Ecological Risk Estimate of Toxic Chemicals Based on Population Extinction.

Yoh Iwasa[1], Hiroshi Hakoyama[2], Mayuko Nakamaru[3] and Junko Nakanishi[4]

[1] Department of Biology, Kyushu University, CREST, Japan Science and Technology Corporation (JST), Fukuoka 812-8581, JAPAN,

[2] Hokkaido National Fisheries Research Institute, Katsurakoi 116, Kushiro 085-0802, Japan

[3] Department of Systems Engineering, Faculty of Engineering, Shizuoka University, Hamamatsu 432-8561, Japan

[4] Institute of Environmental Science and Technology, Yokohama National University, Yokohama 240-8501, Japan

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Abstract

Environmental threats, such as habitat size reduction or environmental pollution, may not cause immediate extinction of a population but shorten the expected time to extinction. We develop a method to estimate the mean time to extinction for a density-dependent population with environmental fluctuation, and compare the impacts of different risk factors. We study a formula for the reduction in habitat size that enhances extinction risk by the same magnitude as a given decrease in population growth rate caused by toxic chemical exposure. This "risk equivalent" is useful in ecological risk management of toxic chemicals. We also discuss the merits and demerits of alternative methods of ecological risk evaluation.

1. Ecological Risk Evaluation Study

A research goal of this CREST project is to establish the cost-benefit analysis of environmental chemicals based on ecological risk. Ecological risk assessment normally lists up diverse aspects, including the changes in energy flow through the ecosystem, the cycling of nutrient and water, modification of soil and vegetation, together with the extinction of animals and plants. However it is not very easy to carry out quantitative management of chemicals based on these. In this CREST project, we have investigated the methods of evaluating ecological risk (hazards to wild animals and plants) and develop the management of environmental chemicals. We must search for the "endpoint" to choose, the procedure of ecological risk estimate, and the data needed to apply it to wild populations.

To do this, [1] Miyamoto and Naito studied the ecological risk estimate of chemicals by applying an ecosystem model (CASM) with many species to Lake Biwa and Lake Suwa. [2] We also estimated the hazard based on the demographic dynamics of a particular species. These two methods can be called "community approach" and "population approach". Among population approaches, [2-1] Tanaka studied the effect of chemicals to population growth rate through laboratory experiments (*Daphnia* and medaka fish) together with literature survey, and [2-2] Nakamaru (herring gull and sparrowhawk) and Murata (cormorant) evaluated the effect of chemicals to bird population growth rate.

In this project, Hakoyama and his colleagues studied the ecological risk concept based on the risk of population extinction of animals and plants (this will be explained below). Tanaka applied it to experimental populations, and Nakamaru carried out the extinction risk estimate of wild populations of herring gull and sparrowhawk.

Matsuda evaluated the population extinction risk caused by various risk factors, including habitat loss, overfishing, and pollution. He classified the endangered species of Japanese vascular plants according to the mean extinction time, and evaluated the loss of a particular habitat in terms of the shortening of the mean extinction time of the species.

In this chapter, we present a method to evaluate the extinction risk of a density-dependent population and discuss the relative impact of the decrease in habitat size and the decrease in survivorship. This gives a basis of the next chapter by M. Nakamaru who evaluated the ecological risk of DDT to a sparrowhawk population.

2. Canonical Model

Suppose that a fraction of the habitat of a population is demolished. This may not cause immediate extinction, but it depresses the population size to a lower level than before, resulting in a shorter mean time to extinction (Fig. 1). Similarly, the mean time to extinction should be shorten if the population is exposed to a toxic chemical in a low concentration that reduces the survivorship or fertility. We developed a method to compare different risk factors by using a common currency of mean extinction time (Hakoyama and Iwasa 2000a, b; Hakoyama *et al.*, 2000; Iwasa *et al.*, 2000). For this purpose we need to consider a population model that incorporates density-dependent population regulation and environmental fluctuations (see also Middleton and Nisbet 1997; Saether *et al.* 1998).

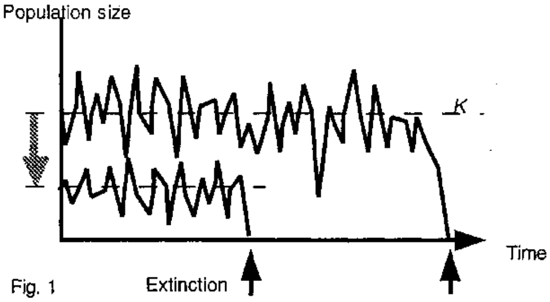

Fig. 1

Many models in conservation biology handle the situation in which the population shows a clear negative trend, and the expected time to extinction is relatively short. In contrast, in evaluating the effect of environmental chemicals at a low concentration or the decrease in the carrying capacity in terms of population extinction risk, we must handle the extinction of a density-dependent population with fluctuating population. In such a case, the mean extinction time is very long (sometimes of 10^{10} generations), and hence it is impossible to obtain it from direct computer simulation. To overcome this difficulty, we derived a mathematical formula of the mean extinction time for a simple soluble case, and evaluate the parameters in the model from available data.

As a simple standard model of population dynamics, we chose a model that includes the minimum number of factors needed to consider the extinction risk of a density-dependent population. Let X be the population size at time t. The dynamics are expressed in terms of the following stochastic differential equation:

$$\frac{dX}{dt} = rX\left(1-\frac{X}{K}\right) + \sigma_e \xi_e(t) \circ X + \xi_d(t) \bullet \sqrt{X}, \qquad (r, K \text{ and } \sigma_e > 0), \qquad (1)$$

where r is the intrinsic rate of population growth, K is carrying capacity, $\xi_e(t)$ and $\xi_d(t)$ are independent white noises for environmental and demographic stochasticities, and σ_e is the intensity of the environmental fluctuation. We call Eq. (1) "canonical model." We here assume Stratonovich-calculus in the environmental fluctuation (denoted by a small open circle) and Ito-calculus in the demographic stochasticity (denoted by a solid circle). This choice is made for the convenience of parameter fitting to time series data (see Hakoyama and Iwasa 2000a for detail).

For a long-term sustainable population, the small size of initial population causes a relatively high extinction rate in the first several generations. Once the population survives through the initial critical period and reaches the carrying capacity, the population may stay around it for a long time before extinction. Thereaftrer, the extinction time follows an exponential distribution, and we can treat extinction events as if they occur at random

(Quinn and Hastings 1987). The extinction risk can then be characterized by a single quantity -- mean time to extinction, which can be calculated as

$$T = \frac{2}{\sigma_e^2} \int_0^{x_0} \int_x^{\infty} e^{-R(y-x)} \left(\frac{y+D}{x+D} \right)^{R(K+D)+1} \frac{dy}{(y+D)y} dx, \tag{2}$$

where $R \equiv \frac{2r}{\sigma_e^2 K}$ and $D \equiv \frac{1}{\sigma_e^2}$ (Hakoyama and Iwasa, 2000a). We use the mean extinction time starting from the carrying capacity $x_0 = K$ (see Lande 1993; Lande *et al.* 1995).

3. Comparison of different risk factors

When the habitat area is reduced, carrying capacity *K* becomes smaller and the average extinction time *T* becomes shorter. The mean extinction time is a power function of the carrying capacity $T \propto K^{2r/\sigma_e^2}$ (Ludwig 1976; Lande 1993). The dependence of the mean extinction time on carrying capacity varies with the environmental fluctuation. When the environmental fluctuation is small, the average time to extinction *T* increases very quickly with carrying capacity *K*. In contrast when environmental fluctuation is large, it increases with *K* slowly.

Consider a population exposed to toxic chemical substances in the environment. Let α be the magnitude of the subsequent reduction in survivorship per generation. Population dynamics have an additional negative term for the loss:

$$\begin{aligned} \frac{dX}{dt} &= rX\left(1 - \frac{X}{K}\right) + \sigma_e \xi_e(t) \circ X + \xi_d(t) \bullet \sqrt{X} - \alpha X, \\ &= \tilde{r}X\left(1 - \frac{X}{\tilde{K}}\right) + \sigma_e \xi_e(t) \circ X + \xi_d(t) \bullet \sqrt{X}, \end{aligned} \tag{3}$$

where $\tilde{r} = r - \alpha, \tilde{K} = K - K\frac{\alpha}{r}$. The decrease in survivorship by α per generation makes both r and K smaller in the canonical model Eq. (1), and its impact on the mean extinction time can be evaluated by using Eq. (2). The same is applicable to the risks caused by pathogens, genetic deterioration, or any process that decreases the survivorship or fertility.

Figure 2a shows the relationship between the decrease in the survivorship per generation α and the mean extinction time *T*. This was calculated for several carrying capacities *K* indicated by different curves. Two other parameters (intrinsic rate of population growth r and environmental noise σ_e^2) are the estimates for the crucian carp in Lake Biwa, Japan, from fishery records (Hakoyama and Iwasa 2000a). Around Lake Biwa, there are many lakelets and small populations of crucian carp that may have similar r and σ_e^2, but different K. The estimate of mean extinction time shown here is the extinction risk of crucian carp populations in these lakelets. Mean extinction time decreases quickly with α and $\log T$ declines with α almost linearly. The decrease in logarithmic mean extinction time is larger for a large population ($K = 10^5$) than a small population ($K = 10^2$). This implies that the decrease in the survival rate per generation α are very effective in threatening large populations that are otherwise quite stable.

Figure 2b illustrate the decrease in the mean extinction time and the habitat loss. A value on the horizontal axis of 50 implies that half of the area is demolished without changing the condition in the remaining part of the habitat. Mean extinction time decreases with habitat area, initially at a slow rate and then rapidly decrease to zero between 90% loss and 100% loss. Note that the curves corresponding to different K are parallel to each other, implying that the decrease in a fixed fraction of habitat area causes the same decrease in $\log T$, irrespective of the total population size K.

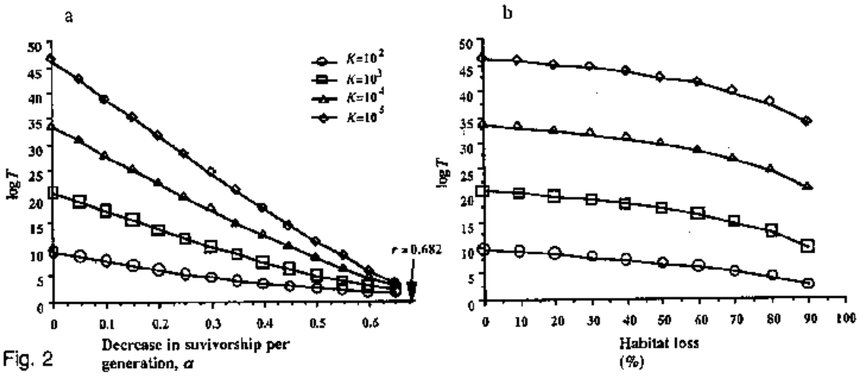

Fig. 2

With a small carrying capacity $K = 10^2$, 25 % decrease in survivorship per generation is needed to cause the risk equivalent to 50 % habitat loss. In contrast with a large carrying capacity $K = 10^5$, 5 % decrease in survivorship per generation ($\alpha = 0.05$) is equivalent to about 50 % habitat loss. In general, the magnitude of the reduction in habitat size in terms of the decrease in the logarithm of carrying capacity $\Delta \log K$ is approximately proportional to the decrease in the survivorship per generation α.

$$\Delta \log K = \left(\frac{\sigma_e^2}{2r} \log T \right) \frac{\alpha}{r}, \tag{4}$$

where T is the mean extinction time (Hakoyama *et al.*, 2000b). The proportionality coefficient increases with carrying capacity K because it increases $\log T$ if the other two parameters are fixed. This implies that the relative importance of the decrease in the habitat area compared to the decrease in survivorship is high for unstable and endangered population (with a small $\log T$); but is low in stable populations (a large $\log T$).

The population growth rate reduction α and the concentration of toxic chemicals z have a nonlinear relationship (Tanaka 1997).

To apply the model to wild populations, four parameters (r, K, σ_e^2, α) have to be estimated. M. Nakamaru (in the same volume) estimated them for herring gulls and sparrowhawks, in which age-specific annual survivorship and fertility are combined in a matrix model. With the assumption that the density-dependence and sensitivity to toxic chemicals work mostly through fertility (which includes the egg number, hatchability, chick survivorship), Nakamaru could calculate the growth rate reduction α from easily available demographic data.

Hakoyama and Iwasa (2000a) also studied the estimate of parameters from time series data, and derived a new method of eliminating bias in the estimation.

4. Different Criteria of Ecological Risk Evaluation

In ecological risk estimate, many alternative methods are available. The best method for a particular situation should depend on many factors -- the availability of the data needed for application, whether the result of evolution is robust against the error of data and the details of the model, whether the criterion is sufficiently sensitive to the hazard of the chemical, whether it represents the different aspects of the environmental hazard of the chemical comprehensively, and whether it does not cause problems in policy decision making.

4.1. Community Approach versus Population Approach

The first problem of choice is between community approach using a model with a number of interacting species or a population approach focusing on a particular species.

Multispecies model is more realistic than a model of a single species, but it includes much more parameters to estimate, and the availability of ecological data to estimate them is problematical. To know the interaction between species quantitatively is quite difficult even in a well studied area. In ecological modeling, we have to "assume" plausible value and plausible functional forms in many places. Important factors controlling the long-term coexistence of species, such as egg dormancy and spatial distribution, are often unknown for most wild populations. Hence the prediction on the species composition may not be reliable. On the other hand, the relative abundance of functional groups, such as zooplanktons, phytoplanktons and benthos, and their response to chemical exposure may be more stable. To know the quantities to be used in risk-benefit analysis, it is needed to identify the aspects of the model's behavior that are robust to the modification of the model and are sensitive to the chemical pollution.

In contrast, the population approach of ecological risk evaluation is based on a much simpler model, in which food resources, competitors and predators are regarded as the environment to the focal species. Although the model in the population approach is less realistic than community approach, it has a fewer parameters to estimate and the data are relatively easy to collect, as demonstrated by the work by Nakamaru and Murata (see their chapters)

Even in those successful examples, some of the data needed in estimation may not be available. In such a case we have to use the data taken from different but related species. The density-dependence of fertility and survivorship would reduce the population growth rate at a high density because of the shortage of food resources and nesting sites, or the spread of infectious diseases. Hence the population growth rate at a low density is not the same as the value observed for the well studied density population. Nakamaru *et al.* (2001) calculated the intrinsic rate of natural population growth of herring gulls from the doubling time of an exponentially growing population. Under additional assumptions that both the density-dependence and the sensitivity to DDTs work at age-specific fertility (rather than mortality), they could estimate the decrease in the population growth rate from available demographic data.

4.2. Intrinsic Population Growth Rate versus Mean Extinction Time

Within the population approach, we may evaluate the reduction of intrinsic population growth rate by the chemical exposure (e.g. Murata) or we may calculate the effect to the population extinction risk (e.g. Nakamaru). To calculation the mean extinction time, we use the estimate of reduction in intrinsic population growth rate and the carrying capacity K and the magnitude of environmental fluctuation σ_e^2. We used several standard value of K and also the standard value of σ_e^2. The merit of expressing the effect of ecological hazard of environmental toxic chemicals is its clarity and the intuitive appeal -- "the risk that is equivalent to the loss of carrying capacity of 40%" should be easier to capture the magnitude of risk than "the risk causing the reduction of population growth rate from r=0.4 to r=0.3."

4.3. Risk of Endangered Species or Risk of Common Species

From the conservation viewpoint, what matters is the extinction of endangered species. Hence the extinction risk to sparrowhawks is much more meaningful than that to herring gulls or cormorants. In addition, if we are to prevent from the extinction of the whole species only, the importance of extinction risk can be much lower if the same species have a stable habitat elsewhere than if it has no other stable habitat.

However the population persistence of species that are sensitive to the chemical exposure (such as waterfowls and raptors) may be an indication of the "ecosystem health" that guarantees the quality of local natural environment. If so, it is important to keep those species in each area, irrespective of the availability of alternative habitats. It is also important that those species exist, whether or not the bird is classified as endangered. To keep those species in each local area (can be of the size of a prefecture in Japan) may be a criterion to use in evaluating ecosystem risk of chemicals.

In conservation, the risk of extinction with an extremely long mean time is meaningless. All the efforts should be focused to the populations with a significant probability of population extinction (say 5%) within a period of reasonable length (say 100 years). In these cases, the effect of environmental toxic chemicals are normally very low and negligible. However if we use the population extinction risk as a measure of ecological hazard of chemicals at a low concentration, estimating the mean extinction time however long it may be is quite useful as it allows us to discuss the magnitude of ecological risk of chemicals and the habitat loss by the same currency.

At this moment, we are not able to identify a single method that is always the best among possible alternatives. But we would like to conclude tentatively that the method of evaluating ecological hazard in terms of the risk equivalent is promising.

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