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THE IMPORTANCE OF TYPE II ERROR AND FALSIFIABILITY

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Abstract. Before intergovernmental consensus under the Rio Declaration in 1992, ignorance of type I errors had been disfavored in science. However, the Precautionary Principle (PP) counsels the avoidance of type II errors, rather than of type I errors. We need a new academic code for the PP. The risk of extinction has usually been evaluated based on conservative estimates of the present population size. I define the weight of evidence as the extinction risk of Japanese vascular plants based on unbiased estimates. Catch quotas in the fisheries are usually decided by precautionary approach. I calculate the long-term yield and risk of stock collapse under a simple stock dynamics model. The weight of evidence depends on the frequency of grids with size unknown. In a few plant species, rankings based on conservative estimates have differed from rankings based on unbiased estimates. In fishery management, a catch quota based on a precautionary approach proved neither sufficient nor necessary to avoid stock collapse. The precautionary approach is one of the reasons that prevent us from maximizing a sustainable yield. We need to clarify the end-point of risks, and check whether it is necessary to adopt a PP. We can obtain the weight of evidence that is measured under unbiased estimates, while the risk based on a PP is measured under conservative estimates.

Key words:
Precautionary approach, Weight of evidence, Risk of extinction, Adaptive management, Sustainable use

INTRODUCTION

In environmental issues, the Precautionary Principle (PP) was established at the Earth Summit in 1992. Principle 15 of the Rio Declaration (1992), reads: "In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" [1]. The PP was explicitly applied by two international conventions in 1992. In the UN Framework Convention on Climate Change, the PP use was: "Where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing such measures, taking into account that policies and measures to deal with climate change should be cost-effective so as to ensure global benefits at the lowest possible cost". In the Convention on Biological Diversity, the PP came out as: "Noting also that where there is a threat of significant reduction or loss of biological diversity, lack of full scientific certainty should not be used as a reason for postponing measures to avoid or minimize such a threat".

Before the Earth Summit, scientists were not expected to make comments without full scientific certainty, and also to defend their results irrespective of public opinion. Many of today's scientists were impressed as children by the episode of Galileo during the Inquisition. Since the Earth Summit, scientists have been encouraged to give some opinions without full scientific certainty. Despite

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the fact that the Rio Declaration states a restrictive condition of "serious or irreversible damage", the PP is often applied to reversible damage, as shown below. In addition, the Rio Declaration did not indicate the weight of evidence needed before applying the PP. There is no new academic code for interpreting what scientists mean by scientific certainty [2].

We thus need to establish some standards if we are not to postpone taking cost-effective measures to prevent environmental degradation. I shall try to use type I errors, for example, we are inclined to diagnosis no cancer unless there is significant evidence of a cancer, and falsifiability as criteria for the PP. I shall also discuss the importance of the weight of evidence and the non-regret policy in the same connection.

THE RELATIONSHIP BETWEEN WEIGHT OF EVIDENCE AND TYPE I ERRORS

The PP is sometimes discussed in reference to type II errors [2]. Consider a particular hypothesis H_1 and its null hypothesis H_0, e.g., H_1 is that a species is endangered and H_0 is that a species is not endangered. If the probability that the type I error (a null hypothesis H_0 is true) occurs is larger than a significant level, e.g., 5%, then traditionally hypothesis H_1 does not obtain. This was a convention in science until the PP was established in the Rio Declaration. After that, it was considered that a hypothesis was not necessarily to be rejected even though the null hypothesis might not statistically be discounted. It is clear that we can make many hypotheses that are neither discounted nor upheld. For example, the risk of leukemia due to electric waves generated by electrical products has not been discounted. Yet many types of electrical products are permitted and many people use these, probably including some who are concerned with human health risks.

According to the PP, type II errors, for example, a species is not listed as endangered when it is in fact endangered, are avoided at the risk of making a type I error, e.g., a species is listed as endangered when it is in fact not endangered. PPs are clearly indispensable on our uncertain and finite planet. They may not serve to avoid all type II errors including cases where the order of magnitude is extremely low. However, the current definition of the PP is not based on the magnitude of type I or type II errors involved.

The risk is usually deduced from assumptions that are not yet verified, and is usually conservatively biased. The World Conservation Union/Special Survival Commission (IUCN/SSC) [3] described the position on page 25: "Assessors should resist an evidentiary attitude and adopt a precautionary but realistic attitude to uncertainty when applying the criteria, for example, by using plausible lower bounds, rather than best estimates, in determining population size, especially if it is fluctuating". Thus, for example, an extinction risk based on conservative estimates is obviously biased. This would serve to avoid type II errors (endangered species not preserved due to lack of full certainty). The contrasts with the policy of basing risks on best estimates, and measuring type I errors (null hypothesis is not discounted).

We can define risks by using type II errors (the actually endangered species is not ranked as endangered). We usually obtain type II errors from the true magnitude of extinction risk of that species. However, the true probability is usually unknown. In order to measure risks, we usually use a conservative assumption instead of the true probability. By contrast, if we can define the weight of evidence, this will serve to assess a potential type I error as measured by the probability of H_0 (a safe species is ranked as endangered) based on unbiased assumptions.

The weight of evidence relates to the certainty of interim assumptions. If best estimates are not known, we use the least biased estimates available for type I errors. I shall proceed to consider the two types of error in the case of extinction risk assessment for Japanese vascular plants.

Suppose an endangered species for which the estimated population size is N_p and the rate of population decline is R. I can evaluate the extinction risk by N_p and R. Despite difficulties of collecting these data for wild plants, the Red Data Book (endangered species list) of Japanese vascular plants (hereafter abbreviated as Japanese plant RDB) [4] is based on the database of the spatial distribution and number of flowering individuals for 2100 suspected threatened taxa (species, subspecies or varieties) of Japanese vascular plants. This database is one of the most intensive field surveys for threatened species [5], and is compiled by the Threatened Species Committee of the Japanese Society of Plant Taxonomists (TSC-JSPT). I estimated N, R, the extinction risk within the next 10, 20 and 100 years, and the mean time to extinction for each taxon using this database and a Monte Carlo simulation [6,7] (see available from: web site http://cod.ori.u-tokyo.ac.jp/~matsuda/redmann.html for a source code of extinction risk evaluation of Japanese vascular plants).

TSC-JSPT used 1:25 000 scale maps of Japan (each ca. 10 • 10 km^2). For every taxon in each of 4457 map grids, taxonomists answered questionnaires regarding the number of local habitats, the number of individuals, the rate of population decline, and the major factors involved in population decline [7]. The questionnaire asked for the number of individuals to be divided into unknown, 1–9, 10–99, 100–999, and more than 1000. The rate of population decline within the past 10 years was categorized as unknown, >99%, 90–99%, 50–90%, 0–50%, and non-positive. The questionnaire also asked about map grids where a taxon once existed, but is now extinct.

Among many known and unknown problems in estimations, one of the biggest problems is ignorance of grids where the number of individuals is unknown. I used the most pessimistic estimate of N and the geometric mean of 1 – R. There is a large uncertainty in R (e.g., 50–90% per decade) and many unknown answers for N and R in the questionnaires. There are many grids whose reported population sizes are unknown. In the Japanese vascular plant RDB, the extant or extinct habitat information for 1325 taxa in some grids was reported. For 29% of 20 877 grids among these 1325 taxa, the reported population size was unknown. In 20 taxa, population sizes of all their extant grids are unknown. In 256 taxa among these 1325 taxa, population sizes of >50% extant or extinct grids are unknown, as is the current extant/extinct status of plants in those grids.

In estimation of the extinction risk for these taxa in the Japanese plant RDB, Yahara et al. [7] ignored the existence of grids whose population sizes were unknown. However, if I assume that population sizes for these unknown grids are randomly chosen from to be the same as the size distribution of grids where the population size of such taxa is known, the estimate of extinction risk becomes lower than the previous estimate. Let the extinction risk within the next t years as the ratio of the extinction risk when grids with unknown population size are ignored be p_t; and the risk when the population size distribution of grids with unknown size is identical to that of grids with known size be p_t^*. According to the above definitions, I define the weight of evidence as p_t^* • p_i^* is usually smaller than $p_{i.}$

A taxon is ranked as critically endangered (CR), endangered (EN) and vulnerable (VU) if $p_{10} > 0.5$, if $p_{20} > 0.2$ and if $p_{100} > 0.1$, respectively [6]. I use p_{100}^* as the weight of evidence for listing this taxon as being threatened.

Table 1. Extinction risk and the weight of evidence for 5 species of Japanese vascular plants based on two scenarios, (A) that ignored the existence of grids whose reported population sizes are unknown and (B) whose size distribution for size-unknown grids is statistically identical to that of size-known grids

	L	%UK	N_p	N_p^*	R	Rank	p_T	p_T^* ranking	p_{100}^* listing
Cynanchum inamoenum	2	81%	35	69	60%	CR	54%	32%	100%
Lycopodium alpinum	6	73%	361	668	50%	EN	27%	11%	100%
Primula tosaensis	15	90%	901	1149	22%	EN	21%	16%	100%
Sedum polytrichoides	19	39%	5979	6524	9%	VU	11%	9%	9%
Carex sacrosancta	6	63%	1613	4775	86%	VU	26%	4%	4%

L – the number of extant grids population size was known by questionnaires.
%UK – percentage of grids whose size was unknown.
N_p and N_p^* – estimates of total population size by scenario A and B, respectively.
R – estimate of population decline rate within the past 10 years.
Rank – ranking in the Japanese plant RDB.
p_T and p_T^* – extinction risk within the years in question in scenario A and B, respectively,
p_{100} – risk within the next 100 years in scenario B.

For 6 CR taxa, 36 EN taxa and 20 VU taxa among 1325 taxa, the weight of evidence within the years in question (p_{10}*, p_{20}* and p_{100}*) was lower than 50%, 20% and 10%, respectively. All of these might be ranked as one rank lower in category (CR to EN, EN to VU and VU to NT). Table 1 shows several typical cases. There are two major reasons for disagreement among most CR taxa, high ratios of size-unknown grids (taxa except for *Sedum polytrichoides* in Table 1) and extinction risks that are close to threshold (the case of *Sedum polytrichoides)*. For the moment I ignore the uncertainty of the rate of population decline, R. If R depends on habitat size and latitude, the weight of evidence differs from the values shown in Table 1. In addition, R may vary with decades. Questionnaires were obtained in the mid 1990s, during which many projects were introduced for golf course construction and other land development. For the year 2000, R might be smaller than it previously was. However, it is unpredictable how R will change throughout the 21st century.

A PRECAUTIONARY APPROACH TO FISHERY MANAGEMENT

Precautionary measures are also applied to reversible damage in the regional environment. Over-exploitation in the fisheries has been interpreted as reversible damage because a fishery collapse usually comes before a stock collapse and stock recovers under zero exploitation. Despite this, Garcia [8] has emphasized the need for precautionary measures in fishery management, probably because of the large uncertainties in fishery management and a long history of fishery mismanagement [9]. This is called the precautionary approach (PA), which is often distinguished from the PP [8,10].

In terms of risk management, we should first detect what is an undesired event. If stock collapse is the only undesired event and the risk of stock collapse caused by overfishing is really negligible, we do not need a PA. A PA in the fisheries usually interprets failure in achieving maximum sustainable yield (MSY) as another undesired event. However, use of pessimistic estimates under a PA is a major factor of failure in achieving MSY itself.

In order to explain the relationship between PA and MSY, I consider a simple stock dynamic model:

$$dN/dt = [r\,(1 - N/K) - f]\,N, \qquad (1)$$

where:

N is the stock abundance as a function of time *t*; *r* is the intrinsic rate of natural stock increase;

K is the carrying capacity which the stock reaches without any exploitation [11];

and f is the rate of exploitation [12,13].

The stable equilibrium state of N, denoted by N*, depends on f, which is a solution of the equation that $dN/dt = 0$. If $f < r$,

$$N^* = K\,(r - f)/r, \qquad (2)$$

and $N^* = 0$ if $f \geq r$. The yield is given by fN. The equilibrium yield, fN*, is obtained by $fN^* = fK\,(r - f)/r$ if $f < r$. It is easily seen that fN* is 0 either when $f = 0$ or when $f \geq r$. In the former case, the natural bioresource is kept at its carrying capacity, K, but humans do not use this natural resource at all. In the latter case, humans do not get any equilibrium yield because the stock is exhausted. If f is between these two extremes, or if $0 \leq f \leq r$, the equilibrium yield is positive. The equilibrium yield, fN*, is maximized at the point where $f = r/2$, and the maximum sustainable yield (MSY) is Kr/4.

Let us also consider MSY under a more general stock-production relationship:

$$dN/dt = R(N) - fN, \qquad (3)$$

where:

R(N) is the production rate of stock without any exploitation, which is a function of N. If $R(N) \geq 0$ for $0 \leq N \leq K$, $R(0) = R(K) = 0$ and $R''(N) < 0$, it is again seen that fN* is 0 either when $f = 0$ or when $f \geq R'(0)$. The MSY is obtained if $R\,(N) = 0$.

This classical MSY theory implicitly assumes permanent information on the stock-production relationship R(N) and the relationship between fishing effort and the rate of exploitation (catchability) and its parameter values, e.g., r, K and f. This also assumed f constant irrespective of N. Because of uncertainties, it is difficult to obtain the MSY, and the actual yield is always lower than the MSY in the long run. The stock may collapse if we overestimate the stock productivity and f is larger than actual R'(0). Thus,

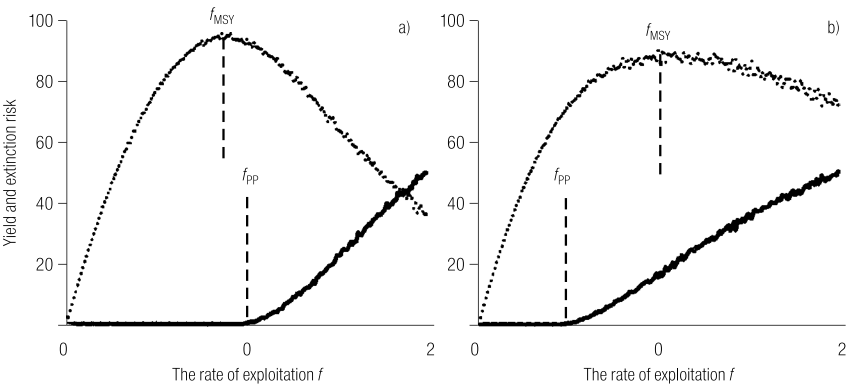

Fig. 1. The average equilibrium yield and extinction risk as a function of the rate of exploitation obtained by a mathematical model (4).

the stock level may either be below or above the desired level.

Fig. 1 shows the relationship between the fishing rate f and the average equilibrium yield, and the extinction risk at which I incorporate uncertainty in R(N) and f. We assumed a generalized logistic model [14] that

$$R(N) = r\,[1 - (N/K)^q]\,N, \quad (4)$$

where:

q is a non-negative constant that indicates the magnitude of density regulation. I used uniformly random variables, ζ, between – 0.5 and 0.5 and assumed that $r = 2\,e^{\upsilon\zeta}$, $K = 200\,e^{\zeta}$, $q = e^{\zeta}$ and $f = f_0\,(1 + \zeta)$. If $r > f$, equilibrium exists and is given by $N^* = K\,[(r - f)/r]^{1/q}$ while the MSY is achieved when $f = f_{MSY} = rq/(1 + q)$. If all the uncertainties are ignored, or if $\zeta \equiv 1$ for r, K, q and f, the MSY is obtained by $f_{MSY}N^* = 100$. Because of large uncertainties in these parameter values, the average MSY is below 100. I consider two reference points with respect to the rate of exploitation, $f \bullet f_{MSY}$ is defined as that f at which the average equilibrium yield is maximized. I talk of the maximum f when the extinction risk is non negligible by f_{PP}. From the PP point of view, we should not allow a larger exploitation rate than f_{PP}. If r has a smaller uncertainty ($u = 0.5$, Fig. 1a), the extinction risk is negligible if $f < 0.95$, or $f_{PP} = 0.95$. If $u = 2$, $f_{PP} = 0.45$ (Fig. 1b). Irrespective of u, the average equilibrium yield is maximized at the point at which $f = 1$.

The PP should be used in order to reduce the risk of extinction, which the PA is effective neither for stock conservation nor for the MSY. Suppose as a guideline that "target" f should be moderately, e.g. 10–20%, smaller than f_{MSY} [10,15]. In the case shown in Fig. 1a, a target f that is smaller than f_{MSY} is unnecessary to avoid stock extinction and is a risk factor in achieving MSY, because the extinction risk when $f = f_{MSY}$ is negligible. In the case shown in Fig. 1b, it is not enough to take 80% of f_{MSY}, because f_{PP} is much smaller than 80% of f_{MSY}. If we adopt a biased and conservative estimate of parameter values, it is impossible to achieve MSY.

In addition, we do not need to assume that the rate of exploitation is constant. Adaptive management [16,17] encourages feedback control, which means that the rate of exploitation changes with the stock level. Let us consider a feedback control rule:

$$dN/dt = r\,[1 - (N/K)^q + \varepsilon]\,N - fN,) \quad (5a)$$

$$df/dt = vf\,(P - 1), \quad (5b)$$

$$P(t) = [N(t)/N_T]e^{sZ}, \quad (5c)$$

where:

Z(t) is the standard normally random variable; ε is a stochastic fluctuation of the renewal process due to environmental stochasticity (I assumed here that $\varepsilon = e^{Z(t)}$); v is a positive constant representing the velocity of feedback control; P(t) is the estimate of current relative stock size; *s* is the standard deviation of measurement error in N; N_T is the target level of stock size. If perfect information is given, N_T should be determined by the stock level, which corresponds to MSY. However, the true MSY stock level is usually uncertain. I assume that $r = 2e^{uz}$, $K = 200e^{\zeta}$, $q = e^{\zeta}$ because of incorporating uncertainty. I also assume

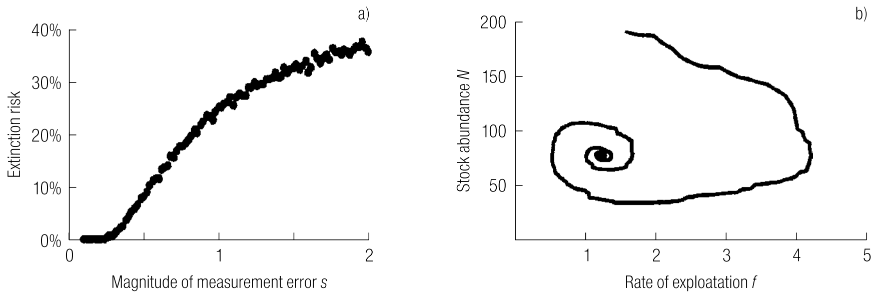

Fig. 2. The relationship (a) between the magnitude of measurement error and the risk of extinction for 5000 sets of randomly chosen r, K, q; and (b) how to reach the equilibrium for particular parameters.

that $N_T = Ke^{s\zeta}/2$, because K/2 is the MSY stock level if q = 1 and $e^{s\zeta}$ is an error factor.

With equations (5a–c), I analyze the differential equation systems for feedback control. In 99% of 5000 sets of randomly chosen parameter values of r, K and q, the system approached the equilibrium point when s = 0.3. At equilibrium, $N = N_T$ and $f = [1 - (N_T/K)^q]r$. Because the rate of exploitation changes with time, the equilibrium *f* increases with r, q and K/N_T. The equilibrium yield increases with r, q and K. If $N_T > K$, the stock will be exhausted. This is possible if $e^{sZ} > 2$. Therefore, the extinction risk does not depend on u (uncertainty in the intrinsic rate of population increase) but depends on s (the magnitude of measurement error in the relative stock size), as shown in Fig. 2.

IMPORTANCE OF FALSIFIABILITY

In the above analysis, I have ignored the effects of ecosystem interactions, decadal climate change and irresponsible fisheries that do not keep to the recommended rate of exploitation. There are probably other risk factors in fishery management and stock collapse. On the other hand, I assumed that the target stock level N_T does not change with time. I could reduce the risk by tuning N_T with the response of fisheries on the monitored stock fluctuation. This is called adaptive learning [17]. Feedback control and adaptive learning characterize adaptive management.

I also ignored the risk that the population dynamics model or the functional form adopted in equations (5a–c) might be incorrect. In the real world, factors that I have not yet incorporated into the risk assessment always exist. We need to incorporate new knowledge (accountability) and to change our actions with the state of bioresources (adaptability). Accountability and adaptability are indispensable in ecosystem management [18]. Furthermore, we need to make a falsifiable prediction within the framework of adaptive management. Even though there is great uncertainty on the future state of adaptive management, we should not accept hypotheses that are neither upheld nor discounted in the future. This is called falsifiability. Accountability, adaptability and falsifiability make up the merit of adaptive management.

One regional government adopted a feedback management policy concerning sika deer in Hokkaido, Japan [19]. I assumed that the deer population size is less than 200 000, despite the lack of full scientific evidence on the deer population size [19]. The fate of management depended on the absolute population size. Before the beginning of deer management in Hokkaido Island, Japan in 1998, I sent an e-mail to a Japanese mailing list provided by a staff member of the Japan Office of the World Wildlife Fund for Nature, in which I stated that if deer management cannot prevent the deer population from increasing, if the deer population size in the eastern Hokkaido is more than 300 thousand. In 2001, many survey data indicated a decrease in the deer population [20]. If management is based on some assumptions that have not yet been verified, we need to explicitly describe a falsifiable prediction.

Matsuda et al. [21] showed another falsifiable prediction relative to Japanese pelagic (surface) fish. Catch statistics for several surface fish (sardine, chub mackerel, anchovy, Pacific saury and jack mackerel) in Japan showed decadal fluctuation (Fig. 3). Decadal change in surface fish stocks is also

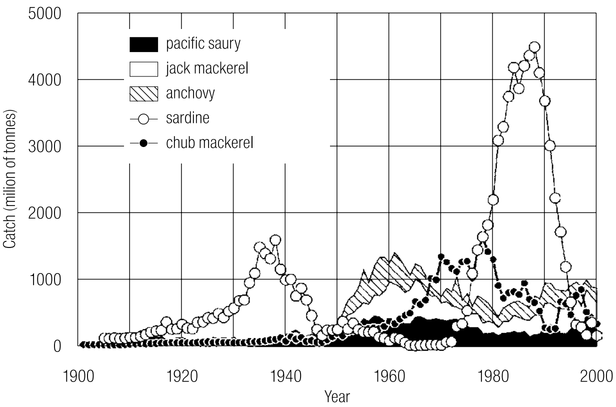

Fig. 3. Catch statistics of 5 surface fish in Japan [24].

known in the California current [22]. Plankton net surveys for egg production indicated a larger amplitude of fluctuation in stock abundances of these species. Catch abundance fluctuations in anchovy, Pacific saury and jack mackerel were positively correlated. Stock abundances among sardine, chub mackerel and anchovy were negatively correlated.

Matsuda et al. [21] proposed the "cyclic-advantage hypothesis" to explain the species replacement pattern among surface fish. If sardine have a stronger competitive ability than chub mackerel, anchovy are stronger than chub mackerel and chub mackerel are stronger than anchovy, a heuristic mathematical model can exhibit permanent oscillation among these stocks. In 1992, I predicted that the next dominant to sardine was anchovy, Pacific saury and jack mackerel. That prediction came true in the mid 1990s. I also predicted that the second next dominant was chub mackerel. Chub mackerel in the northwestern Pacific water has been over-exploited since the 1990s. If the cyclic advantage hypothesis is true, the sardine recovers only after the chub mackerel increases. Overexploitation of chub mackerel prevents the chub mackerel from recovering [23]. Therefore, if the sardine recovers before the recovery of chub mackerel, the cyclic advantage hypothesis must be rejected. This is falsifiable. Overfishing of chub mackerel would be interpreted as an experiment on this hypothesis [24].

DIOXIN CONTROVERSY IN JAPAN

In Japan, industrial waste is usually disposed of burning or burying. Burning waste is a source of dioxin. Since the impact of endocrine disrupters on human health and ecological risks is well known [25], small incinerators have been removed from many apartments in Japan. Some people were also annoyed by dioxin from small fallen-leaf bonfires. Small fallen-leaf bonfires have been traditional in Japan. Despite the fact that the risk from such bonfires is probably as high as the risk from incinerators, Kitoh [26] disagreed with stopping this traditional custom, and emphasized the importance of comparing ecological and health risks with social, economic, religious and cultural values. The present human health risk is much smaller than that in the past and that of wild living things. In nature, wild living things are exposed to many kinds of risk, especially during early life and the breeding season.

Nakanishi [27] emphasized the comparison of health and ecological risks with economic benefit. Gamo et al. [28] compared population risks from use and non-use of chlordane using the expected loss in life expectancy. Matsuda et al. proposed using the database of Japanese threatened plants as a yardstick of ecological risk [4,6]. Oka et al. [29] assessed the ecological risk and economic benefit from the development of a wetland, where many threatened plants inhabited.

Over-response against small incinerators was alleged. If residents do not burn plastics, the health risk to neighbours is probably quite small. The health risk from dioxin contamination in seafood is much higher than the risk from incinerators as it stands. Some chemists recommend that pregnant women eat meat rather than fish because of dioxin contamination [25]. However, fish is known as a healthy food owing to its rich unsaturated fatty acids. Higher consumption of fish and omega-3 fatty acids is associated with a lower risk of coronary heart disease in men and women [30]. Japanese people are known as fish-eaters and for having the longest life expectancy in the world (Table 2). We need to compare the risk from a food item with its benefit. Thus Colborn et al. [25] still recommend breast-feeding, despite endocrine disrupter contamination.

RISK COMMUNICATION BASED ON A NON-REGRET POLICY

Even though the economic benefit is insufficient compared with the risks, the next question is how to evalu-

Table 2. Mean longevity, daily caloric intake and % intake from cereals in the USA, Sweden, Japan and China

	Mean longevity (yrs)		Daily caloric intake (kcal/day/person)	% intake from cereals
	Male	Female		
USA	72.0	78.9	3732	21.7
Sweden	75.3	80.8	2972	21.1
Japan	76.6	83.0	2903	39.7
China	69.1	72.4	2727	69.7

Source: Britannica Yearbook, 1997.

ate the social, religious and cultural values. There is no algorithmic or systematic method to measure these values. The only way is democracy. It is important to appreciate and preserve social links in environmental conservation policies. Kitoh [26] considered conservation movement as a result of disconnected social links by modernization. Consumers eat meat and sashimi, but they rarely imagine that their food is made from living things.

A non-regret policy is probably an ingenious idea to rationalize inconsistencies between risks and values. A non-regret policy in terms of energy resources is a downsizing of energy consumption, rather than development of new energy resources from agricultural products. If a shortage of food resources comes earlier than lack of an energy resource, this technology becomes regretful. A non-regret policy is defined as a policy that will not be a source of regret irrespective of future uncertainty. I here define this word as a policy that is acceptable to the public irrespective of future uncertainty. In order to meet with public consensus, we need to explain the quantity and quality of risks and social, economic and cultural consequences from the various environmental policies. This is called risk communication. Risk communication should not incorporate only risk, but also social concerns and values into the public process of decision making.

In conclusion, we need the PP for our uncertain world, and also some academic codes for the relationship between the PP and the degree of scientific certainty that is indispensable in the classical scientific standard. The risk and weight of evidence should refer to type II and type I errors. The weight of evidence based on best estimates as defined in this paper would afford a good relationship between the PP and the classical scientific standard.

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