Risk/Benefit Analysis and Risk Management

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Abstract

Results of cost-effectiveness analysis for environmental chemical regulations are summarized. Their implications are discussed from the perspectives of cost-effectiveness analysis and cost-benefit analysis. The reasons why more expensive policies are permissible for environmental regulation than for other health-related public policies are discussed, and a cost-effectiveness analysis concerning ecological risk is presented. Methods of integrating health risks and ecological risks in cost-effectiveness analysis are suggested.

1. Results of Cost-Effectiveness Analysis

The aim of this part of our research project is to provide a framework for assisting decision-makings in chemical risk management, focusing on the quantities of risks reduced as well as the costs for the risk-reductions. Since the cost for risk-reduction is the benefit that would be lost due to the risk-reduction or the benefit that is enjoyed in exchange for the risk, the framework is called 'risk-benefit analysis'.

Risk-benefit analysis is a form of 'cost-effectiveness analysis (CEA)'. The goal of CEA is to obtain a ratio of the cost for risk-reduction to the quantity of the risk reduced. Table 1 shows the ratios for several human health risk control policies in terms of cost per life-year saved (CPLYS) based on our researches.

2. Implications of the CPLYS Results

What are the implications of the CPLYS results? First, these values can be used for assessing priorities among risk control policies. The priority-setting is based on the idea that total costs for reducing a given amount of risks can be minimized or total quantity of risk-reductions with a given

Table 1. Cost per life-year saved in chemical control policies

Policy	CPLYS (million yen)	Source
Prohibition of chlordane	45	Oka et al. 1997
Mercury regulation in caustic soda production	570	Nakanishi et al. 1998
Mercury removal from dry batteries	22	Nakanishi 1995
Regulation of benzene in gasoline	230	Kajihara et al. 1999
Dioxin control (emergency counter measures)	9.5	Kishimoto et al 2001
Dioxin control (long-term counter-measures)	125	Kishimoto et al 2001
Regulation of NO_x for automobiles	86	Oka 1996
Collection of CFCs from waste refrigerators	150	Original

costs can be maximized by prioritizing the policy according to CPLYS. For example, if the control of particulate matters from diesel-engined vehicles costs not more than 44 million yen per life-year saved (Nakanishi 2000), there is no reason why not to introduce this control when dioxin emissions are to be reduced by the 'long-term countermeasures'. Such a utilization of CEA would not be very controversial.

The second use of the CPLYS results is the comparison of the values with those in other health-related public policies. Kishimoto (1999) collected the values of CPLYS not only for environmental chemical controls but also safety controls and health care projects. The result is summarized in Table 2.

Table 2. CPLYS of lifesaving interventions in Japan

Category	No. of cases	CPLYS (million yen)	
		Mean	Median
Safety control	5	44	30
Environmental control	16	1100	320
Health care (prevention)	37	3.9	2.4
Health care (treatment)	27	2.7	0.76
All	85	200	3.3

Kishimoto (1999).

An implication of this result is that CPLYS is much higher in environmental control than in other areas and that it would save more lives to withdraw resources from environmental controls and to reallocate them to safety controls or health care programmes. However, such an extension of CEA may not be unanimously agreed upon. One may argue that environmental regulation should not be compared with safety control or health care programmes, and that higher cost for environmental regulation is justifiable.

What is the reason why more resources should be spent per unit of risk-reduction on avoiding health risks from environmental pollution than on other sources of risk? Difference in the characteristics of risk may be cited as a justification. It is often alleged that involuntary risks should be treated with greater weights than voluntary risks. Certainly many programmes may be included under the categories of 'health care' and 'safety' which mainly targeted at reduction of voluntary risks. Provided that management of voluntary risks should be left to individuals' voluntary decision, voluntary risk-reductions should not be included in comparing the cost-effectiveness of risk control programmes as public policies. However, even when excluding voluntary risk controls, the CPLYS are still higher in environmental regulations than other areas.

Another reason for the high CPLYS of the environmental regulation is the indirectness in the health effects of the environmental regulation. The direct objective of the environmental regulation is the improvement in the quality of the environment, and the improvement of human health is an indirect objective, while the direct target of safety control and health care is the improvement of human health. Therefore, the high cost-effectiveness of the latter policies is reasonable. The indirectness of the environmental regulation implies that it may have other effects than the human health effect. The ecological effects may be the most important one. This point will be discussed below.

The third use of the CPLYS results is to combine them with the 'benefit per life-year saved' and to perform cost-benefit analyses (CBA). CEA can assess priorities among risk control policies, but it does not determine whether the society should perform a particular regulation or a project or not, nor how far the society should proceed to reduce risks. CBA could answer these questions. In the application of CBA to human health risk management, a programme is regarded as economically efficient when people's willingness to pay (WTP) for reducing a unit of risk, i.e. for extending a life-year, is greater than the corresponding cost, namely CPLYS.

WTP for reducing a unit of risk has been estimated mainly in the US and the UK (Fisher et al. 1989). However, we have not attempted to estimate WTP. The reason for not doing so is as follows. WTP was expected to be too low to justify the existing risk control policies. When a unit of risk is taken as a statistical life, the WTP for reducing it is called the value of a statistical life (VSL), whose estimates ranges from US$1.6 million to US$8.5 million (in 1986 dollar) (Fisher et al. 1989). VSL in the CBA of Clean Air Act in the US was US$4.8 million (USEPA 1997). In the UK, the VSL of £0.9 million is used in evaluations of road safety (DETR 1998). From these instances, it would be safe to regard VSL is not larger than 1 billion yen also in Japan. It means that value of a life-year is unlikely to exceed 100 million yen, provided that a statistical life is equivalent to about 40 life-years (Oka 1999), whereas CPLYS often exceeds 100 million yen as shown in Table 1.

It is often argued that WTP for reducing involuntary risks is greater than that for reducing voluntary risks, and the VSL estimates are too low to evaluate the environmental risk-reduction because those estimates are usually derived from the situations where people revealed or stated their willingness to pay for reducing risks voluntarily. In fact, Jones-Lee and Loomes (1995) argued that they observed people are willing to pay greater amount for reducing involuntary risks than for reducing voluntary risks. However, WTP is only observed for a change that individuals can choose voluntarily, and cost-benefit analysis is, from the outset, a tool for determining economic efficiency for public programmes concerning the supply of public goods or the reduction of public bads, by using data on people's willingness to pay for such changes when they can choose them voluntarily. A public good is consumed collectively and its private supply is not imaginable, but the benefit of a public good has been evaluated on the basis of how much individuals are willing to pay for purchasing the good or services of the good voluntarily. This should also be the case for a public bad such as health risk. Individuals cannot often avoid suffering a public bad or cannot often be excluded from the removal or reduction of the public bad, but its evaluation must be based on their voluntary willingness to pay for removing or reducing the bad on the assumption that they can choose whether or not to remove it or to what extent to reduce it. This discrepancy between the involuntariness in the supply of goods or bads to be evaluated and the voluntariness intrinsic to WTP concept is inevitable in cost-benefit analysis of public policies. It is impossible to resolve it by coining a fallacious notion of WTP for reducing involuntary risks, and VSL should not be adjusted quantitatively to cope with the high CPLYS and the low VSL.

3. High CPLYS and Ecological Risks

The most persuasive reason why higher CPLYS for environmental regulations can be accepted than that for other health-related policies is that environmental regulations have other effects than human health effects, as mentioned above. Improvement in the quality of the environment would reduce risks from harmful effects of pollutants on animals, plants and other living things, i.e., 'ecological risks' should also be taken into account for environmental programmes.

That is why this project has attempted to develop methods for ecological risk assessment.

We have developed a method for assessing ecological risks in terms of expected loss of biodiversity (ELB) that can be applied to habitat losses (Oka et al. 2000). ELB is defined as the weighted sum of the increments in the probabilities of extinction of the species living in a habitat due to its loss. The weighting for a particular species is calculated according to the length of the branch on the phylogenetic tree that will be lost if the species becomes extinct. The length of the branch on the phylogenetic tree is regarded as reflecting the history of speciation and the extent of contribution of the species to the taxonomic diversity of the world of living things.

We applied this measure of ecological risk to the case of the development of Nakaikemi Wetland (Oka et al.2000). The increments in the probabilities of extinction of the plant species in Nakaikemi were calculated by a simulation used for making the Red List for vascular plants in Japan. There are 15 threatened species in Nakaikemi, and the impact of the loss of Nakaikemi on their nationwide extinction probabilities ranges from 1.9×10^{-6} to 3.6×10^{-4} per year as shown in Table 3. These values were calculated as the reciprocal of the decrease in the time for extinction. The contribution of a species to the global biodiversity was calculated as the expected value of the reciprocal or the number of nodes in the phylogenetic tree for the vascular plants between the terminal node for the species and the root node of the whole tree. It is expressed in terms of the length of time. The values range from millions to tens of millions years as shown in Table3. ELB is the sum of these values weighted by the extinction probabilities. The resulted value of ELB for the entire area of Nakaikemi is 9,200 years.

This result was combined with the economic costs for conservation of the wetland to produce a value for the indicator of the 'cost per unit of biodiversity saved'. Depending on the scenario, the value is 13,000 yen/year-ELB or 110,000 to 420,000 yen/year-ELB.

Table 3: ELB of the plant species in Nakaikemi

No	Species	Increment in the probability of extinction per year due to the loss of Nakaikemi ΔP_i	Contribution of species to the global biodiversity Y_i (year)	ELB $\Delta P_i Y_i$ (year)
1	Isoetes japonica	8.9×10^{-6}	29328994	261
2	Marsilea quadrifolia	6.4×10^{-5}	19514737	1254
3	Salvinia natans	5.7×10^{-6}	28278915	161
4	Azolla japonica	4.1×10^{-5}	30881499	1267
5	Persicaria foliosa	4.3×10^{-5}	7101914	303
6	Trapa incisa	1.4×10^{-4}	12341354	1755
7	Eusteralis yatabeana	3.6×10^{-4}	3406671	1214
8	Prenanthes tanakae	5.1×10^{-5}	2124976	108
9	Sagittaria	4.4×10^{-6}	11085960	49
10	Najas japonica	1.5×10^{-4}	11618822	1782
11	Monochoria	6.7×10^{-5}	12010897	802
12	Iris laevigata	6.3×10^{-6}	6297533	40
13	Sparganium erectum	1.9×10^{-6}	12588373	24
14	Sparganium japonica	1.1×10^{-5}	12588373	139
15	Habenaria sagittifera	1.5×10^{-6}	2226034	3
				9163

4. Integration of Human Health and Ecological Risks in CEA

Once the ecological risk is quantitatively assessed for chemical pollution, cost-effectiveness analysis for environmental risk management can incorporate both human health risks and ecological risks. However, it remains a question of how to incorporate them.

One way is to construct an index of the total risk including human health and ecological risks. However, it requires weighting of human health risks and ecological risks. It must be determined how many years of human life, for example, is equivalent to one year of ELB. After that, one can obtain as an index of cost-effectiveness the ratio of the cost to the total risk, i.e. the ratio

$$\frac{Cost}{W_h \Delta(LLE) + W_e \Delta(ELB)},$$

where W_h is the weight for human health risk and W_e is the weight for ecological risk. This method is straightforward, but there is no agreed way for determining the weights.

A second way is to attach monetary value to human health risk and to subtract it from the cost side, and to obtain the ratio

$$\frac{Cost - V\Delta(LLE)}{\Delta(ELB)},$$

where V represents the monetary value of a life-year. For constructing this index of cost-effectiveness, to estimate the value of a life-year is necessary.

5. Other Issues on CEA for Environmental Risks

1) Index of Human Health Risk

We have used LLE for the index of human health risk. It reflect the idea that the endpoint in human health risk assessment should be the death. LLE, however, takes nonfatal health effects into account by capturing them as increases in the probability of death caused by the deterioration in the health state. An alternative, more popular, way of expressing nonfatal health effects is to use 'quality of life (QOL)' weights for health states and to adopt loss of 'quality-adjusted life-year (QALY)' in stead of LLE as a measure of risk. Still another way may be to attach monetary values to nonfatal health effects and to subtract them from the cost side. The question of which is the best way remains unresolved and requires further research.

2) Application of CBA and CEA in a Changing World

It is often pointed out that CBA measures only static efficiency. Decision-making based on CBA would certainly brings about an efficient allocation of resources under the present knowledge about technology and people's preferences. If, however, technology changes and the cost for risk-reduction is lowered, a regulation that was regarded as inefficient may become efficient. Furthermore, introduction of a regulation may stimulate the progress of technology. If this is the case, introduction of a strict regulation which seems inefficient in a short-run may be justified from the point of 'dynamic efficiency'.

However, cost for developing new technology must be taken into consideration. Let us suppose that a regulation incurs a cost of C and produces a benefit of B, but that when the regulation is introduced, investment in research and development, I, is expected to reduce the cost by ΔC. When $B<C$, the regulation is regarded as inefficient statically, but if $\Delta C-I>C-B$, it becomes dynamically efficient. The last inequality is equivalent to

$$\frac{B}{C}+\frac{\Delta C}{C}>\frac{I}{C}+1.$$

This means that the sum of the ratio of the benefit to the cost plus the rate of the cost reduction must be greater than the ratio of the research and development investment to the cost plus one. If the ratio of the research and development investment to the cost is known empirically, this inequality may provide a guidance to the judgement on the expected dynamic efficiency of regulations.

On the other hand, when an analyst confines himself/herself within CEA and do just a priority-setting, it matters whether reversals in the order of the cost/risk ratio are expected to occur before and after the regulation. If there is no evidence of such reversals, CEA is freer from dynamic considerations than CBA.

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