Risk Analyses in Ecosystem Management and Environmental Impact Assessment

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1. Abstract

Discounted mean time to extinction is a useful index of evaluating ecological impacts. In continuously declining populations, extinction risk depends on the current size and decline rate of the population. Human impact on a population reduces either the current size or decline rate. We use ecological discount rate in extinction rate, because one-day saving of threatened species this year is much more important than one-day saving 1 century after. Ecological discount rate should be in range of 0.01 to 0.03 per year, which is much smaller than economic discount rate. In spite of the fact that regulations of tributyltins in antifouling paints paid US$5.7 billion per year in the world, this is not costly in comparison with value of ecosystem services. We introduce adaptive management for deer population in Hokkaido Island, Japan, that is based on risk assessment and includes adaptability and accountability with uncertainty in fluctuating environment, life-history parameters and absolute population size.

2. Introduction

There are two major factors of species extinction in nature (Caughley & Gunn 1996), continuously population declining and stochastic processes to small sized populations. I focus on the former, rather than the latter. In a continuously declining population, its extinction risk depends on the decline rate and absolute size of the population.

I consider risk assessment of locally threatened species as a useful tool for conservation of biological diversity. Although habitat destruction may not directly lead to local extinction of some species instantly, this will increase future extinction risks of these species. Extinction is deterministic event, while extinction risk is measured by probability that a species in question will extinct within a specific period.

Extinction risk of some species depends not only on direct impact on this species, but also on impacts on other species or abiotic environmental factors. This is because population size of a particular species depends on any other species and physical factors via ecosystem process. Biological diversity, or biodiversity, is not a medley of species. Even if we prevent most of threatened species from extinction, we might lose some ecosystem service. For instance, species conservation by transplantation or off-site mitigation is often compatible with habitat destruction. Ecosystem management is a key concept for conservation of biodiversity (Christensen *et al.* 1996).

Therefore, I consider extinction risk as an indicator for integrity of the ecosystem. I do not consider conservation of threatened species as a goal of ecosystem management. There will be many loopholes to result in the species persistence and loss of ecosystem integrity. We need many rules for ecosystem management and mitigation. In this paper, (1) I propose the concept of discount mean time to extinction as a measure for extinction risk assessment. (2) I show some case studies on risk assessment using the discount mean time to extinction. (3) I discuss risk and benefit of tributyltins regulation and adaptive management of deer population in Hokkaido Island, Japan.

3. Discount mean time to extinction

Loss of mean human longevity is a useful index for human health risk (Gamo *et al.* 1995). I have no reason that saving a man is less valuable than saving a woman. I have no reason that 1-day loss of an 80 year-old person is less valuable than 1-day loss of a newly born baby. Mean longevity is not only a measure of human health risk but also a reference of the human right to life.

In contrast to loss of mean human longevity, biodiversity conservation is not a goal of the workings of human, but contributes to intergenerational sustainability on ecosystem services. I need not consider that extinction of Japanese crested ibis (*Nipponia nippon*) is equally important to extinction of Japanese wolves (*Canis lupus*). Impact of species extinction on loss of biodiversity is weighted by the phylogenic distance with the most related taxon (Weitzman 1993).

I need not consider that 1-year loss of a secure species is equally valuable to 1-year loss of a threatened species. Mean time to extinction is greatly different between species. Since an extinct species never comes back and since our descendents will be wiser to make a more effective saving plan for threatened species, then 1-year saving in the near future is more important than 1-year saving in the far future. Namely, 1-year loss of threatened species is more important than 1-year loss of secure species. In order to evaluate the near future as the more important, I introduce the discount mean time to extinction as a measure for ecological risk impact assessment.

In accordance with IUCN red list categories (IUCN 1994), if the extinction probability of a species within the next 1 century is larger than 10%, this is ranked as vulnerable (VU). However, we should not make a threshold in extinction risk. We can consider a lower risk in farther future than 1 century.

Let the probability that a particular species or population goes to extinction t years after the present be $g(t)$. Mean time to extinction of this species or population, denoted by T, is

$$T = \int_0^\infty t g(t) dt \quad (1)$$

Because of the reason mentioned above, I introduce the discount rate D for future surviving of a species. If present value of persistence in t years after the present is e^{-Dt}, the discount mean time to extinction, denoted by $T(D)$, is

$$T(D) = \int_0^\infty t e^{-Dt} g(t) dt \quad (2)$$

4. Two factors of impact that reduces mean time to extinction

Let the extinction probabilities of a particular species with and without a particular human impact be $g_1(t)$ and $g_0(t)$, respectively. Let the discount mean time to extinction when $g(t)=g_i(t)$ be $T_i(D)$, for i=0, 1. If I obtain $g_1(t)$ and $g_0(t)$, I can evaluate loss of discount mean time to extinction of this species by this human impact: $T_1(D)$- $T_1(D)$.

First, I consider a stationary process under which the extinction probabilities with a human impact and without the impact are constant, say a_1 and a_0, irrespective of t. In this case, $g_i(t) = a_i \exp(-a_i t)$. The discount mean time to extinction is $1/(D+a_i)$, whereas the mean time to extinction, $T_i(0)$, is $1/a_i$. When $D>>a_1>a_0$, the loss of discount mean time to extinction, denoted by $\Delta T(D)$, is:

$$\Delta T(D) \approx (a_1{*} - a_0{*})/D^2. \quad (3)$$

Therefore, the extinction risk of the human impact is approximately proportional to the difference in the instantaneous extinction probability, a_i.

When $D<<a_0<a_1$, the loss of discount mean time to extinction is:

$$\Delta T(D) \approx 1/a_0 - 1/a_1 = T_0(D) - T_1(D). \quad (4)$$

Therefore, the extinction risk is approximately proportional to the difference in the mean time to extinction.

The instantaneous extinction probability $g'(t)$ may not be constant. A wild species usually goes to extinction after a continuous decrease in population size. In 1996, 118 taxa of 48 families, 18 orders of marine fish species were listed in red list. Among these, 83 taxa were listed in threatened based only on the population decline rate. This is the reason why I have focused on continuously declining population as a major factor of human-induced mass extinction. In continuously declining populations, the instantaneous extinction risk will drastically increase with time. For the simplest case, if the extinction probability of a population until year t is 0 and if this population decidedly go extinct in year t, the discount mean time to extinction is $T(D)=(1-e^{-DT(0)})/D$. Loss of $T(D)$, $\Delta T(D)$, is approximately given as

$$\Delta T(D) = [\exp(-DT_1)-\exp(-DT_0)]/D. \quad (5)$$

In a continuously declining population, the mean time to extinction depends on the current size and decline rate of population. Despite of it, criterion A in the IUCN red list categories is linked only to population decline rate, which resulted in listing of apparently secure species, such as southern bluefin tuna (Matsuda *et al.* 1997). Impact of a transient event decreases the population size at the present or in the near future, but unlikely increase the population decline rate in the future. In contrast, impact of a permanent and repeatable factor increases the population decline rate in the future, as shown in Fig. 1. Some factors affect both loss of the current size and increase of future decline rate.

The simplest way to estimate the mean time to extinction, $T(0)$, is

$$T(0) = \log(N_c/N_0)/r, \quad (6)$$

where N_c is the critical population size; N_0 is the current population size; r is the population decline rate. Since demographic stochasticity will be a major role of extinction process when the population size is sufficiently low, we assume that N_c is bigger than 1. If we know the variance and autocorrelation of annual population decline rate, we estimate the mean time to extinction by some stochastic models (e.g., Lande & Orzack 1988, Lande 1996).

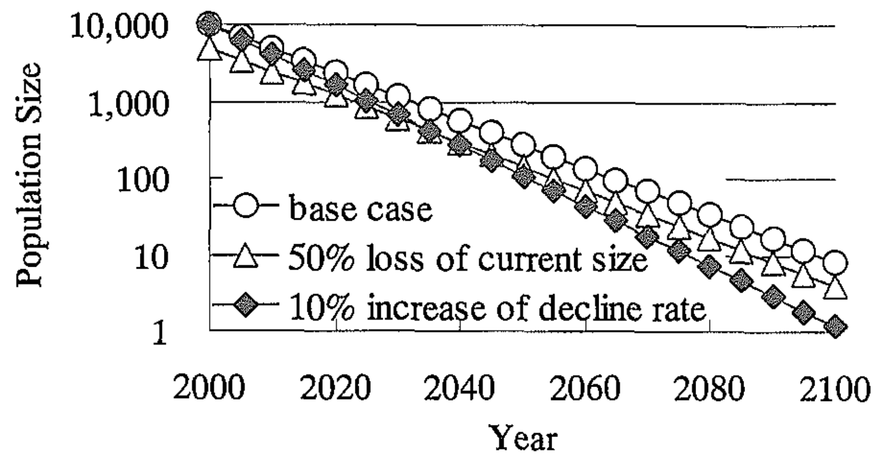

Fig. 1. Schematic impact on extinction risk. Suppose an imaginary population whose size is 10000 in the year 2000 and decreasing by 30% per decade ("base case", circles). A 10% increase of decline rate (squares) makes a larger impact on extinction risk than a 50% loss of current population size (triangles).

Table 1. Loss of the mean time to extinction of various populations and these impact assessment.

Case	$T_0(0)$	$T_1(0)$	$\Delta T(0.01)$	$\Delta T(1/30)$	$\Delta T(1)$	$\Delta(1/T)$	$\Delta\log T$
(1) bluefin tuna	10^6	90	40.66	1.494	10^{-40}	0.011	9.316
(2) star magnolia	250.6	249.9	0.057	10^{-4}	10^{-109}	0.00001	0.003
(3) bell flower	70.8	70.5	0.148	0.028	10^{-32}	0.00006	0.004
(4) an "nt" sp.	1000	500	0.669	10^{-6}	10^{-218}	0.0010	0.693
(5) an "EN" sp.	15.6	1	13.45	11.18	0.368	0.936	2.747
(6) an orchid	15.6	15.3	0.257	0.179	10^{-8}	0.0013	0.019
(7) Sea lions	2400	31	73.34	10.68	10^{-14}	0.032	4.349

(1) The southern bluefin tuna (SBT, *Thunnus maccoyii*) with and without the recent decline rate (7%/yr, Matsuda *et al.* 1997), assuming that $T_0=10^6$ years; (2) loss of 400 individuals of Star magnolia (*Magnolia tomentosa*) by the former plan of the 2005 World Exposition, Aichi, Japan; (3) An imaginary loss of the largest patch (about hundreds individuals) of a widely distributed and rapidly declining plant (Yahara *et al.* 1997), Japanese bellflower (*Platycodon grandiflorum*); (4) Imaginary impact in the case that $T(0)$ of a near threatened ("nt") species decreases from 1000 to 500 years; (5) the case of extinction of an endangered orchid (Yahara *et al.* 1997), *Cypripedium macranthum speciosum*; (6) Imaginary loss of the smallest patch of *Cypripedium macranthum*; and (7) Catch of Steller sea lions (*Eumetopias jubatus*).

The magnitude of discount mean time to extinction, $T(D)$, depends on the discount rate, D. Table 1 shows $\Delta T(D)$ of the several populations, where $T_1(0)$ and $T_0(0)$ mean the mean times to extinction with and without some human impact, respectively. I obtain $\Delta T(D)$ for various values of discount rate, D=0.01/yr., 0.033/yr. and 1/yr. I also show $\Delta(1/T)=1/T_1(0)-1/T_0(0)$ and $\Delta(\log T)=\log[T_0(0)/T_1(0)]$.

The relative magnitudes of impacts are quite different among these measures, $\Delta T(0.01)$, $\Delta T(1/30)$, $\Delta T(1)$, $\Delta(1/T)$ and $\Delta\log T$. I consider that extinction risk assessment should reflect the degree of emergency and current importance for making conservation actions. $\Delta\log T$ when $(T_0(0), T_1(0)) = (10^6, 1000)$ is very large and $\Delta\log T$ is not appropriate indicator for this purpose. If extinction of an endangered orchid in the next year is interpreted as the largest impact among the cases in Table 1, $\Delta T(0.01)$ is not appropriate either. Discount rate D in extinction risk assessment does not reflect the economic discount rate. The latter is usually much larger than the former.

5. Risk and benefit of TBTs-regulation

We have very few data of ecological impacts by endocrine disrupting chemicals (EDCs) on wild populations. It is easy to evaluate ecological impacts by commercial fishing and by habitat destruction. In contrast, it is relatively difficult to evaluate ecological impacts by pollution of environmental chemicals. However, these three factors are equally important and cooperatively affected on threatened species (Campbell & Hutchinson 1998). Catch of a shellfish (seta-shijimi in Japanese), *Corbicula sandai*, drastically decreased by 98% within the past 40 years. The decline may be caused by eutrophication, habitat destruction, pollution and invasion of exotic plants. During 1959-1964, catch of *C. sandai* decreased by 55%, probably because of pollution of a herbicide, pentachlorophenol (PCP). During 1972-1974, catch of *C. sandai* decreased by 34%, probably because of pollution of polychlorinated biphenyl (PCB) and mercury (Shiga Prefecture 1974). Now, the fishing effort is too large to maintain this population (Nishimori, pers. comm.).

We suspect that physiological functions of many animals are influenced by EDCs. EDCs often promote feminization especially in vertebrates. However, in several species of gastropods, it is known that female has male genitalia, which is called imposex. In severe cases, these females are sterile. Warty murex (ibonishi, *Thais clavigera* Küster) and Japanese ivory-shell (bai-gai, *Babylonia japonica* Reeve) in Japan were greatly influenced by tributyltins (TBTs) in antifouling boat-bottom paints. Almost 100% of female warty murex distributed in 94 points out of 97 survey points, had male genitalia. In addition, average fertility of Japanese ivory-shell per adult at an oyster farm was about 10g during 1981 to 1985 and was about 3-5g during 1988 to 1990 (Horiguchi 1998). Restrictive use of TBTs was regulated since ca.1990. This regulation has reduced TBT levels in environment and tissues of mussels and oysters (Rouhi 1998). In Japan, Mizuguchi (1998) reported restoration of these species in Japan.

Cost of TBTs-regulation in antifouling paints is estimated to be about US$5.7 billion per year (Rouhi 1998). TBTs-prohibition may have a direct effect to save at least 36 species of gastropods. The "willing-to-pay" (WTP) of individuals for ecosystem services and goods in the world, most of which is outside the market, is estimated to be in the range of US$16-54 per year (Costanza *et al.*. 1997). The WTPs for nutrient cycling and biological control in shelf area (2660 million ha in the world) are estimated to be US$3810 and 56 billion per year, respectively. If TBTs make these species sterile and if these gastropods are indispensable for ecosystem functions in their habitat, I disagree that TBTs-regulation does not pay and "is shortsighted or foolish" (Rouhi 1998).

6. Population declining of Steller sea lions

TBTs also distribute tissues of marine mammals (*e.g.*, Steller sea lions, *Eumetopias jubatus*, Kim *et al.*. 1996). Marine mammals accumulate many other EDCs, such as polychlorinated biphenyl (PCB, Tanabe 1998). The Krill Islands population of Steller sea lions drastically decreased by about 80% during 1960 to 1984 (Loughlin *et al.*. 1992, Takahashi &

Wada 1998) due to decrease of survival rate and reproduction rate. The sea lions have been caught surrounding Hokkaido Island, Japan. In addition, Pascual & Adkison (1994) concluded that sea lion declines have been caused by a long-term or catastrophic change in conditions, and that the magnitude of this change is equivalent in effect to a 30-60% reduction in juvenile survival or 70%-100% reduction in female fecundity. No data of reduction of reproduction rate explain the drastic population declining in the northern sea lions during 1975 to 1985 (York 1994).

Time series of the past population decline of Steller sea lions suggest that the mean time to extinction is 66 years (after 1984). If we assume that 30% reduction of juvenile survival in York's (1994) life table, the mean time to extinction is estimated to be 44 years, which is shorter than the mean time to extinction estimated from the past population declining. This is because reduction of juvenile survival has a delayed effect on population declining.

7. Discussion

Christensen *et al.*. (1996) reported on scientific basis for ecosystem management. They suggested the following several keywords for ecosystem management (EM). (1) EM includes intergenerational sustainability as a precondition. (2) EM has measurable goals that specify future processes and outcomes. (3) EM has sound ecological models and understanding that are reflected by all levels of ecological organization. (4) EM recognizes biodiversity and complexity that strengthen ecosystems against disturbance. (5) EM acknowledges dynamic character of ecosystems, instead of attempts to "freeze" ecosystems in a particular state or configuration. (6) EM recognizes that there is no single appropriate scale or time frame for management. (7) EM includes humans as ecosystem components. (8) EM has adaptability and accountability that prepares to change and test current ecosystem status and knowledge by research and monitoring programs.

We made a management program of sika deer (*Cervus nippon*) in Hokkaido Island, north Japan (Matsuda *et al.*. 1999). The population size of sika deer rapidly increased. Hokkaido Prefectural Government adopted a "feedback management program" of sika deer. This management includes sustainability, dynamic characters in deer population, hunters' behavior, adaptability and accountability in changing hunting pressure with monitoring population size. This is probably the first example of "adaptive management" (Holling 1978) for wildlife management in Japan. However, this management does not incorporate ecosystem viewpoints into mathematical models, despite of compiling ecosystem impacts of deer overabundance. Know complexity, say simply. Deer overabundance is known as a factor threatening some endangered herbs and trees. I cannot investigate quantitative influence of deer overabundance on these threatened species.

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