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Evaluation of Two Approaches to Defining Extinction Risk Under the U.S. Endangered Species Act

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Abstract

The predominant definition of extinction risk in the conservation biology involves evaluating the cumulative distribution function (CDF) of extinction time at a particular point (the "time horizon").

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Using the principles of decision theory, this paper develops an alternative definition of extinction risk as the expected loss (EL) to society resulting from eventual extinction of a species. Distinct roles are identified for time preference and risk aversion. Ranges of tentative values for the parameters of the two approaches are proposed, and the performances of the two approaches are compared and contrasted for a small set of real-world species with published extinction time distributions and a large set of hypothetical extinction time distributions. Potential issues with each approach are evaluated, and the EL approach is recommended as the better of the two. The CDF approach suffers from the fact that extinctions that occur at any time before the specified time horizon are weighted equally, while extinctions that occur beyond the specified time horizon receive no weight at all. It also suffers from the fact that the time horizon does not correspond to any natural phenomenon, and so is impossible to specify non-arbitrarily; yet the results can depend critically on the specified value. In contrast, the EL approach has the advantage of weighting extinction time continuously, with no artificial time horizon, and the parameters of the approach (the rates of time preference and risk aversion) do correspond to natural phenomena, and so can be specified non-arbitrarily.

Keywords: Extinction risk, Endangered Species Act, decision theory, time preference, risk aversion

1. INTRODUCTION

The U.S. Endangered Species Act (ESA) requires the Secretary (of Commerce or the Interior, as appropriate) to "determine whether any species is an endangered species or a threatened species" (§4(a)(1)), but its definitions of *endangered* and *threatened* are fairly ambiguous. An *endangered species* is defined as "any species which is in danger of extinction throughout all or a significant portion of its range…" (§3(6)), and a *threatened species* is defined as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range" (§3(20)). So, *threatened* is defined in terms of "endangered," and *endangered* is defined in terms of "danger of extinction," but *danger of extinction* is not defined. In the interest of simplicity, the focus of this paper will be on the "endangered" (*en*) category, but the concepts developed here could potentially be applied to the "threatened" (*th*) category also (see Section 4.4.5).

Absent a rigorous definition of the phrase "danger of extinction," it is difficult to apply the ESA consistently.[(1)] Therefore, assuming that consistency of application is a desirable goal, it is necessary to adopt a standard definition of the phrase "danger of extinction." *Danger* is a term that is not often quantified, but *risk*, one of the most common dictionary synonyms of "danger," is routinely quantified. If "danger of extinction" can be equated with *extinction risk*, the problem becomes more tractable from a quantitative perspective.

The objective of this paper is to compare two approaches to defining extinction risk and evaluate their potential use in making listing decisions under the ESA. Although there are also approaches to evaluating extinction risk that bypass a quantitative definition of the term altogether and instead

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provide only qualitative rankings (e.g., many of the criteria specified by the IUCN[(2)]), these are not considered here. The paper's focus on the ESA derives from the fact that one of the two definitions arose from a succession of committees established jointly by the U.S. National Marine Fisheries Service and the U.S. Fish and Wildlife Service for the purpose of developing quantitative criteria for making listing decisions under the ESA. Although the comparisons presented here were developed explicitly with ESA consistency in mind, both definitions are applicable in other contexts as well.

2. METHODS

2.1. Description of the approaches

Fundamental to both approaches is the existence of a probability density function (PDF) of extinction time T, which will be measured in units of years throughout this paper. In general, the PDF will be conditional on an n-dimensional vector of parameters $\boldsymbol{\theta}$, and will therefore be referred to as the *conditional* distribution of extinction time, denoted $f_T(T|\boldsymbol{\theta})$. It is important to understand that, given conventional interpretations of the ESA wherein the benefits and costs associated with speculative management actions cannot be considered in the listing process (see further discussion in Section 4.4.1), the elements of $\boldsymbol{\theta}$ *do not* include management actions that might be taken in the future to aid in the species' recovery, unless they are sufficiently certain to occur that their consideration is admissible under the ESA[1].

Two cases will be considered here:

A. The case in which only a statistical point estimate of $\boldsymbol{\theta}$ is available, which is used as though it were the true value.

B. The case in which posterior PDFs of one or more elements of $\boldsymbol{\theta}$ are available.

In the field of applied population viability analysis, as in applied ecological modeling generally, uncertainty in parameter estimates is usually treated by estimating those parameters statistically, and proceeding as described in Case A. Although Case B provides a fuller accounting of the true uncertainty regarding extinction time (or any function of extinction time) and is therefore preferable (at least from a Bayesian perspective), because Case A is more common,[(3)] it is included here in order to provide a pragmatic alternative to Case B.

For Case B, the appropriate PDF of extinction time for this case is the *marginal* distribution $g_T(T)$, defined as:

$$g_T(T)=\int_{-\infty}^{\infty}\cdots\int_{-\infty}^{\infty} f_T(T|\boldsymbol{\theta})\cdot h_{\boldsymbol{\theta}}(\boldsymbol{\theta})\; d\theta_1\ldots d\theta_n \quad ,$$

where $h_{\boldsymbol{\theta}}(\boldsymbol{\theta})$ represents the joint posterior PDF of $\boldsymbol{\theta}$.

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[1] See FWS/NOAA "Policy for Evaluation of Conservation Efforts When Making Listing Decisions," http://www.nmfs.noaa.gov/pr/PDFs/fr/fr68-15100.PDF.

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2.1.1. The cumulative distribution function approach to extinction risk

The most common way to define extinction risk is to equate it with the cumulative probability of extinction at some future point in time (i.e., the probability that extinction will occur at, or any time before, the specified point). Because it consists of evaluating the cumulative distribution function (CDF) at a single point, this approach can be referred to as the "CDF approach." Hooper appears to have been the first to advocate the CDF approach, writing, "The probability of random extinction of any species increases with time and there is no real minimum viable population size *unless* a period of time is defined and a small probability of extinction accepted" (emphasis added).[4] The CDF approach gained considerable momentum when it was expanded upon by Shaffer, who repeated Hooper's claim that choice of a fixed time frame was not merely an option but a logical necessity.[5]

To quantify extinction risk in the CDF approach, only a single parameter is needed, viz., the point in time at which the CDF is to be evaluated (the "time horizon"). In the context of the ESA, however, quantification of extinction risk is only the first step. The next step is to determine whether the amount of extinction risk is sufficient to warrant listing, for example, as *en*. In order to use the CDF approach to determine whether a species qualifies for an *en* listing, an additional parameter must be specified, viz., a cutoff level of cumulative probability. Thus, letting T_{en} represent the *en* time horizon and CDF_{en} represent the *en* cutoff probability, a species would qualify for an *en* listing if and only if the CDF evaluated at $T = T_{en}$ met or exceeded CDF_{en}. In more formal terms, the CDF approach defines extinction risk as

$$CDF\left(T_{en}\right) = \int_0^{T_{en}} f_T(T|\boldsymbol{\theta})\, dT \quad \text{(1A)} \quad \text{or}$$

$$CDF\left(T_{en}\right) = \int_0^{T_{en}} g_T(T)\, dT \quad \text{(1B)} \quad ,$$

depending on whether Case A or Case B obtains, and the species qualifies for an *en* listing if and only if $CDF(T_{en}) \geq CDF_{en}$.

For example, Fig. 1 shows a hypothetical extinction time PDF (upper panel) and corresponding CDF (lower panel) for a species that would qualify for an *en* listing under the CDF approach given T_{en} = 100 and CDF_{en} = 0.05, because the species has exactly a 5% probability of going extinct by year 100.

Because the ESA is usually interpreted as disallowing species-specific values when making a listing decision, both parameters in the CDF approach are assumed to take the same values for all species.

The online Supporting Information applies the CDF approach to two simple models of extinction dynamics (Section S2.1.2.3), considering both Case A (point estimates of model parameters treated as though they were the true values) and Case B (uncertainty in model parameters integrated out).

2.1.2. The expected loss approach to extinction risk

Despite the claims of its original proponents, adoption of the CDF approach is not a logical necessity. A clear alternative is provided by decision theory, which views risk not as probability *per se*, but as *expected loss* (*EL*), involving both the probability of potential outcomes and the relative undesirability of those outcomes.[6-8] Specifically, a decision-theoretic alternative to the CDF approach can be developed by defining extinction risk as the expected loss to society resulting from

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the eventual extinction of a species. Unlike the CDF approach, the EL approach does not require specification of a fixed time horizon.

The EL approach has been under development for several years. A few aspects of the EL approach described below have been mentioned or used previously.[3,9-10] However, those previous papers did not show how the approach could be derived from first principles, and fell short of providing either a full description or technical evaluation of the approach in comparison to the CDF approach.

In developing the EL approach, the first step is to conceptualize a measure of the nominal *wealth* (W) that the species confers to society, where W is a random variable. Note that W does not have to consist of actual cash or even physical assets, but it must be measurable, at least in principle, using some sort of objective quantity such as units of currency or non-monetary expressions of relative value.

Existence benefits[11] will be assumed to be constant over all time T, and wealth will be defined as the *present discounted value of future existence benefits*. The latter will be computed by applying a *discount rate* (α) to all future existence benefits and integrating over time from the present (T=0) until such time as the species goes extinct.

The next step is to specify a *utility* function (U) for nominal wealth. One of the most frequently used forms for utility is the "constant relative risk aversion" (CRRA) utility function, which is the form that will be assumed here. The CRRA utility function includes a single parameter (ρ) representing the level of *relative risk aversion* as defined by Pratt and Arrow.[12-14] Qualitatively, levels of relative risk aversion are typically partitioned into the categories *risk averse* ($\rho < 0$), *risk neutral* ($\rho = 0$), and *risk prone* ($\rho > 0$). Risk averse, risk neutral, and risk prone examples of the CRRA utility function are shown in Fig. 2. An intuitive interpretation of the parameter ρ is provided in the online Supporting Information (Section S2.1.2.1).

In order to complete the development of the decision-theoretic definition of extinction risk, it is necessary to switch from measuring *utility* (i.e., how society perceives the present value of future existence benefits) to measuring *loss*, which is simply the converse of utility. The upper panel of Fig. 3 shows loss as a function of wealth for the same three values of ρ used in Fig. 2. As shown in the online Supporting Information (Section S2.1.2.2), the assumptions of time-invariant existence benefits, exponential discounting with discount rate α, and CRRA utility with relative risk aversion ρ imply that loss can be written as the following function of extinction time T:[3,10]

$$L(T|\alpha,\rho)=1-\left(1-\exp(-\alpha\cdot T)\right)^{1-\rho} \quad . \quad (2)$$

The lower panel of Fig. 3 shows loss expressed as a function of extinction time for the same three values of ρ used in the upper panel, with α set at a value of 0.02 as an example.

Equation 2 exhibits an inflection point at $T = \ln(1-\rho)/\alpha$. Note that the inflection point occurs at a positive value of T if and only if $\rho < 0$. That is, the existence of an inflection point in the loss function (when expressed as a function of T) implies risk proneness, and vice versa.

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The EL approach defines extinction risk as

$$EL(\alpha,\rho) = \int_0^\infty L(T \mid \alpha,\rho) \cdot f_T(T \mid \boldsymbol{\theta})\, dT \quad (3\text{A}) \quad \text{or}$$
$$EL(\alpha,\rho) = \int_0^\infty L(T \mid \alpha,\rho) \cdot g_T(T)\, dT \quad (3\text{B}) \quad ,$$

depending on whether Case A or Case B obtains.

Thus, in the EL approach, extinction risk is defined as the expected loss to society resulting from eventual extinction of a species, which is a function of both society's time preference (α) and attitude toward risk (ρ).

As suggested by its name, decision theory involves choosing an optimal decision from among alternatives. The typical decision-theoretic solution is to choose the alternative that minimizes the risk (*EL*). However, given conventional interpretations of the ESA, wherein the benefits and costs associated with alternative management actions cannot be considered in the listing process, risk *minimization* is not an allowable approach, so another is necessary. Although falling short of a full optimization, the EL approach can proceed just as in the CDF approach by specifying a cutoff level of risk EL_{en}, so that any value of $EL \geq EL_{en}$ implies that an *en* listing is warranted. For example, Fig. 4 shows probability-weighted loss (upper panel) and its integral (lower panel) for the same extinction time PDF shown in the upper panel of Fig. 1, given α=0.02, ρ=0.5, and EL_{en}=0.03. This species would qualify for an *en* listing under the EL approach, because the integral is equal to 0.03 as *T* approaches infinity.

Application of the EL approach thus requires specifying values for three parameters: the discount rate α, the relative risk aversion ρ, and the risk cutoff EL_{en} (in comparison, note that the CDF approach requires only two parameters, T_{en} and CDF_{en}—Fig. 1).

As in the CDF approach, because the ESA is usually interpreted as disallowing species-specific values when making a listing decision, all parameters are assumed to take the same values for all species.

The online Supporting Information applies the EL approach to the same two simple models of extinction dynamics used to illustrate the CDF approach (Section S2.1.2.3), considering both Case A (point estimates of model parameters treated as though they were the true values) and Case B (uncertainty in model parameters integrated out). Comparisons of the EL approach as described here are compared with previous uses of decision theory in the context of endangered species in Section S2.1.2.4.

2.2. Candidate values for the parameters

2.2.1. Use of "consensus" species to bound the universe of acceptable parameter values

Regan et al. used the extinction time PDFs corresponding to a set of 13 "consensus" species to determine candidate values for policy parameters related to ESA listings.[3,10] The consensus species were chosen from a larger set of 20 species, all of which were the subjects of previously published population viability analyses capable of generating extinction time PDFs (all but three of these were taken from Akçakaya et al.[15]). Eight scientists ("subjects") familiar with the ESA listing process were

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provided with the extinction time PDFs for all 20 species and were asked to assign each species to one of three categories: 1) a listing of *en* is warranted, 2) a listing of *th* (but not *en*) is warranted, or 3) no listing is warranted. There were 13 species for which either all or all-but-one of the 8 subjects agreed as to whether some sort of listing (either *en* or *th*) was warranted, and these 13 were adopted as the consensus species. Extinction time PDFs for the consensus species are shown in Fig. 5.

Although there was unanimous or near-unanimous agreement as to whether *some* sort of listing was warranted for the consensus species, the level of agreement regarding the more specific question of whether a species warranted an *en* listing was often considerably lower. Table I lists the votes in favor of an *en* listing for each subject and species. The right-hand column shows the correlation between each subject's listing decisions (*en* or not *en*) and the average listing decisions across the remaining 7 subjects.

For the present study, three "candidate" values for each parameter in each of the two approaches were chosen, consisting of a "base" value and two alternative values, one above the base value and one below. The results of the consensus classifications were used to guide the selection of the alternative values, conditional on the base values described below in Sections 2.2.2 and 2.2.3. Given those base values, alternative values for the parameters associated with each approach were chosen so as to satisfy the following three criteria:

1. The alternative values for each parameter had to contain the respective base value as the midpoint.
2. All possible combinations of values for all parameters had to produce correlations (between the resulting set of listing decisions and the average listing decisions across the 8 subjects) greater than a threshold value of 0.603, representing the upper end of the lower quartile of the correlation distribution estimated by a two-stage bootstrap.
3. The alternative values for each parameter had to be as far apart as possible.

Details of the two-stage bootstrap and the method used to satisfy the above criteria are described in the online Supporting Information (Section S2.2.1).

2.2.2. Candidate parameter values for the CDF approach

For the CDF approach, two parameters need to be specified: the point in time at which the CDF is to be evaluated, T_{en}, and the *en* risk cutoff, CDF_{en}.

Shaffer originally proposed {T_{en}=1000, CDF_{en}=0.01} as "tentative" values for the parameters of the CDF approach, although he noted that reasonable values for T_{en} could range anywhere from 100-10,000 years and that reasonable values for P_{en} could be set anywhere in the 0.00-0.05 range or at "any other level."[(5)] Shortly thereafter, the parameter combination {T_{en}=100, CDF_{en}=0.05} was variously used "for purposes of illustration," labeled an "arbitrary criterion," and described as providing "high level mid-term security."[(16-18)] This parameter combination has since become something of a standard.[(19-21)] The literature contains numerous examples where this combination of parameter values has been used;[(17-18, 22-52)] although it should be noted that these examples have not

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always tied the {100,0.05} combination explicitly to the definition of *en* status under the ESA (i.e., these values have sometimes been used to describe other points of conservation concern); also, some of these examples have addressed “quasi-extinction” rather than absolute extinction.[53-55]

Following tradition, the base values for the parameters used in the CDF approach were set at {T_{en}=100, CDF_{en}=0.05}.

The alternative T_{en} values were set at the base value +/– 60 years, giving the following set of candidate values: {40, 100, 160}. The alternative CDF_{en} values were set at the base value +/– 0.03, giving the following set of candidate values: {0.02, 0.05, 0.08}.

2.2.3. Candidate parameter values for the EL approach

For the EL approach, three parameters need to be specified: the discount rate α, the relative risk aversion ρ, and the *en* risk cutoff EL_{en}. Our choices for the base values of α and ρ are 0.02 and 0.5,[3,10] and our choice for the base value of EL_{en} is 0.03. Each of these choices was motivated by a pair of considerations, described below.

The two main considerations motivating our choice for the base value of α (0.02) were:

1. Guidance from the U.S. Office of Management and Budget states that, although regulatory analysis should typically use discount rates between 3% and 7%, analysis of government policies with “important intergenerational benefits or costs” may consider a discount rate lower than 3%, with estimates of the appropriate discount rate in such situations ranging from 1% to 3%.[56] Preservation/loss of a species obviously has important intergenerational benefits/costs, so a discount rate between 1% and 3% would comply with the OMB guidance.

2. Further support for a discount rate of 2% comes from a survey of 2,160 Ph.D. economists who were asked to estimate the appropriate discount rate for use in mitigating the possible effects of global climate change.[57] Like species extinction, global climate change would be expected to have intergenerational effects, so this survey might be expected to have applicability to the extinction risk problem. The modal response (21% of all responses) was a discount rate of 2%, which is also the midpoint of OMB’s recommended range for problems of this type. We therefore recommend a discount rate of 2%.

The two main considerations motivating our choice for the base value of ρ (0.5) were:

1. U.S. government policies regarding management of fish and wildlife resources have been moving toward a precautionary approach,[58-61] which is consistent with a positive level of risk aversion.

2. A value of ρ near 0.5 has been validated as a typical value for personal risk aversion in numerous empirical studies.[62-67]

With respect to choosing a base value for EL_{en}, it is important to remember that any particular value of *EL* is interpretable only in the context of the associated value of ρ (when loss is viewed as a

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function of the present value of future existence benefits) or the associated values of both α and ρ (when loss is viewed as a function of extinction time). The two main considerations motivating our choice for the base value of EL_{en} (0.03) were:

1. The current "background" rate of extinction implies a lower limit on the admissible range of EL_{en} values.[9] The current rate of extinction for birds and mammals translates to 0.65% of all known species per century.[68] As shown in the online Supporting Information (Section S2.1.2.3.1), if the extinction time PDF is exponential and α and ρ are set at their base values, this background extinction rate corresponds to a background *EL* of 0.002. Assuming that Congress did not intend for the ESA to result in listing of species that exhibit extinction risks similar to the background level, this implies that the cutoff for listing should be substantially greater than 0.002 (i.e., finding that a randomly selected species qualifies as *en* should be the exception rather than the norm). Our suggested base value of 0.03 is approximately 15 times the background level of *EL*.

2. After experimenting with a large number of simulations and extinction time PDFs (including those of the consensus species), and assuming the base values of α and ρ justified above, we found that an EL_{en} value of 0.03 comported well with our subjective understanding as to what Congress intended when it passed the ESA. In other words, we found that this combination of values tended to be consistent with what we would characterize as "common sense" regarding species that ought, and ought not, to warrant an *en* listing.

The alternative α values were set at the base value +/– 0.006, giving the following set of candidate values: {0.014, 0.020, 0.026}. The alternative ρ values were set at the base value +/– 0.35, giving the following set of candidate values: {0.15, 0.5, 0.85}. The alternative EL_{en} values were set at the base value +/– 0.015, giving the following set of candidate values: {0.015, 0.03, 0.045}.

2.3. Experimental design

The two approaches were explored by applying them to the extinction time PDFs for the 13 consensus species and a large number of hypothetical species, under various combinations of the candidate parameter values. The purposes of this experiment were: 1) to understand how each approach behaves under various scales and shapes of the extinction time PDF, 2) to determine the extent to which the two approaches are consistent with each other, and 3) to provide a basis for users of either approach to develop their own preferred values for the parameters.

2.3.1. Comparison based on consensus species extinction time PDFs

The first part of the experiment consisted of making a determination, for each approach, as to whether each of the consensus species warranted an *en* listing under each possible combination of candidate parameter values. For each approach and each possible combination of candidate parameter values, the correlation between the resulting set of listing decisions (*en* or not *en*) and the average listing decisions across the 8 subjects in the original consensus species exercise was also calculated.

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2.3.2. Comparison based on hypothetical extinction time PDFs

A possible shortcoming of the extinction time PDFs associated with the consensus species is that, for most practitioners, the appropriate decision regarding an *en* listing is likely fairly obvious for several of those species. In order to provide more "gray areas" that might demonstrate greater contrast between the two approaches, the 13 extinction time PDFs associated with the consensus species were supplemented with a large set of hypothetical extinction time PDFs (see online Supporting Information, Section 2.3.2). Each PDF took the form of a three-parameter generalized *F* distribution characterizable in terms of an expected value (*ET*), a coefficient of variation (*CVT*), and a shape parameter (*k*). However, in the interest of brevity, the comparison based on hypothetical extinction time PDFs did not examine each possible combination of candidate parameter values. Instead, for each approach, the three candidate parameter values for each parameter were examined conditionally on the remaining parameter(s) being set at its/their base value(s).

3. RESULTS

3.1. Comparison based on consensus species extinction time PDFs

Tables II and III show how the *en* ratings associated with the candidate parameter values for the CDF and EL approaches compare to those of the subjects in the original consensus species experiment. Recall that the candidate parameter values were chosen, in part, such that the correlation between the resulting set of listing decisions (*en* or not *en*) and the average listing decisions across the 8 subjects in the original consensus species exercise was at least 0.603, representing the upper end of the lower quartile of the distribution of the consensus species correlations, as estimated by a two-stage bootstrap (see online Supporting Information, Section S2.2.1).

For the CDF approach (Table II), the correlations ranged from 0.639 to 0.830, with a mean of 0.731. The base parameter values (T_{en} = 100, CDF_{en} = 0.05) resulted in a correlation of 0.826. The *en* ratings for all 9 CDF candidate parameter combinations agreed with the majority of subjects for 6 of the 13 species (A-C and K-M). For the remaining 7 species (D-J), ratings were mixed across the CDF candidate parameter combinations.

For the EL approach (Table III), the correlations ranged from 0.639 to 0.880, with a mean of 0.750. The base parameter values (α = 0.02, ρ = 0.5, EL_{en} = 0.03) resulted in a correlation of 0.740. The *en* ratings for all 27 EL candidate parameter combinations agreed with the majority of subjects for 6 of the 13 species (A-C and K-M). For the remaining 7 species (D-J), ratings were mixed across the EL candidate parameter combinations.

Table IV shows the correlations between the two approaches for each possible pair of candidate parameter combinations. The minimum correlation was 0.3 (obtained under 14% of the 243 possible pairs of candidate parameter combinations), the maximum correlation was 1.0 (obtained under 15% of the possible pairs), 70% of the correlations were greater than 0.5, and the mean correlation was 0.632. The correlation for the case in which the parameters for each approach were set at their respective base values was 0.854.

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3.2. Comparison based on hypothetical extinction time PDFs

Results of the comparison based on hypothetical extinction time PDFs are shown in Figs. 6 and 7. The horizontal axes in all panels of both figures all span the same range (0-800 years). Three curves are shown in each panel of each figure, corresponding to three values of a particular parameter. In all cases, the solid curve corresponds to the lowest parameter value, the dashed curve corresponds to the middle parameter value, and the dotted curve corresponds to the highest parameter value.

Conditional on the base parameter values for the respective approach, Fig. 6a shows combinations of *ET* and *CVT* that satisfy the base value for the respective *en* cutoff (CDF_{en} = 0.05 for the CDF approach and EL_{en} = 0.03 for the EL approach) exactly for three values of the shape parameter *k* in the generalized *F* distribution (4, 16, and 64). In contrast, Figs. 6b and 6c consider only a single value of *k* in the interest of brevity. One of the main purposes of Fig. 6a, therefore, is to provide a caution against over-interpreting the results in Figs 6b and 6c, by showing that a substantial amount of potential variability can be masked by focusing on a single value of *k*.

For any given curve in either panel of Fig. 6a, points lying above and to the left of the curve imply that the extinction risk (as defined by the respective approach) exceeds the *en* cutoff. Note that the feasible range of *ET* values for a given approach and a given *k* value is bounded both above and below (each curve approaches a vertical asymptote). The feasible ranges of *ET* values tend to be broader for the CDF approach than for the EL approach. For both approaches, the locus of (*ET*,*CVT*) combinations tends to move upward and to the left as *k* increases.

In Figs. 6b and 6c, the parameter *k* in the generalized *F* distribution was set at a value of 16, corresponding to the dashed curves in Fig. 6a.

Conditional on the base value for *one* of the parameters of the CDF approach, each panel of Fig. 6b shows combinations of *ET* and *CVT* that match the *en* listing criterion for the CDF approach exactly for three values of the *other* parameter of the CDF approach. The upper panel shows how the combinations vary when CDF_{en} is fixed at the base value (0.05) while T_{en} takes on each of its three candidate values (40, 100, 160), and the lower panel shows how the combinations vary when T_{en} is fixed at the base value (100) while CDF_{en} takes on each of its three candidate values (0.02, 0.05, 0.08). With CDF_{en} held constant at a value of 0.05, the locus of (*ET*,*CVT*) values tends to move downward and to the right as T_{en} increases (upper panel), and with T_{en} held constant at a value of 100, the locus of (*ET*,*CVT*) values tends to move upward and to the left as CDF_{en} increases (lower panel).

Conditional on the base values for *two* of the parameters of the EL approach, each panel of Fig. 6c shows combinations of *ET* and *CVT* that match the *en* listing criterion for the EL approach exactly for three values of the *other* parameter of the EL approach. The upper panel shows how the combinations vary when ρ and EL_{en} are fixed at their base values (0.5 and 0.03, respectively) while α takes on each of its three candidate values (0.014, 0.02, 0.026), the middle panel shows how the combinations vary when α and EL_{en} are fixed at their base values (0.02 and 0.03, respectively) while ρ takes on each of its three candidate values (0.15, 0.5, 0.85), and the lower panel shows how the combinations vary when α and ρ are fixed at their base values (0.02 and 0.5, respectively) while EL_{en} takes on each of its three candidate values (0.015, 0.03, 0.045). In all three panels, the locus of

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(*ET*,*CVT*) values tends to move upward and to the left as the value of the respective parameter is increased.

Fig. 7 is constructed similarly to Fig. 6, except that the vertical axes no longer represent *CVT*.

In Fig. 7a, all parameters for both approaches are set at their respective base values, with the exception that the *en* cutoff for one approach is ignored in each panel (specifically, EL_{en} is ignored in the upper panel, and CDF_{en} is ignored in the lower panel). Each panel contains three curves corresponding to the same three values of *k* (4, 16, and 64) used to generate Fig. 6a. As in Figs. 6b and 6c, Figs. 7b and 7c consider only a single value of *k* in the interest of brevity. As with Fig. 6a, one of the main purposes of Fig. 7a, therefore, is to provide a caution against over-interpreting the results in Figs. 7b and 7c, by showing that a substantial amount of potential variability can be masked by focusing on a single value of *k*.

Any given point on any curve in Fig. 7a corresponds to a (*k*,*ET*,*CVT*) combination defining an extinction time PDF that satisfies the *en* cutoff for *one* of the two approaches exactly, where all parameters for that approach are set at their base values (*k* is identified in the embedded legend, *ET* is shown on the horizontal axis, and *CVT* is implicit at the value for the corresponding point in Fig. 6a). The value on the vertical axis shows the extinction risk for the *other* approach corresponding to the same (*k*,*ET*,*CVT*) combination, with all parameters for that approach except the *en* cutoff set at their base values (i.e., α and ρ are set at their base values in the upper panel, and T_{en} is set at its base value in the lower panel).

For example, given *k*=64, the upper panel of Fig. 6a indicates that the combination (*ET*=155,*CVT*=0.261) satisfies the base value of the *en* cutoff in the CDF approach exactly (i.e., $CDF=CDF_{en}=0.05$ when $T=T_{en}=100$)—note that this is the same parameter combination used to generate Figs. 1 and 4. In Fig. 7a, the curve for *k*=64 crosses *ET*=155 at an *EL* value of 0.03 (given that α and ρ are set at their base values), which is the base value for the *en* cutoff in the EL approach, meaning that this particular (*k*,*ET*,*CVT*) combination satisfies the base value for the *en* cutoff in both the CDF and EL approaches exactly. However, for any lower value of *ET* (with *CVT* set at the corresponding value shown in Fig. 6a, with *k*=64), *EL* will be greater than the base value of 0.03, and for any higher value of *ET*, *EL* will be less than the base value of 0.03, meaning that moving leftward and upward from the point (*ET*=155,*EL*=0.03) along the curve for *k*=64 in Fig. 7a implies that the species qualifies for an *en* listing under both approaches, while moving rightward and downward along the curve implies that the species qualifies for an *en* listing under the CDF approach but not under the EL approach.

Figs. 7b and 7c are similar to Fig. 7a, except that the shape parameter *k* is held constant at a value of 16 and one of the approach-specific parameters is varied instead (*CVT* is implicit at whatever value is necessary to solve the respective equation).

Fig. 7b shows *EL* values associated with PDFs that satisfy the *en* cutoff for the CDF approach exactly (the method used to construct the curves in Fig. 7b is described in the online Supporting Information (Section S2.3.2)). The upper panel shows how the combinations vary when CDF_{en} is fixed at the base value (0.05) while T_{en} takes on each of its three candidate values (40, 100, 160), and the lower panel

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shows how the combinations vary when T_{en} is fixed at the base value (100) while CDF_{en} takes on each of its three candidate values (0.02, 0.05, 0.08). With CDF_{en} held constant at a value of 0.05, the locus of (ET,EL) values tends to move downward and to the right as T_{en} increases (upper panel), and with T_{en} held constant at a value of 100, the locus of (ET,EL) values tends to move upward and to the left as CDF_{en} increases (lower panel).

Conversely, Fig. 7c shows *CDF* values associated with PDFs that satisfy the *en* cutoff for the EL approach exactly. The curves in Fig. 7c were constructed analogously to those Fig. 7b. The upper panel shows how the combinations vary when ρ and EL_{en} are fixed at their base values (0.5 and 0.03, respectively) while α takes on each of its three candidate values (0.014, 0.02, 0.026), the middle panel shows how the combinations vary when α and EL_{en} are fixed at their base values (0.02 and 0.03, respectively) while ρ takes on each of its three candidate values (0.15, 0.5, 0.85), and the lower panel shows how the combinations vary when α and ρ are fixed at their base values (0.02 and 0.5, respectively) while EL_{en} takes on each of its three candidate values (0.015, 0.03, 0.045). In all three panels, the locus of (ET,CDF) values tends to move upward and to the left as the value of the respective parameter is increased. The curves are all monotonic except for the curve corresponding to α=0.014 in the upper panel, which shows a peak at an *ET* value of about 133; in addition, it may be noted that the direction of the relationship between *CDF* and *ET* in the middle panel switches from positive to negative somewhere between ρ=0.5 and ρ=0.85.

4. DISCUSSION

4.1. Behavior of the two approaches with respect to each other

In the comparison based on consensus species extinction time PDFs, both approaches, using their respective base parameter values, resulted in listing decisions (*en* or not *en*) that exhibited a reasonable correlation with those of the original 8 subjects. The correlations were 0.826 and 0.740 for the CDF and EL approaches, respectively (Tables II and III; by design, the correlations for all possible combinations of the other candidate parameter values exceeded a value of 0.603, representing the upper end of the lower quartile of the correlation distribution estimated by a two-stage bootstrap (see online Supporting Information, Section S2.2.1)). The correlations between the two approaches themselves were also uniformly positive. For example, 70% of the possible pairings of candidate parameter combinations resulted in a correlation between the two approaches of at least 0.5, and no combination resulted in a correlation less than 0.3 (Table IV).

However, agreements of this magnitude for the consensus species should not be surprising, because, as noted previously, the appropriate decision regarding an *en* listing is likely fairly obvious for several of those species. The comparison based on hypothetical extinction time PDFs was more revealing, as that set of PDFs was much broader and more finely grained than the set of consensus species PDFs.

Several patterns illustrating behavioral differences between the two approaches are evident in Fig. 6a:

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- For a given value of the shape parameter k, the range of ET values spanned by the respective curve tends to be larger in the CDF approach than in the EL approach.
- For a given value of CVT, the horizontal distance between any pair of curves (i.e., the curves associated with any two values of k) tends to be larger in the CDF approach than in the EL approach.
- For a given value of ET, the vertical distance between any pair of curves tends to be larger in the EL approach than in the CDF approach.

Fig. 6b shows that a wider range of mean extinction time can result in an *en* classification under the CDF approach as T_{en} increases or CDF_{en} decreases, while Fig. 6c shows that a wider range of mean extinction time can result in an *en* classification under the EL approach as any of the parameters decreases.

As shown in Fig. 7a, for the range of extinction time PDFs considered here, the values of EL_{en} that map into the base value of CDF_{en} (upper panel) and the values of CDF_{en} that map into the base value of EL_{en} (lower panel) both vary widely; specifically, an EL_{en} range of 0.012-0.044 can map into CDF_{en} = 0.05 and a CDF_{en} range of 0.004-0.149 can map into EL_{en} = 0.03.

In Fig. 7b, the upper panel shows that, with CDF_{en} held constant, the values of EL that match the *en* listing criterion for the CDF approach exactly tend to move downward and to the right as T_{en} increases, because, in order for a higher value of T_{en} to map into the same value of CDF_{en}, the extinction time PDF would have to be concentrated more toward higher values of T, which would tend to cause EL to decrease. The lower panel shows that, with T_{en} held constant, the values of EL that match the *en* listing criterion for the CDF approach exactly tend to move upward and to the left as CDF_{en} increases, because, in order for a higher value of CDF_{en} to map into the same value of T_{en}, the extinction time PDF would have to be concentrated more toward lower values of T, which would tend to cause EL to increase.

In Fig. 7c, all three panels show that the values of CDF that match the *en* listing criterion for the EL approach exactly tend to move upward and to the left as the respective parameter of the EL approach increases. In the cases of α and ρ (upper and middle panels), this can be explained as follows: Increasing the value of either of these parameters causes the loss function to shift downward, meaning that the extinction time PDF would have to be concentrated more toward lower values of T in order for EL to match the same value of EL_{en}, which would tend to cause the CDF (evaluated at the same value of T_{en}) to increase. In the case of an increase in EL_{en} (lower panel), because the loss function is not affected, the extinction time PDF would have to be concentrated more toward lower values of T in order for EL to match the new, higher, value of EL_{en}, which would tend to cause the CDF (evaluated at the same value of T_{en}) to increase.

4.2. Irreconcilability of the two approaches

It might be hoped that the difference between the two approaches is merely semantic, and that the listing decisions resulting from adoption of either approach could be made to match those resulting from adoption of the other simply by choosing the parameter values appropriately. However, the

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results shown here demonstrate that such wishful thinking is unlikely to be correct (as confirmed, for example, by Fig. 7a—see discussion above).

There is one limiting case in which the two approaches are equivalent: If α is set at the value $\ln(1-\rho)/T_{en}$ for any negative value of ρ, then, in the limit as ρ approaches negative infinity, Equation 2 converges on a descending "stair-step" loss function, where the edge of the "stair" occurs at T=T_{en} (Fig. 8), in which case the values of extinction risk in the two approaches are equal. However, this requires both utter risk proneness and an immense discount rate, neither of which seems plausible.

Given that the two approaches cannot be reconciled (except in the pathological case described above), it is necessary to choose between them. Potential issues with each approach are discussed in the remainder of this section.

4.3. Potential issues with the CDF approach

Four issues with the CDF approach are addressed below.

4.3.1. Is the CDF approach internally consistent?

Although extinction can occur at any point in time, the CDF approach typically calls for evaluating the CDF at only *one* point in time.[4-5] It is easy to imagine cases where different listing decisions could be made for the same species simply by switching between sets of {T_{en}, CDF_{en}} values to which society (or a policy maker) is indifferent.[69-71]

For example, Fig. 9 shows hypothetical extinction time distributions for three species (solid, dashed, and dotted curves), with the PDFs plotted in the upper panel, the CDFs in the middle panel, and the log CDFs in the lower panel. Some hypothetical values of T_{en} (90, 100, 110) and the candidate values of CDF_{en} (0.02, 0.05, 0.08) are also shown in the lower panel (the latter on a log scale). Given that all possible combinations of candidate parameter values are broadly consistent with the results of the consensus species exercise (Section 2.2.1), it is not unreasonable to suppose that a policy maker might be indifferent between the combinations {T_{en}=90, CDF_{en}=0.02}, {T_{en}=100, CDF_{en}=0.05}, and {T_{en}=110, CDF_{en}=0.08}, as they are all feasible (by the consensus species criterion) and the values of T_{en} and CDF_{en} vary directly across these combinations. However, whether a species exceeds the *en* cutoff depends critically on which of those combinations is chosen. For example:

- If T_{en} = 90 and CDF_{en} = 0.02, only the *solid* curve exceeds the *en* cutoff.
- If T_{en} = 100 and CDF_{en} = 0.05, only the *dotted* curve exceeds the *en* cutoff.
- If T_{en} = 110 and CDF_{en} = 0.08, only the *dashed* and *dotted* curves exceed the *en* cutoff.

Although the above are just examples, there surely must exist an entire "indifference curve" of {T_{en}, CDF_{en}} pairs that society (or a policy maker) would find equally acceptable, and there is absolutely no reason to expect an extinction time CDF that intersects such a locus at *one* point will intersect it at *every* point, which leads inevitably to the conclusion that the CDF approach cannot be made to be internally consistent.

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4.3.2. Is the value of the parameter in the CDF definition of extinction risk inherently arbitrary?

The fact that the CDF approach cannot be made to be internally consistent might not be a significant problem if it were possible to identify a single value of T_{en} that has logical priority over all the others. Unfortunately, finding such a value appears to be an impossible task, as many authors have recognized that classifications of extinction risk based on the CDF approach are inherently arbitrary.[(72-75)]

It is important here to distinguish between "arbitrary" and "subjective" choices. A typical dictionary definition of *arbitrary* is, "existing or coming about seemingly at random or by chance or as a capricious and unreasonable act of will;" while a typical dictionary definition of *subjective* is, "peculiar to a particular individual: personal ('subjective judgments'); modified or affected by personal views, experience, or background." In short, an arbitrary choice is one that has no reasonable rationale; while a non-arbitrary choice is one that has a reasonable rationale, where such reasonable rationale can be either subjective (dependent at least in part on personal judgment) or objective (completely independent of personal judgment).

In order for the CDF approach to make sense, T_{en} must be the value of T below which all extinctions are equally undesirable and above which all extinctions are completely irrelevant.[(9)] This is a double-edged sword: as T_{en} *increases*, the claim of equal undesirability for all extinctions at any $T < T_{en}$ becomes increasingly untenable; but as T_{en} *decreases*, the claim of complete irrelevance for all extinctions at any $T > T_{en}$ becomes increasingly untenable. Asking policy makers to specify the value of T_{en} is like asking someone to specify a level of income such that: 1) all incomes below that level are equally desirable, and 2) all incomes above that level are completely irrelevant. No one would claim that such a level exists. In short, there does not seem to be any sense in which T_{en} corresponds to any real phenomenon, in which case there does not appear to be any recourse but to set the value of T_{en} arbitrarily.

In contrast, the parameters of the loss function developed here for the EL approach (Equation 2) *do* correspond to real phenomena, viz., the discount rate and the level of relative risk aversion. Each of these two quantities has been the subject of numerous studies. Their values can be estimated statistically or prescribed normatively (Section 2.2.3). People might argue about the "right" value of α or ρ, but the *existence* of time value and risk aversion/proneness are both well established.

Recognizing this inherent problem with the CDF approach, some recent listing decisions have attempted to remedy the situation by specifying multiple pairs of values for T_{en} and CDF_{en}, particularly if identified threats are perceived to have different impacts over different time periods[2]. Of course, specification of multiple pairs of values for these parameters complicates the approach considerably, not only because it multiplies the number of parameters, but it adds the requirement of creating a decision rule to determine what happens if one or more parameter pairs imply that listing *is* warranted while one or more other parameter pairs imply that listing *is not* warranted. For

[2] Angela Somma, U.S. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Endangered Species Conservation Division; personal communication received via e-mail, April 27, 2015.

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example, the listing decision might be based on the most pessimistic outcome, the most optimistic outcome, the unweighted average outcome, or some sort of weighted average outcome. More importantly, though, simply increasing the number of arbitrarily chosen parameter values obviously does not solve the problem of relying on arbitrarily chosen parameter values in the first place.

However, if one were to take the limiting case in which the listing decision were made by comparing the entire extinction time CDF to the entire $\{T_{en}, CDF_{en}\}$ indifference curve (e.g., as seems to have been suggested implicitly by D'Elia and McCarthy[71]), this would indeed solve the problem of making the listing decision dependent on a finite number of arbitrarily chosen points. However, it would also require estimating the entire indifference curve, and it would still require creation of a decision rule (perhaps involving still more parameters) specifying what to do in case the CDF were to exceed the indifference curve over one or more portions of the range but not others. An additional difficulty is that this "generalized" CDF approach is simply a mechanical operation with no apparent meaning, in contrast to the EL approach, where the listing policy may be stated simply as, "list a species as endangered if and only if the expected loss to society resulting from eventual extinction of the species exceeds the appropriate cutoff."

4.3.3. Does theoretical justification for the CDF approach exist?

Unlike the EL approach, which rests on a firm theoretical foundation, the CDF approach is entirely *ad hoc*. Although usually not stated explicitly, it might be guessed that the primary theoretical appeal of the CDF approach lies in its "sound-alike" relationship to null hypothesis testing,[9] particularly when CDF_{en} is set equal to 0.05.

Emlen et al. provided one of the few explicit attempts to equate the CDF approach with null hypothesis testing when they argued that a CDF_{en} value of 0.05 should be chosen because it is "in keeping with standard statistical procedure for distinguishing between a null hypothesis (extinction) and its alternative."[76]

However, this argument has two difficulties. First, it begs the question of how to specify the null hypothesis. More specifically, it is not clear that the null hypothesis should be equated with extinction rather than persistence, especially given that persistence is the current state, whereas extinction would constitute a change from the current state.

Second, the usefulness of null hypothesis testing in the biological sciences has been seriously questioned, with many authors noting that the traditional cutoff value of 0.05 is arbitrary.[77-85] In particular, it has been argued that null hypothesis testing is inappropriate for ecological risk assessment.[86] The only justification typically cited for the 0.05 value in null hypothesis testing is its frequency of use, but this is circular reasoning; for example, the preface to the American Statistical Association's "Statement on *p*-values" describes the rationale for use of 0.05 as follows: "We teach it because it's what we do; we do it because it's what we teach."[87]

4.3.4. Does a standardized scale of risk aversion exist in the CDF approach?

Even if it were possible to specify the parameters non-arbitrarily, the CDF approach has another shortcoming: In contrast to the EL approach, where the sign of ρ unambiguously determines whether the loss function is risk averse or risk prone, it is impossible to partition risk attitudes in the

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CDF approach into “risk averse” and “risk prone” ranges; all that can be said is that, for any given value of T_{en}, a particular value of CDF_{en} is more or less risk averse or risk prone than some other value. For example, given some value of T_{en}, CDF_{en}=0.10 is more risk averse than CDF_{en}=0.15 and more risk prone than CDF_{en}=0.05, but there is no way to judge whether CDF_{en}=0.10 is risk averse or risk prone in and of itself, even when placed in the context of the given T_{en}. As noted previously, U.S. government policies regarding management of fish and wildlife resources have been moving toward a precautionary approach,[58-61] which is often characterized in terms of being risk averse rather than risk prone. In order to use these terms meaningfully, it would be very helpful to be able to tell one from the other objectively.

4.4. Potential issues with the EL approach

Six issues with the EL approach are addressed below, and a seventh, more technical, issue is addressed in the online Supporting Information (Section S4.4.7).

4.4.1. Does the ESA allow consideration of foregone benefits in a definition of extinction risk?

One area where the EL approach may raise concerns is its reliance on societal costs associated with future extinction of a species (i.e., foregone existence benefits). The basis for such concerns will likely be a particular interpretation of the ESA wherein “costs” or “economics” cannot influence a listing determination, with the U.S. Supreme Court’s 1978 landmark decision in the case of Tennessee Valley Authority v. Hill, and also House of Representatives Report 97-567, being commonly cited in support of this interpretation.[88-90] Note, however, that the text of the ESA itself contains no such prohibition.

In Tennessee Valley Authority v. Hill, the Court’s opinion contained three relevant uses of the term “cost:”

A. “Whether a dam is 50% or 90% completed is irrelevant in calculating the social and scientific costs attributable to the disappearance of a unique form of life;”

B. “The plain intent of Congress in enacting this statute was to halt and reverse the trend toward species extinction, whatever the cost;” and

C. “The 1973 Act is substantive in effect, designed to prevent the loss of any endangered species, regardless of the cost.”

Note that use A does not actually disallow calculating the costs of extinction (in fact, it seems to presume that such calculations will be made); it simply disallows certain factors from impacting that calculation. Uses B and C, which are essentially identical, clearly refer to the cost *associated with listing* (or perhaps recovery); not the cost *of extinction itself*. If the societal costs of extinction were truly irrelevant, one must wonder why Congress would have bothered passing the ESA in the first place.

The ESA was amended in 1982 by clarifying that listing determinations are to be made “solely” on the basis of the best scientific and commercial data available. With respect to this amendment, House of Representatives Report 97-567 states, “Whether a species has declined sufficiently to

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justify listing is a biological, not an economic, question;" and, "The Committee strongly believes that economic considerations have no relevance to determinations regarding the status of species…. Applying economic criteria … to any phase of the species listing process is … specifically rejected by the inclusion of the word 'solely' in this legislation." The House report thus seems to consider "scientific and commercial data" to mean "biological data." Given this, at least two interpretations of the House language are possible: 1) the determination of whether a species has met the listing criteria (i.e., the "listing process" referenced in the Report) is to be based solely on biological data, and 2) *both* the development of the listing criteria themselves *and* the determination of whether a species has met those criteria are to be based solely on biological data (in which case the "listing process" is expanded to include development of the listing criteria themselves). Interpretation #1 is by far the more sensible of the two, because there is simply no way that the listing criteria themselves can be derived solely from biological data.

In short, the language in both the Tennessee Valley Authority v. Hill opinion and House of Representatives Report 97-567 is most sensibly interpreted as being directed against including *the costs associated with listing a species* (e.g., negative impacts on future economic development) as a factor in making a determination. Interpreting this prohibition to extend to any and all consideration of the societal costs of extinction itself is not required. Moreover, such an interpretation would make it impossible to develop listing criteria in the first place.

4.4.2. Is the order of the discount and utility operators correct?

One area where the approach developed here appears to diverge from most of the economics literature on decision-theoretic treatment of future benefits is the order of the discount and utility operators. Looking at the problem from the perspective of utility rather than loss, the approach taken here has been to focus on the *expected utility of discounted benefits* (EUDB). However, most of the literature has tended to focus on the *expected discounted utility of benefits* (EDUB).

The EDUB approach appears to have originated with Samuelson.(91) The EDUB approach was later adopted by Yaari,(92) who appears to have been the first to allow for stochasticity in the time at which future benefits cease. Like Yaari, Levhari and Mirman(93) followed the EDUB approach with allowance for stochasticity in the time at which future benefits cease and, like the present paper, they assumed a constant discount rate and a CRRA utility function.

The EUDB approach, although apparently in the minority, is also an old one, dating back at least to Markowitz.(94) The EUDB approach was adopted by Weinstein and Zeckhauser,(95) who described it as "a common and convenient practice" and also by Smith.(96) The problem addressed by Lopes and Michaelides,(97) who likewise used the EUDB approach, was more complicated than the one addressed here, but their approach nevertheless bears several similarities, such as weighting future benefits by survival probability and assuming both a constant discount rate and constant relative risk aversion.

Although both the EDUB and EUDB approaches find support in the literature, Baucells and Sarin(98) showed that the EDUB approach is, in fact, inconsistent with any positive values of risk aversion and discount rate. Here is another way to look at the issue: If the benefits of future existence are constant over time (conditional on the species being extant) and normalized at a value of unity, and

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if the utility function is of the CRRA form, then utility is independent of the level of risk aversion (i.e., $U=1^{1-\rho}=1$). Then, if risk is defined as 1 minus EDUB (rather than 1 minus EUDB), it follows that risk is also independent of risk aversion, which makes no sense.

4.4.3. Should future existence benefits be discounted?

The approach developed here rests in part on the assumption that future existence benefits should be discounted. The concept of existence benefits, a form of non-market value, was first put forth by Krutilla.(11) Existence value implies that an individual derives benefits from the mere *existence* of a natural resource, independent of any direct use of the resource by that individual. This is closely related to the concept of "option value" described by Weisbrod,(99) which is the benefit that an individual derives from retaining the *option* of future use.

Discounting of future existence values or option values derived from plant and animal species is a common practice.(100-109) However, this practice is not without its critics. Criticisms typically focus on the issue of inter-generational equity, the perception being that discounting is unfair to future generations. Summaries of the arguments, both pro and con, have been provided elsewhere,(110-111) and it is beyond the scope of this paper to evaluate all of these arguments. Instead, the following three reasons are offered in support of the decision to discount future existence benefits: 1) This practice is well established in the literature (as in the examples listed above). 2) The approach developed here is intended to be consistent with U.S. government policy, and U.S. government policy currently calls for discounting future existence benefits, albeit at a lower rate than would be the case if the welfare of future generations were not an issue.(56) 3) Although existence benefits accruing to future generations are discounted in the EL approach, they are discounted at the same rate as existence benefits accruing to the present generation; so in this sense, all generations are treated equally.

4.4.4. Should time be measured in units of species-specific generations rather than years?

It might be suggested that extinction time should be measured in units of species-specific generations rather than years.(72) Regan et al.(3,10) briefly summarized arguments for and against use of generation length as the appropriate unit of extinction time, the argument in favor being an assertion that short-lived species would be treated preferentially to long-lived species if extinction time were measured in units of years. However, in the context of the approach developed here, the opposite is true: Measuring extinction time in units of generation length would give preferential treatment to long-lived species, because this is equivalent to dividing the discount rate by the generation length, thus implying, for any future point in time, that the existence benefits to be received from a long-lived species are worth more than those to be received from a short-lived species. In conventional interpretations of the ESA, such preferential treatment is prohibited.

One possible concern with measuring extinction time in units of years rather than species-specific generations can be illustrated by means of a hypothetical example: Suppose that a population with a very long generation length has been reduced to a very small number of individuals. If the cutoff for protection is not set sufficiently low, the species will not be protected, even though there may be a high probability that it will go extinct within a few generations. However, the solution to this problem is simple: If a species like this is among those to which Congress intended to afford protection, then the cutoff should be set low enough that they are protected (this is not to suggest

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that different cutoff values should be set for different groups of species, merely that the cutoff should be set such that species meriting protection actually receive it).

4.4.5. Can the EL approach be applied to the ESA's "threatened" category?

Under the definitions prescribed by either the CDF approach or the EL approach, extinction risk is a continuous variable. For the sake of simplicity, the discussion so far has focused on a single category of extinction risk, viz., the *en* category defined by the ESA. In principle, however, any number of categories could be delineated under either approach simply by designating multiple cutoff values of extinction risk (using the definition appropriate to the respective approach).

For example, we suggest tentatively that, using the EL approach, an extinction risk cutoff of EL_{th} = 0.01 would correspond to our subjective understanding of Congress' general intent with respect to the "threatened" (*th*) category in the ESA. This cutoff value is approximately 5 times the background level of extinction risk (see Section 2.2.3). In Fig. 5, the pair of curves labeled "H" and "J" represent the only two of the 13 consensus species that would qualify as *th* under the base α and ρ values of 0.02 and 0.5 and a EL_{th} value of 0.01.

While we believe that an extinction risk cutoff of EL_{th} = 0.01 would correspond to the general intent of Congress, we note that the exact text of the ESA may not permit this as a definition of "threatened." Under the ESA, a *th* species is one that "is *likely* to become an endangered species within the *foreseeable future* throughout all or a significant portion of its range" (§3(20), emphasis added). Thus, regardless of whether extinction risk is defined under the CDF approach or the EL approach, the exact language of the ESA implies that *extinction risk* is not the operative variable for the *th* category at all, but rather the *probability* of becoming *en*.

In other words, for the *th* category, the statutory language seems to require specifying a cutoff probability CDF_{th} and comparing it against the probability of the species qualifying for *en* status within a specified time horizon T_{th}. For example, using the EL approach to define *en* status, a strict interpretation of the statutory definition of *th* status would imply that a species qualifies for *th* status if and only if both of the following two conditions are met: 1) extinction risk at T=0 is less than EL_{en} and 2) the probability that extinction risk will exceed EL_{en} at $T=T_{th}$ is greater than CDF_{th}.

If this is the case, the following questions arise: 1) What level of probability (CDF_{th}) should be associated with "likely?" and 2) what time horizon (T_{th}) should be associated with "the foreseeable future?" For the answer to the first question, we agree with DeMaster et al.,[9] who concluded that the most reasonable definition of "likely" is a cumulative probability of at least 50%. For the answer to the second question, our preference is to reverse-engineer the problem so that application of the chosen value of T_{th} (given CDF_{th} = 0.50 and the base α, ρ, and EL_{en} values of 0.02, 0.5, and 0.03, respectively) approximates our tentatively suggested criterion of a 1% extinction risk for *th* status to the extent possible. In other words, we suggest choosing the value of T_{th} that, when applied according to a strict interpretation of the statutory definition of *th*, would best approximate a decision rule in which a species is listed as *th* if and only if the following set of inequalities is satisfied:

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$$EL_{th} \leq \int_0^\infty L(T \mid \alpha, \rho) \cdot f_T(T \mid \boldsymbol{\theta})\, dT < EL_{en} \quad \text{or}$$

$$EL_{th} \leq \int_0^\infty L(T \mid \alpha, \rho) \cdot g_T(T)\, dT < EL_{en} \quad ,$$

depending on whether Case A or Case B obtains.

Based on explorations with a simple model of extinction (see online Supporting Information, Section S4.4.5), our best estimate of this value (rounded to the nearest 10 years) is T_{th} = 50 years.

4.4.6. Is the EL approach too complicated?

The EL approach is more complicated than the CDF approach. Although the mathematics are not necessarily more complicated (both approaches involve the same number of integrations), the EL approach requires three parameters, as opposed to only two for the CDF approach. Moreover, the *meanings* of the parameters used by the EL approach are more complicated than those used by the CDF approach, at least when considered at face value. The added complexity of the EL approach can be considered from at least a couple of different perspectives: First, is a simpler approach always preferable? Second, does the added complexity have implications for communicating the results of listing decisions to the public? [3,10]

With regard to the first question, it is important to keep in mind the primary objective of any ESA listing procedure, which is to list species that merit listing while not listing species that do not merit listing. As might be expected, there is a fundamental tradeoff inherent in developing an ESA listing procedure: One can develop a procedure that is extremely easy to understand but utterly fails to accomplish the objective, or one can develop a procedure that is harder to understand but more nearly accomplishes the objective. For example, one could argue that the CDF approach is needlessly complicated, because a procedure consisting of, “List a species if and only if it has a population size smaller than 1,000 individuals,” is much easier to understand, yet this procedure would easily be trumped in turn by a procedure consisting of, “List a species if and only if it is a mammal.” However, the criteria used in these latter two procedures are related only very indirectly (if at all) to a species’ extinction risk, which is why they would not be very good procedures.

Turning to the second question, it must be admitted that the average member of the public might claim to understand probability, which would seem to confer an advantage to the CDF approach. However, many studies have demonstrated that people often misinterpret statements about probability.[112-118] This leads to the question of whether it is better to adopt a technically simple approach that people *think* they understand even though they actually do not, or a technically more complicated approach involving some terms that most members of the public, were they do delve into the technical details, would likely have to admit that they do not understand.

Of course, communication is also a matter of phrasing. For example, if the EL approach were explained to the public by reciting Section 2.1.2 of this paper, it would not be surprising to find that the average member of the public was confused. However, if the EL approach were explained as, “We looked at the probability of extinction from now until far into the future and, after considering what it would mean to society were the species to go extinct soon versus much later, we determined that the extinction risk was too high,” the reaction might be much more positive. On the other

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hand, if the CDF approach were explained as, “We looked at the cumulative probability of extinction 100 years from now and determined that it was too high,” it would not be surprising to find members of the public asking, “So what? Why is 100 years so special?”

4.5. A potential issue common to both approaches

Both the CDF approach and the EL approach rely on the extinction time PDF, and are therefore usable only in cases where the responsible agency has agreed that a scientifically acceptable estimate of that PDF exists. Of course, obtaining such an estimate is no small task and, as a practical matter, a large proportion of listing decisions will inevitably be made using methods (some entirely qualitative) that do not map directly into a quantitative definition of extinction risk under either approach. For example, a listing decision may need to be based on a model describing the time to *quasi*-extinction rather than absolute extinction.[53-55] We suggest that such methods be tested and refined through simulation studies to ensure that they relate as closely as possible to whatever quantitative definition of extinction risk is adopted.[3,9,10] If estimates of extinction risk based on the extinction time PDF need to be supplemented with other information because the model from which the PDF was derived omits key features of the species’ population dynamics (i.e., a “weight of evidence” approach[119]), we likewise suggest that attention be paid to maximizing consistency with whichever overall approach (CDF or EL) is adopted.

More pessimistically, it should also be recognized that some scientists have expressed a high level of skepticism regarding the reliability of extinction time PDFs in many or most circumstances,[73,120-125] stressing that naïve acceptance of point estimates of the parameters from which the PDFs are generated is seldom justified, owing to the large amount of uncertainty that typically accompanies such point estimates. We agree that extinction time PDFs should always be used with an appropriate degree of humility. To the extent possible, uncertainty in parameter estimates should be integrated out of the PDF (i.e., in the terminology of Section 2.1., Case B should be preferred to Case A). Model averaging can also be useful when multiple estimates of the PDF exist.

5. CONCLUSIONS

This paper has compared and contrasted two approaches to defining extinction risk under the ESA: the cumulative distribution function (CDF) approach that has been the standard for several decades, and an expected loss (EL) approach that is relatively new.

We suggest that the EL approach is the better of the two. The EL approach defines extinction risk as the expected loss to society resulting from eventual extinction of a species, and is firmly grounded in decision theory. In computing extinction risk, distinct roles are identified for time preference and risk aversion. The EL approach provides consistent and coherent listing decisions, in contrast to the CDF approach, which:

- is *ad hoc*
- considers time preference on a binary basis only
- provides no objective means of distinguishing between risk proneness and risk aversion

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- can easily result in different listing decisions simply by switching between sets of parameter values to which society (or a policy maker) is indifferent.

Tentative values for the parameters of the EL approach have been identified (i.e., the base values α = 0.02, ρ = 0.5, and EL_{en} = 0.03), and we stress the word "tentative." While these values are not arbitrary, they are subjective, and we realize that it may be possible to improve upon them. In particular, EL_{en} is clearly a policy parameter, and we do not presume to usurp agency discretion in setting an appropriate value. For consistency with the ESA, any parameter values that are adopted will need to be constant across species.

Suggested directions for future research include fully vetting, and refining as appropriate, the tentative parameter values suggested here; development of effective strategies for communicating the EL approach to policy makers and the general public; and simulation testing of methods that can be used to approximate the EL approach in cases where a usable estimate of the extinction time PDF is not available.

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Note: Supporting Information may be found in the online version of this article at the publisher's website.

TABLES

Table I. Listing decisions (*en* = 1, not *en* = 0) for the 13 consensus species (columns labeled A-M) and 8 subjects (rows labeled 1-8) in the original consensus species exercise. Row labeled "*en* prop." represents the proportion of subjects that voted to list the respective species (column) as *en*. Right-hand column ("Corr.") shows the correlation between each subject's listing decisions (*en* or not *en*) and the average listing decisions across the remaining 7 subjects.

Subject	A	B	C	D	E	F	G	H	I	J	K	L	M	Corr.
1	0	1	1	0	1	0	0	0	0	1	0	0	0	0.530
2	1	1	1	1	0	0	0	0	0	0	0	0	0	0.662
3	1	1	1	1	1	0	0	0	0	0	0	0	0	0.801
4	1	1	1	1	1	1	0	0	1	0	0	0	0	0.762
5	1	1	1	1	1	1	0	0	0	1	0	0	0	0.897
6	1	1	1	1	1	1	0	0	1	1	0	0	0	0.854
7	1	1	1	1	1	1	1	0	0	1	0	0	0	0.786
8	1	1	1	1	1	1	1	1	1	1	0	0	0	0.651
en prop.:	0.88	1.00	1.00	0.88	0.88	0.63	0.25	0.13	0.38	0.63	0.00	0.00	0.00	

Table II. Listing decisions (*en* = 1, not *en* = 0) for the 13 consensus species (columns labeled A-M) and 9 candidate parameter combinations in the CDF approach. First two columns contain the candidate parameter combinations (shaded row represents the base parameter combination). Row

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labeled “*en* prop.” represents the proportion of subjects in the original consensus species exercise that voted to list the respective species (column) as *en*. Right-hand column (“Corr.”) represents the correlation between the respective candidate parameter combination and the “*en* prop.” row.

T_{en}	CDF_{en}	A	B	C	D	E	F	G	H	I	J	K	L	M	Corr.
40	0.02	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
40	0.05	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
40	0.08	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
100	0.02	1	1	1	1	1	1	1	0	1	1	0	0	0	0.830
100	0.05	1	1	1	1	1	1	0	0	1	0	0	0	0	0.826
100	0.08	1	1	1	1	1	1	0	0	1	0	0	0	0	0.826
160	0.02	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
160	0.05	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
160	0.08	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
en prop.:		0.88	1.00	1.00	0.88	0.88	0.63	0.25	0.13	0.38	0.63	0.00	0.00	0.00	

Table III. Listing decisions (*en* = 1, not *en* = 0) for the 13 consensus species (columns labeled A-M) and 27 candidate parameter combinations in the EL approach. First three columns contain the candidate parameter combinations (shaded row represents the base parameter combination). Row labeled “*en* prop.” represents the proportion of subjects in the original consensus species exercise that voted to list the respective species (column) as *en*. Right-hand column (“Corr.”) represents the correlation between the respective candidate parameter combination and the “*en* prop.” row.

α	ρ	EL_{en}	A	B	C	D	E	F	G	H	I	J	K	L	M	Corr.
0.014	0.15	0.015	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.014	0.15	0.03	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.014	0.15	0.045	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.014	0.5	0.015	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.014	0.5	0.03	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.014	0.5	0.045	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.014	0.85	0.015	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.014	0.85	0.03	1	1	1	1	1	1	0	0	0	0	0	0	0	0.880

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0.014	0.85	0.045	1	1	1	1	0	0	0	0	0	0	0	0	0	0.742
0.02	0.15	0.015	1	1	1	1	1	1	1	1	1	1	0	0	0	0.726
0.02	0.15	0.03	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.02	0.15	0.045	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.02	0.5	0.015	1	1	1	1	1	1	1	0	1	1	0	0	0	0.830
0.02	0.5	0.03	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.02	0.5	0.045	1	1	1	1	1	1	0	0	0	0	0	0	0	0.880
0.02	0.85	0.015	1	1	1	1	1	1	0	0	0	0	0	0	0	0.880
0.02	0.85	0.03	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
0.02	0.85	0.045	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
0.026	0.15	0.015	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.026	0.15	0.03	1	1	1	1	1	1	0	0	1	0	0	0	0	0.826
0.026	0.15	0.045	1	1	1	1	1	1	0	0	0	0	0	0	0	0.880
0.026	0.5	0.015	1	1	1	1	1	1	1	0	1	0	0	0	0	0.740
0.026	0.5	0.03	1	1	1	1	1	0	0	0	0	0	0	0	0	0.854
0.026	0.5	0.045	1	1	1	1	0	0	0	0	0	0	0	0	0	0.742
0.026	0.85	0.015	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
0.026	0.85	0.03	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
0.026	0.85	0.045	1	1	1	0	0	0	0	0	0	0	0	0	0	0.639
en prop.:			0.88	1.00	1.00	0.88	0.88	0.63	0.25	0.13	0.38	0.63	0.00	0.00	0.00	

Table IV. Correlations between the listing decisions (*en* = 1, not *en* = 0) for the 13 consensus species corresponding to each pairing of candidate parameter combinations under the CDF approach (columns) and EL approach (rows). The first two rows represent the T_{en} and CDF_{en} values for the candidate parameter combinations under the CDF approach. The first three columns represent the α, ρ, and EL_{en} values for the candidate parameter combinations under the EL approach. Shaded cells represent the base parameter combinations for each approach.

			40	40	40	100	100	100	160	160	160
			0.02	0.05	0.08	0.02	0.05	0.08	0.02	0.05	0.08
0.014	0.15	0.015	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000

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0.014	0.15	0.03	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000
0.014	0.15	0.045	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000
0.014	0.5	0.015	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000
0.014	0.5	0.03	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000
0.014	0.5	0.045	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.014	0.85	0.015	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.014	0.85	0.03	0.592	0.592	0.592	0.617	0.857	0.857	0.507	0.507	0.507
0.014	0.85	0.045	0.822	0.822	0.822	0.444	0.617	0.617	0.365	0.365	0.365
0.02	0.15	0.015	0.300	0.300	0.300	0.822	0.592	0.592	1.000	1.000	1.000
0.02	0.15	0.03	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.02	0.15	0.045	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.02	0.5	0.015	0.365	0.365	0.365	1.000	0.720	0.720	0.822	0.822	0.822
0.02	0.5	0.03	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.02	0.5	0.045	0.592	0.592	0.592	0.617	0.857	0.857	0.507	0.507	0.507
0.02	0.85	0.015	0.592	0.592	0.592	0.617	0.857	0.857	0.507	0.507	0.507
0.02	0.85	0.03	1.000	1.000	1.000	0.365	0.507	0.507	0.300	0.300	0.300
0.02	0.85	0.045	1.000	1.000	1.000	0.365	0.507	0.507	0.300	0.300	0.300
0.026	0.15	0.015	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.026	0.15	0.03	0.507	0.507	0.507	0.720	1.000	1.000	0.592	0.592	0.592
0.026	0.15	0.045	0.592	0.592	0.592	0.617	0.857	0.857	0.507	0.507	0.507
0.026	0.5	0.015	0.433	0.433	0.433	0.843	0.854	0.854	0.693	0.693	0.693
0.026	0.5	0.03	0.693	0.693	0.693	0.527	0.732	0.732	0.433	0.433	0.433
0.026	0.5	0.045	0.822	0.822	0.822	0.444	0.617	0.617	0.365	0.365	0.365
0.026	0.85	0.015	1.000	1.000	1.000	0.365	0.507	0.507	0.300	0.300	0.300
0.026	0.85	0.03	1.000	1.000	1.000	0.365	0.507	0.507	0.300	0.300	0.300
0.026	0.85	0.045	1.000	1.000	1.000	0.365	0.507	0.507	0.300	0.300	0.300

FIGURES

Fig. 1. Illustration of the CDF approach. The upper panel shows an example extinction time PDF and the lower panel shows the corresponding CDF. Vertical dashed lines represent T_{en} = 100. Horizontal dashed line in lower panel represents CDF_{en} = 0.05. Note that the CDF crosses the intersection of the vertical and horizontal dashed lines, indicating that this species would satisfy the criterion for an *en* listing exactly under the CDF approach with the specified values of T_{en} and CDF_{en}.

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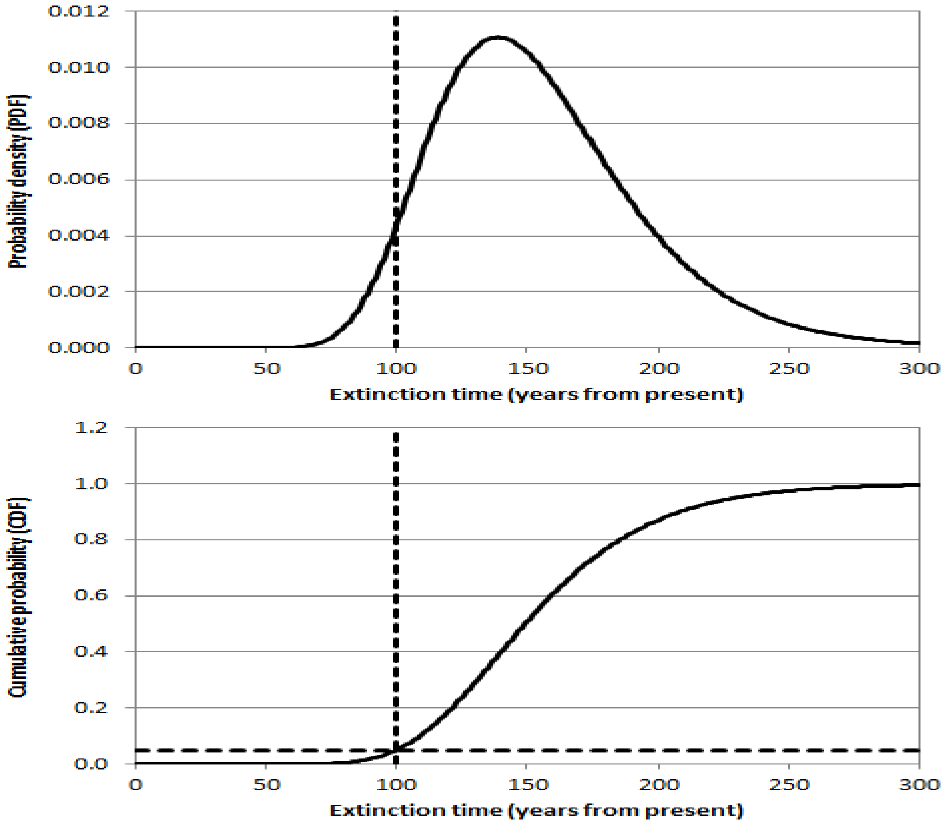

Fig. 2. Examples of utility as a function of nominal wealth for three values of relative risk aversion (ρ): a risk averse value ($\rho = -1$), the risk neutral value ($\rho = 0$), and a risk prone value ($\rho = 0.5$).

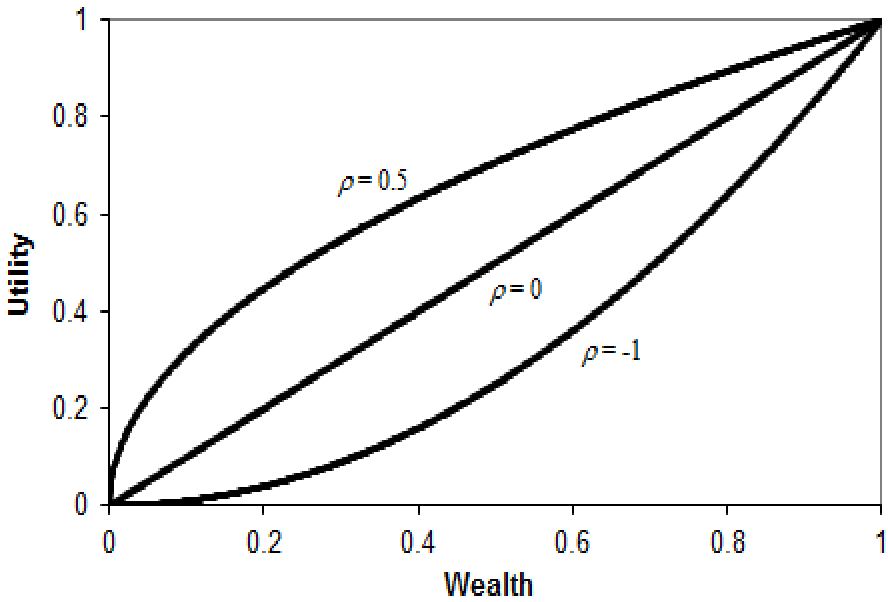

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Fig. 3. Upper panel: examples of loss as a function of nominal wealth for the same values of relative risk aversion (ρ) used in Fig. 2. Lower panel: loss functions corresponding to those shown in the upper panel when plotted as a function of extinction time.

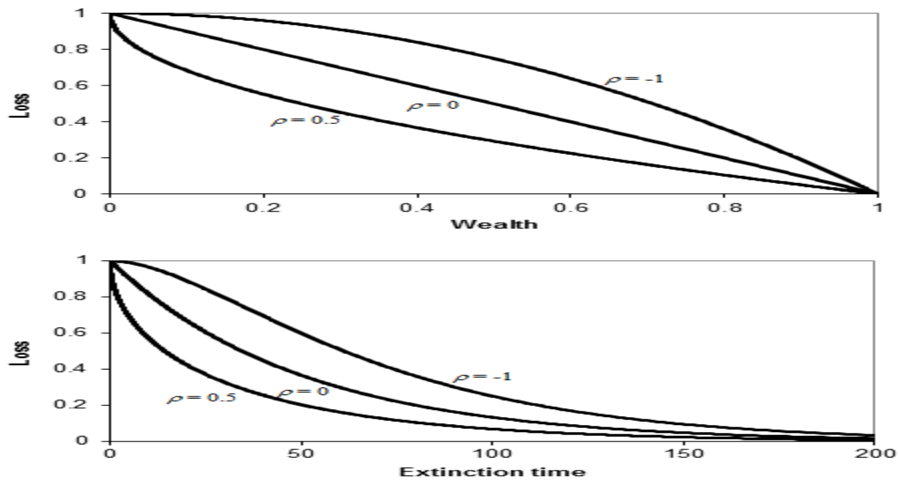

Fig. 4. Illustration of the EL approach. The upper panel shows the product of the extinction time PDF shown in the upper panel of Fig. 1 and the loss function (Equation 2) with parameters α = 0.02 and ρ = 0.5. The lower panel shows the integral of the curve in the upper panel. The horizontal dashed line represents EL_{en} = 0.03. Note that the curve in the lower panel reaches an asymptote corresponding to the horizontal dashed line, indicating that this species would satisfy the criterion for an *en* listing exactly under the EL approach and the specified values of α, ρ, and EL_{en}.

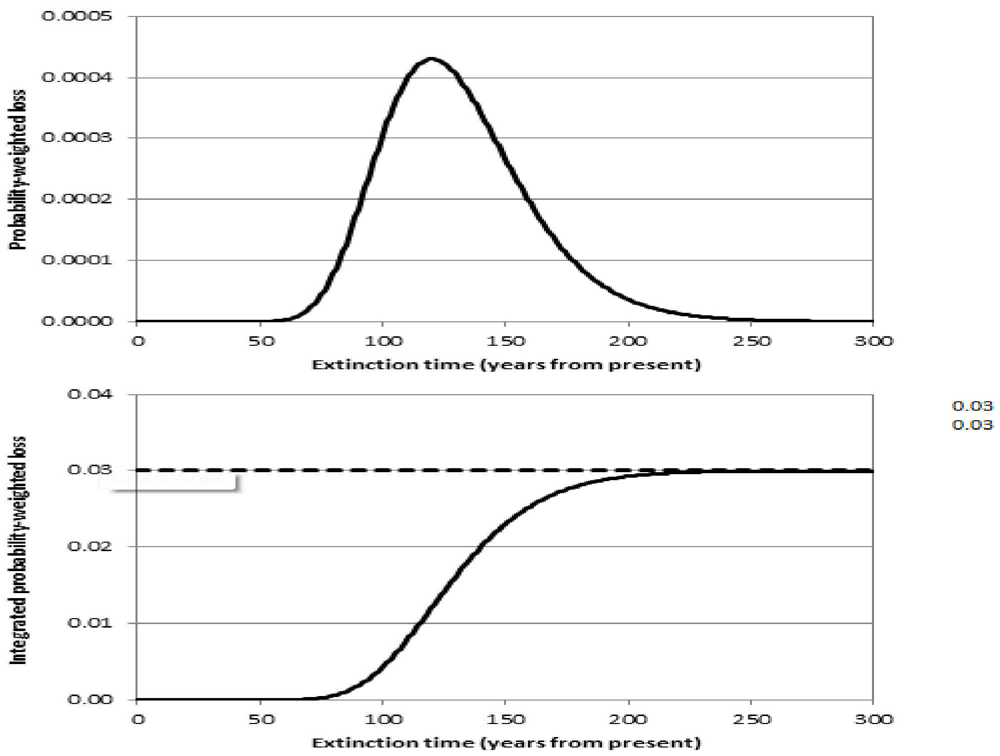

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Fig. 5. Extinction time CDFs for the 13 consensus species. Note that the scale of the horizontal axis in the upper panel is different from that of the two lower panels. All of the species in the upper panel were categorized as qualifying for an *en* listing by at least 7 of the 8 subjects in the original exercise. Species F and J were the only two species in the middle panel that were categorized as qualifying for an *en* listing by a majority of the 8 subjects. None of the species in the middle panel were categorized as qualifying for an *en* listing by any of the 8 subjects.

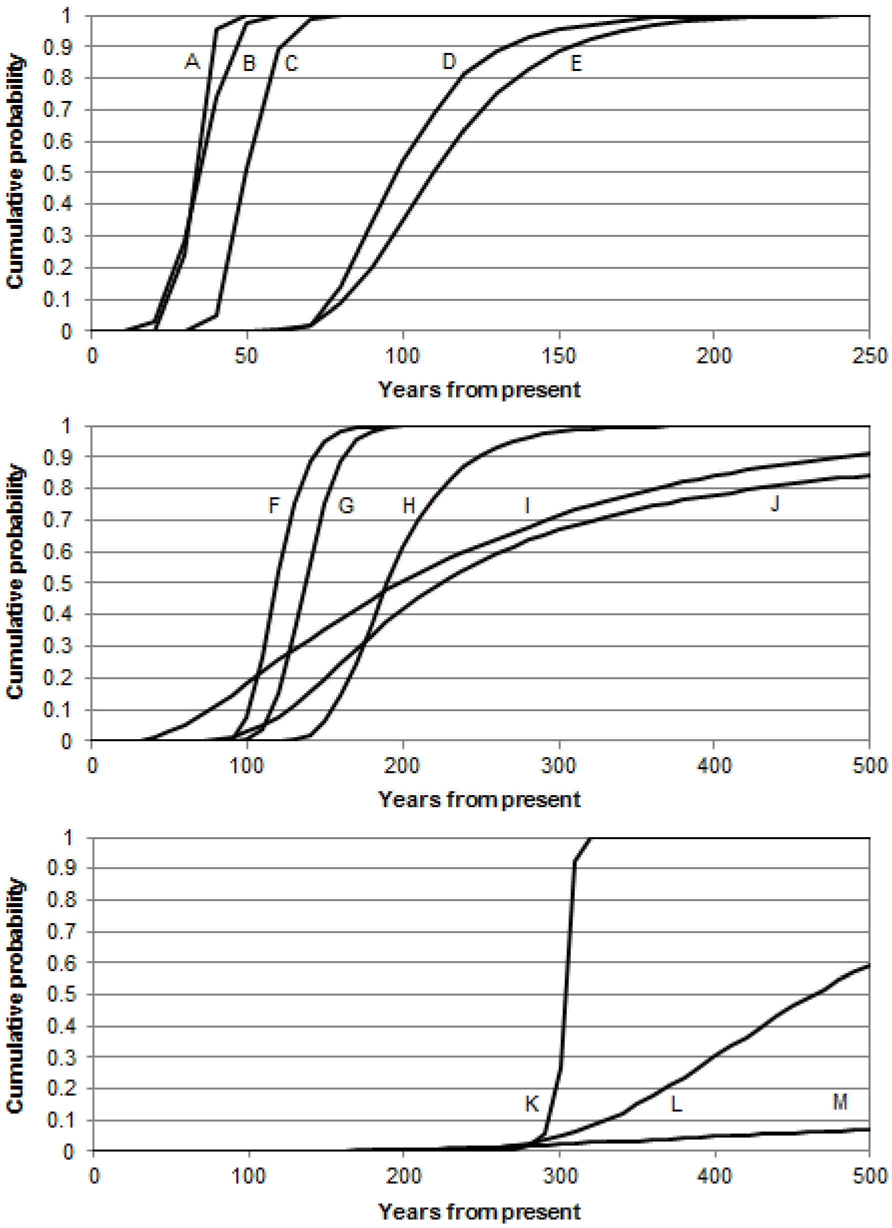

Fig. 6. A. Combinations of expected extinction time (*ET*) and extinction time coefficient of variation (*CVT*) that satisfy the base value for the respective *en* cutoff (CDF_{en} = 0.05 for the CDF approach and EL_{en} = 0.03 for the EL approach) exactly for three values of the *k* parameter in the generalized *F*

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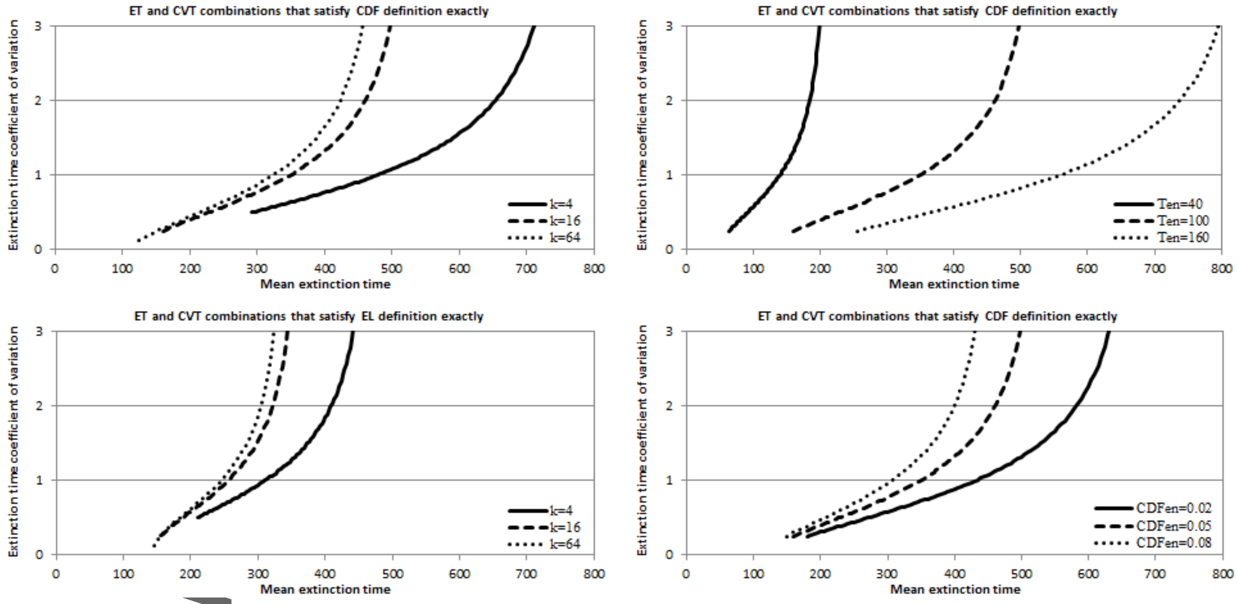
distribution (4, 16, and 64). **B.** Conditional on the base value for *one* of the parameters of the CDF approach, each panel shows combinations of *ET* and *CVT* that satisfy the respective base *en* cutoff for the CDF approach exactly for three values of the *other* parameter of the CDF approach. Dashed curves are identical to the dashed curve in the upper panel of Fig. 6a. **C.** Conditional on the base values for *two* of the parameters of the EL approach, each panel shows combinations of *ET* and *CVT* that satisfy the respective base *en* cutoff for the EL approach exactly for three values of the *other* parameter of the EL approach. Dashed curves are identical to the dashed curve in the lower panel of Fig. 6a.



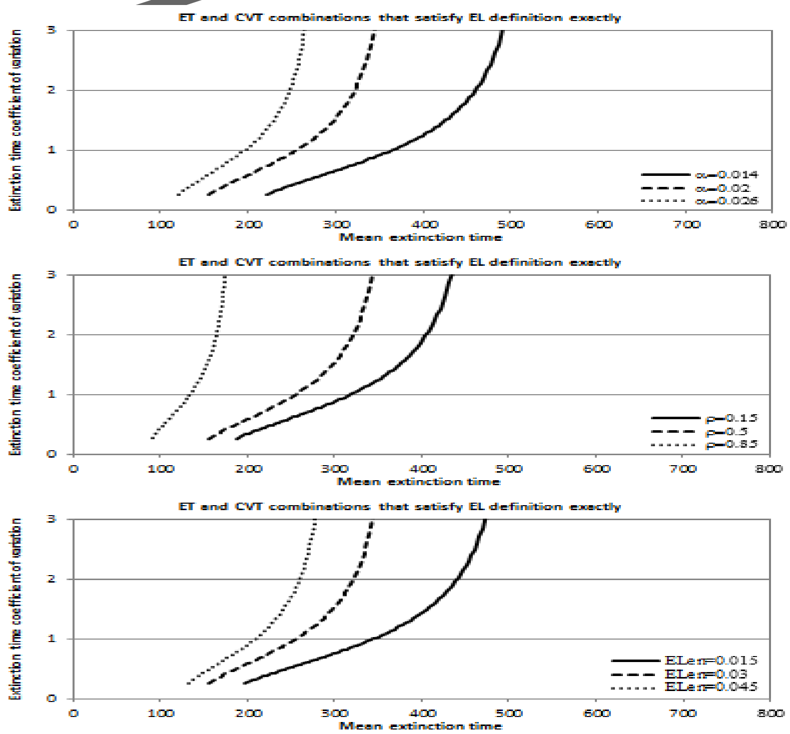


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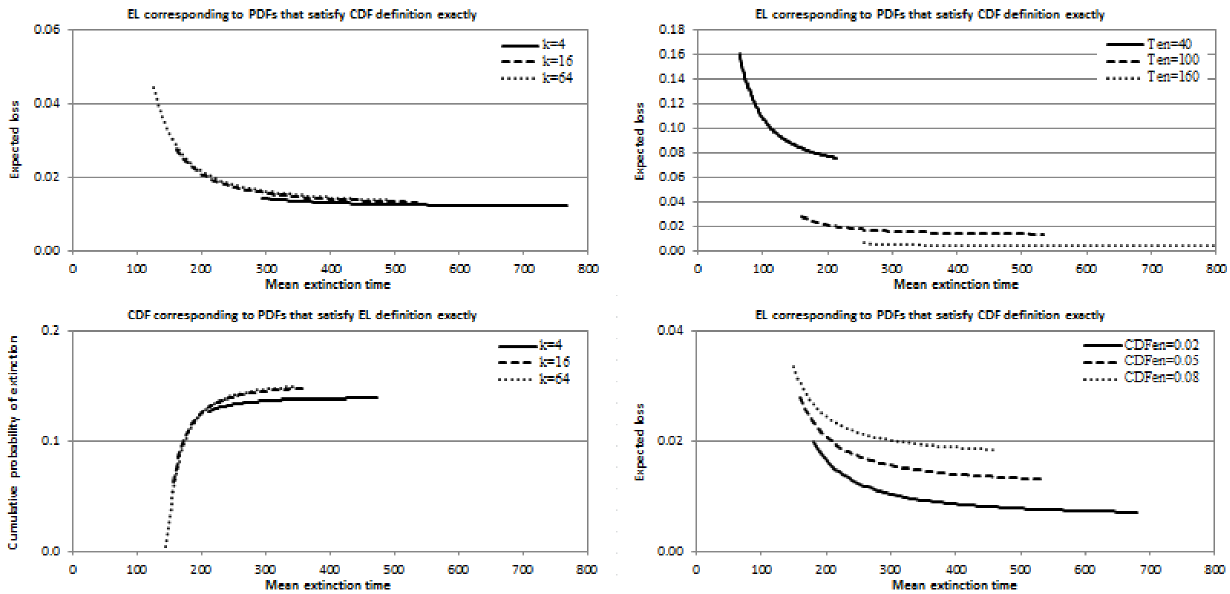
Fig. 7. A. Values of extinction risk for each of the two approaches (EL in the upper panel, CDF in the lower panel) that satisfy the base cutoff value of extinction risk for the *other* approach, under a range of parameter values for the extinction time PDF (mean extinction time is shown on the horizontal axis, shape parameter k is identified in the embedded legend, and the coefficient of variation of extinction time is implicit at the value for the corresponding point in Fig. 6a). **B.** *EL* values associated with PDFs that satisfy the *en* cutoff for the CDF approach exactly, with shape parameter k held constant at a value of 16 and *CVT* implicit at whatever value is necessary to set $CDF = CDF_{en}$ given the other parameter values. The upper panel shows how the combinations vary when CDF_{en} is fixed at the base value (0.05) while T_{en} takes on each of its three candidate values (40, 100, 160), and the lower panel shows how the combinations vary when T_{en} is fixed at the base value (100) while CDF_{en} takes on each of its three candidate values (0.02, 0.05, 0.08). Dashed curves are identical to the dashed curve in the upper panel of Fig. 7a. **C.** CDF values associated with PDFs that satisfy the *en* cutoff for the CDF approach exactly, with shape parameter k held constant at a value of 16 and *CVT* implicit at whatever value is necessary to set $EL = EL_{en}$ given the other parameter values. The upper panel shows how the combinations vary when ρ and EL_{en} are fixed at their base values (0.5 and 0.03, respectively) while α takes on each of its three candidate values (0.014, 0.020, 0.026), the middle panel shows how the combinations vary when α and EL_{en} are fixed at their base values (0.020 and 0.03, respectively) while ρ takes on each of its three candidate values (0.15, 0.5, 0.85), and the lower panel shows how the combinations vary when α and ρ are fixed at their base values (0.020 and 0.5, respectively) while EL_{en} takes on each of its three candidate values (0.015, 0.03, 0.045). Dashed curves are identical to the dashed curve in the lower panel of Fig. 7a.



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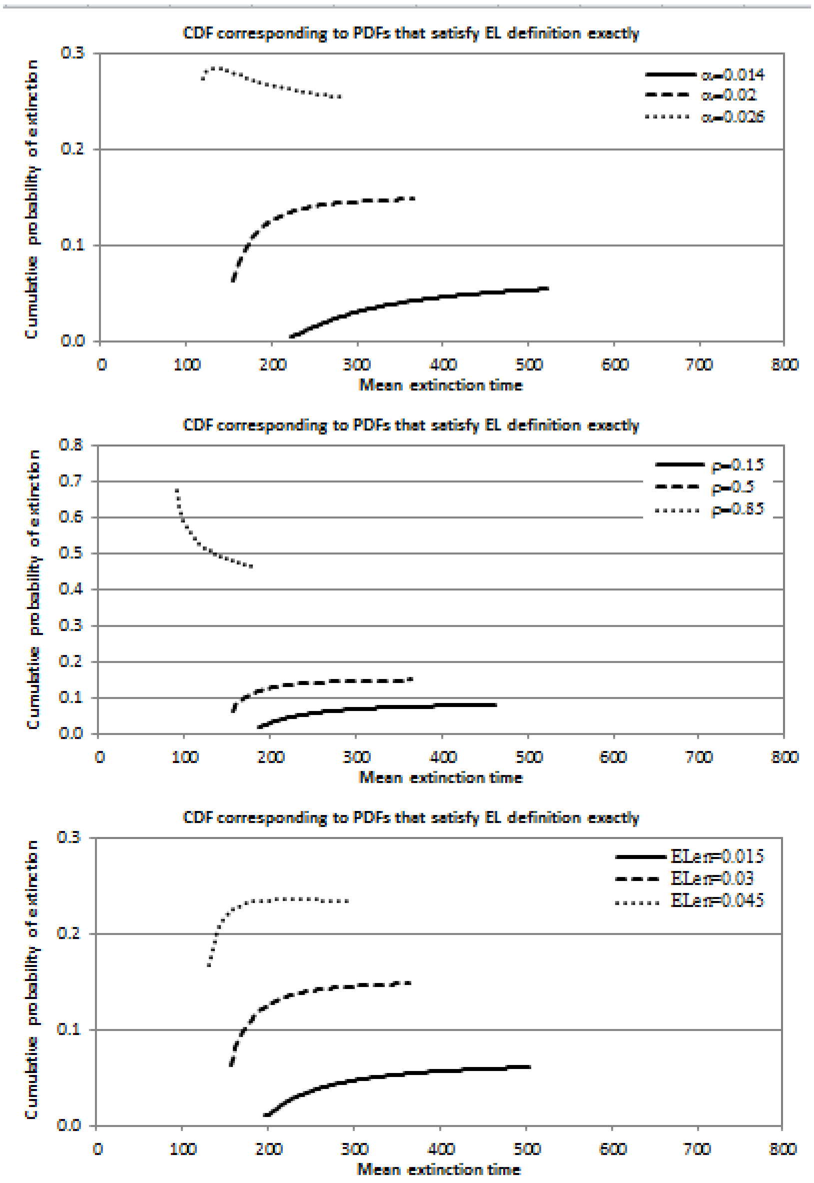

Fig. 8. Illustration of how the loss function (Equation 2) converges on a descending "stair-step" function with the edge of the "stair" occurring at $T=T_{en}$ when α is set at the value $\ln(1-\rho)/T_{en}$ and ρ approaches negative infinity. The curves marked by the circles, squares, diamonds, and triangles correspond to $\{\rho,\alpha\}$ values of {–E+01,0.024}, {–E+02,0.046}, {–E+04,0.092}, and {–E+08,0.184}, respectively; with the dashed line representing the limiting case.

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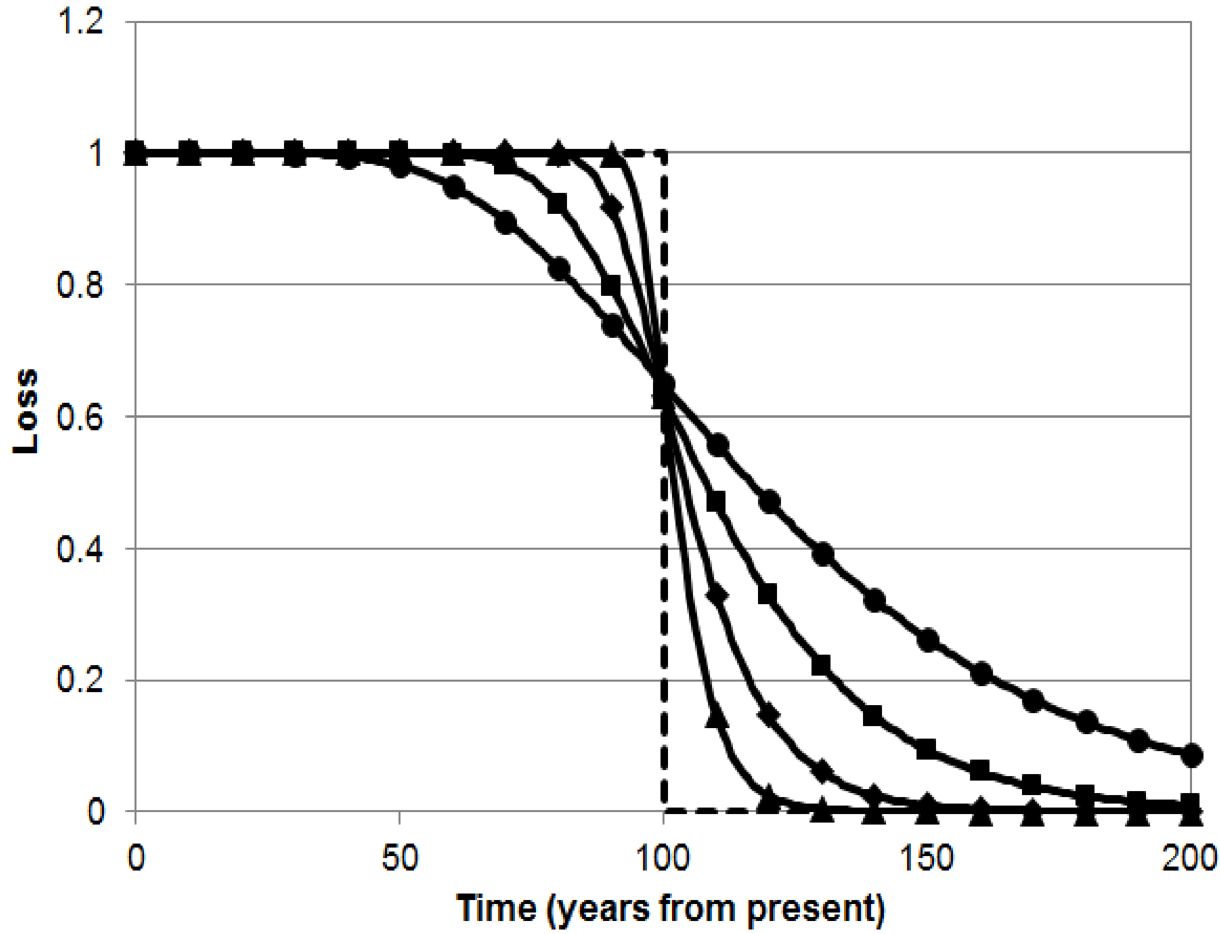

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Fig. 9. Hypothetical example of internal inconsistency in the CDF approach, where a policy-maker is indifferent between the candidate parameter pairs {T_{en} = 90, CDF_{en} = 0.02}, {T_{en} = 100, CDF_{en} = 0.05}, and {T_{en} = 110, CDF_{en} = 0.08}. Three hypthetical extinction time PDFs are shown in the upper panel, corresponding to the three CDFs in the middle panel. The area enclosed by the box in the middle panel is shown close up in the bottom panel, with the vertical axis expressed on a log scale. For {T_{en} = 90, CDF_{en} = 0.02}, only the solid curve is above the cutoff, whereas for {T_{en} = 100, CDF_{en} = 0.05}, only the dotted curve is above the cutoff; and for {T_{en} = 110, CDF_{en} = 0.08}, only the dashed and dotted curves are above the cutoff.

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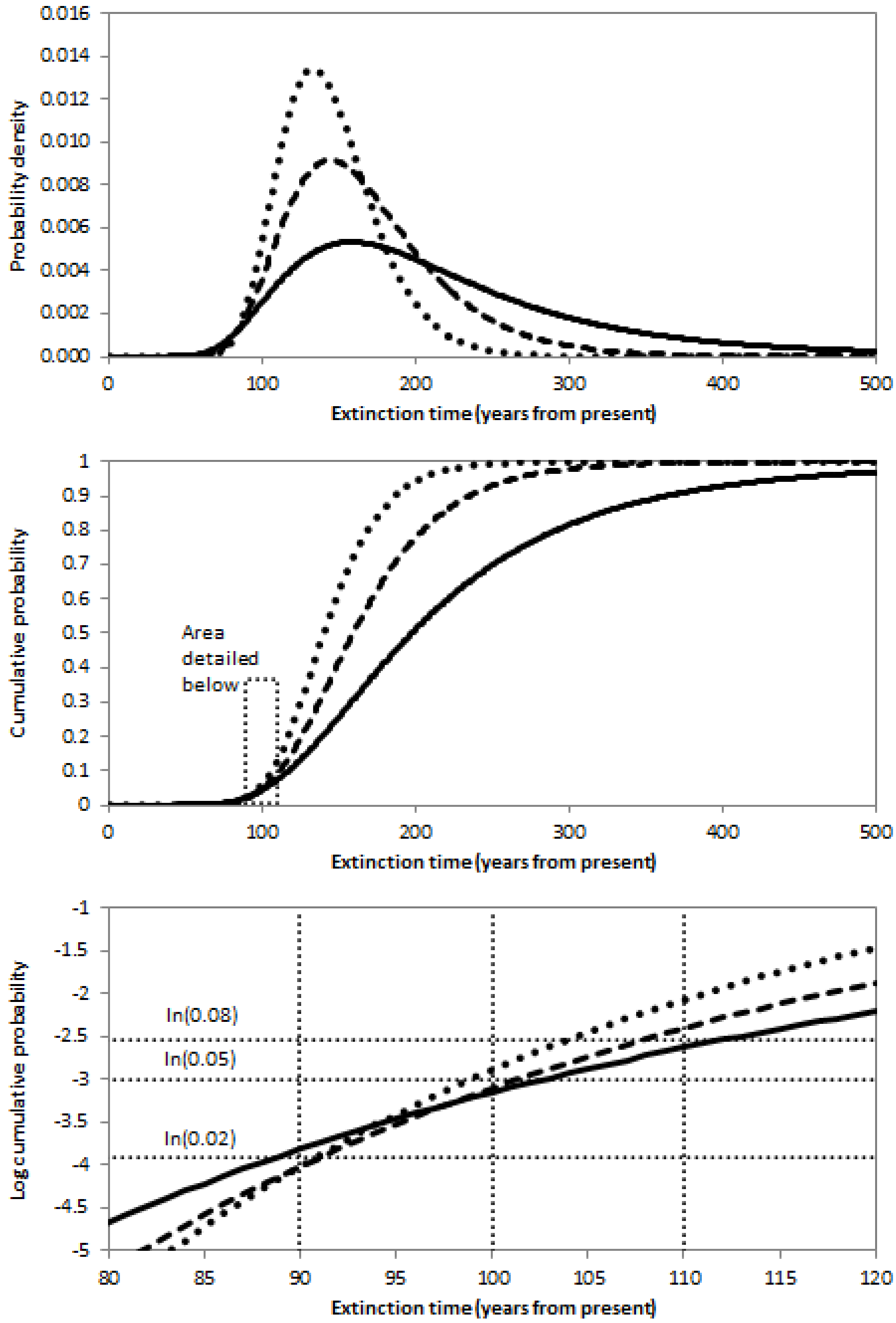

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