasingly used to develop species conservation plans (e.g., McKelvey et al. 1992, Pulliam et al. 1992). In comparison to the spatially explicit, individual-based models of wildlife dynamics in McKelvey et al. (1992) and Pulliam et al. (1992), the metapopulation model that I use greatly simplifies the basic demographic processes of birth, survival, and movement. It should be possible to incorporate more complex models of population dynamics in this decision framework to better predict both the impacts of forest management on population persistence and the impacts of viability standards on economic costs.

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