Marginal Reduction Cost Approach to Eco-Efficiency

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1 Introduction

Public and private purchasers with preference for environmentally friendly products are often faced with trade-offs between the environment and the economy. They have to decide whether to buy or not to buy a product that is more environmentally friendly but more expensive. Life Cycle Assessment (LCA) is an analytical instrument that can be used to help us with this decision.

However, we have several obstacles to overcome before LCA becomes able to serve this purpose. First, LCA usually quantifies several parameters of environmental burden, such as CO_2, NO_X, COD, heavy metals, toxic chemicals, etc., over the lifetime of a particular product or an activity. In order for LCA to help us to decide whether to adopt environmentally friendly products or activities, however, those individual parameters have to be aggregated into a single index of environmental impact. Unless we have such a single index, we cannot determine the superiority of a product with less CO_2 emission but with more NO_X emission. Second, the value in physical terms has to be converted into monetary terms in order to be compared with the higher costs of such products.

The marginal reduction cost approach to eco-efficiency is a method that can contribute to overcoming these obstacles.

2 Cost-Benefit Approach

A standard approach provided by the environmental economics is the cost-benefit approach (or cost-benefit analysis: CBA). In the cost-benefit approach, LCA parameters are converted into values for adverse environmental effects such as the adverse effects on human health, on amenity, and on the natural environment. It is assumed that people are willing to pay some amount of money to reduce those adverse effects. The amount is called 'willingness to pay (WTP)', and is regarded as the benefit in monetary terms of reducing those adverse effects. The sum of WTP for the reduction of all the adverse effects through the reduction of the pollutants by introducing the product in question is regarded as the environmental benefit of the product.

It is not until we obtain this monetary benefit that we can judge the efficiency of introducing environmentally friendly products in the sense that their benefits exceeds the costs.

The cost-benefit approach is, however, beset with a number of difficulties. The difficulties are classified into two categories: difficulties in monetization and difficulties in the link between the pollutants and its adverse effects.

Most of the environmental adverse effects are nonmarket 'bads'. The activities reducing those bads are usually not traded at the market, and do not have market prices. We cannot, therefore, use market prices, which are observable data having close relation to individuals' WTP. In many cases, the only method to estimate people's WTP is by using hypothetical markets. This method is called 'contingent valuation method (CVM)', 'questionnaire method' or 'stated preference method'. In this method, researchers elicit, by questionnaire, respondents' WTP to get hypothetical improvement in the environmental quality in

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question. The hypothetical nature of this method, however, undermines the reliability of its estimation of WTP. Respondents are often not so serious to state their true WTP, or they may not know their true WTP for such unfamiliar goods. Furthermore, they may not have any WTP at all, or they may not have any stable preference among all the goods including unfamiliar environmental goods that they have never bought before.

In some cases, however, market data, though indirectly, are available to estimate people's WTP for improving environmental quality. For example, higher health or safety risks in workplaces may be associated with higher wages. The method to estimate WTP by using this relation is called wage-risk approach. The WTP obtained by this method is often applied to evaluate the improvement in the environmental quality affecting human health, but there is great controversy about the legitimacy of applying the WTP derived from people's behaviour in the labour market to the evaluation of the change in environmental qualities. Involuntariness in the health risks due to pollution has attracted researchers' attention.

WTP obtained from the data in workplaces reflects people's will where they can control the level of their safety and health. Workers choosing a workplace with higher risk in exchange for higher income are thought to do so voluntarily. In contrast, people usually have no control in suffering from environmentally generated health defects.

If WTP derived from people's voluntary behaviour cannot be applied to the evaluation of the supply of public goods over which people usually have no control, the applicability CBA will become very small. That is because CBA is a tool for evaluating public projects or regulatory programmes, which are usually intended to provide public goods or to reduce public bads, and because involuntariness is intrinsic to public goods or bads, whereas WTP exists, in principle, in people's voluntary choice.

The fact that improvement of an environmental quality is a supply of a public good brings a further problem when WTP is elicited by the hypothetical market method. Questionnaire usually includes a scenario providing respondents with a hypothetical situation in which they are buying an improved environmental quality. Since the good to be bought is a public good, it is unrealistic to provide a scenario where a respondent can obtain the good by paying a certain amount of money by himself. In the scenario, therefore, it is usually assumed that the respondent can only partly contribute to the provision of the good by donating some amount to a fund or paying some amount of tax, which is just a small part of the total money necessary for supplying the public good. The amount of money thus elicited is the respondent's willingness to contribute to the fund, not a WTP for obtaining the good in question.

In addition to these problems in the stage of monetization, there are difficulties in the preceding stage of relating the reduction in the emission of pollutants to the improvement in environmental qualities which concern human beings. In order to relate the reduction in the emission of CO_2 by one tonne, for example, to the adverse effects for human beings, we have to identify how much the reduction contributes the reduction in the global average temperature, how much it affects regional climate, and how much, in turn, the change in regional climate affects human lives. Since this chain of effects have not yet been made clear, we cannot assign a WTP for the reduction in the emission of CO_2.

3 Cost-Effectiveness Approach

Cost-effectiveness approach (or 'cost-effectiveness analysis: CEA') is an approach in which only the costs are expressed in monetary terms and the environmental impacts are expressed not in monetary but in physical terms. In CEA, we cannot compare costs and benefits on a same scale. We have only a ratio of the costs to the effectiveness of a programme. This ratio can be used to assess relative efficiency among alternative activities whose environmental effects are expressed on a single physical scale.

CEA has been successful in some domains of environmental policy appraisal. For example, in the management of hazardous chemical pollutants, the environmental effects of policy programmes, such as regulations of certain chemicals, are assessed by a single index, such as 'reduced cancer case', 'avoided number of death', or 'life-year saved'. We have obtained the ratios of costs to life-years saved for several regulation programmes, shown in Table 1.

Table 1 shows the ratios for several chemical risk control policies based on our research.

However, this success is confined to the domain of hazardous chemical control. The reason of the success is that those policy programmes have a common scale to measure their effects, namely 'life-year saved'. The more confined the area of the activities under consideration, the easier to find a common

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

Policy programme	CPLYS (¥ million)	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 countermeasures)	9.5	Kishimoto et al., 2001
Dioxin control (long-term countermeasures)	125	Kishimoto et al., 2001
Regulation of NOx for automobiles	86	Oka, 1996

scale of policy effects, and the more successful the CEA. If we need to assess the relative efficiency among alternative programmes to reduce dioxin emission, the quantity of the reduction would be sufficient as a measure of effectiveness. If, however, we need to compare the regulation of dioxin emission and the regulation of benzen emission, then broader measure would be necessary such as the number of cancer cases or deaths avoided. Further, when we need to compare the regulatory programmes of benzen and mercury, then much more broader measure such as life-year saved would be necessary. When we need to assess the regulation of alternative pesticides, taking into account both the risks to human health and the risks to ecosystems, then a broader measure including both aspects would need to be used, but it would be difficult.

4 Marginal Reduction Cost Approach as a Cost-Effectiveness Analysis

In the case of introduction of a product or an activity, which is under investigation in LCA, we have to assess many pollutant parameters that have different environmental impacts. It is diffilult to assess on the same scale the impact of NO_X on human health and the impact of phosphorus and nitrogen, which cause eutrophication in lakes or inland seas, on ecosystems, and it is further more difficult to assess on the same scale those impacts and the multiple impacts of CO_2 emission on the global climate.

Therefore, to apply CEA based on a single scale of effectiveness to the assessment of a particular product or an activity would not be hopeful.

Our 'marginal reduction cost approach (MRC approach)' avoids this hard task of creating a common measure of effectiveness over all the environmental impacts, while achieving the objective of CEA, i.e. to assess the relative efficiency of particular activities. MRC allows the LCA parameters remain disaggregated. Instead of aggregation, it attaches to each pollutant reduced in the activity in question a value of marginal reduction cost of the other activities carried out somewhere in a society for reducing that pollutant.

Marginal reduction cost means the cost for reducing additional unit of a pollutant emitted into the environment in a society. In our research, we adopted as the marginal reduction cost the unit cost, i.e., the cost per kilogram of pollutant reduction, of the activity which has the highest unit cost among all the activities to reduce the same pollutant over Japan.

The marginal reduction cost is a cost determined by people's and industries' decision intending to comply with governments' regulations, to earn good reputation or to obey their own moral sense, which may reflect public decisions or many people's opinion about to what extent the society should pay cost for reducing pollutants.

In MRC, the marginal reduction cost times the quantity of the corresponding pollutant reduced by the activity in question is added up over all the pollutants relevant to the activity. That is, in MRC, we aggregate LCA parameters with the weighting of marginal reduction costs. Let us call this total sum 'avoidable reduction cost (ARC)' of the activity, in the sense that the sociey can avoid, if they wish, the efforts elsewhere incurring that amount of cost by introducing the activity in question without increasing the emissions of any pollutants.

ARC is compared with the cost for the activity or the introduction of the product in question. If the latter cost is smaller than the former cost, namely ARC, then the activity or the product is regarded as a relatively efficient means compared with the other means in a society for reducing the relevant pollutants.

5 Application of Marginal Reduction Cost Approach

Table 2 shows the quantities of pollutants emitted from a conventional pump and from an environmentally friendly pump over their lifetime from their production to their retirement. The environmentally friendly pump emits less amount for all pollutants except for dioxins.

Table 2: LCA for alternative pumps

Pollutant	Emission (kg)		
	Conventional pump	Environmentally friendly pump	Reduction
CO_2	17900	13100	4770
NO_X	41.7	30.7	11.1
SO_2	54.9	41.6	13.4
TOD	4.47	3.05	1.42
SPM	17.0	6.29	10.7
DXN	6.30×10^{-9}	1.25×10^{-8}	-6.20×10^{-9}
HM	3.15	1.35	1.80

1) Emission is the emission during the lifetime of the pump including its production, operation and disposal. It is assumed the pump is used for 15 years from its production and then disposed of, and the discount rate of 5% is applied to calculate present values of emission.

2) TOD represents the load of pollutants causing eutrophication in closed water areas taking into account endogenous production, and is defined as TOD=2 COD+(21 TN + 310 TP)/2.

We have obtained the values of marginal reduction cost for the pollutants relevant to this case, which are shown in the first column of Table 3. The marginal reduction costs times the pollution reduction due to the introduction of the environmentally friendly pump are shown in the second column. The sum ARC, is 170 thousand yen (1 euro ≈140 yen, 1 US dollar ≈ 110 yen).

Table 3: Marginal reduction cost and avoidable environmental cost

Pollutant	Marginal reduction cost (yen/kg)	Marginal cost times emission reduction (yen)
CO_2	7.0	33400
NO_X	2500	27700
SO_2	43	575
TOD	1600	2200
SPM	6700	72000
DXN	1.9×10^{10}	-118
HM	20000	36000
ARC		170000

The price of the environmentally friendly pump is 500 thousand yen, while the price of the conventional pump is 320 thousand yen. However, when using the environmentally friendly pump, less electricity is needed. The electricity consumption during 15 years of its use is 34300 kWh, while that of the conventional pump is 46500 kWh. Under the electricity price of 20 yen/kWh and under the discount rate of 5%, this energy saving brings about the cost saving of 176 thousand yen. Offsetting this cost saving, the introduction of the environmentally friendly pump incurs additional cost of 4000 yen.

This additonal cost of 4000 yen for reducing 7 kg of CO_2, 2500 kg of NO_X, etc. is much smaller than the ARC of 170 thousand yen, which our society is now spending to reduce the same amounts of pollutants. In this sense, the introduction of the pump is regarded as relatively efficient.

6 Discussion

6.1 What is Marginal Reduction Cost?

Marginal reduction cost is the highest unit cost of pollutant reduction in our society. In our research, we adopted as the value of marginal reduction cost of, for example, NO_X, the unit cost in terms of the loss to the user of diesel vehicles caused by the compulsory shortening of the amortization period to meet the standard imposed by the Automobile NO_X Law. We regarded the average unit cost of this loss over all the regions where this regulation is applied as the marginal reduction cost. Strictly speaking, there must be many users whose costs are higher than the average, and the highest value should be much greater than the average. However, it is difficult to find the actually highest cost, and the highest cost among exceptionally few users is not so important. Therefore, we selected activities which brings considerably large part of the emission reduction over the society, classified those activities into five categories, and regarded the average unit cost of a category with the highest value as the marginal reduction cost.

6.2 Environmental Benefit?

The marginal reduction cost is not the benefit of pollution reduction. The benefit should be based on people's WTP for reduction. The marginal reduction cost depends on regulations or other public policies, or on producers' condition, and has no relation to peoples' WTP for reduction.

However, if by introducing the environmentally friendly product in question, some activity with the highest unit cost which has previously been taken for pollution reduction is actually abandoned, then the cost saving from the abandonment can be regarded as the benefit which is actually enjoyed by the society.

Since it is quite unlikely that will happen, the marginal reduction cost cannot be regarded as the benefit. Therefore, MRC should be regarded as a cost-effectiveness analysis rather than a cost-benefit analysis.

6.3 Dependence on Regulation

Marginal reduction cost is determined by the costs actually expended by some agents in a society. It is, therefore, dependent on the social regulations or public opinion. When some regulations are revised, the marginal reduction cost can also be changed.

Let us suppose product A is assessed by MRC and regarded as an inefficient means because its cost is greater than its ARC, but no cheaper method is available for reducing the pollutants that can be reduced by product A, and the majority of people want to reduce the pollutants even at more cost. Under these circumstances, product A may be accepted and even become prevalent among the public. At this stage, the unit costs of product A for some pollutants will become the marginal reduction costs that should be used in MRC for other products.

As is evident from this example, MRC is a method applicable to particular products or activities. It cannot be used for assessing social decision on introducing a new regulation for certain pollutants, because such a regulation may change the value of marginal reduction cost.

This may be a shortcoming of this method, but at the cost of this inapplicability to social regulations, MRC has obtained substantial effectiveness in assessing individual products and activities.

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