Risk-Benefit Analysis for Environmental Policy
---What Has Been Done and What Remains to be Done---

Tosihiro Oka[1,2]

[1] Fukui Prefectural University, 4-1-1 Kenjojima, Matsuoka-cho, Fukui 910-1195, JAPAN
[2] CREST, Japan Science and Technology Corporation

Key Words: risk-benefit analysis, cost-effectiveness analysis, environmental policy, human health risk, ecological risk

Abstract

The characteristics of risk-benefit analysis as a special case of cost-effectiveness analysis is made clear. The three forms of risk-benefit analysis are explained with the examples of the studies on the regulation of chlordane, mercury and benzene. The method of aggregating the probabilities of the species extinction that enables us to interpret the resulting measure of ecological risk as 'expected loss of biodiversity' is applied to a case of land-use conversion. It is proposed to apply risk-benefit analysis to the cases where how to select among alternative policies is at issue.

1. What Is Risk-Benefit Analysis?

'Risk-benefit analysis' for environmental risk management is an analysis that estimates the ratio between the benefits of using chemicals or of releasing pollutants into the environment and the environmental risks from the chemicals or pollutants. Let us call this ratio the `B/R ratio'. Since some parts of those benefits would be lost when the risks are reduced through any policies limiting the use of the chemicals or limiting the discharge of the pollutants, the B/R ratio is equal to the ratio between the costs of reducing risks and the magnitudes of the reduced risks when policies to reduce risks are concerned. In this case, risk/benefit analysis is used to estimate the `unit cost of risk reduction'.

Risk-benefit analysis is a tool to assist policy-makers in pursuing efficient environmental regulation. That is because by setting priorities among environmental regulatory programmes according to the B/R ratio they can achieve a certain amount of risk reduction at the lowest cost or achieve the largest risk reduction at a given cost.

Risk-benefit analysis is a form of `cost-effectiveness analysis' as distinguished from `cost-benefit analysis'. In cost-benefit analysis, all the effects of a policy should be quantified in monetary terms and the `net benefit', which is the total benefits minus the total costs, is calculated. Policy makers can base their decision on whether the net benefit is larger than zero or not. However, the effects of a policy cannot always be quantified in monetary terms. Some effects are difficult to attach monetary values to, and others are regarded as having no monetary values from the outset. In these cases, since not all the effects are evaluated on a common scale, no value of net benefit can be obtained. Even so, if all the effects that are not evaluated in monetary terms can be quantified on a common physical scale, then we can obtain two dimensions of values, one in monetary terms and the other in physical terms, and we can determine the ratio between them. Risk-benefit analysis is a special case of cost-effectiveness analysis where the effects that are not evaluated in monetary terms are expressed in terms of risk-reduction and all the other effects are expressed as `costs' in monetary terms.

2. Risk-Benefit Analysis in Practice

2.1 *Human health risk*

1) *The basic form of risk-benefit analysis for human health*

As is shown above, risk-benefit analysis involves the assessment of risk reduction and the assessment of costs for reducing risks. The most prominent feature of the method our research group has adopted for assessing risks to human health is to use loss of life expectancy (LLE) as a measure of risk (Gamo, Oka and Nakanishi 1996). We have been using three types of model for relating LLE to human exposure to chemicals. The first model is concerned with the risk from carcinogens. In this model a certain level of exposure to a carcinogen for one year at a certain age is related to some increases in the subsequent annual death rates from cancer over a lifetime, which are, in turn, related to the value of LLE. We can determine an average value of the LLEs across the ages of the people being exposed as a measure of risk from one-year-exposure to the carcinogen. The second model is the one we have used for assessing the risk from methylmercury. This model is the same as the first one except that it relates a certain level of exposure to methylmercury for one year at a certain age with some increases in the annual death rates from *general causes* over a lifetime. The third model has been used for assessing the risk from neurotoxins. In this model, exposure to a neurotoxin for one year at any age is related to an increase in the annual death rate *within just the same year* from general causes.

The common characteristic of all the three models is that they relate one-year-exposure for a population of various age groups to an average value of LLE for the population. This characteristic is crucial for connecting the risk with the cost for reducing it. Owing to this characteristic, the quantity of risk-reduction (which occurs at the same time as exposure-reduction takes place) in year t can be expressed as $\Delta LLE(t)$, which represents the average reduction of LLE for the whole affected population due to the reduction of exposure within year t and we can obtain a stream of risk-reduction, $\Delta LLE(1)$, $\Delta LLE(2)$,..., $\Delta LLE(T)$, from which we obtain a discounted initial (or present) value,

$$\sum_{t=0}^{T} \Delta LLE(t)\,(1+r)^{-t}$$

where r is the time discount rate. Costs are also usually obtained in the form of a stream over years, $C(0)$, $C(1)$, ..., $C(T)$, which produces a discounted initial (or present) value,

$$\sum_{t=0}^{T} C(t)\,(1+r)^{-t}$$

From these initial values we can easily obtain the value of the B/R ratio.

2) *Application to the cases of the regulations of chlordane, mercury and benzene*

This is the basic form of our risk-benefit analysis. The risk-benefit analysis of the prohibition of chlordane, a termiticide, was carried out according to this basic form. As a substitute for chlordane, which was regarded as a carcinogen and prohibited in 1986, organophosphorus termiticides such as chlorpyrifos came to be used, which have neurotoxicity. The risk of the substitutes for chlordane was estimated according to the third model described above (Gamo, Oka and Nakanishi 1995), and we obtained a stream of risk reduction due to this regulation as shown in Table 1. On the other hand, we estimated a stream of its costs also shown in Table 1 on the basis of information on the rise in the price of termiticide and according to our forecast for the increase in termite control treatments (Oka, Gamo and Nakanishi 1997). The discounted initial value of the stream of the risk-reduction with the discount rate of 5% is 22,741 years of LLE, and that of the stream of the cost is 1033 billion yen. The B/R ratio is, thus, 45 million yen per year of LLE.

Table 1: Risk-reduction and cost in the prohibition of chlordane

Year	Risk-reduction LLE(year)	Cost (billion yen)
1987~1991	103	140
1992~1996	6165	289
1996~2001	11612	452
2002~2006	11690	452
2007~2011	11842	454
2012~2016	11998	457

In some cases, it is more convenient to use the constant annual value equivalent to the discounted initial value, because risk-reduction is estimated more easily in the form of annual value in a stationary state. The initial value, say P, is converted to a constant annual value, A,

$$A = Pr/[1-(1+r)^{-N}]$$

a stream of which for N years produces an initial value that is equal to P according to the formula where r is the discount rate. In the risk-benefit analysis for the regulation of benzene concentration in gasoline, the initial cost for reducing the concentration, 100 billion yen, was converted to the annual cost of 7 billion yen under r=0.05 and N=25, which was added to the running cost of 13 billion yen per year to produce a total annual cost of 20 billion yen (Kajihara et al. 1998). The reduction in cancer risk due to exposure to benzene was estimated to be 7.3 cases of leukemia per year (Kajihara et al. 1998), which was estimated to have reduced 81 years of LLE per year. Therefore, the B/R ratio of this regulation is 250 million yen per year of LLE.

Another variation of the basic form was used when the prohibition of the mercury electrode process for production of caustic soda was analyzed (Nakanishi, Oka and Gamo 1998). In this analysis, avoided LLE per annum was estimated by comparing the background level of methylmercury intake with its level in a hypothetical case where the production of four million tonnes of caustic soda led to the discharging of 4.8 tonnes of mercury into ten hypothetical bays in Japan. The estimated value was 75.2 years of LLE per annum. On the other hand, the cost for complying with the regulation per gram of mercury reduced was estimated by dividing the constant annual value of the stream of expenditures from 1973 to 2005 by the constant annual value of the stream of reduction of mercury discharged for the same period. The resulting unit cost was 8950 yen/g. This, multiplied by the 4.8 tonnes of mercury prevented from being discharged into the hypothetical bays, produced 43.0 billion yen, which was regarded as the amount spent to reduce the risk by 75.2 years of LLE. That means 570 million yen per year of LLE is the value of the B/R ratio for this regulation.

3.1 *Ecological Risk*

1)*How to aggregate the probabilities of species extinction*

The basic form of risk-benefit analysis should also be the same for ecological risks. However, a big challenge for ecological risk-benefit analysis is the development of methodologies for assessing ecological risk. This research group is energetically undertaking research to evaluate it in terms of the probability of species extinction. In ecological risk, in contrast to health risk, risk assessment does not end with the estimation of extinction probability. It is necessary to aggregate the extinction rates of the different species influenced by the environmental changes in question. Here emerges the question about whether all the species should be treated equally or discriminately.

This question is related to the question of how to measure biological diversity. Some taxonomists (and an economist) have proposed that *taxonomic diversity*---a term including both inter- and intra-specific diversity--- can be measured by using phylogenetic information (Willams et al. 1991, 1994; Weitzman 1992; Faith 1995). I have developed a practical method to apply their proposal to the assessment of the risk from land use conversion.

The basic idea is to use the length of the branch of the phylogenetic tree that would be lost if a species were extinct as a weight for aggregation of the extinction probabilities of species. It would be desirable if we could determine the branch length by the length of the real time that has passed since the species i diverged from its sister. It is, however, quite rare that it is known.

Therefore, I treated the reciprocal of the number of nodes between the terminal node of a species and the root as a surrogate for the length of the time that has passed since the divergence from its sister species.

In addition to the problem of the lack of knowledge about the ages of species, there is another problem, that is to say, a fully resolved phylogenetic tree is not always available for a group of species. Faced with this problem, I adopted the approach of using a phylogenetic tree from the root to a certain upper taxon including the species in question and to estimate the expected value of the reciprocal of the number of nodes between the terminal node for the species and the root, on the basis of the number of nodes above the upper taxon and the number of species included in the upper taxon.

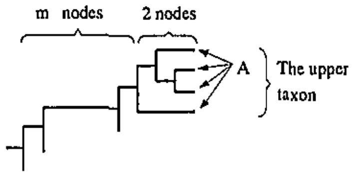

Fig. 1 Node counting when there are four species in the upper taxon

For instance, when there are four species within an upper taxon, 15 phylogenetic trees can occur within the upper taxon. In three cases, a species, say A, has one node between the root of the upper taxon and itself, in six cases, A has two nodes between the root and itself, and in six cases it has three nodes. Hence, when the upper taxon itself has m nodes between itself and the root of the whole tree (Fig.1), the expected mean value of the reciprocal of the number of nodes between the terminal node for species A and the root of the whole tree is $3/[15(m+1)]+6/[15(m+2)]+6/[15(m+3)]$.

When there are n species within an upper taxon, the number of phylogenetic trees in which a species A has k nodes between itself and the root of the upper taxon is represented by $f_k(n)$ that forms the equation

$$\begin{cases} f_1(n) = f(n-1) \\ f_k(n) = \sum_{i=1}^{n-k} {}_{n-1}C_i f(i) f_{k-1}(n-i), \quad k = 2,3,\cdots,n-1 \end{cases}$$

where ${}_nC_i$ is the combination of n things taken i at a time and $f(n)$ represents the total number of phylogenic trees when there are n species, i.e.,

$$f(n) = \sum_{i=1}^{n-1} f_i(n).$$

The expected mean value of the reciprocal of the number of nodes between the terminal node for species A and the root of the whole tree is, therefore,

$$E_n[1/(m+k)] = [1/f(n)] \sum_{k=1}^{n-1} f_k(n)/(m+k)$$

where m is the number of nodes between the upper taxon and the root of the whole tree.

To obtain the value of $E_n[1/(m+k)]$ using these equations requires huge amounts of calculations when n is large. Hence in practice, when n is larger than 100, I used $(m+1/E_n[1/k])^{-1}$

as an approximation for $E_n[1/(m+k)]$, where $E_n[1/k]$ is equal to $E_{n-1}[1/k](2n-4)/(2n-3)$.

2) *Application to the case of wetland development*

I applied this approach to the case of land use conversion in the Nakaikemi wetland. There are 16 `threatened' plant species in the wetland, which are widely distributed in the phylogenetic tree of vascular plants. For the relationships among Psilophyta, Licopodophyta (including Isoetaceae, Lycopodiaceae and Selaginellaceae), Equisetophyta, true ferns, and seed plants, I assumed the tree shown in Fig. 2 according to Bremer et al. (1987). The relationships among the three families within Licopodophyta are based on the result of a molecular analysis by Manhart (1995).

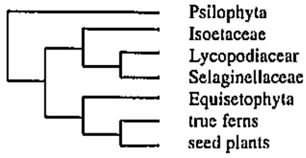

Fig. 2 Phylogenetic tree for higher taxa of vascular plants

As a result, the number of nodes for Isoetaceae between the terminal and the root is three, which is shown in the column 'number of nodes above the upper taxon' for *Isoetes japonica* in Table 2 (Isoetaceae is the selected upper taxon for *Isoetes japonica*).

Table 2: Contribution to biodiversity and ELB of the plant species in Nakaikemi

No.	Species	Selected upper taxon	No. of nodes above the upper taxon	No. of species within the upper taxon	Contribution to biodiversity
1	Isoetes japonica	Isoetaceae	3	68	0.07332
2	Marsilea quadrifolia	Marsiliaceae	9	67	0.04879
3	Salvinia natans	Salviniaceae	10	10	0.07070
4	Azolla japonica	Azollaceae	10	6	0.07720
5	Persicaria foliosa	Polygonaceae	20-21	1000	0.01775
6	Anemone hapatica	Ranunculaceae	16-19	2000	0.01465
7	Trapa incisa	Trapaceae	25-29	15	0.03085
8	Eusteralis vatabeana	Lamiaceae+Verbenaceae	29-33	580	0.00852
9	Prenanthes tanakae	Asteraceae	28-29	20000	0.00531
10	Sagittaria	aginashi Alismatales	17-19	249	0.02771
11	Najas japonica	Najadales	17-19	205	0.02905
12	Monochoria korsakowii	Pontedariaceae	22-26	34	0.03003
13	Iris laevigata	Iridaceae	18-18	1400	0.01574
14	Sparganium erectum	Sparganiaceae	22-27	20	0.03147
15	Sparganium japonica	Sparganiaceae	22-27	20	0.03147
16	Habenaria sagittifera	Orchids	17-21	20115	0.00557

For the true ferns, I determined the number of nodes above the families according to the molecular phylogenetic tree from Hasebe et al. (1995, p.146). The nodes for the seed plants were counted according to the phylogenetic tree presented by Chase et al. (1993). The resulting numbers of nodes are shown in the column `number of nodes above the upper taxon'. The numbers for the flowering plants were not determined uniquely because the phylogenetic tree I used contains unresolved relationships between some taxa and because the tree does not include all the families of flowering plants.

The numbers of species within the upper taxa were obtained from Cronquist (1981) for the dicotyledons, from Dahlgren et al. (1985) for the monocotyledons and from Kramer and Green (1990) for the ferns and fern allies respectively. The resulting values of $E_n[1/(m+k)]$$ or $(m+1/E_n[1/k])^{-1}$ depending on whether $n \leqq 100$ or $n > 100$, are also presented in Table 2. When m is not determined uniquely, the value of $E_n[1/(m+k)]$ or $(m+1/E_n[1/k])^{-1}$ is calculated by using the probability of a particular value of m. The average value of $(m+1/E_n[1/k])^{-1}$ with these probabilities as weights can also be seen in Table 2.

These values of the contribution to biodiversity can be combined with the estimations of the increments in the probability of extinction of the species to produce a value for 'loss of expected biodiversity' due to the loss of the wetland. This can then, in turn, be combined with the cost for the conservation of the land or the benefit from the development of the land to produce a value for the B/R ratio.

3. What Remains to Be Done

In many cases of decision-making for environmental regulation, the costs are actually taken into account, but only *implicitly*. Risk-benefit analysis reveals explicitly the cost per unit of the policy effect and enables us to select efficient policies. However, to persist in risk-benefit analysis may be risky because of the static or conservative nature of benefit evaluation. It is usual that when a regulation is implemented, technological innovation takes place to reduce compliance cost. It is, however, very difficult to take into account the possibility of such innovation in the assessment of the cost. Moreover, failing to adopt a regulation according to the result of a risk-benefit analysis is likely to discourage innovation that would take place otherwise.

To avoid this drawback of risk-benefit analysis, it is important to have a programme to reduce overall environmental risks in which we can compare alternative policies. If the direction of reducing risks is taken for granted and risk-benefit analysis is used only for establishing the order of priority among alternative regulations, then the ability of the regulations as a whole to provide incentive to undertake technological innovations in risk reduction will not be undermined.

It is not cost-effectiveness analysis but cost-benefit analysis that is appropriate for evaluating regulations or public projects individually. Cost-benefit analysis can determine whether a particular regulation should be implemented or not independently from other regulations, but cost-effectiveness analysis can originally only do relative comparison among regulations. This has been considered a drawback of cost-effectiveness analysis, but it may be an advantage in terms of the 'dynamic' efficiency of regulations.

Risk-benefit analysis, therefore, should take the form of comparison among alternatives rather than one-by-one analysis for individual regulations. For instance, for reducing the release of dioxins into the atmosphere from incineration of solid wastes, about 2.6 billion yen per year has been spent since 1996 by publicly owned incineration plants in order to comply with the new emission standards in force from 1998 to 2002, which has resulted in a reduction of the emission by about 700g-TEQ/year of dioxins. After 2002, a set of more stringent standards not only for emissions but also for design and operation will be enacted whereas an incentive policy using subsidies to encourage regionalization of waste treatment is being implemented. Comparisons have not been made of the alternative policies for publicly owned incineration plants, or of the policies for reducing emissions from various sources, or of the policies relating to emission control and intake control.

Risk-benefit analysis should be used for such comparisons. Of course, it is desirable that it is used for selecting among the policies covering a broader area, but to use risk-benefit analysis for selecting just among the alternative policies to reduce dioxin risks is meaningful as long as it gives rise to reallocation of resources and enhances efficiency of the policies.

4. Acknowledgments

This work has been supported by CREST of the Japan Science and Technology Corporation.

5. References

Bremer, K, Humphries, C. J., Mishler, B. D. and Churchill, S. P.: *Taxon*, **36**, 339-349 (1987).

Chase, M. W. et al.: *Annals of the Missouri Botanical Garden*, **80**, 528-580 (1995).

Cronquist, A.: *An Integrated System of Classification of Flowering Plants*, Columbia U. Pr.(1981).

Dahlgren, R. M. T., Cliford, H. T. and Yeo, P. F.: *The Families of the Monocotyledons: Structure, Evolution, and Taxonomy*, Springer-Verlag (1985).

Faith, D. P.: *Biodiversity: Measurement and Estimation*, ed. by Hawksworth, 45-58 (1995).

Gamo,M., Oka,T. and Nakanishi,J.:*Regulatory Toxicology and Pharmacology.*,**21**,151-157 (1995).

Gamo, M., Oka, T. and Nakanishi, J.: *Environmental Science*, **9**, 1-8 (1996) in Japanese.

Hasebe, M. et al.: *American Fern Journal*, **85**, 134-181 (1995).

Kajihara, H., Ishizuka, S., Masunaga, S. and Nakanishi, J.: SRA Annual Meeting (1998).

Kramer, K. U. and Green, P. S. eds.: *The Families and Genera of Vascular Plants: Volume I Pteridophytes and Gymnosperms*, Springer-Verlag (1990).

Manhart, J. R.: *American Fern Journal*, **85**, 182-192 (1995).

Nakanishi, J., Oka, T. and Gamo, M.: *Environ. Engg. and Policy*, **1**, 3-9 (1998).

Oka, T., Gamo, M. and Nakanishi, J.: *Japanese Journal of Risk Analysis*, **8**, 174-186 (1997).

Weitzman, M. L.: *Quarterly Journal of Economics*, **107**, 363-406 (1992).

Williams, P. H., Humphries, C. J. and Vane-Wright, R. I.: *Aust. Syst. Bot.*, **4**, 665-679 (1991).

Williams, P. H., Gaston, K. J. and Humphries, C. J.: *Biodiversity Letters.*, **2**, 67-78 (1994).