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Cost-Benefit Analysis of the Clean-Up of Hazardous Waste Sites

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1. Introduction

Hazardous waste, defined as any material that poses a substantial threat to human health, can potentially contaminate all the environmental media: atmosphere, groundwater, surface waters and soil, and through these media can be harmful or even fatal for human health. The prolonged exposure to toxic pollutants such as benzene derivatives, dioxins and trichlorophenol has been associated with acute health effects such as narcosis, skin irritation, or respiratory diseases such as asthma and allergies. Hazardous waste exposure has also been associated with chronic health effects such as leukaemia, liver tumour, lymphomas and, in the case of methylene chloride, premature mortality.

Since the case of Love Canal, New York State, in 1980 an increasing number of cases of hazardous waste mismanagement have been reported. Studies suggest that children are the most vulnerable victims of toxic pollutants. Exposure to compounds increases the likelihood of miscarriage and birth defects. In the Love Canal, for instance, birth defects were found to be twice as likely to occur among those living near the dump site (Goldman et al.1985). In Canada, a large study conducted by Goldenberg *et al*. (1999), suggested that individuals living close to landfill sites have an increased risk of liver, kidney, pancreas cancers and non-Hodgkin's lymphomas.

Another study conducted by Pukkala (2001) in Finland found that the prevalence of asthma was significantly higher in individuals living near landfill sites.

Lack of resources requires policy makers to prioritise competing alternatives. Despite the potential gains for both environmental and human health, it remains uncertain whether the benefits of interventions to clean-up hazardous sites would outweigh the costs. The analytical tool of cost-benefit analysis provides a powerful and transparent method to evaluate and select risk management strategies. Nevertheless, cost-benefit analysis has rarely been used to assess hazardous waste site cleanup interventions. There are several reasons for this: the effects of hazardous waste exposure are often ignored; there are difficulties indentifying the causal link between waste exposure and health effects; and estimating the value of the potential impacts resulting from cleanup interventions. Costs of cleanup interventions are also subject to great uncertainty because it is difficult to quantify them *a priori*, especially where more than one media has been affected by hazardous pollutants. The aim of this chapter is to provide an overview of the major steps necessary to conduct a cost-benefit analysis of cleanup interventions.

2. Economic evaluations of benefit and cost of hazardous site cleanup

Cost-benefit analysis evaluates the social gain associated with a given intervention by comparing the benefits (any increase in welfare) and the costs (any decrease in human well being). The aim of cost-benefit analysis is to maximize the net social benefits:

Max B(Q)-C(Q)

Cost benefit (CB) analysis is used in environmental regulation to determine acceptable levels of risk. Acceptable risk denotes a level that maximizes the difference between total social cost and total social benefits, or in other words, where the marginal social benefits associated with the risk reduction are equal to the marginal social costs of pollution abatement.

In the case of the cleanup of hazardous waste sites, cost benefit analysis is used both to distinguish between interventions offering higher net benefit (difference between cost and benefits) and to identify priority sites for intervention, as in the case of the US Superfund.

CB analysis involves six steps: quantifying the health outcomes associated with waste exposure before and after regulation (hazardous waste site cleanup); assigning monetary values to the number of cases potentially averted by regulation; quantifying the cost of regulation; accounting for the timing of costs and benefits; and comparing the resulting estimates. The final step of CB analysis is to perform sensitivity analysis to evaluate the effect of parameter uncertainty on the study results.

2.1 Health benefits analysis

Several types of benefits result from hazardous waste cleanup. These are: direct benefits, for example reduction in the number of health effects (e.g. asthma cases, lung cancer, malformations); aesthetic benefits, such as decreases in odour; and indirect benefits, such as productivity increase of real estate properties. This chapter focuses on describing how the direct benefits to human health can be quantified using a damage function approach.

As shown in Figure 1 the damage function approach framework uses three types of data: environmental data to identify the potential hazards/pollutants present in the hazardous waste sites; epidemiological data to identify and quantify the health effects associated with the regulatory intervention and economic data to assign a monetary value to negative health outcomes associated to waste exposure.

The first step involves the estimation of the health effects due to pollutant exposure. The second step evaluates the number of health outcomes that can be averted by site cleanup. And the third step multiplies the estimated number of avoidable health outcomes as a result of the regulatory strategy (number of deaths averted per year) by the economic value per health unit (e.g. value of a statistical life).

2.1.1 Quantifying cleanup health benefits

In the majority of cost benefit analyses conducted to evaluate the effects of an environmental regulatory strategy (e.g. air pollution control intervention) the baseline number of health outcomes attributable to pollution exposure is determined using a dose-response function. This function is "an estimate of risk per unit of exposure to pollutant" (EPA, 2010a). The dose-response functions can have different shapes. They can be linear (any change in the pollutant concentration will produce a corresponding change in the health outcome), non-linear (e.g.it can be a sigmoidal curve that starts with an increasing slope but after reaching a maximum value it levels off) and/or can present a threshold dose. For example a study
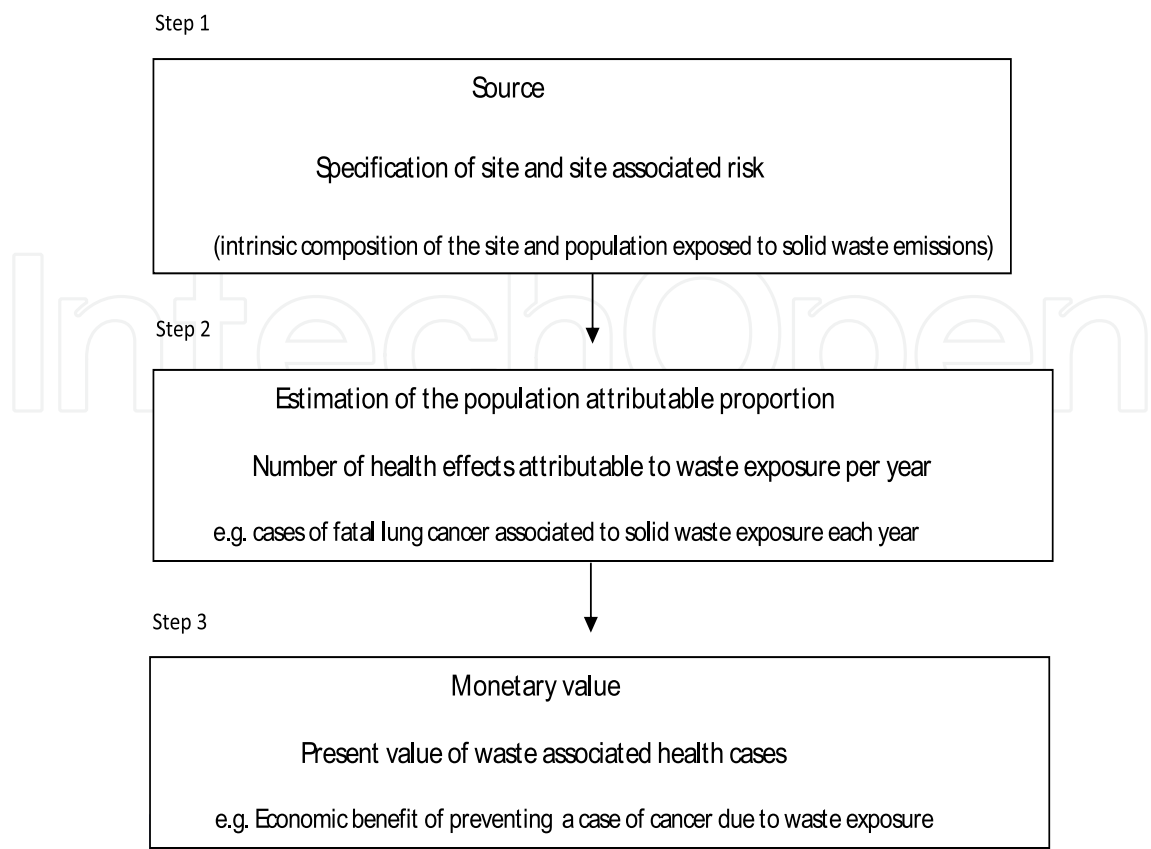
Step 1

Source Specification of site and site associated risk (intrinsic composition of the site and population exposed to solid waste emissions)

Step 2

Estimation of the population attributable proportion Number of health effects attributable to waste exposure per year e.g. cases of fatal lung cancer associated to solid waste exposure each year

Step 3

Monetary value Present value of waste associated health cases e.g. Economic benefit of preventing a case of cancer due to waste exposure

Fig. 1. Damage Function Approach

conducted by Grosse *et al.* (2002) on the relationship between blood lead level and intelligence quotient (IQ) estimates that there is a linear relationship between the blood lead level and the decrease in IQ points (2.57 IQ points for each 10 mg/dL).

Where the effects on health of hazardous waste disposal result from exposure to a single pollutant (e.g. asbestos), the population attributable proportion (PAP), the number of cases that would have not occurred in the absence of pollutant, is estimated using the following formula:

$$PAP = (p - (RR - 1)) / (1 + p * (RR - 1))$$

Where RR is the relative risk of developing the health outcome given pollutant concentration, and p the proportion of the population exposed (e.g. children only).

In the majority of cases, identifying the individual pollutants responsible for the health effects observed in the exposed population is problematic. In the case of landfills or illegal waste disposals, impacts are likely to result from different compounds discharged in the same site. Thus, the PAP is estimated using primary epidemiological data with the following formula:

$$PAP = Observed\ number - Observed\ number / SHR$$

Where SHR is Standardised mortality/hospitalisation ratios (SMR, SHR) that are estimated by dividing the observed cases (e.g. individuals with lung cancer) by the expected cases.

2.1.2 Monetizing health benefits

There are two main methods for placing a monetary value on changes in health: the human capital; and the willingness to pay approach. (Table 1) The human capital approach assumes that the value to society of an individual's life can be measured in terms of future production potential. The willingness to pay (WTP) approach measures how much individuals are willing to pay to decrease the likelihood of a negative outcome.

Basic approach	Main subsets	Evaluation methods
Human capital		Cost of illness
Willingness To Pay	Revealed Preferences	Hedonic wage method
		Averted expenditures
	Stated Preferences	Contingent Evaluation
		Stated Choice

Source: Enhealth 2003

Table 1. Methods for valuing health

Based on the human capital approach, the Cost of Illness (COI) method is a measure of the monetary losses due to a negative health outcome (e.g. case of liver cancer). The COI has several advantages. It is straightforward and objective as it both considers all the direct monetary costs of a given health outcome and it does not depend on personal preferences. However, COI tends to underestimate the true value of a health outcome because it does not include the intangible aspects of being ill such as stress, pain and suffering. Additionally, given that the COI values can be estimated only *a posteriori* it is impossible to elicit with this method the values that individuals assign to future environmental risk reductions.

As a result, the most popular approach adopted in cost-benefit analyses is the WTP approach. The WTP method can be divided in two main categories: revealed and stated preferences. The revealed preference method derives values from observed actions of individuals while the stated preference method elicits valuations by asking individuals how much they are willing to pay to reduce the risk of a given health outcome.

Contingent valuation and the Hedonic Wage method have been widely used to estimate the value of saving a statistical life. As can be seen in Table 2 there is great uncertainty regarding the value to adopt for analysis. Estimates vary dramatically among studies and between regions. The meta-analysis by Mrozec and Taylor (2002), for example, suggested a value of a statistical life (VSL) of $2.4 million (in 1998 US$). While in the meta-analysis conducted by Kochi *et al*. (2006) with an empirical Bayes approach the estimated value of a statistical life was $5.4 million.

However, as Pearce (2000) suggests, not all deaths are valued equally and different evaluation techniques can lead to different and often misleading estimates of the VSL.

For example, it has been shown that adopting the human capital approach for assigning a monetary value to mortality risk would underestimate its cost. Although this method is easier to apply as it relies on a simple calculation of visible and easily quantifiable costs it does not consider individual preferences, and willingness to pay for a risk reduction and individual aversion towards death.

Thus, the approach mainly used to estimate the value of a statistical life in environmental health studies has been the willingness to pay approach and in particular, the hedonic wage, and contingent valuation methods.

Estimate	Source	Method	N of studies
$2.4 million (1998US$,1990 income)	Mrozec and Taylor (2002)	Hedonic Wage	47
$5.4 million (2000US$,1990incomes)	Kochi *et al*. (2006)	Hedonic Wage Contingent Valuation	47 14
$7.6 million (2002US$,1990incomes)	Viscusi and Aldy (2003)	CV,HW	33

Table 2. Value of a statistical life

The hedonic wage (HW) method has been widely used in the last decades to estimate the value of a statistical life. The estimation of the WTP (or WTA) in the HW method involves two stages. First, by controlling for productivity and intrinsic quality of the job, the hedonic wage determines the wages associated with the different types of risk according to the equation below:

$$W_k = \alpha + \beta \, Risk_k + \Sigma \, \lambda_n X_{kn} + \Sigma \, \gamma_m D_m + \varepsilon$$

Where W_k is the wage of the worker k, $Risk_k$ is the risk of death of the worker, n describes human capital and demographic characteristics of the individual X_{kn}, and D_m describes the job characteristics of the individual. The coefficient β (occupational fatal risk) of the risk variable is the additional wage the worker would require to assume an incremental risk of death on the job. Thus according to the hedonic wage method the VSL is estimated as:

$$VSL = \left(\delta \, w \, / \, \delta r \right) * \text{ mean annual wage} * \text{units of fatal risk}.$$

Although this method is widely used in US environmental health studies it presents several disadvantages. The first main disadvantage is that HW does not seem to provide robust and unbiased estimates as it is sensitive to the specification of the wage equation. According to Mrozec and Taylor (2002) studies that control for inter-industry wages have an 85% lower VSL. In addition, it is unclear whether this can be applied only to the occupational risk or whether it can be generalized to the entire spectrum of mortality risks that individuals can face.

Another limitation of the HW method is that it doesn't take account of the characteristics of the person who faces the risk of death nor the risk context. The value assigned to a risk reduction with the HW method is the value for a risk that is immediate, or quite soon in time. While, especially in the context of environmental-related health effects the risk is latent for several years. It is likely that the value that individuals assign to reducing the risk of death in the future is lower than their willingness to pay for a current reduction of risk.

The contingent valuation method on the other hand is a more flexible tool to elicit individuals WTP for fatal risk reduction. According to this method, individuals are asked how much they would be willing to pay for an improvement in their health status or their willingness to accept values for an increased risk. Compared to the COI, this method has the advantage of taking into account the intangible consequences: premature death, the

suffering from an illness. In addition, it can be applied also to individuals who are not in the labour force, and can easily account for different types of risk context.

Contextual factors, such as age, health status, income and cultural differences, have been shown to influence how much individuals are willing to pay for a reduction in the risk of an adverse outcome. Several studies demonstrated that older individuals have a decreased willingness to pay for a reduction in mortality risk often referred as the "senior discount" phenomenon. According to Shepard and Zeckauser (1984) the relationship between VSL and age is not linearly decreasing as might be expected but it is an inverted U-shaped relationship which means that the WTP increases until individuals are 40-45 (as their savings increase as well as their income level) and after that peak it decreases with age because the income level decreases and also because their probability of survival declines. Also, the nature of health outcome (death from cancers) and the time of death have been proven to affect individual WTP. Several studies report that individual WTP to avert a case of immediate death (road traffic accident) is lower than for chronic degenerative disease because of the fear and the pain associated with it. As Pearce (2000) suggests, the WTP to avoid cancer is higher than with other types of diseases because of the dread and pain effects associated with this pathology. According to the European Commission (2001) in cases of cancer related mortality VSL should be inflated by 50% to account for the "Cancer premium".

2.2 Cost analysis

Once the potential benefits arising from remediation have been established, it is necessary to quantify the cost of the cleanup, to decide both the stringency of cleanup standards and who should pay for remediation.

It is difficult to evaluate *a priori* the effectiveness of a given remediation strategy and the cessation lag, the time necessary to observe the improvement in health condition of the population exposed (e.g.decrease in the number of malformations).

In general, remediation expenditures can be divided into three main categories: transaction costs borne by agencies (for example EPA in the US superfund) and private parties/polluters (e.g. oil companies); removal actions and long term remediation costs.

Long term remediation cost constitutes the bulk of the overall cost and is highly dependent on the degree of permanence attainable with the cleanup intervention and on the size of the area to reclaim. According to Gupta *et al*.(1998) in the US, it has been estimated that the average cleanup cost is $27 million per site. However, the cost varies according to the type of media that have been contaminated and to the concentration of compounds in the media.

The choice of the technology is also very important. It determines the permanence of the clean-up intervention. In the case of contaminated ground water, the choice of the technology is restricted. The typical method is "to pump and treat" the contaminated water. Following treatment, the water is either released into the aquifer again or released in a river or stream. In the case of contaminated soil remediation there are several alterative options. The first decision is whether to cap the site. Capping soil is the least permanent option (depends on the shelf life of the cap) and has an average cost of $79 per cubic yard (1996 values: Gupta *et al*, 1998). A more permanent option consists of treating the soil *in situ* (costs $231 per cubic yard) (Gupta et al 1998). The third and most permanent option is excavation. In the case of excavation, the removed soil can be transferred to another landfill site or can be further treated and the organic element incinerated. Excavation with offsite treatment is the most expensive option with costs per cubic yard $1,428 (Gupta et al 1998).

2.3 Time adjustment for environmental benefits and costs

The cost and the benefit of a hazardous waste site cleanup, especially in the case of permanent cleanup, materialise over lengthy periods. Thus, discounting plays a crucial role in the estimation of the value of future costs and benefits. Where different types of interventions are compared, discounting future costs and benefits to present values renders them more easily comparable. Discounting implies that the further in the future the benefits and the costs occur, the lower the weight that should be attached to them.

Thus, the general formula of discounting is the following (Pearce *et al.* 2006):

$$W_t = \frac{1}{(1+s)^t}$$

Where w_t is the discount factor for time t and s is the discount rate.

Thus, the conversion of future benefits to a present value can be estimated with the following formula:

$$Present\,Value = \sum Future\,Value_t \times w_t$$

Where economists use discounting to adjust the value of costs and benefits occurring in the future, the standard approach is to assume a constant discount rate common to both costs and benefits. For example, since 1992 the US discount rate suggested as base case for cost-benefit analyses was a fixed at 7% for both cost and benefit estimates. A 3% discount rate was also suggested for sensitivity analysis. The European Commission (2001) recommends for environmental cost benefit analyses the use of a discount rate of 4% and to perform sensitivity analyses using a discount rate of 2 and 4%. However, there has been extensive discussion of whether the discount rate for health benefits should be lower than that applied to monetary costs. Also, where the effects under consideration are long-lived the case for discount rates declining over time has been made.

Mainly due to the lack of empirical studies, there is uncertainty regarding the discount rate to be adopted in the economic evaluation of toxic waste cleanup interventions. A recent study conducted by Alberini *et al*. (2007) in four Italian cities with significant toxic waste problems applied a contingent choice methodology and evaluated that individuals discount future risk with a 7% rate. Recent studies also suggest that the discount rate might not be fixed and that s should be varying with t. According to Viscusi and Hubert (2006) the discount rate shown for improvements in environmental quality does not follow the standard discounted utility model but its pattern is consistent with the hyperbolic model.

Time lag between the cleanup policy and its related benefits is also an important issue. The annual number of health outcomes (for example number of asthma cases) observable in a given area increases after the creation of a waste site which is producing toxic emissions. After a latency period, which denotes the lag between emissions and onset of the negative health effects, the number of health effects will increase at either a proportional or non-proportional rate. Eventually, if both the emission dose and the population exposed remain constant over the years, the incremental number of health outcomes attributable to pollution exposure is likely to remain the same. When a cleanup policy is implemented, there are no immediate reductions in the number of health outcomes. This is referred to as the "cessation lag". Following the cessation lag, there will be a gradual (proportional/non proportional) decline in the effects of the reduced emission on health up to the point where the number of health outcomes is the same as observed before the creation of the waste site.

The formula used to account for both discounting and latency of benefits is the following:

$$Present\ value\ of\ Benefits = \lambda * X_a * 1 / (1+d)^l * (1 - 1 / (1+d)^t) / d$$

Where: X_a is the number of health endpoints averted by the cleanup, t is the number of years over which the benefits accrue, and d is the discount rate. λ is the WTP for the health outcome a and latency period l, which is the time occurring between the reduction of the exposure and the improvement in the health of the population.

2.4 Cost-benefit evaluation

The main condition for the adoption of a clean-up intervention is that the present value of the benefit exceeds the present value of the cost or that the: Net present value >0. The Net present value (NPV) rule is usually adopted to decide whether to accept or reject an option, to rank different projects and to choose between mutually exclusive projects. An equivalent feasibility test is the benefit cost ratio (BCR) test (Pearce *et al.* 2006):

$$PVB / PVC > 1.$$

However, there are differences between the two tests. The first evaluates the excess in benefits and is a more direct way of measuring the social benefits of a cleanup intervention. The second evaluates the benefits per dollar of cost incurred. For example, a cost ratio of 2.2 means that for each dollar invested $2.20 of social benefit is realized (Pearce *et al.* 2006). There is general agreement that BCR can be misleading when used outside the rationing context (when only one project should be evaluated: implemented versus rejected).

2.5 Risk and uncertainty

As mentioned in the previous paragraphs, cost and benefits are difficult to ascertain. In this context, it is important to define risk and uncertainty given that these are often used as interchangeable elements in the literature. Risk denotes the possibility of attaching a probability to costs or benefits that are not known with certainty. Uncertainty denotes a case in which the probability distribution is not available, but crude end points like the min and max are known.

If the decision maker is risk neutral, the expected values of benefits and cost are evaluated. In this case, the net present value equation is as follows (Pearce *et al.* 2006):

$$NPV = (\sum_1 p_i \times B_i) - (\sum_1 p_j \times C_j)$$

Where P_i is the probability that the benefit B_i occurs and p_j is the probability that the cost j occurs.

A recent study evaluating the potential benefit of reducing the pollution exposure in the two industrial areas of Gela and Priolo (Sicily) adopted, for the first time, cost benefit acceptability curves to assess uncertainty in benefit/cost estimates. To build cost benefit acceptability curves Guerriero *et al.* (2011) assign to each parameter a probability distribution (e.g. gamma for cost, normal for excess cases). Then, from each distribution they generate 10,000 Monte Carlo simulation samples. Cost benefit acceptability curves are built plotting the proportion of simulations producing a positive net benefit given a range of remediation cost.

4. Conclusion

Hazardous waste sites are a major environmental problem. There is a large body of literature showing an association between hazardous waste (mis)management and negative health outcomes. Substances resulting from industrial production (e.g. arsenic, cadmium and mercury) once released into landfills without proper treatment can be fatal for the populations exposed. In the US, the public has ranked toxic wastes sites as the number one national environmental priority. A recent study of a contaminated site in the Italian region of Campania, found that 87% of survey respondents believed that they are going to suffer from cancer because of waste exposure (Cori & Pellegrino 2011). Responding to public concerns, national reclamation projects have been created in several countries, e.g. Superfund program in the US, and programma nazionale di bonifica in Italy. The objective of these programs is collecting public and private resources to prioritize the clean-up of hazardous waste sites. Cost benefit analysis is a transparent decision informing procedure to prioritize the cleanup of those sites that for a given remediation budget would allow to produce the highest benefit in terms of negative health outcomes averted.

Despite the potential benefits resulting from the application of cost benefit analysis in waste management there are few empirical studies using this tool. The study conducted by Hamilton and Viscusi (1999) evaluating the cost effectiveness of EPA Superfund decisions showed that the majority of clean-up decisions are ineffective and highlights the importance of conducting site level analysis. Further studies conducted in US found that other factors such as media coverage were prevailing in determining the stringency of clean-up standards and the selection of clean-up sites/size. As long as the true benefits and costs of cleanup interventions are ignored resources will be allocated inefficiently. Despite measurement problems and the equity issues, cost-benefit analysis should be conducted routinely to address National Superfund's decisions. (Zimmerman and Rae, 1993).

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