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AN INVESTIGATION INTO
SUNSET CHEMICALS AND THE ECONOMIC DIMENSION

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with the assistance of Terry Hatton and Dave Murphy

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Pollution Probe

FINAL REPORT

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1. EXECUTIVE SUMMARY

A. OVERVIEW

This report is a critical review of both the theory and practice of economics in the evaluation of the costs and benefits of changes in environmental quality, with special focus upon the virtual elimination of persistent toxic chemicals.

B. RESULTS

The report has three sections. Section 1 (comprising chapters 3 and 4) describes the underlying economic theory of cost-benefit analysis. Benefits are measured as the maximum amount an individual would be willing to pay in order to obtain some good or service, and costs are measured as the minimum compensation that must be given to an individual in order for him or her to accept a burden. A public policy increases efficiency if there is an excess of benefits over costs when these are summed over all individuals in society.

Chapter 4 concentrates upon the task of describing the various methods used to obtain estimates of the benefits associated with changes in the environment. Since these benefits are associated with non-market goods, they must be inferred either indirectly (using either Averting Behaviour, Hedonic Market, or Travel Cost Models) or directly (using a Contingent Valuation Methodology survey). While the indirect methods can all measure the use benefits from environmental improvements, only the latter can measure the non-use (existence value and option value) benefits.

Section 2 (comprising chapters 5 and 6) presents and evaluates the results of past efforts to assess the magnitudes of costs and benefits associated with a change in environmental quality. Chapter 5 shows that human health and recreational non-use benefits are empirically important, but are difficult to measure. To date, costs associated with environmental protection laws have not imposed a significant burden on North American economies. However, environmental regulations have the potential to redirect private sector investment away from contributing towards increases in marketed outputs and towards the production of non-marketed outputs such as improvements in environmental quality.

Chapter 6 comprises three case studies that examine the effects of government efforts to regulate emissions of DDT, lead, and dioxins. This review shows that the use of economic analysis to assess proposed environmental policies has been uneven. Nonetheless, the cost-benefit analysis conducted by the USEPA strongly supported the banning of lead in gasoline.

Section 3 (comprising chapters 7 and 8) gives a final assessment of the role of economic theory and cost-benefit analysis in the evaluation of environmental issues. While the approach has conceptual and empirical limitations and entails a number of difficulties, it is worthwhile pursuing. It provides the researcher significant insights into the multitude of issues surrounding changes in environmental regulations.

C. CONCLUSIONS AND IMPLICATIONS FOR VIRTUAL ELIMINATION OF PERSISTENT TOXIC CHEMICALS IN THE GREAT LAKES REGION

The scope of the report does not include the estimation of actual benefits and costs from the virtual elimination of persistent toxic chemicals in the Great Lakes region. However, judging by the numbers that other researchers have found for benefits and costs of environmental changes, the potential costs and benefits are likely to be very large. There is some limited information regarding clean-up costs. One report from 1991 (VHB Research and Consulting) that estimated the abatement and control expenditures required under Ontario's MISA program suggests that the annualized capital and operating costs for Ontario industries alone will be in excess of $1 billion and may reach $2 billion. There is less information regarding the scale of benefits from environmental remediation in the Great Lakes but the available evidence suggests that they too can be expected to be important. Annual recreational benefits in Ontario stemming from water quality improvements under the Great Lakes RAP plan are estimated to exceed $270 million. In addition, the annual health benefits for Ontario residents stemming from the proposed revisions to Ontario's air quality regulations are estimated to range from $1 to $4 billion. Finally, USEPA risk assessment calculations are reported in order to show that eliminating PCB's and dioxin from the Great Lakes could result in as many as 300 to 400 avoided premature fatal cancers annually for the population residing in the Great Lakes basin. Avoiding these premature cancer deaths could imply an annual benefit to residents of the Great Lakes basin of $700 million to $1 billion.

The report sees a clear need for further empirical research into the costs and benefits of virtual elimination of persistent toxic chemicals. This research will need to be conducted under a variety of scenarios regarding the actual methods chosen by regulators to achieve the target.

2. INTRODUCTION AND TERMS OF REFERENCE

A. INTRODUCTION

This project examines the economic aspects of the regulation of toxic chemicals in the environment. Particular emphasis is placed upon the role of virtual elimination, the phase out and banning of the most hazardous chemicals, so called sunset chemicals.

The report has eight chapters and a bibliography. Chapter 1 contains the executive summary of the project. Chapter 2 introduces the topic and gives the terms of reference for the project. Chapter 3 gives a brief overview of the theory of cost benefit analysis, which is the method used most frequently by economists to evaluate public policy changes. Chapter 4 presents the theoretical background for the methodologies for the determination of benefits from public policies that alter the quality of the environment. Chapter 5 discusses the empirical work to date that has focused upon the estimation of the benefits and costs associated with environmental regulations. Chapter 6 presents the results of three case studies chosen by Pollution Probe and the researchers. The first case is a retrospective examination of the banning of DDT. The second case analyzes the effects of legislation that has reduced the amount of lead in both gasoline and drinking water. The third case is an examination of current and past efforts of controlling the amount of dioxin that enters the Great Lakes ecosystem.

Chapter 7 takes a critical look at the cost benefit methodology. Chapter 8 presents conclusions arising from an assessment of both the current theoretical techniques and empirical applications as relating to environmental quality regulations.

B. TERMS OF REFERENCE

The object of the project is to look at the costs and benefits of sunsetting chemicals. The study is composed of three parts.

a) Methodology : What is the methodology (assumptions, assessment techniques, extrapolation issues) of assessing the long-term costs and benefits of phasing out the worst chemicals? This part will lay out a more comprehensive approach to understanding what kind of costs and benefits should be taken into account.

b) Case Studies : The case studies will examine and attempt to shed some light on the methodology developed under (a) in terms of concrete examples. Three case studies have been agreed to by Pollution Probe and the researchers. They are the actual phase-out of DDT, the proposed phase-out dioxins, and the proposed phase- out of lead.

c) Findings and implications : This section will summarize the findings and suggest the implications of those findings, such as : areas for further research, how to develop further methodology, what the case studies suggest.

3. THE THEORY OF COST-BENEFIT ANALYSIS

A. OVERVIEW

Cost/benefit analysis is a widely used methodology employed to evaluate the change in society's welfare (or well-being) from a proposed public policy change. The change in welfare is defined with respect to the benefits gained and the costs incurred as a result of the adoption of a new public policy. The benefits and costs are calculated in comparable terms; generally, in terms of dollars of benefits gained or dollars of new costs incurred. These dollars are intended to act as a proxy for changes in the welfare of individuals obtaining the benefits or incurring the costs. A cost/benefit analysis takes into account all benefits and all costs of a new policy. In particular, it includes both private benefits and costs (i.e., those borne directly by individuals making decisions), and external benefits and costs (i.e., those borne by third parties not involved in decision making). When both the private and external benefits and costs are summed, the result is a measure of the degree of net gain to society as a whole from the adoption of the policy. This is called the net social benefit. If more than one policy is being considered, the policy that maximizes the net social benefit to society is identified as the best.

In order to establish the existence of a net gain for society from the adoption of a new policy, we must first identify the net social benefit for the status quo. Then, a new post-policy equilibrium is found, and the net social benefit is compared to that of status quo. If the policy change results in a higher net social benefit, then the policy should be adopted since it is favourable to society as a whole.[1]

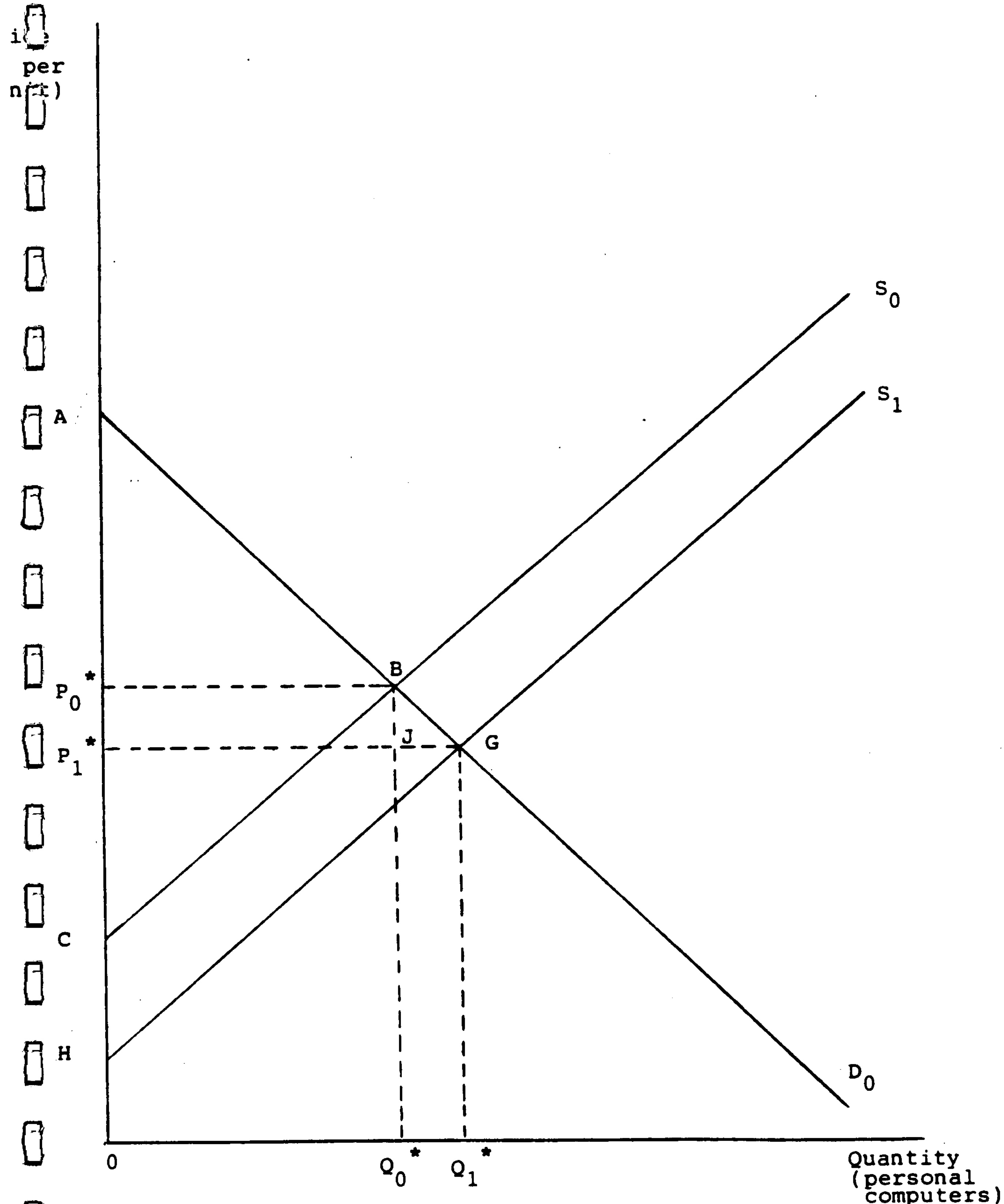
Figure 3.1 illustrates the cost/benefit analysis in graphical terms. To be concrete, we will assume that Figure 3.1 represents all relevant information about the market for a particular good. Furthermore, this analysis will be a partial equilibrium one; linkages to other markets will assumed to be negligible, so that the change in society's overall well-being can be measured only by looking at the change in well-being in this market.

Let us say for the moment that Figure 3.1 represents the market for personal computers. This is a private good; when an individual purchases this good, it is no longer available for someone else to enjoy. The purchaser gets private satisfaction from ownership of the good and there are no external benefits or costs. The downward sloping line marked D_0 is the market demand for personal computers conditioned upon the current market prices for other goods that can be purchased, population, the per capita income, and tastes/preferences (likes/dislikes). The upward sloping line marked S_0 is the market supply of personal computers conditioned upon the prices of other goods that could be produced by the computer firms, the costs of inputs, and the technology available. This figure assumes that the industry producing personal computers is perfectly competitive.

A market equilibrium is established at the intersection of the supply and demand curves resulting in an equilibrium price for personal computers of P_0^* and equilibrium quantity traded of Q_0^*. The net social benefit to society of this status quo situation is found as the area in triangle ABC. This is composed of two parts : area of triangle ABP_0^*, commonly known as the consumer surplus, and area of triangle CBP_0^*, commonly known as the producer surplus.

Consumer surplus, simply stated, is the difference between what individuals would have been willing to pay in order to get the personal computers and what they must pay

Figure 3.1 : Maximization of Net Social Benefit



according to the market price. In this interpretation, the demand curve is also called the "willingness-to-pay" curve because it identifies the maximum price consumers would be willing to pay in order to obtain one more unit of the good. The difference between the maximum willingness to pay, and what the consumer must actually pay in the market, represents a gain to the consumer. Thus, the total benefit to consumers from consuming Q_0* of the good is $OABQ_0$*, while they have to pay an amount equal to OP_0*BQ_0* to purchase that quantity. Hence, the net benefit (consumer surplus) is the difference between these two values, or the area of triangle ABP_0* as previously stated.

The producer surplus is defined in a similar way. The supply curve is the marginal cost curve. It shows the additional cost of producing one more unit of the good. If the supplier receives a unit price equal to this marginal cost, then the supplier is willing to supply one more unit of the good. Thus, one can interpret the supply curve as the minimum willingness-to-receive. Any price received that is greater than this minimum represents a "profit"[2] to the supplier. Thus, the total payment received for Q_0* of the good is the area of rectangle OP_0*BQ_0*. The total variable cost of producing this is given by the area $OCBQ_0$*. The remainder is the area of triangle CBP_0* or the producer surplus. (Included in the producer surplus is a normal return on capital).

With the establishment of an initial status quo position for net social benefits[3], it is now possible to discuss how cost/benefit analysis works. We begin by assuming that the government has adopted a new public policy which leads to a new equilibrium in Figure 3.1. We next compare the sum of consumer and producer surpluses in the new situation to that of status quo. Cost/benefit analysis says that if the sum in the new situation is greater than that of status quo, then the policy initiative is beneficial to society as a whole and should be adopted. That is, that the welfare change for society, as a whole, in going from status quo to the new equilibrium, is positive.[4]

To make this concrete, assume that the government decides to fund some research and development that leads to a technological improvement in the production of personal computers. This has the effect of shifting down the supply curve to S_1 - that is, the additional cost of producing one more unit is lower than it was previously before the technological advance. The new equilibrium price will be established at P_1* and equilibrium quantity traded will be Q_1*. So, consumers will be paying lower prices per unit for computers and buying more of them as a group. Cost/benefit analysis answers the question of the effect that the adoption of the policy has on society as a whole. Note that society includes not only consumers but also firms (or their owners). Overall, society has gained - the sum of consumer and producer surpluses is bigger than it is for status quo, i.e., the sum is the area of triangle AGH. Consumers enjoy a consumer surplus of AGP_1*, while firms get HGP_1* in producer surplus. Consumers have gained an amount equal to $P^*_0BGP^*_1$ while producers have lost an amount of producer surplus equal to P_0*BJP_1* and gained an amount equal to CBGH. As long as the latter is greater than the former, producers enjoy a net gain in producer surplus. Suppose, however, that producer surplus actually fell after the adoption of the policy. As long as the gain in consumer surplus was greater than the loss in producer surplus, then the project should go ahead.

To complete this analysis, we would need to add to the net social benefit found above an amount equivalent to the costs incurred by the government in its research and development undertakings. Assuming that the net social benefits (from above) minus the government expenditures are positive, then the project would be worthwhile from society's point of view, and should be adopted.

The above analysis rests upon several key assumptions. First, the price change caused by the technological improvement is a marginal one; the new price is only incrementally different from the previous price. This follows from the emphasis in economics upon marginal analysis. Techniques used by economists consider solutions that are in close approximation to status quo. These techniques are often not appropriate when the price changes are large. This is discussed in chapter 4, with reference to specific methods used in the assessment of benefits from improvements in environmental quality.

Secondly, the benefits and costs flowing to different individuals or groups in society are assumed to have the same weight. Some might argue that benefits to lower income groups ought to carry more weight. Within the standard treatment of cost benefit analysis, it is possible to change the weights on the various cost and benefit components (Dasgupta and Pearce, 1972; Pearce and Nash, 1981). The real difficulty lies in determining the values of these weights. The choice of a particular set of weights implies a certain ethical view regarding the ideal distribution of benefits and costs. A number of alternative ethical positions can be included into this type of analysis. They will be reviewed in chapter 7 of this report which is concerned with a critical assessment of cost/benefit analysis.

Thirdly, regardless of the weights attached to the various cost and benefit components, at the basis of cost/benefit analysis are the pillars of the Pareto principle and the compensation principle. The first adopts the notion that each individual in society knows what is best for him/her according to personal preferences, and costs and benefits are assessed in this context (Dasgupta and Pearce, 1972). The second requires only that gainers under the public policy <u>be able</u> to compensate the losers (Hicks, 1939; Kaldor, 1969). It does not, however, require that actual payments must exchange hands.

Fourthly, the analysis presented above is static in the sense that only costs and benefits for the current period are being assessed. In practice, many costs and benefits flowing from public policies are felt in the future. It is common to discount future benefits and costs according to a discount rate, and to re-express the rule concerning the adoption of a policy as the following. Namely, if the present value of net benefits is positive, then the project is worth doing (Pearce and Nash, 1981). Furthermore, the project with the largest present value of net benefits is the best one. The reason for discounting is that members of society express a preference for the present over the future; this is the social rate of time preference. Why is the present preferred? Two obvious answers spring to mind : the risk of death, and the belief that future generations will be wealthier than the current one (Dasgupta and Pearce, 1972). The discount rate generally incorporates a 'social rate of interest', or the opportunity cost of resources currently being used for the project.

Fifthly, the analysis does not incorporate the possibility of uncertainty or risk with regard to the flow of benefits or costs. The standard treatment of uncertainty is to assume that the decision makers maximize either expected utility or expected profit (where expected utility is a function describing a weighted sum of utilities in alternative states of the world, where the weights are the probabilities of those states occurring) (Johansson, 1987). Chapter 4 discusses the role of uncertainty in two specific instances of benefit estimation.

Finally, the analysis presented above uses partial equilibrium techniques. A single, isolated market is considered. However, markets are interdependent and one can expect that there might be spillover effects in other markets. For example, markets that produce complementary products to personal computers would probably see gains in benefits, while markets producing substitutes would see losses (or costs) that must be taken into account in order to evaluate the policy from a general equilibrium (or economy-wide) framework. If the costs of the research and development and/or the costs imposed upon consumers in the other markets (in the form of lost benefits or lost consumer surplus) are greater than the net benefits in the personal computer market, then the policy would not pass the test of maximizing the net social benefits to society as a whole.

To this point, the discussion has centred on the impact upon society of a public policy that has lowered the price of a good. Using the same general cost/benefit framework, it is also possible to analyze the benefits to society of a change in environmental quality arising from the adoption of a public policy. However, from the standpoint of economic theory, the consumer surplus measure discussed above has a number of shortcomings. These problems and their solutions are discussed in the next section, which will be more technical than this section. Instances of policy changes resulting in changes in price and changes in environmental quality will be considered.

B. MEASURING WELFARE CHANGE

B.1. Benefits to Consumers from Price Changes[5]

The analysis presented above uses the market, or Marshallian demand (named after Alfred Marshall, the father of modern economics and marginal analysis), to evaluate changes in consumer welfare. There are several theoretical problems associated with the use of this demand curve to measure welfare change. In order to understand more fully the difficulties, the theoretical derivation of this demand curve is presented.

We begin with the assumption that an individual consumer derives utility from a set of N market goods, **X**. Thus, the utility function is given in equation (1).

$$U = f(X_1, X_2, \ldots, X_N) \tag{1}$$

This is an ordinal utility function that merely ranks bundles of goods as preferred, not preferred, or indifferent. It does not represent intensity of preferences. The consumer is assumed to have a fixed income, **y**, and to face fixed market prices for the goods, P_1, P_2, etc. Then, the solution to the consumer's utility maximization problem (maximize utility subject to the constraints of income and prices) is a set of N Marshallian demand curves shown in equation (2).

$$X_i* = m(P_1, P_2, \ldots, P_N, y) \tag{2}$$

These demand curves are said to be uncompensated; when prices change, income is not adjusted to compensate for the resulting change in utility. Thus, when the gain in consumer surplus resulting from a price decrease is measured (as is shown for all consumers in Figure 3.1), it includes both the effect of lower prices (the substitution effect which leads the consumer to purchase more of the cheaper good) and also the effect of the consumer's real income or purchasing power being higher (thereby, permitting the consumer to purchase more of all goods according to his/her preferences). Strictly speaking, when measuring the welfare gain, we only want the first effect. The second effect is not part of the original public policy change.

There is a second problem with using Marshallian demand curves to measure welfare effects. If more than one market is involved and a series of price changes occurs, then the measure of the welfare gain can be altered by the order in which the price changes are considered. This property is called path-dependency (Johansson, 1987). In other words, for the same set of price changes, one could find instances of overall social welfare loss and social welfare gain depending on the order in which the price changes were considered. Clearly, this is a serious problem. A second consequence of path dependency is that the consumer surplus measure might indicate a positive change (thereby lending support for the public policy) when the true underlying change in utility is negative. The reverse may also occur. There is one instance in which the consumer surplus measure is path-independent. This occurs when the utility function is homothetic (Johansson, 1987), meaning that the income elasticities on all goods are one. This is, however, a very restrictive assumption.

Sir John Hicks (1940/41, 1945/46) provided a solution to the first problem in the 1940's. He suggested the construction of a compensated demand curve where the demand for a good would depend upon the prices of goods, and a given level of utility, as shown in equation (3). The Hicksian (compensated) demand functions show how the

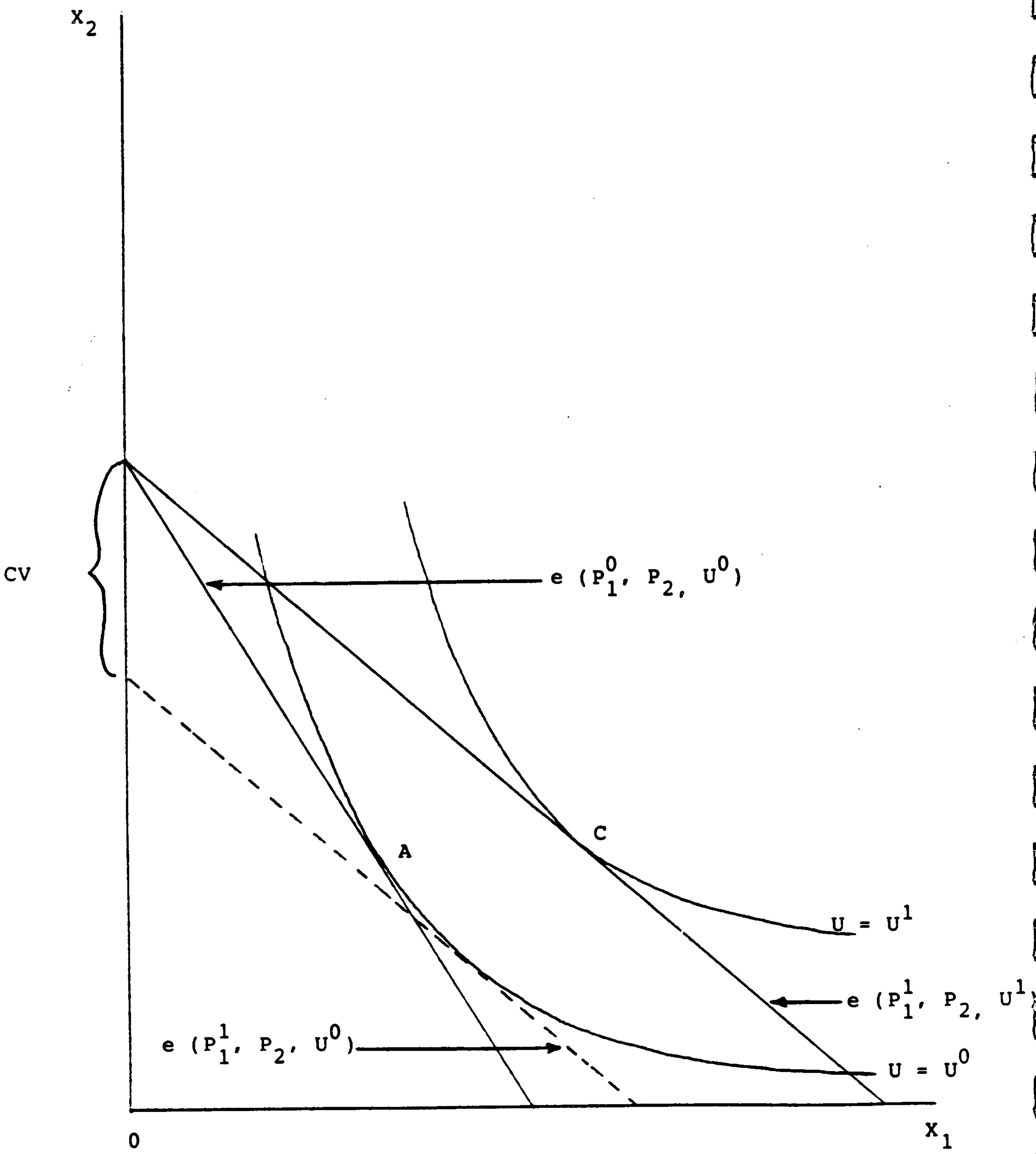
$$X_1^H = h(P_1, P_2, \dots, P_N, U^0) \tag{3}$$

quantity demanded of the good is affected by prices when consumers' incomes are adjusted (either positively in the case of price increases or negatively in the case of price decreases), so as to keep the consumers at the same level of utility as before the price change. Generally, if goods are normal (so that the consumption increases as income increases) the compensated demand curves will be steeper than the Marshallian ones. Note that Hicksian demand curves are conditioned upon specific levels of utility. Thus, for each level of utility, there is an unique Hicksian demand curve for the good relating the desired quantity demanded at different prices.

The importance of the Hicksian demands lies in the fact that one can define two alternative measures of welfare change resulting from a price change. Suppose there is a decrease in price. One way to measure the welfare change is to calculate the compensating variation. This is the maximum amount of income that could be taken away from a consumer, while leaving him/her just as well off as he/she was prior to the fall in price. (One could call this the willingness to pay to remain at one's status quo position.)[6] There are two ways of showing the size of this income adjustment - the first is in the context of an indifference curve and budget line analysis, and the second is as an area related to the Hicksian demand curve. It is valuable to examine both ways.

Figure 3.2 (adapted from Johansson, 1987) shows one consumer's compensating variation for a price decrease. Initially, the consumer is consuming at point A with associated utility U^0. Nominal income is y^0, and the prices of the two goods are P_1^0 and P_2. In this example, assume only two goods are consumed, and the price of the second good and nominal income are held constant. When the price of the first good falls to P_1^1 then the consumer could get to point C with associated utility U^1. However, recall that we only want to measure the welfare change associated with the price reduction. To do so, we must adjust for the fact that the price change increases the consumer's real income (i.e. purchasing power of the nominal income). Thus, if an amount of money is taken away

Figure 3.2 : Compensating Variation, Decrease in Price of X_1



from the consumer that will just put him or her back on the status quo indifference curve with utility U^0, he or she will be content. This amount of income is the Compensating Variation (CV). It is measured by the vertical distance CV in Figure 3.2. Note that the compensating variation approach to welfare change assumes that status quo utility is the appropriate reference point. In other words, there is an assumption that the consumer has the right only to status quo utility.[7]

More formally, the Compensating Variation is defined as the difference between the expenditure needed to achieve U^0 level of utility at the two different prices for good X_1 (Braden and Kolstad, 1991). The **e** functions below are expenditure functions conditioned upon prices and the given level of utility, U^0. These functions represent the minimum expenditure necessary to achieve a utility level of U^0 at different prices. This is shown in equation (4).

$$CV(P_1^0, P_1^1, P_2) = e(P_1^1, P_2, U^0) - e(P_1^0, P_2, U^0) \tag{4}$$

The alternative measure of welfare change derived from the Hicksian demand curve is the Equivalent Variation (EV). It takes the new level of utility as the reference point. In other words, the consumer has the right to the new level of utility, U^1, after a price change occurs. This is shown in Figure 3.3. Again, assume that price falls. Previously the consumer was on U^0 at point A, but now can get to point C with utility level U^1. The Equivalent Variation is the amount of income that would have to be given to the consumer to put him/her at the same level of utility as the price reduction. This can be thought of as willingness to accept monetary compensation in lieu of the price change.

More formally, the Equivalent Variation is defined as the difference between the expenditure needed to achieve U^1 level of utility at the two different prices for good X_1. Equation (5) shows the exact definition. In Figure 3.3 the Equivalent Variation is shown

$$EV(P_1^0, P_1^1, P_2) = e(P_1^1, P_2, U^1) - e(P_1^0, P_2, U^1) \tag{5}$$

as is the vertical distance marked EV.

These two measures, CV and EV, can be compared in a single diagram (Johansson, 1987; Braden and Kolstad, 1991) along with the Marshallian consumer surplus measure described in the previous section. Figure 3.4 shows a Marshallian (uncompensated) demand for good X_1 (identified as m(P,y) and two Hicksian (compensated) demand curves for the same good (identified as h(P, U^0) and h(P, U^1). The Hicksian demand curves are distinguished by two different utility levels, U^0 and U^1. The Compensating Variation associated with a reduction in the price of good X_1 from P_0 to P_1 is shown as the area to the left of the Hicksian demand curve associated with utility level U^0, while the Equivalent Variation for the same price reduction is shown by the area to the left of the other Hicksian demand curve associated with the utility level U^1. These areas are bounded by P_0ABP_1 and P_0CDP_1 respectively. The consumer surplus measure is found to the left of the uncompensated Marshallian (shallower sloped) demand curve and is identified as area P_0ADP_1. Note that the consumer surplus lies between the Compensating Variation (CV) and the Equivalent Variation (EV) measures, and that EV is greater than CV.[8]

In comparing these two measures, several comments can be made. First, the Compensating Variation measure does not measure the utility change directly, but rather the offsetting income change necessary to prevent a utility change. However, the

Figure 3.3 : Equivalent Variation, Decrease in Price of X_1

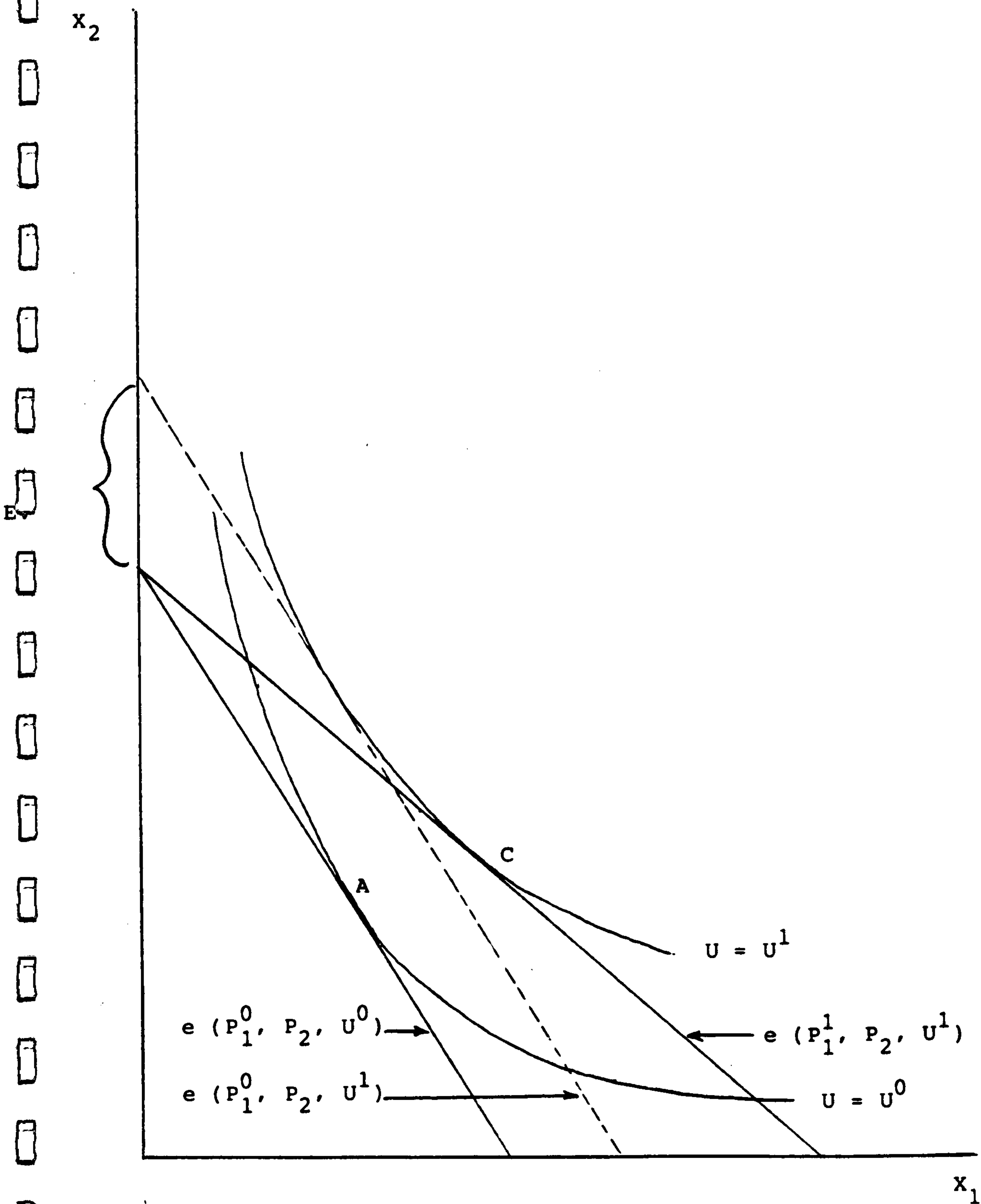
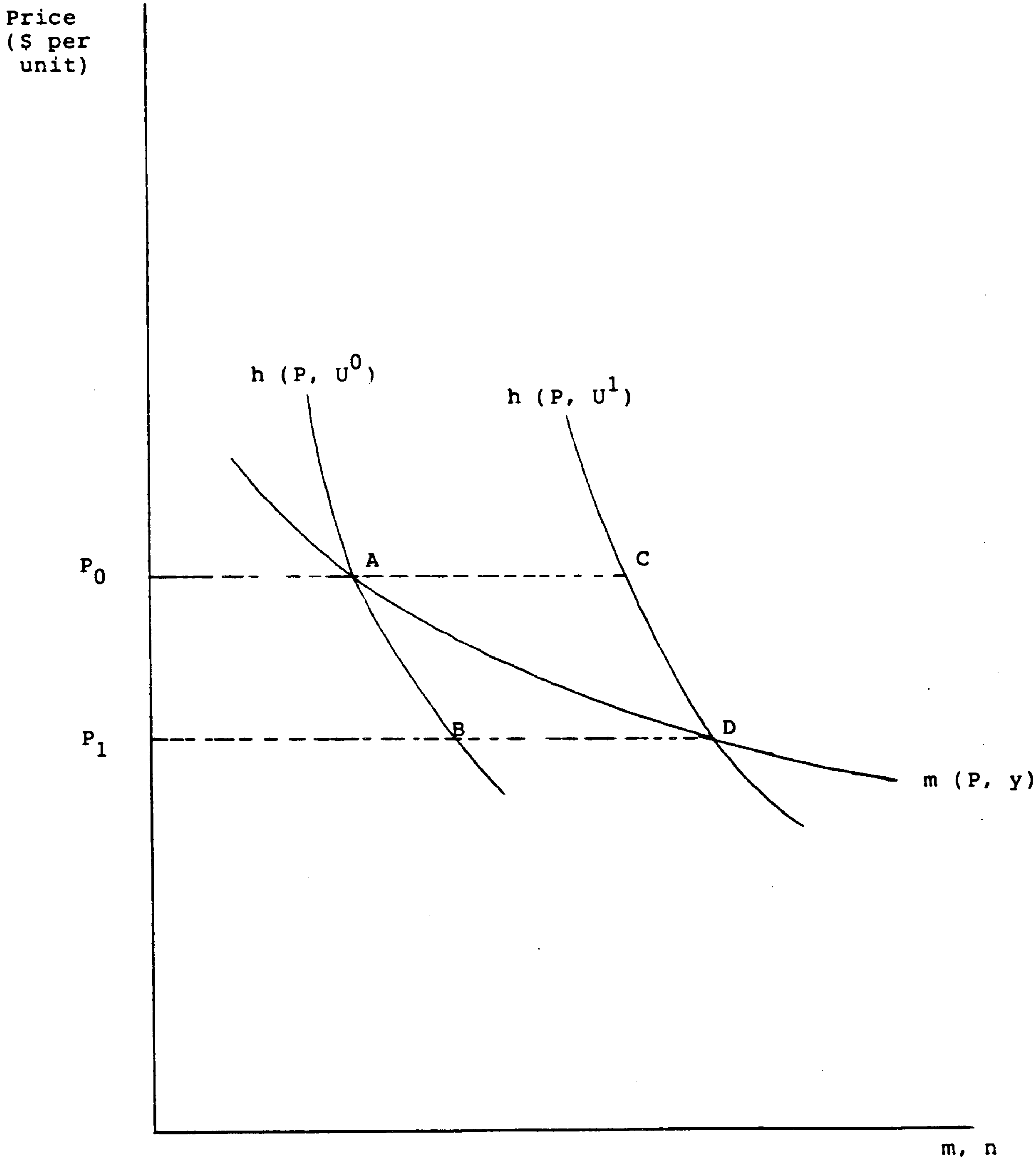


Figure 3.4 : Comparison of Consumer Surplus, Compensating Variation, and Equivalent Variation



Equivalent Variation measures directly the money equivalent of a price change; it is the income change necessary to have the same effect on utility as the price change brought about by the public policy. In choosing which measure to adopt, the researcher should be sensitive to the property rights implied in the measures.

Secondly, both the Compensating Variation and the Equivalent Variation measures are path-independent when a series of price changes are considered. Thus, they are independent of the order in which the multiple price changes are evaluated (Johansson, 1987). However, in cases where more than one public policy is being evaluated and ranked, the EV will give a complete ranking that is consistent with individual preferences, that is, in a manner that is consistent with the way in which the consumer ranks the projects, while the CV will not (Johansson, 1987). However, CV will do so, provided that the utility function is either homothetic or quasi-linear.[9]

Thirdly, as shown above, the consumer surplus measure is bracketed by the CV and EV measures. The advantage to the consumer surplus measure is that is can be easily obtained from knowledge of a well-specified single equation demand function that is readily observable from market data. The Hicksian demand functions needed to measure CV and EV are not directly observable, although there are a few ways to obtain them (Braden and Kolstad, 1991). Rather than describing the methods for obtaining the Hicksian demands (Blackorby, Primont, and Russell, 1978), it is more useful to ask just how far off is the consumer surplus measure? Two comments are pertinent. First, if the income elasticity of demand for the good is zero, the three measures coincide. In this case, consumer surplus provides an exact measure of both CV and EV. Secondly, even in the case where the income elasticity of demand is not zero, Willig (1976) has shown that the maximum error in using consumer surplus as an approximation for either CV or EV is about 5 %. Consequently, on a practical level, the consumer surplus measure is a reasonable approximation to the size of welfare change resulting from a public policy action.

The fourth observation in comparing the EV and CV methods requires that one ask how large a measure of benefits could a researcher could obtain. In particular, could it be infinite? Recall, in the case of the price reduction, that the Equivalent Variation is the amount of income a consumer would be willing to accept in order to forego the price reduction. In theory there could be no bound on this value. In order to put a bound on this EV it is sufficient for the good to be what Willig (1978) defines as a non-essential good. This says that any bundle of goods including the non-essential good can be matched by a bundle of goods excluding it. (In other words, there is some degree of substitution between the non-essential good and some other good or goods). Clearly, this may be problematical in the case of certain natural resources.

Fifthly, the discussion of welfare changes has concentrated upon describing how to obtain one individual consumer's measures of CV and EV. An important issue is whether we can aggregate consumer surplus measures; that is, add up individual benefits to get society's overall level of benefits. The issue of aggregation is concerned with the conditions under which it is theoretically possible to use the areas under Marshallian and Hicksian <u>market</u> (not individual) demand curves to measure the gain or loss to society as a whole.

With regard to the Marshallian consumer surplus measure, if everyone in society has the same (constant) social marginal utility of income[10], then one can interpret the area under the uncompensated market demand curve, and above the market price, as a measure of welfare change. Otherwise, the sum of individual consumer surpluses need not have the same change in sign as the welfare change. This requirement is quite stringent. The social marginal utility of income is the product of one's welfare weight (weight in the function that aggregates up each person's utility) and the marginal utility of income (Johansson, 1987). The marginal utility of income is the measure of how much extra utility an individual gets from one more dollar of income. Assuming that each member of society has the same constant marginal utility of income means assuming that one dollar more given to a person with $100,000 income gives that person the same additional utility as one dollar more given to a person with $1,000 income. In addition, it must be possible to compare utility levels across people, that is, 1000 units of utility to person A would be equivalent to 10 times the utility of person B getting 100 units of utility.

Compensating and Equivalent variation measures, because they are based on an expenditure function defined in a common monetary unit, lend themselves to aggregation. For example, suppose the we add up the Compensating Variations attributable to a particular project and the sum is positive, thereby indicating that the project should be carried out. For the aggregate Compensating Variation to be positive, it must be the case that the money necessary to keep losers at their original utility levels is less than the amount of money that can be extracted from the gainers in order to keep them at their original utility levels. The aggregate measure corresponds with the sum of the individual measures, thereby ensuring consistency.

B.2. Benefits to Consumers from Quality Changes

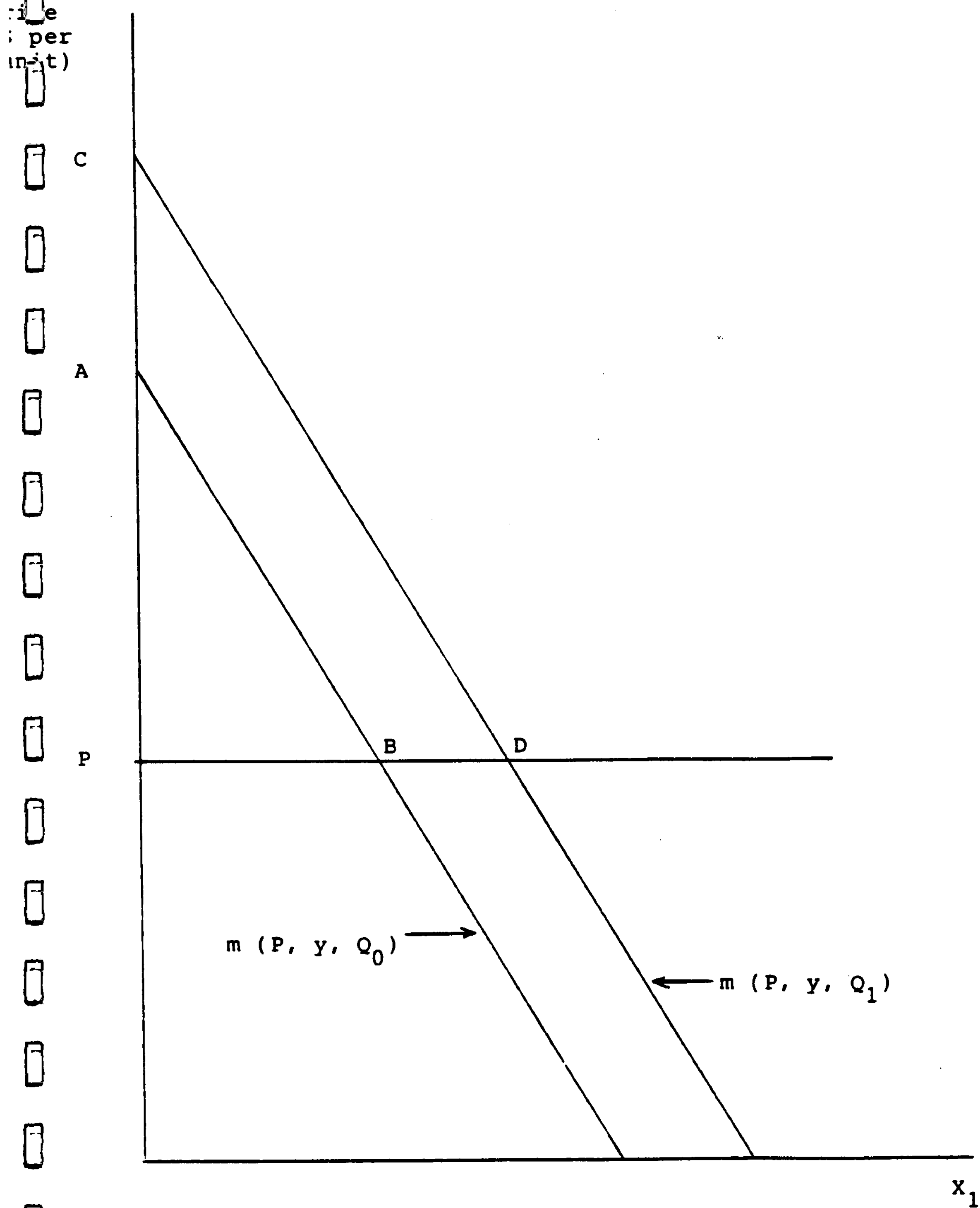
Our focus in this report is upon the effects of a public policy that changes the environment. In addition to the possibility that a changing environment may change the prices of goods purchased by consumers, there is the direct route by which the public policy can affect welfare. The policy can affect either the quantity of an environmental good, or the quality of the environment itself, and in this way have an impact upon an individual's and society's well-being.

In order to examine the welfare changes associated with changes in the quality of the environment or quantity of environmental goods available to consumers, it necessary to augment the utility function discussed previously. We assume that the utility function for an individual includes a variable, Q, that measures environmental quality (assuming that this can indeed be measured in a meaningful way). This is shown in equation (6).

$$U = f(X_1, X_2, \dots, X_N, Q) \qquad (6)$$

Furthermore, we assume that a higher level of Q is better than a lower one but at a diminishing rate. Then, the Marshallian demand for a good, X_1 that is complementary to the level of Q will increase when the quantity (or quality) of Q increases after the introduction of a public policy. Figure 3.5 illustrates this change. The Marshallian demand is identified as $m(P, y, Q_0)$. Suppose the price of the good, X_1, is unchanged by the change in environmental quality. The gain in consumer surplus associated with the increase in Q is measured as the area ABDC. This is the implicit value that the consumer places on the change in the quality of the environment. It represents the amount of money the consumer would be willing to pay, in order to have the better quality environment.

Figure 3.5 : Consumer Surplus Associated with an Improvement in Environmental Quality

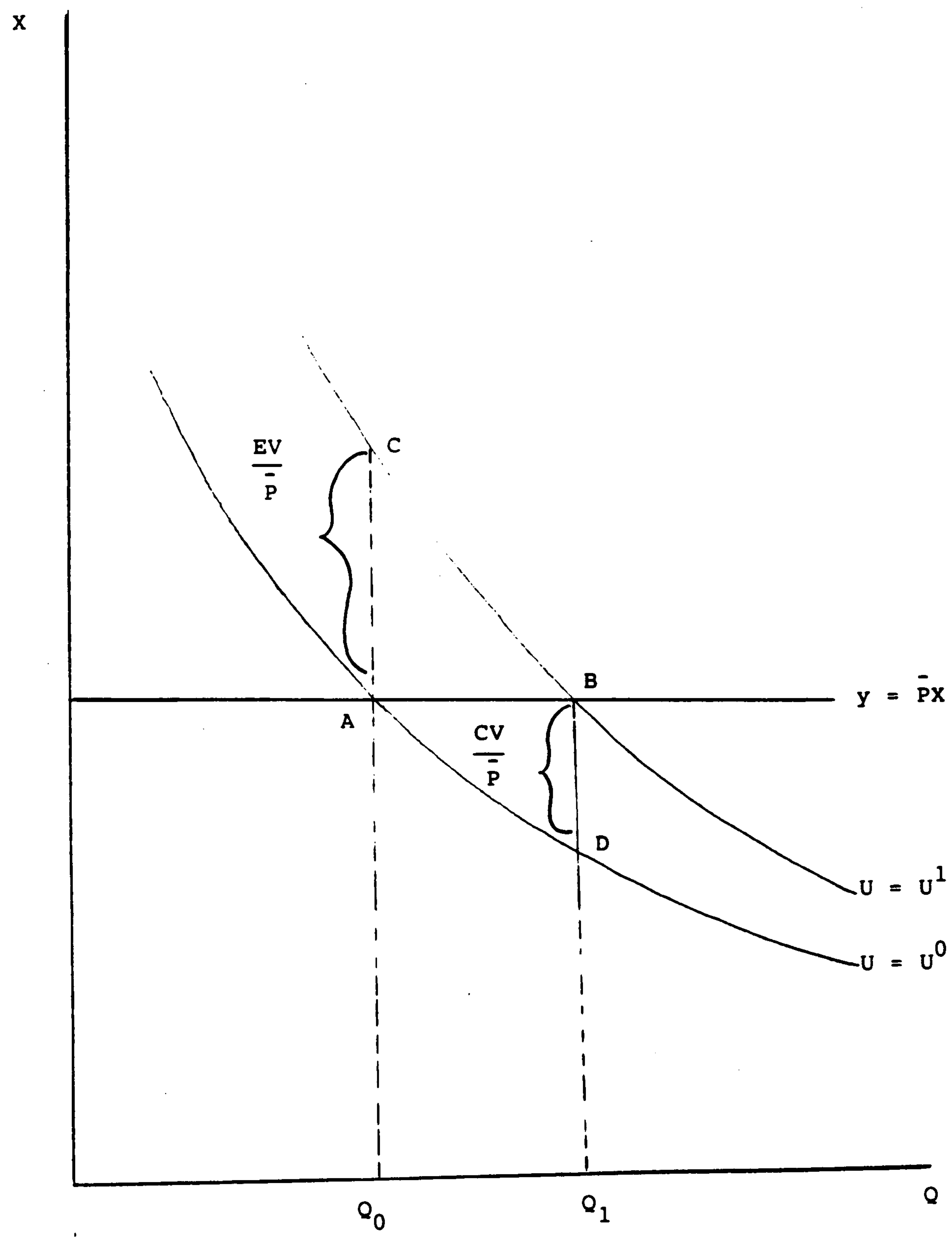


Alternatively, we can define the Compensating and Equivalent variations associated with a quality change. It is common to assume that most environmental goods are available in a fixed quantity at a zero price. Figure 3.6 shows the indifference curves between a composite commodity consisting of all the market goods, X, and environmental quality, Q. The horizontal line PX = y represents the consumer's budget line. Suppose initially, if the quantity of the environmental good is given by Q_0, then a public policy causes an increase in this quantity to Q_1. Originally, the consumer is at equilibrium at point A with utility U^0. A higher quantity of the environmental good moves him or her to point B on a higher indifference curve with utility U^1. Compensating Variation is the reduction in real income needed to put the consumer back to the old utility level, U_0. This is the vertical distance BD. Equivalent Variation is the amount of income increase that the consumer would accept in order to be willing to forego the opportunity to experience the increase in Q. This is given by the vertical distance AC.[11] Once again, EV and CV will not be equal unless the income elasticity of demand for Q is zero. When Q is a normal good (i.e., as one's income increases, one spends more on Q), EV will be greater than CV.

We can also use the expenditure function approach to define the CV and EV measures. The Compensating Variation associated with an increase in the quality of the environment, Q, is shown in equation (7), while the Equivalent Variation is shown in equation (8).

$$CV = e(P, Q_1, U^0) - e(P, Q_0, U^0) \tag{7}$$

Figure 3.6 : Comparison of Compensating and Equivalent Variations Associated with an Improvement in Environmental Quality



$$EV = e(P, Q_1, U^1) - e(P, Q_0, U^1) \tag{8}$$

B.3. The Effects of Price and Quality Changes Upon Firms

The measurement of the welfare effects of a policy change upon firms is straightforward, unlike the measurement of benefits to consumers just discussed. This is so for two reasons. First, the welfare of an individual consumer depends upon that consumer's utility function. Since utility is inherently unobservable, economists have searched for a method to translate utility changes into something observable, measurable, and comparable across different individuals. The search has led them to adopt dollar values of gains and losses. It is common to talk of income gains or income losses. Secondly, the presence of real income gains or losses can cause us to overestimate, or underestimate, the true measure of the consumer's gain or loss of utility resulting from a particular policy change. These difficulties simply are not present when the subject is the welfare effects of policy changes upon firms.

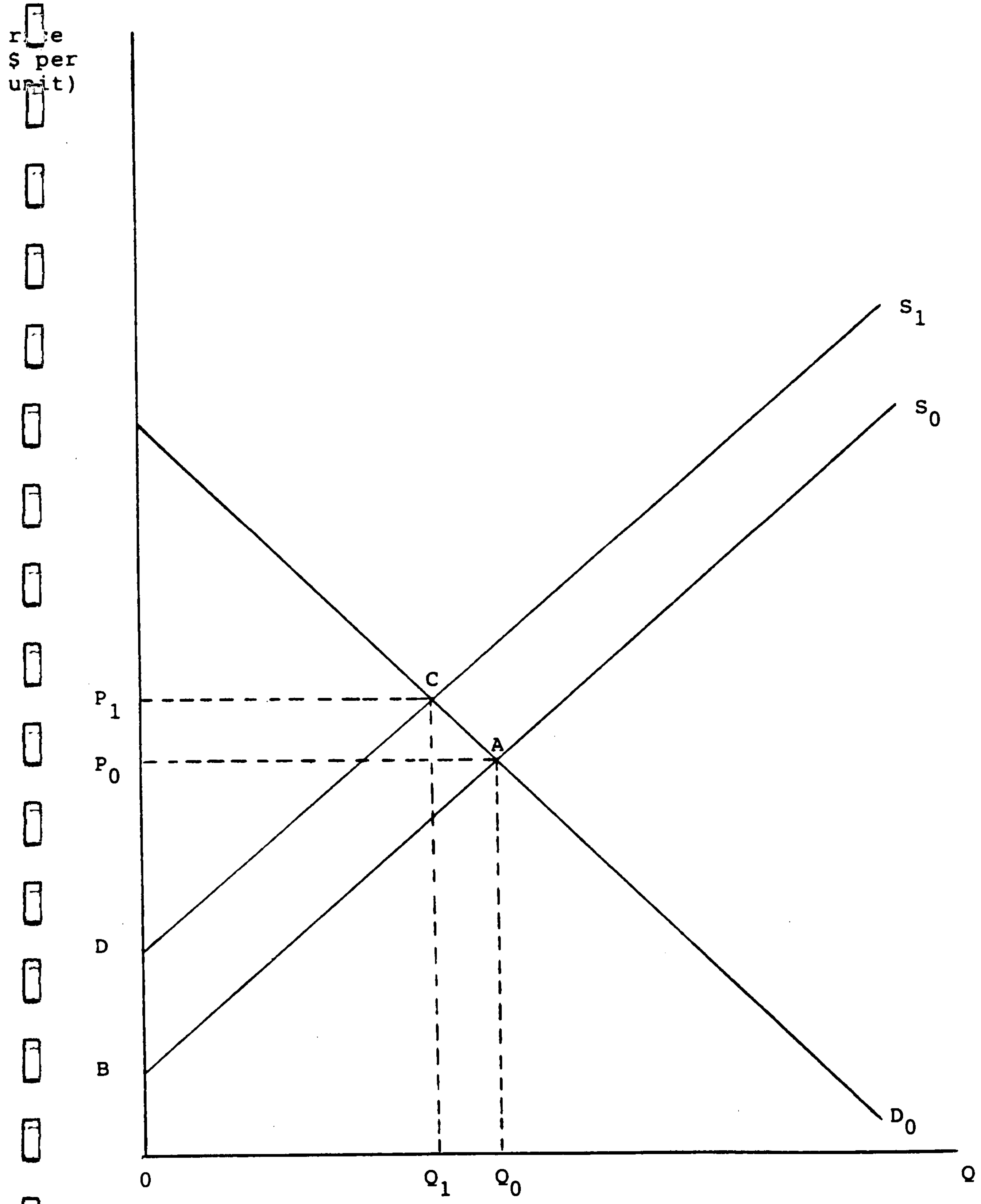
Recall from Section A that the effects of a policy change upon firms are measured by changes in producer surplus (or profits). Since surplus or profit is measured in terms of dollars, and dollars are comparable across firms, changes in producer surplus are used to measure the costs[12] upon firms of changes in public policy. While the consumer surplus measurement has some theoretical shortcomings arising from two cited sources, this is not the case for the producer surplus measure.

As Figure 3.1 shows, producer surplus is measured with reference to the industry's supply curve (simply the sum of each firm's marginal cost curve). Thus, when there is a change in public policy affecting firms, one must ask how this change alters the marginal cost function. This will depend upon the role of the environment in the firm's production process. On the one hand, the firm may use the environment as a waste receiving body. On the other hand, the firm may use the environment as a valuable input into the production of the final market product. For example, beverage companies use a large amount of water. Prior to use, these companies spend a great deal of money treating the water to ensure that it is of an acceptable standard. In general, an improvement in the quality of the environment will increase production costs (per unit input costs) for firms of the first type who must adopt higher cost technologies to comply with the regulations. But that same improvement in the quality of the environment can decrease production costs (per unit input costs) for firms of the second type.

Figure 3.7 illustrates a case in which the adoption of an environmental regulation forces firms to adopt a more expensive production technology. In this case, the supply curve for the industry will shift upward to the left from S_0 to S_1; the new equilibrium price for the product will be P_1, while the initial price was P_0, and the equilibrium quantity sold will have decreased from Q_0 to Q_1. Ignoring the effect upon consumers, the cost to the firms is simply the lost producer surplus. This is shown by area ABDC. This could be measured either by examining the increase in total costs of production in order to comply with the regulation, or by calculating the lost profits to the firms involved. Assuming that the economist has information on the costs and revenues of each of the firms, these decreases in profits (or increases in costs) can be easily calculated for each firm. Since they are denoted in dollar terms, it is easy to add up the costs over all firms to arrive at the total cost to the industry.

This completes the overview of cost/benefit analysis. The following chapter focuses upon the theoretical development of four methods used to measure the various kinds of benefits obtained by consumers when there are changes in environmental quality.

Figure 3.7 : Loss in Producer Surplus Associated with a more Stringent Environmental Regulation



Endnotes

1. If two or more policies are being considered, the one with the highest net social benefit should be adopted.

2. Strictly speaking, this profit is variable profit, or the difference between total revenues and total variable costs. One would then subtract from variable profit a normal return on any capital used in the production process (the so-called fixed costs) in order to obtain an estimate of the firm's economic profit. This is profit in excess of the opportunity costs of all inputs used in the production of the good.

3. The maximization of net social benefits is equivalent to the maximization of the sum of consumer and producer surpluses.

4. Note, however, that there may be gainers and losers from the adoption of the project. It is simply the case that dollar gains to gainers outweigh the dollar losses to losers.

5. The discussion is conducted in terms of policy changes that may be largely beneficial to consumers. However, the methodologies presented can also be used to examine the costs imposed upon consumers of policies that are detrimental to consumers' well-being. It is important to keep this symmetry of the approach in mind.

6. In the case of a price increase the consumer would need to be compensated a sum of income sufficient to return the consumer to status quo. This is called the willingness to accept compensation.

7. Note that, while compensating variation is bounded by the person's income in the case of a price decrease, there is no similar bound on the amount of compensation a consumer might require in the case of a price increase.

8. The difference between these two measures is attributable to the income effect. The ordering of the measures as stated assumes that the income elasticity for the good is positive. For a price increase, the order of relationship between the measures will be reversed. That is : CV > CS > EV.

9. If the utility function is homothetic and income is fixed, the consumer surplus measure previously discussed will also rank outcomes correctly (Johansson, 1987).

10. One instance in which the social marginal utility of income is constant occurs when the marginal utility of income is decreasing in income. This requires, however, that the weight given to high income individuals is larger than the welfare weight to low income individuals.

11. Both the equivalent variation and the compensating variation measures are defined in real terms by dividing them by the numeraire price, P.

12. Changes in producer surplus may be positive, thereby indicating benefits to firms from the adoption of a public policy. Again, as is the case for consumers, one methodology may be used to measure either benefits or costs.

4. METHODOLOGY FOR APPLYING CBA TO ENVIRONMENTAL ISSUES

A. OVERVIEW OF THE THEORY OF BENEFIT MEASUREMENT

Given the pervasiveness of the environment, it is reasonable to assume that the welfare of individuals in a society is either directly affected by the quality of the environment or indirectly affected through the enjoyment of some activity that uses the environment. An underlying assumption in benefit estimation associated with environmental quality is that it is possible to measure an improvement (or reduction) in environmental quality, and to trace through the ways in which the consumers valuation of the environment or related goods changes. That is, we assume that there is a unique line of causation between changes in the level of environmental quality and household behaviour. This view says that the benefits of improved environment are valued according to an anthropocentric view of the world. Consumer sovereignty is paramount, and the benefits express society's explicit or implicit preferences.[1]

Normally, in measuring benefits economists take as the reference point the market values of the items in question. For example, suppose there is a technological improvement in the production of computers. Using market data on supply and demand, it is straightforward to calculate the predicted reduction in the price of computers and, therefore, the gain in benefits to the consumers of computers. However, when talking about environmental quality, most of the benefits accrue from so-called non-market goods or services. That is, markets for their provision do not exist, and hence, economists do not have direct observations on the demand or supply of the goods, and no information on what the prices of such goods might be in a market setting.

In a broad sense the non-market good we are talking about when we look at the sunsetting of chemicals[2] is environmental quality (however defined). We believe that sunsetting will increase the amount of the good available, and that this will increase the benefits flowing from the use of the good by humans and other species on earth. We will make reference to that "good" in what follows. Thus, we assume the existence of a function that looks like $S = S(Q)$ where S is something valued by consumers, and Q is the quality of the environment. (This assumes that Q is not directly valued in and of itself, but this can be incorporated.)

When dealing with non-market goods, economists have taken two alternative approaches to obtain the benefits of goods or services. The INDIRECT APPROACH takes as its starting point that the non-market good (i.e., the environment) can be valued indirectly through its relationship to observable market goods. The use of the market is not discarded, but rather, proxy markets are developed to infer prices for non-traded environmental resources. There are three variants to this approach.

The Averting Behaviour Method observes consumers expenditures associated with the avoidance of a particular pollutant. These expenditures allow the economist to infer an implicit willingness to pay to avoid the contaminant (contingent valuation). The purchase of bottled water as a replacement for tap water is an example of this type of behaviour. This method exploits the fact that the consumer views bottled water as a good substitute for tap water, which is seen as being tainted.

The Hedonic Method assumes that the price of a marketed good can be broken down into the prices of the attributes making up that good. For example, suppose one's house is located near to a hazardous waste site; the market price of the house will have as one of its components this (negative) attribute. Presumably, proximity would lower the market price of this house relative to an identical house in a neighbourhood located further away from the waste site. The difference in the two market prices would reflect the reduction in benefits to the homeowner of having the waste site located nearby. In this way, the economist can calculate a willingness to pay to locate away from the hazardous site. This would represent a benefit from a relocation of the site. This method is also used in the labour market setting to examine the premia that must be paid to workers operating in a risky or hazardous job.

The Travel Cost Method starts with the assumption that the environment is complementary to one's enjoyment of recreational amenities. Thus, a cleaner aquatic environment will lead to more fishing and/or swimming. An approximation to the increased benefits flowing from this cleaner environment is calculated from the actual expenditures laid out to travel to the recreational site.

In contrast to each of the indirect methods for benefit estimation discussed above, there is the DIRECT QUESTIONNING or CONTINGENT VALUATION METHOD. The underlying assumption is that people, when questionned about how much they would value having a cleaner environment, will tell the truth (assuming that they can translate the effects of a cleaner environment into the direct effects upon themselves and then value those effects). There is a vast sociological and psychological literature dealing strictly with how to structure survey questions to get truthful and realistic responses.

This method has an advantage over the indirect methods in that it can be used to obtain values for environmental goods that are currently not being used, but which the householder might wish to use in the future, or to reserve for the use of one's progeny. These are non-use values, and fall into several categories : existence and/or preservation value and option value.

B. SPECIFIC METHODOLOGIES TO DETERMINE BENEFIT MEASURES

B.1. Indirect Methods

B.1.1 Averting Behaviour

The Averting Behaviour Method can be used to value benefits, to either consumers or firms, from an improved level of environmental quality.[3] In either case, we begin with a damage function that relates that quality of the environment to something that is valued by consumers or firms. For consumers, that something might be one's health, and the quality of the environment may refer to the type of air quality one breathes. For firms, the something might be crops, and the quality of the environment may be the purity of the water used for irrigation.

A typical damage function is shown in equation (1). The symbol S refers to the good being valued, Q is the quality of the environment (a non-market good), and A is a private good that can be used as a substitute for the non-market good, Q. For example, in the case of a consumer, the S could be thought of as one's health, Q could be the quality of drinking water available from a local stream, and A could be bottled water available from the local store. Alternatively, for firms, S could represent crop yield,

$$S = S\ (Q, A) \tag{1}$$

Q the amount of rainfall, and A the amount of irrigation water available. Equation (1) is simply then a production function showing how different levels of Q and/or A will alter the crop yield (S). For consumers, the model is called a household production function, and is widely used in models of non-market goods to explain consumer behaviour. In what follows, the valuation of benefits from the consumer's point of view will be discussed. Averting behaviour studies are most commonly used in this context.

The household production function model assumes that households do not derive utility directly from purchased goods, but use these goods as inputs to produce outputs of value to the household. Thus, S is interpreted as a final service flow that gives utility to the consumer. Both A and Q contribute to the level of the service flow enjoyed by the consumer. Thus, the environment enters indirectly into a consumer's utility function.

In order to complete the household production function for the consumer, we must add the utility function in equation (2).

$$U = U(C, S) \tag{2}$$

In this equation, U is the utility level, C is a composite index of all other goods consumed, and S is the something the consumer ultimately values. In what follows, it is assumed that this something is one's health. Thus, henceforth, H will replace S in equations (1) and (2).

We are now able to discuss averting behaviour. It is assumed that consumers can alter their health levels, for some given level of environmental quality, by purchasing market goods that allow them to avoid the deleterious effects associated with the given level of Q. Equation (1) reflects this degree of substitution between averting behaviour and the level of environmental quality to which one is personally exposed. As written, equation (1) assumes that all detrimental effects of pollution are avertible. An extension that relaxes this assumption will be examined later.

The simplest case of averting behaviour to examine is one that assumes that there is perfect substitution between Q and A (the level of environmental quality and the market good used to defend against poor environmental quality). This is clearly a very restrictive assumption, but it does illustrate simply the main point behind the notion of averting behaviour. In order to incorporate the assumption of perfect substitution, we rewrite equation (1) -- which can be thought of as a damage function -- as equation (3). This says that A and Q are complete substitutes for one another and their rate of substitution is "r".

$$H = A + rQ \tag{3}$$

In order to examine the degree to which a consumer would undertake averting expenditures in this case (and, therefore, to derive a value for the benefits of a higher quality of the environment), let us ask what value a consumer would place on having one more unit of environmental quality (Q). Clearly, in the case of perfect substitution, this value is equal to the amount the consumer will save in expenditures on A, assuming that the consumer stays at the original utility level and that prices are unchanging. Thus, the marginal value of environmental quality is the savings in defensive or averting expenditures (P_A*r) as shown in equation (4) (Braden and Kolstad, 1991). This is the unit price of A (P_A)

$$W = P_A * r \tag{4}$$

times the rate at which A is substituted for Q. That is, a one unit increase in Q means "r" less A is required to produce the same level of health (H) as before.

It is, of course, possible to look at a one unit decrease in environmental quality, then the consumer would buy "r" more of A in order to adjust the service flow of H to the original level.

In the first case, of an improvement in environmental quality, this would represent a willingness to pay to obtain the better quality environment. In the second case, it would be a willingness to avoid the deleterious effects of a worsening level of Q. In either case, what is being measured is a compensating variation type measure. That is, the reference utility level is the original one.

The discussion so far has assumed that there is perfect substitution (at least in the consumer's mind), between A and Q. We can look at how the compensating variation measure is modified when less-than-perfect substitution possibilities are available. To do this, we will set up a formal utility maximization model for a representative consumer. This is shown in equation (4). This equation will include a 'full-income' constraint; both income and one's time are assumed to be fixed. In this equation, **C**, is a composite numeraire good

$$Max_{A,Q}\ \mathcal{L} = U(C,H) + \lambda[I + wT - Cq_C - Aq_A - wT_L] \qquad (5)$$
$$where\ q_i = (P_i + wT_i)$$

whose market price is P, I is asset income, w is the market wage rate, q_A is the per unit cost of medical care (or averting expenditure). Finally, T_i is the amount of time lost through consumption of a good. So, the full price to a consumer is both the market price plus the lost wages through consumption.

Maximization of this equation yields the first order conditions for optimization of the consumer's activities, in particular, the activities regarding averting behaviour. The first order conditions can be manipulated to provide a rule regarding the extent of averting behaviour that the consumer should undertake for a marginal change in the quality of the consumer's personal environment. Namely, the consumer should equate the marginal value of personal environmental quality to its marginal cost.

It is the benefits of a marginal reduction in the level of pollution (improvement in the quality of the environment) which are of interest. This can be found by defining and working with the indirect utility function.[4] It shows the maximum level of utility (unobservable) for given levels of income, wages, prices and environmental quality, when the quantities of the composite good and averting behaviour are chosen optimally.

$$v = V(P, w, I, Q) \tag{6}$$

For a marginal change in the level of Q, it is possible to define a compensating variation measure (i.e., if Q increases, what is the amount of income that can be taken away in order to keep the consumer on the same utility level as before). This is a measure of the willingness to pay to have a higher quality environment. Quite simply, this income loss is equal to the savings in averting expenditures needed to reach the original level of environmental quality. This value can be found from knowledge of the technical characteristics of the damage function (H(A, Q)) as shown in equation (6) (Courant and Porter, 1981; Gerking and Stanley, 1986; Harrington and Portney, 1987).

$$\frac{\partial I}{\partial Q}\Big|_{fixed} = - \frac{H_Q}{H_A} q_A \qquad (7)$$

This equation differs from equation (4) in two respects only. First, the rate of substitution between A and Q is fixed at "r" in equation (4), whereas in equation (7) this rate of substitution (given by the ratio of partials of the damage function with respect to Q and A) will vary according to the levels of A and Q. Second, the price of the averting good includes the full price of the good which includes both its market price (P_A), and the amount of time used by the consumer to purchase the good. This simply says that the compensating variation is not constant. Furthermore, the willingness to pay could conceivably be calculated only from knowledge of the damage function (or clean-up production function).

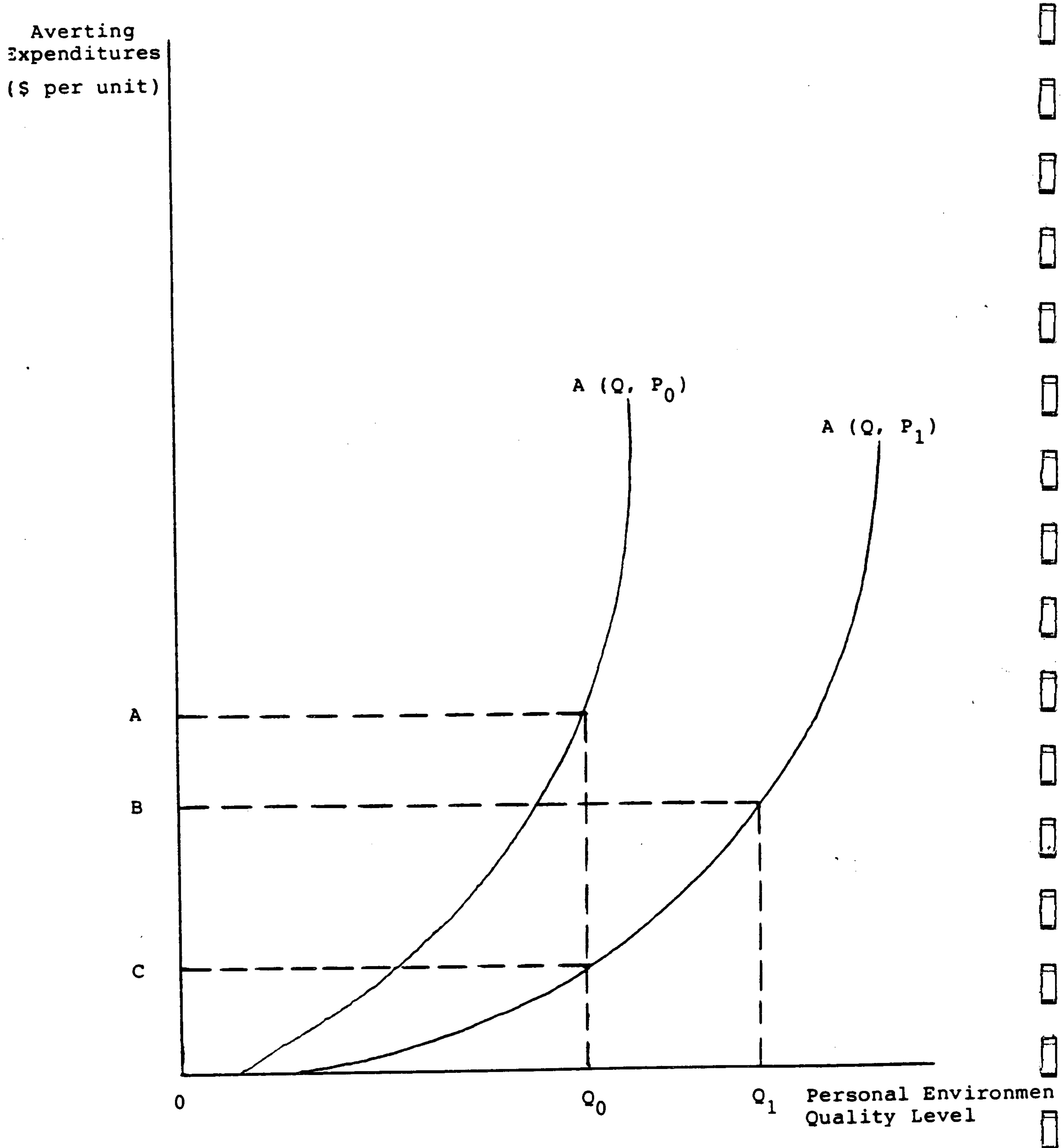
This marginal willingness to pay will depend upon the characteristics of the utility function and the values of the parameters faced by the consumer, but a few comments can be made. First, for a given improvement in Q, the consumer is willing to give up more income, the greater the associated improvement in health, the higher the cost of averting expenditures, and the lower the productivity of those averting expenditures. However, this value only represents the marginal willingness to pay to avoid the deleterious health effects of lower Q for one period in a perfectly certain world (Gerking and Stanley, 1986). Thus, it may provide an understatement of the true benefits of a better quality environment.

This, then, is the theory behind the use of averting expenditures to measure the benefits of an improved level of the non-market good called environmental quality. The question remains as to whether a researcher can take actual observed decreases in spending on substitute goods as a measure of the benefits from increasing Q? Unfortunately, the answer is no. The difficulty is that when Q is increased, there is both a substitution effect (more Q means less A must be purchased) and an income effect (more Q means that the same utility level as previously can be maintained with a smaller expenditure on A and hence, a new higher utility level can be attained by adjusting expenditures on all goods) (Freeman, 1979). Thus, the observed decrease in spending on A will be less than the decrease required to stay on the original indifference curve (at the original level of utility). This is illustrated in Figure 4.1 adapted from Abdalla (1990).

Figure 4.1 depicts averting expenditures as a function of the level of health a consumer enjoys. Thus, averting expenditures are represented on the vertical axis and a consumer's health on the horizontal axis. Two averting expenditure functions are shown for two different levels of environmental quality (Q_0 and Q_1, where Q_1 is greater than Q_0). Suppose the consumer is initially enjoying a health level of H_0. Associated with this is a given level of utility, U_0, when the level of environmental quality is Q_0. The consumer is currently paying A in averting expenditures. Now, suppose that the environment improves to Q_1. This means the new relevant averting expenditure function lies below the old one. To achieve the same utility level, U_0, the consumer does not have to spend as much on averting behaviour.

The theoretically correct compensating variation measure is the distance AC, i.e.,

Figure 4.1 : Averting Behaviour

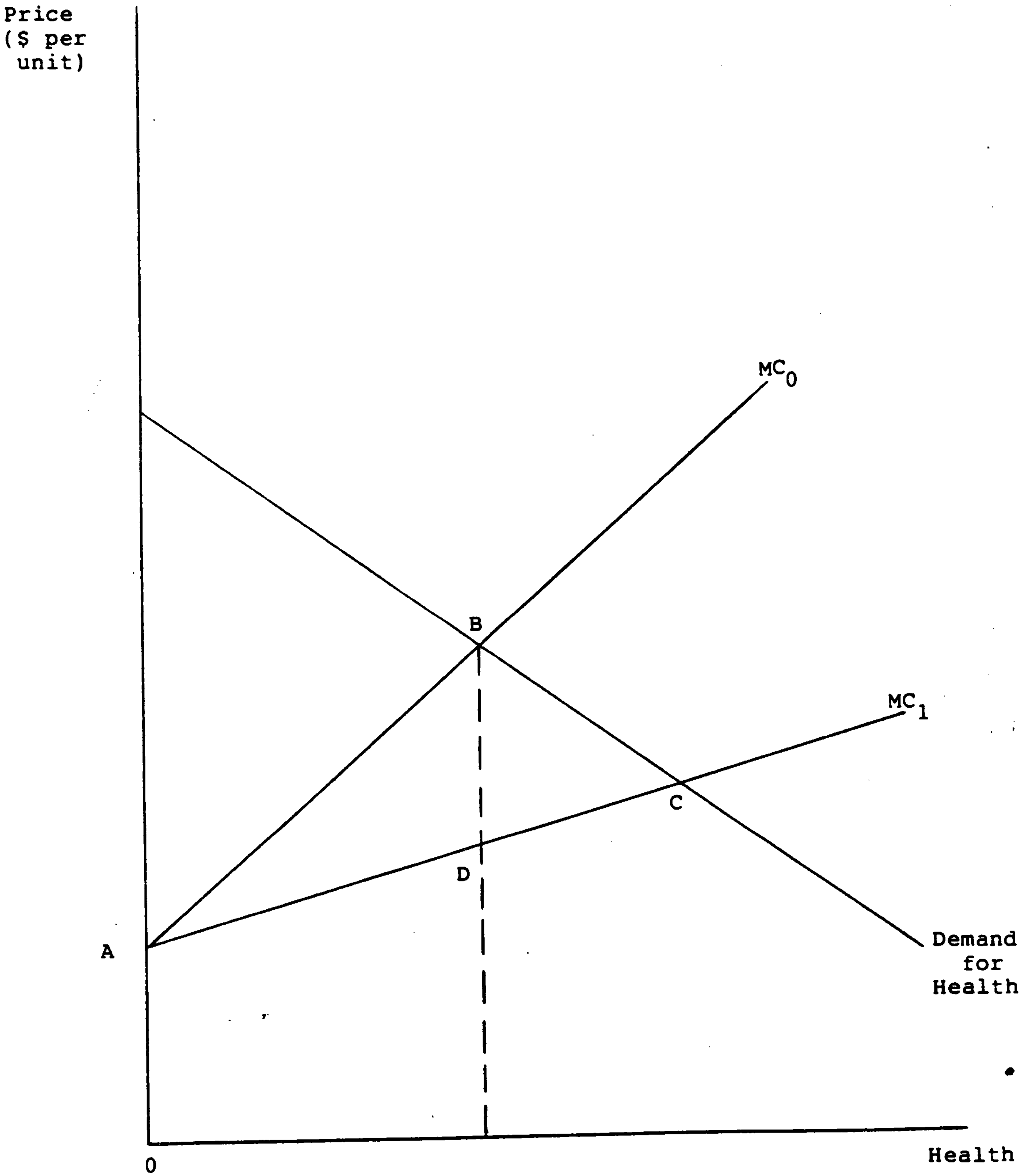


the savings in averting expenditures holding the person at the original health level, H_0 and associated utility level, U_0. However, at the higher level of Q_1, what we actually observe is the consumer operating on the new averting expenditure function, choosing the higher level of health H_1, and the higher utility level U_1. The consumer is spending B on averting behaviour. So, if we use actual observed averting behaviour expenditures to obtain the benefits from improved environmental quality, we would measure only the difference between averting expenditure in the first instance (A), and in the second instance (B), for a savings of AB. Hence, actual expenditures are less than the true benefits to the consumer from a better quality environment.

The problem is that when we observe actual averting expenditures, they may represent an underestimate of the consumer's true benefits (compensating variation) from environmental quality improvement. On the other hand, if the environment Q, and the market good A are complements, then the change in averting behaviour expenditure will provide an upper bound (but, not a lower bound) to the estimate of willingness to pay. That is, if there is an increase in the level of pollution, then averting expenditures may actually improve one's health compared to the original position. In this case, actual averting expenditures overestimate true benefits (Courant and Porter, 1981, Harrington and Portney, 1987).

This analysis has assumed that the change in the level of environmental quality is marginal (very small). We must modify the analysis for when the change is non-marginal; it is not enough to know the marginal value of one more unit of environmental quality. Rather, we would need to know the entire demand curve (or marginal value function) for

Figure 4.2 : Measuring the Health Benefits of a Non-Marginal Change in Pollution



the good affected by pollution, along with the knowledge of the cost function for the good (Cropper and Oates, 1992).[5] Then, as Figure 4.2 shows, a large increase in Q will shift the marginal cost of the good (e.g., health) to the right (Cropper and Oates, 1992). The value of the increase in Q is given by the area between the two marginal cost curves, bounded by the demand curve for health. However, health is not a market good, so it is difficult to estimate its demand curve. The problem with a good like health is that it is produced by the household itself (through the household production function). Thus, the price (or marginal cost) of the good is not observed by researchers; it must be inferred from the marginal cost function. Furthermore, this price is endogenous since it depends upon one's health. An alternative (Bartik, 1988a) suggests using the change in the cost of producing the original level of health (H_0), or area ABD in Figure 4.2, as an approximation to the value of environmental quality change. Recall, that the increase in Q lowers the marginal cost of producing H. This is only an approximation, however. For improvements in Q, this understates the value of the change because it does not permit consumers to increase the chosen value of health.[6] However, this measure needs information from the cost function alone.

So far, the analysis has assumed that pollution damages are completely avertible in the sense that the quality of the environment does not enter directly into the utility function itself. Once Q is assumed to affect utility directly, then the utility function of equation (2) must be modified to look like equation (8) (Courant and Porter, 1981; Harrington and Portney, 1987).

$$U = U(C, H, Q) \tag{8}$$

In this case, the averting expenditure is not necessarily related to the true willingness to pay and it is not necessarily even a lower bound. Now, the averting expenditure can be greater than, equal to, or less than the willingness to pay according to the type of complementary or substitute relationships there are between C, X, and Q. *A priori*, it is not know what these relationships might be. Furthermore, the derivation of the true expression for willingness to pay (benefits) includes an expression which is a derivative of the unobservable utility function. So, one cannot precisely measure what the benefit might be.

One frequently used empirical alternative to averting expenditures for valuing reduced morbidity is the cost-of-illness approach (Lave and Seskin, 1977). This says that benefits from reduced pollution are valued by the savings in direct expenses incurred because of injury or illness, as well as the opportunity costs associated with the injury or illness (largely foregone earnings). Usually, one begins with a dose-response relationship and then uses this information to predict the effect on health status of a change in the quality of the environment. Medical expenses are assessed and increases in sick days are valued at the opportunity wage rate. Among other problems, the cost of illness approach does not include any disutility arising from illness or injury. Nor does it include the averting or defensive expenditures that people undertake.

Harrington and Portney (1987) argue that even if cost of illness and averting expenditures are added together, they will still underestimate true willingness to pay, as long as the increase in pollution lowers averting expenses. The sum will overestimate true benefits, if and only if, individuals respond to increased pollution by undertaking more defensive expenditures and thereby improving their health level.

The remainder of the section describing the theory of the Averting Behaviour Method is devoted to a discussion of the strengths and weaknesses of the approach from a theoretical standpoint. The section on empirical work catalogues further strengths and weaknesses of a more practical nature.

Strengths of the Averting Behaviour Methodology

Averting expenditures are generally relatively easy to measure. Furthermore, it is possible to examine different types of averting expenditure : e.g. those concerned with cleanup costs, those intended to allow avoidance of specific goods/resources, and finally, averting expenditures employed to diminish the effects of short run pollution episodes (Smith and Desvousges, 1986).

More complex models can include an equation describing investment in health, and, therefore, deal with multi-period expenditures. In addition, it is possible to make H (one's health) multidimensional (Gerking and Stanley, 1986). This can allow for different willingnesses to pay for different types of illnesses or health problems attributable to different pollutants. Finally, more complex models can incorporate leisure time in the utility function (Harrington and Portney, 1987), thereby allowing one further avenue through which consumers value having a cleaner environment, a cleaner environment means better health, which translates into more enjoyment of leisure time.

In addition, the model can be formulated to allow for zero-one outcomes (Berger, Blomquist, Kenkel and Tolley, 1987). The outcomes are sick or not sick, and the objective is to maximize expected utility where there are probabilities of each outcome. These decisions can be examined within a random utility framework.[7] This allows for uncertainty to enter the decision-making process. However, the methodology used to measure the benefits of improved environment is the same as described for the earlier models.

Weaknesses of the Averting Behaviour Methodology

The most serious weakness with the averting behaviour approach is that actual observed expenditures can either underestimate, or overestimate, the true willingness to pay (Courant and Porter, 1981; Harrington and Portney, 1987). In order to distinguish which is the case, the researcher must know the person's perceived damage function, since it is the marginal rate of substitution that ultimately defines the imagined benefit of purchased good, and hence the degree of willingness to pay for environmental improvements.[8] Without complete knowledge of the exact relationship between A (market averting expenditures) and Q (non-market environmental good), it is impossible to tell the direction of the bias. Related to this is the issue of perceptions, and how they differ from one individual to another. The models described above hold for a single person, but are often assumed to hold for all people. We know that there is heterogeneity among individuals; their subjective damage functions are, therefore, not likely to be identical.

In addition, there are further theoretical problems with the averting behaviour model. First, most models assume a single period world with certainty. Thus, expenditures are calculated only for the single period. They would tend to be an underestimate of the true lifetime willingness to pay to obtain a better quality environment.

Secondly, the averting behaviour models assume that there is non-jointness in production for the household. In other words, averting behaviour provides no utility to households other than in its defensive role. However, it is often the case that items purchased for an averting purpose fill other needs of the consumer. For example, an air conditioner may reduce the level of air pollutants inside someone's home, but it also provides cooling. While it is hard to separate the benefits of these two disparate roles, one can say that the actual averting expenditure would be an overestimate of willingness to pay to avoid air pollutants.

Thirdly, some effects of pollution cannot be controlled by defensive expenditures (Bartik, 1988a). The models above generally assume that this is the case. In addition, the models assume that there are no significant adjustment costs associated with reducing the level of investment in defensive expenditures. It may in fact be expensive to adjust, so we observe no or little defensive expenditures, when in reality the true benefits of an improved environment be large. For example, one type of averting behaviour would be to move away from the source of pollution. This, however, means uprooting the family, and there may be a psychic threshold cost of moving.

B.1.2 Hedonic Market Method

B.1.2.1 Property Values and Environmental Amenity Benefits

The Hedonic Market Method takes as its starting point the assumption that the market price of a good can be decomposed into its constituent attributes, and thus a marginal willingness-to-pay for one more unit of each attribute can be determined. The hedonic method has been used most successfully in valuing environmental disamenities in urban areas (i.e., air quality, noise), and in valuing the morbidity and mortality risks attached to particular jobs. It has also been used, although less successfully, in conjunction with the travel cost method to value different attributes at recreational sites. (See subsequent discussion on the Travel Cost Method.)

The basic theoretical framework is that the price one observes in the market can be expressed as a function whose arguments include the various attributes of the good. For example, if the good purchased is a house, then attributes of the house include the number of rooms, the lot size, the number of bathrooms, etc., as well as a set of attributes that are related to the location of the house, and the noise or air quality surrounding the location. This allows the construction of the so-called hedonic, or implicit, price equation shown in equation (9). The variables in this equation include P_H is the observed market price; S,

$$P_H - P_H(S, N, Q) \tag{9}$$

a vector of site characteristics (e.g., number of rooms, pool, etc.); N, a vector of neighbourhood characteristics, and Q, a measure the quality of the environment. This, too, can be defined as a vector of different characteristics.

Then, the marginal implicit price (value) of a characteristic (e.g., a marginal improvement in air quality) to the consumer is the partial derivative of the hedonic market price function with respect to that attribute. This represents the amount (benefit) that a

$$\frac{\partial P_H}{\partial Q} - P_Q(Q) \tag{10}$$

consumer would be willing to pay, for an environmental improvement in air quality, for example. As shown, this says that the marginal willingness to pay depends upon the quantity of the attribute.

In making this step, this model adopts a strong assumption regarding the behaviour of consumers. It assumes that each household is in equilibrium with respect to a given set of housing prices that just clears the market for a given stock of housing and attributes. If this is the case, this implies that for each household the marginal implicit prices associated with the housing bundle (house plus characteristics) actually chosen are equal to the calculated marginal willingness to pay for each of the characteristics.

The next step is to use the information about the quantity of the characteristic, along with the marginal implicit price for a characteristic, to obtain a inverse demand function for that characteristic.[9] The inverse demand function says that the price (or willingness to pay) is a function of the quantity of the characteristic and variables that explain tastes: e.g., socioeconomic variables such as income, age, etc.[10] This is shown in equation (11).

$$WTP_i = WTP(Q_i, Y, A, \ldots) \tag{11}$$

The final stage belongs more in the realm of the empirical but has some theoretical aspects, so will be included here. In general, in order to determine the demand (or inverse demand) function, one needs to know something about the supply function of the implicit good. The three possibilities are : 1) that the supply of housing with given bundles of characteristics is perfectly elastic at observed prices, 2) that the supply is perfectly inelastic, and 3) that supply is neither of the above two. In the first case, one regresses the observed level of the characteristic on the observed implicit prices defined by equation (10), along with actual incomes and other socioeconomic characteristics. This identifies the demand function. This assumes that the quantities adjust to the given market price. In the second case, one simply regresses each household's marginal willingness to pay (as measured by (10)) against the actual quantity of the characteristic, along with the incomes and socioeconomic variables. This assumes that the quantity of housing (and attributes) is fixed, so that the price must adjust. Finally, the third case is the most complex because the quantities supplied and demanded of the characteristic are both functions of prices. This requires the simultaneous estimation of both the demand and supply functions. In order to make this operative, a housing supply function must be developed.

In all three cases, in order to determine the benefit associated with a marginal change in the level of the environmental quality Q, one simply has to look at the WTP from equation (11). Aggregate benefits would involve summing over each household's WTP. Note, however, that because these values are derived from Marshallian (uncompensated) demand functions, that the measures are of consumer surplus, not compensating or equivalent variation. However, for a non-marginal change in Q, the normal consumer surplus approach is adopted. That is, one plots the inverse demand function, showing willingness to pay against the quantity of the environment as in Figure 4.3. Suppose environmental quality increases from Q_0 to Q_1. Then, the consumer surplus is equal to the area ABCD. Unfortunately, this is simply a short-run value, for in the long-run the change in the environmental amenity may cause housing prices (equation (10)) themselves to change. To value this long-run change, one must include the household's adjustment to the amenity change along with any other price changes that result (Bartik, 1988b). However, the area identified above is a lower-bound value for the long-run benefits of amenity change. An upper bound can be given by knowledge of the hedonic price function itself (Kanemoto, 1988).

Strengths of the Property Value Approach to Environmental Disamenities

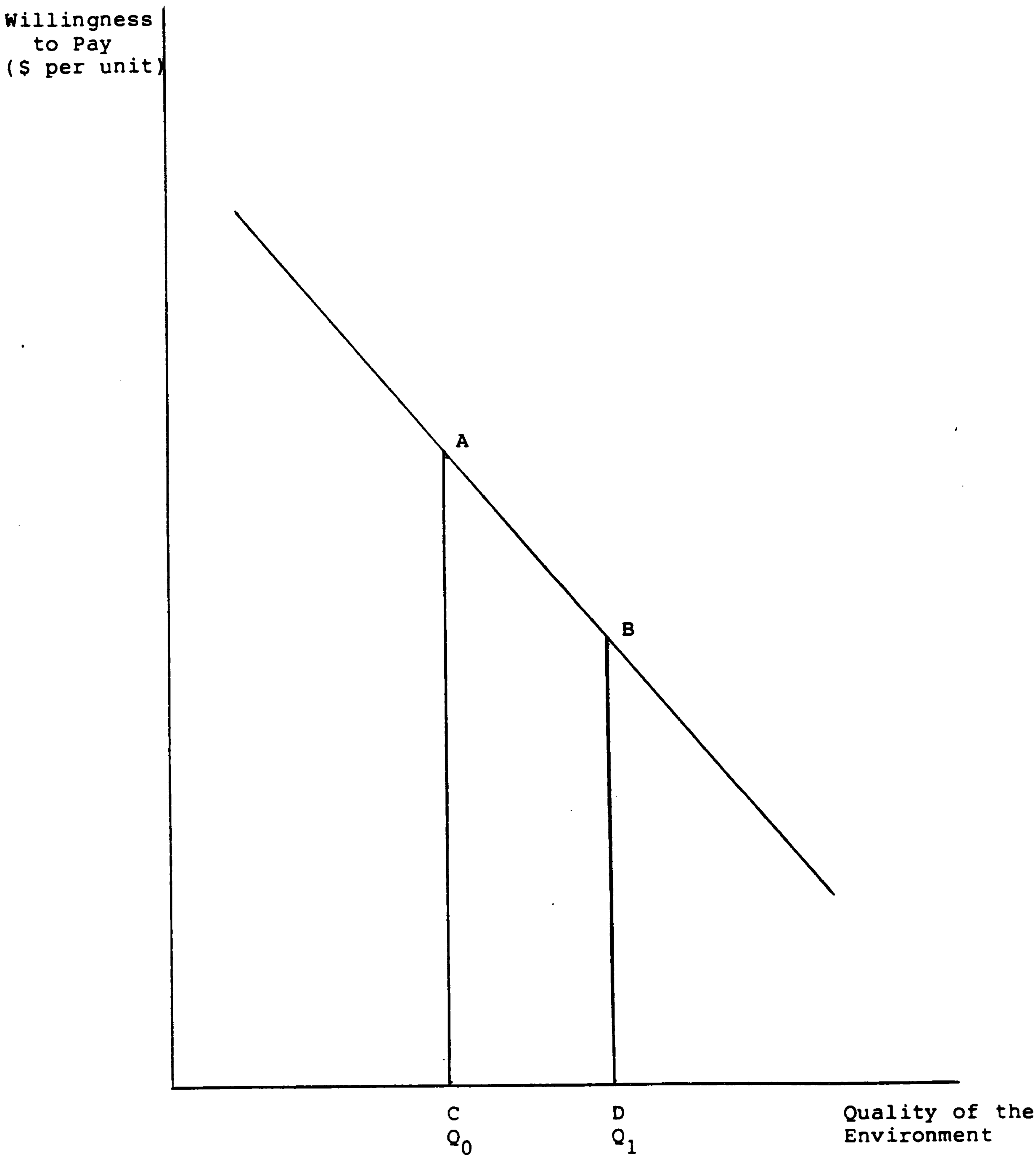
According to Cummings, Cox Jr., and Freeman (1986) the major strength of the Hedonic Market Model is that, in theory, it will give the researcher the market equilibrium value of a public good (e.g., the natural environment). Given that economists believe that market values provide the best information, this is an important item.

Weaknesses of the Property Value Approach to Environmental Disamenities

There are, however, a number of weaknesses with the Hedonic Market Approach, especially in the context of the evaluation of the benefits from environmental changes. First, households are assumed to have perfect information on all prices and attributes in the market. Transaction and moving costs are assumed to be zero. Also, markets are assumed to clear instantaneously. If these assumptions are not met, then the estimates of benefits are likely to biased either upward or downward. Secondly, the models assume that households can perceive differences in air quality, and incorporate this information into their purchase decisions (Freeman, 1979). If they are unable to do this, then the model is no longer valid and the results suspect.

Thirdly, for every household to be in equilibrium, there must be a sufficient selection of properties. Any household settling for a house that may not meet their preferred criteria fully will not maximize utility. The large urban areas usually chosen for empirical studies generally have enough variation in housing units to ensure that the smooth continuous implicit price function is a reasonable approximation of reality.

Figure 4.3 : Using the Hedonic Market Method to Measure the Non-Marginal Change in Environmental Quality



Fourthly, Cummings, Cox Jr., and Freeman (1986) state that the benefits calculated from a housing property study may understate the full value of a cleaner environment because the benefits miss out on benefits of a cleaner work environment, as well as a living environment.

B.1.2.2 Hedonic Wage Models and Morbidity Valuation

Hedonic price models have also been used in the labour market. From the workers point of view, a job is simply a bundle of different characteristics, including working conditions, prestige, and degree of risk of injury or exposure to toxic chemicals. The purpose of the hedonic wage model is to isolate that portion of labour rental payments which compensates employees for changes in on-the-job risk (Thaler and Rosen, 1975). That is, certain jobs are more risky than others; in order to induce employees to take on more risky jobs, they are generally compensated through differences in wage rates. Use of hedonic price function allows the researcher to calculate the implicit marginal prices for various levels of risk. This information can be used further to determine the value of safety on the job or the saving a life.

The model (Thaler and Rosen, 1975) begins with an expected utility function where expected utility is a function of Y, the prospect of a certain income. Workers are assumed to be risk averse, so the first derivative of U(Y) is positive, i.e., more income is better, and the second derivative of U(Y) is negative, the rate at which more income is better diminishes with greater income. Expected utility is then simply a weighted average (where the weights are the probabilities (p) of different states of the world occurring, i.e., no accident versus an accident) times the incomes in the different states.

$$E(U) = (1-p)\cdot U\,[W(p) - \frac{p}{1-p}\cdot I] + p\cdot U\,[(1-k)\cdot W(p) + I] \tag{12}$$

In this equation W(p) is the wage rate associated with each value of p; I is the amount of insurance purchased; (p/(1-p)) is the price of insurance for a job with risk p, k is the constant cost of injury (1 > k > 0). The expression W(p) - (p/(1-p))I is the worker's net income if an accident does not occur; the expression (1-k)W(p) + I, is the net income if an accident does occur.

The objective is for the worker to choose p (probability of a 'standard' accident) and I (amount of insurance purchased) in order to maximize expected utility. Differentiating (12) with respect to I, setting the result equal to zero and substituting this back into (12) yields the following expression for expected utility of income:

$$E = U[(I-pk)\cdot W(p)] \tag{13}$$

This simplified expected utility function depends only on the optimal choice of p, conditional upon the prior choice of optimal insurance coverage. This says that the worker chooses p optimally, such that the change in insurance payment is equal to a change in income. An acceptance wage (i.e., the portion of labour rental payments which compensates for changes in on-the-job risk) is the payment necessary to make the worker indifferent between two jobs offering different risks (conditioned on the purchasing of optimal insurance coverage). Thus, the wage differential between a more risky and less risky job reflects only actuarial differences in the risks between the jobs. This may be considered an estimate of how much a worker is willing to compensated in order to accept a little more on-the-job risk.

Strengths of the Hedonic Wage Approach

This model may be useful in conjunction with the property value model in order to generate estimates of the full benefits of a cleaner environment. This is particularly important since wage levels vary from city to city, and variations in wage rates help to compensate individuals for environmental disamenities.

The model can be extended to look at situations in which there is a positive probability of death related to job activity. In this case, the researcher can measure the risk premia associated with jobs offering a higher risk of death than other jobs.

Weaknesses of the Hedonic Wage Approach

Hedonic Wage Models requires that the researcher adopt a number of restrictive, and not necessarily realistic assumptions. First, workers are assumed to have the same skills and personal characteristics. In particular, workers are assumed to have identical attitudes toward risk of death or injury. This attitude is also independent of their exogenously acquired skills. Interpersonal differences in physical capacity to cope with job risks are zero.

Secondly, jobs exhibiting equal risks are identical. In other words, the identity of the firm/employer is irrelevant. Furthermore, risks are described with certainty and a known probability. Death and 'pain and suffering' are ignored because the model uses only a 'standard accident'. Thaler and Rosen (1979) relax some of these assumptions, but the basic result remains unchanged.

Thirdly, it is assumed that there is equilibrium in the worker/risk market. That is, there are an equal number of workers applying for jobs at each value of risk and the numbers of jobs offered at each risk.

Fourthly, a perfect insurance market is assumed to exist. That is, the cost of insurance equals its actuarial value and there is perfect information on risks/probabilities. In addition, the model discounts the possibility of a moral hazard problem, whereby a person purchases insurance, then changes his/her behaviour to act in a more risk-loving manner.

B.1.3 Travel Cost Method

The Travel Cost Method has been used for a more than 30 years to value benefits associated with environmental resources used by consumers for recreational activities (Trice and Wood, 1958; Clawson, 1959). The important insight that initially drove this method is that the consumer must incur both explicit out-of-pocket costs to travel to the chosen site and participate in particular activities, and implicit opportunity costs in the form of lost work time (Hotelling, 1949). Thus, the determinants of the demand for visits to a recreational site include the costs (explicit and implicit) of travelling to, and entering the recreation site, along with another costs associated with doing the recreation, and, more importantly, the quality of the recreation site. Once a demand function is established that includes site quality as a factor influencing the number of visits, then it is possible to use the demand function to measure the benefits associated with site quality improvement. This is shown in Figure 4.4.

Figure 4.4 : Benefits of an Improvement in the Quality of a Recreational Site

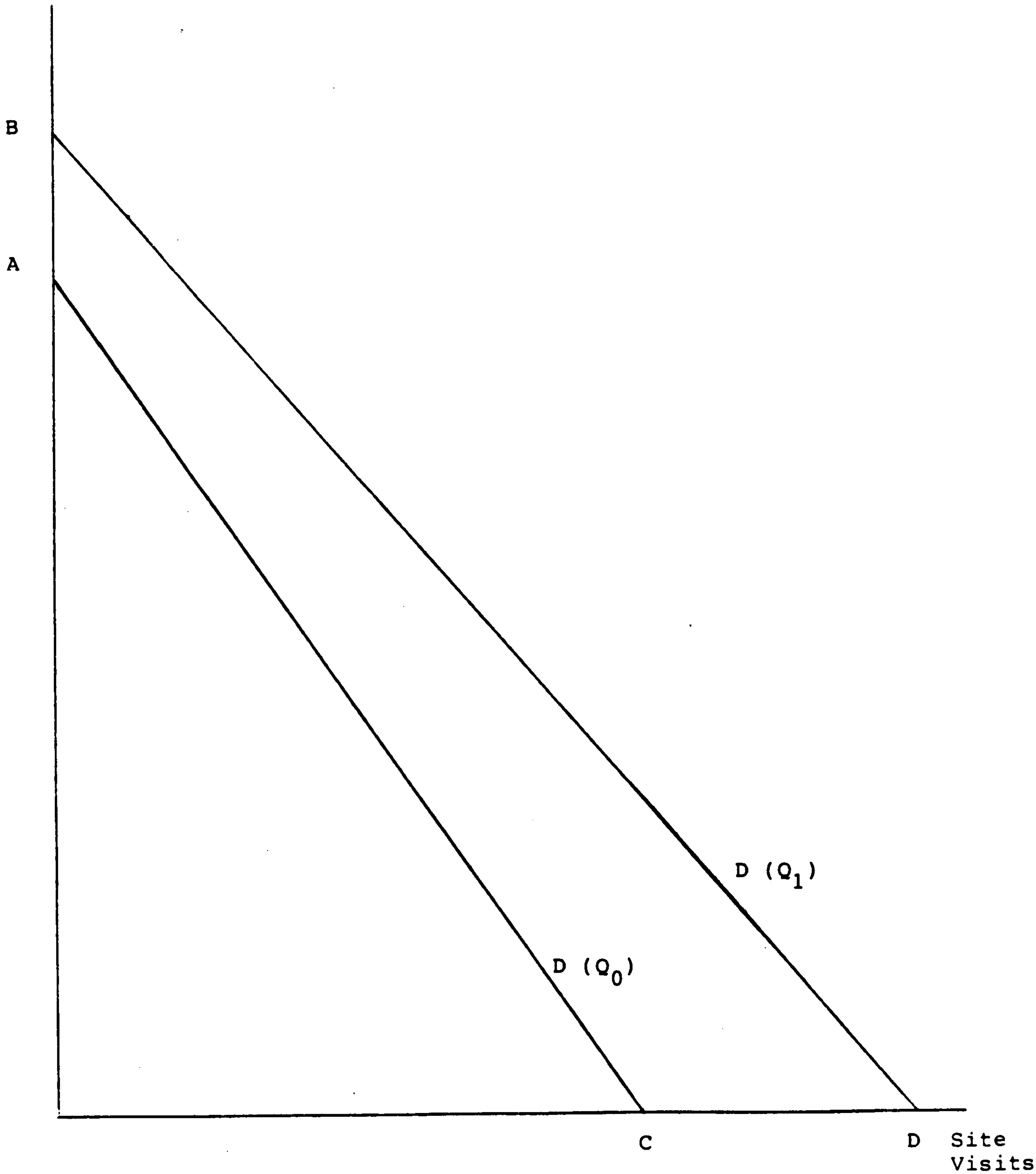


Figure 4.4 shows the demand for site visits to a particular recreational site as a function of the price of site visits. This includes all costs, both explicit and implicit, incurred by the consumer to travel to the site. While there is no explicit market price, the consumer nevertheless must pay to enjoy the recreational activity. The figure shows two demand curves. The first is the demand for visits, assuming some initial level of environmental quality of the site (Q_0), and the second shows a new higher level of demand predicated upon an improvement in the quality of the environment (Q_1). Benefits from the improved environment are then measured by the area ABCD.

The key assumption that allows the researcher to go from the development of the demand curves in Figure 4.4, to the measurement of benefits like area ABCD is the assumption of weak complementarity (Maler, 1974). This is central to the Travel Cost Method. This assumption says that environmental quality is a weak complement to the good or activity observed (i.e., recreational activity such as boating, fishing, swimming, canoeing, etc.). This assumption has two important implications. First, the marginal utility of environmental quality is zero if none of the good is purchased (i.e., there are no visits to a particular site). Secondly, there will be some price (a choke price like B in Figure 4.4) above which no recreational trips will be undertaken; if the travel costs are greater than B, no one will visit the site. Several other assumptions are made by users of this model. First, there is a homogeneity of the on-site outputs of trips across site visitors. Secondly, consumers take trips for sole purpose of visiting the site; they do not derive utility from any other aspect of the visit.

The Travel Cost Method of valuing environmental amenity benefits begins with the specification of a utility function. The first models assumed that trips to a recreational site enter the utility function directly, as in equation (9) (McConnell, 1975; Freeman, 1979). Since the early models also concentrated upon finding the demand for a single site without substitutes, it is repeated here.

$$U = U(X, V, Q) \tag{14}$$

In equation (14), utility depends directly upon a composite market good (X), the number of visits (V) to the recreational site, and the environmental quality of the site (Q). The consumer has a budget constraint of the form given in equation (15), and a time constraint shown in equation (16).

$$Y = P_X \cdot X + P_V \cdot V + c \cdot Z \tag{15}$$

$$T = a \cdot V + t \cdot Z \tag{16}$$

In these equations, Y is nominal income, P is the price of the market good, P_V represents the "price" of the recreational good (e.g., entrance fees, equipment costs, etc.), V is the number of visits to the site, Z is the total distance travelled to the site, c is the unit cost per mile of travel, T is the total time available for recreation, t is the travel time per mile to the site, and a is the time on site per visit.

Constrained utility maximization yields the demand curve for the recreational site. Note that this demand curve depends upon the environmental quality of the site, as well as travel costs, and time spent, etc., as shown in equation (17). Note in equation (17), that D is the one time return distance to the site, so that Z (total distance travelled) is equal to the product of D (distance) and V (number of trips). Note also that the time on site per visit is assumed to be fixed for the site.

$$V = V(P,\ P_v,\ D,\ c,\ t,\ h,\ Q,\ Y) \tag{17}$$

The demand curve in equation (17) is for a single individual using the site. In order to obtain the aggregate demand curve presented in Figure 4.4, we need to sum horizontally the demand curves of all individuals using the site. It is important to note several points. First, the number of visits (or demand for a particular recreational site) depends upon Q, the quality of the environment at the site. Thus, an increase in Q will shift the demand curve to the right. One can then recalculate the area ABCD in Figure 4.4.

The second important point is that this is a Marshallian demand curve, so that we are measuring benefits using the consumer surplus approach from Chapter 3. So, the benefit measure obtained is neither of the theoretically correct measures, the compensating and equivalent variations. Recall, however, that Willig (1976) shows that in most cases, the consumer surplus differs from either of the true benefit measures by about 5 %.

The final comment is that this approach implicitly assumes either that there are no substitute sites, or that distances/travel times to other sites and quality levels at other sites are all held constant. This is a serious problem with the early Travel Cost Models (known as the Clawson-Knetsch Travel Cost Method). Namely, it is site specific. It is reasonable to believe that a consumer's decision to visit a particular recreational site may depend upon not only prices and qualities of that site, but also the prices and qualities of alternative sites; this information must be incorporated into the demand function. Thus, the value measured by area ABCD could be the result of extra benefits flowing from the particular site *plus* diverted activity from other lower quality sites, i.e., there is a price-quantity relationship and a quality-quantity relationship which the site specific model does not distinguish between. The Clawson-Knetsch technique must be generalized for estimating systems of demand functions for several sites and for multiple quality changes.

Since the first models, there have been modifications and improvements to the basic travel cost notion. In particular, the later models have tried to incorporate multiple site selection explicitly into the consumer's decision making. An important aspect of these attempts have been increased emphasis on the use of time and its valuation in the models.

One class of models (Smith, Desvousges, and McGivney, 1983; Smith, 1989) assumes the existence of a household production function; more specifically, of a household recreation production function. In this case, recreational activities are merely inputs into the production of services from which consumers derive utility. So, the relevant utility function looks like that in equation (18). In equation (18) utility is a function of the service flows from recreational activities (S_r) and other non-recreational activities (S_o). It does not contain the number of visits to recreational sites directly.

$$U = U(S_o, S_r) \tag{18}$$

There is also a household recreation production function like equation (19).

$$S_r = S(X_r, V_1, V_2, t^1{}_{V_1}, t^1{}_{V_2}, t^2{}_{V_1}, t^2{}_{V_2}, Q^1{}_{V_1}, Q^1{}_{V_2}, Q^2{}_{V_1}, Q^2{}_{V_2}) \tag{19}$$

In equation (19), we assume that for the household to produce the recreational service flow (S_r), it uses a market purchased good (X_r), along with two visits to two different sites (V_1 and (V_2), the time spent at each of the two recreation site on each of two visits (t^1_{V1}), and the quality of the two recreation sites at each of two visits (Q^1_{V1}). There is a companion non-recreational production function that depends upon only market goods and time. Finally, the model needs a full income budget equation describing how one's income and time allotment can be no larger than the expenditures and time spent producing recreational and non-recreational service flows. This equation looks like the following:

$$Y = w\cdot\bar{T} + I - \{ \sum_{j=1}^{2} [(c\cdot d_j + r t_j) V_j + w \sum_{l=1}^{V_j} t^j{}_l] + w t_o \} \tag{20}$$
$$= P_r \cdot X_r + \sum_{k=1}^{C} P_k \cdot X_k$$

Equation (20) says that one's income (Y) is equal to one's wage income (w^*t_w) and some nonwage income (I). Total time available is T and this must be used to travel to and spend time at a recreation site as well as used on the acquisition of non-recreational good (t_o). Travel time to each site is given by t_j and the opportunity cost of this time is a constant r. The symbol c represents the per mile vehicle-related costs of travel and d_j is the round-trip distance to each site. Finally, P_r is the cost of the market good used in producing recreation (X_r) and P_k is the per unit cost of the non-recreational items purchased by the consumer (X_k). In essence this constraint says the times spent on non-recreational service flow activities must add up to total time available) and that one's income is equal to the expenditures on market goods.[11]

Again, the objective is to use constrained utility optimization with the above equations (18)-(20), in order to derive the set of demands for recreational visits to various sites. The first order conditions can be rearranged to obtain a system of two demand equations, one for each of the two recreational sites, as in equation (21)

$$V_1 = V(P_r, \sum_{k=1}^{C} P_k, c, r, d_1, d_2, t_1, t_2, Q_1, Q_2, Y) \tag{21}$$

Once again, equation (21) represents the demand curve for a recreational site by a single consumer. In order to obtain the aggregate market demand, each of the individual demands would be summed horizontally. One would continue as before with Figure 4.4, perform a comparative statics exercise that changes Q, and observe where the new demand would lie. Benefits would again be like area ABCD, using the consumer surplus approach. However, now when examining the benefits from increasing Q at a particular site, we would need to be cognizant of the effects upon other sites. That is, if demand for site 1 increased, this might occur directly at the expense of a reduction in the demand at site 2. The net benefits of the change in Q would then have to be calculated; site 1 would have a larger consumer surplus, but site 2 a smaller one. The difference would be the net benefit of altering Q.

The second method adopted to deal with the issue of multiple sites represents a departure from the previous models. This is the random utility model. Instead of determining a demand function for the number of trips, it predicts the probability that a consumer will choose a particular site from among a set of choices. It makes a number of important assumptions. First, it does not assume that one's decisions to undertake recreational activities are continuous (as do the previous models). Rather, it is a discrete choice model; it concentrates upon the choices among substitute sites for a single recreational trip (so it cannot predict how many trips are taken).[12] Secondly, it assumes that decisions are independent across trip occasions. Thirdly, it assumes that a consumer compares the utility that could be realized from all other related decisions, conditional on the selection of a recreation site. Fourthly, the conditional utility function (conditional upon choice of site) is assumed to be stochastic in so far as the researcher is concerned.

The random utility model begins with the specification of a utility function. In this case, it is typical to use the indirect utility function. This is the function depends upon income, prices, and characteristics of the good. [13] The problem is that the consumer faces a set of choices (e.g., recreational sites that she could visit on a particular trip), but is actually observed to choose only one of the options. Implicitly, this tells the researcher something about the non-chosen options. One assumes that the choice made is the one that gives the consumer the higher utility. However, there is an element of randomness. It may arise either on the researcher's side, in the sense that the characteristics of the consumer or the site are not measured correctly, or it can enter on the consumer's side. They too can make errors in the calculation of utility from particular sites. Thus, the perceived (indirect) utility from the choice of site j can be written as follows:

$$\hat{V}_i = V(P_i, Q_i, Y) + \epsilon_i \tag{22}$$

In equation (22) V is the utility of location i and it has both a deterministic component fully described by site qualities (Q), site prices (P), the consumer's income (Y), and a stochastic or random component, epsilon$_i$. Each site has a different utility value, and the largest is the one the consumer actually obtains.

The random utility model calculates the probabilities of choosing different locations according to personal and site characteristics.[14] Simply, the probability of choosing site i is given by the following. Then, assuming that the error terms are independent and

$$Prob\ (site = i) \ = (Prob\ (V_i - V_k > \epsilon_k - \epsilon_i)) \tag{23}$$

identically distributed extreme value distributions, one can rewrite the probability of a consumer choosing site i as in equation (24) (Bockstael, McConnell, and Strand, 1991).

$$prob\ (i) \ = \ \frac{exp^{V(P, Q_i, Y)}}{\sum_{j=1}^{N} exp^{V(P,\ Q_j,\ Y)}} \tag{24}$$

Equation (24) is a multinomial logit model. One of its major drawbacks is that it implies that the probabilities of choosing site i, over site k, depends exclusively on the qualities and prices of the two alternatives alone and no other sites. This can be dealt with by using a nested version of the multinomial logit model. In this version, there is a sequence of nested alternatives. For example, if my decision is to go fishing as opposed to boating, then I next consider a target species for the trip, following that I make my choice of a fishing site.

Since we do not estimate a demand curve with a random utility model, the procedure to obtain welfare or benefit measures is somewhat different from that described above (Small and Rosen, 1981; Hannemann, 1982). The fact that it is assumed that the decision to choose a site is dependent upon the indirect utility function is useful. Thus, instead of using a consumer surplus measure, it is possible to go directly to the theoretically preferred compensating variation measure (see Chapter 3). Then, suppose there is an increase in the quality of environment at site i. Further, suppose that site i is chosen both before and after the change in quality. Then, the compensating variation of this quality change is given by C in the following expression (Bockstael, McConnell, and Strand, 1991).

$$V_i(P_i, Q_1, Y - C) + \epsilon_i = V_i(P_i, Q_0, Y) + \epsilon_i \tag{25}$$

The estimation procedure would have estimated coefficients for the various arguments in the indirect utility function, and these would be used with values for the prices, income, and quality levels to find a value of C.

The difficulty with the above is that it is not known *a priori* whether the consumer will choose site i. What if the consumer chooses site j instead? In this case, the consumer does not obtain additional benefit from site i's quality improvement. We can only predict the probability of a site being chosen, not whether it actually will be chosen.

To deal with this uncertainty, researchers have developed a measure of compensating variation that equates the expected value with the maximum of the indirect utility functions. If the marginal utility of income is non-constant, then the compensating variation can be approximated by equation (26) (Bockstael, McConnell, and Strand, 1991).

$$C = \frac{\sum_{j=1}^{N} exp^{V_j(Q_j^0)} - \sum_{j=1}^{N} exp^{V_j(Q_j^1)}}{\sum_{j=1}^{N} \gamma_j exp^{V_j(Q_j^1)}} \tag{26}$$

Note that if the probability of choosing site i is small and the quality change occurs there, then the expected compensating variation will be small.

In each case, when one is using random utility models, it is important to note that the welfare measure is the compensating variation for a single trip. There are various approaches used to obtain seasonal compensating variation measures including multiplication of the total number of days in the season by the per trip measures (Caulkins, Bishop, and Bouwes, 1985). Ideally, one should allow for the number of trips to be a choice variable as well. One way to do this is to estimate both a random utility model of site selection and a traditional Travel Cost Model in order to get an estimate of the number of trips taken following upon a quality change. The estimated number of trips and the per trip expected compensating variation measure in equation (26) are then multiplied to get the seasonal benefits.

While the above discussion is straightforward in theory, there have been difficulties with putting it into practice. One difficulty lies in the fact that while there is variation in the costs of travelling to the site for different consumers, there is no variation in the site quality among persons who visit the site.[15]

There are a number of approaches that attempt to solve the quality variation problem. In the first, one could estimate an equation like (21) using cross-sectional data for recreational demand at several sites where site qualities differ. This is not the preferred approach. The first problem is that it does not incorporate the notion of substituting to alternative sites into the parameter estimate, and thereby biasing them, and any benefit measures calculated using them. Secondly, there is no guarantee that this type of pooled regression will not incorporate other biases into the estimation process.

In the second approach, the researcher can try to collect time series data for a given site. One hopes that the time series will provide sufficient variation in the site quality variable so that estimates can be obtained for the parameters in the demand function. This is very difficult and expensive to do.

In the third approach, the varying parameters model (Vaughan and Russell, 1982; Smith and Desvousges and McGivney, 1983), addresses the issue of site quality variation by assuming that site quality enters recreation demand functions multiplied by travel cost or income, both of which vary across households. In this way, the researcher can detect changes in demand due to quality differences among sites. A simplified demand equation is given in (27), where V_j is the number of trips to site j, Q_j is the quality at site j, I_i is the income of the ith person, C_i is the travel cost per unit of the ith person, and a, b, and c are the parameters to be estimated. As shown, equation (27) does not allow for multiple sites, although this can be incorporated through the inclusion of more parameters relating to alternative sites.

$$V_j = a + b(Q_j) \cdot C_i + c(Q_j) \cdot I_i \tag{27}$$

A final approach taken to the valuation of quality in the context of the travel cost approach is a union of the Hedonic Market Models and Travel Cost Models. The objective of the Hedonic Travel Cost Model is to estimate the demands for site characteristics (Brown and Mendelsohn, 1984). Notice the change in focus. Previously, one estimated the number of trips or the probability of a trip to a particular location. As with all hedonic models, this begins with the assumption that attributes that are valued are priced at the margin. Thus, one regresses travel costs incurred in visiting a site on the quality characteristics of the site. The marginal cost of each quality characteristics is determined from the estimated equation and then used in estimating the demands for each characteristic. There are several problems with this approach. First, while the hedonic method works reasonably well for goods in which buyers and sellers actually participate, this is not the case for the provision of recreational services. There is no market for the good. Secondly, the relationship between travel costs and the level of quality characteristics is determined by nature, not by a market. Thirdly, the costs should include opportunity costs which are inherently unobservable and different for each person. This approach has not been vigorously pursued.

Strengths of the Travel Cost Model

The Travel Cost method focuses upon observable purchase of inputs in a series of markets to determine the benefits of amenities at chosen sites. Thus, actual transactions reflect real preferences are the basis for the analysis. Furthermore, the measures of benefits derived from these models are obtained under the assumption that each individual consumer is maximizing his or her utility through independent decision-making. Thus, each person's preferences are being respected.

It is a relatively low cost method for obtaining benefit estimates in the case of a single site.

The random utility model greatly expands the types of recreational behaviours that can be evaluated. Furthermore, the measure of benefits is the theoretically preferred compensating variation.

The Hedonic Travel Cost Method allows the researcher to value each of several characteristics changes on a given site. This can be useful since consumers frequently desire different characteristics in different ways.

Weaknesses of the Travel Cost Model

The most serious deficiency of the Travel Cost method is that it fails to measure the full value of environmental resources. It cannot measure the non-use benefits associated with a recreational site. In order to measure benefits, it requires that individuals actually make visits to the sites in question. Related to this deficiency is a second serious one. In general, the more remote or inaccessible or unused a site is, the lower the value a Travel Cost Model will put on the site. This is related to the fact that the Travel Cost Model relies upon actual visits to measure benefits. One can generalize this difficulty of the Travel Cost Model in the following way. It is suited only to valuing sites *ex post*, and not at all useful for valuing sites *ex ante*. (For example, it would not provide good measures for the recreation opportunity costs associated with the costs of flooding a remote/unused valley associated with a proposed hydro-electric scheme.)

With the Travel Cost Method, it is sometimes difficult for the researcher to determine what effects changes in the relevant measure of environmental quality have on the site demand curve. Researchers frequently take observed physical changes in the environment as being the appropriate measures; however, what ultimately matters is how individuals perceive these quality measures.

The model pertains only to single purpose visits, and thus, does not do a good job of allocating costs for multipurpose visits. This has implications for the benefits of a change in the environmental quality of one of the sites on a multipurpose visit.

A further weakness in the Travel Cost Model is that it assumes that the supply of attributes in a site is fixed. This is because the time frame in which the data are available does not allow for major changes in the site to occur. However, short term effects such as overcrowding, probably do have real affects on demand. They have recently been incorporated into the Travel Cost Model (Wetzel, 1977; McConnell, 1980).

In addition, the benefit estimates generated by the Travel Cost Model turn out to be very sensitive to a number of key parameters in the model, such as, the geographic definitions of the site(s), assumptions about who the relevant users of the site are (e.g., does the researcher include only individuals living within a certain distance of the site), and the values imputed to the individual's time spent travelling as opposed to on-site.

There are two aspects to the problem of travel time and its valuation. First, Smith, Desvousges, and McGivney (1983) have been critical of the use of the average wage rate (w) as a proxy for value of time. They argue that not all time is the same to all individuals, e.g. Monday mornings vs. Saturday afternoons have different values. Secondly, the opportunity cost of time varies disproportionately with distance. The differences are especially great when one is comparing benefits from 'local' and 'national' sites. Travelling to distance parks (Miami to Yellowstone) requires a 'different kind of time' than travelling to local sites (Toronto to the Muskokas). They should not be valued in the same fashion. Finally, there is the issue of the treatment of workers versus non-workers. The use of wages, either average wages or estimated hedonic wages, excludes valuing the time of those who do not work (but may visit the site).

B.2. Direct Method : Contingent Valuation Method

The indirect methods already discussed have focused on the consumer's actual choices as providing the most important "window" to individuals values. The direct questionning method, or contingent valuation method, simply asks individuals about their preferences and runs contrary to the previous focus. The major advantage of the contingent valuation method over the indirect actual choice-oriented methods is that it can, in principle, measure non-use or existence values of environmental resources.

The contingent valuation method relies on questionnaires/personal interviews in soliciting information on people's willingness to pay for environmental goods. The underlying assumption is one of a contingent market. That is, respondents are asked to value a proposed or hypothetical change in the commodity. Then they are asked to say what the maximum amount they would be willing to pay for the proposed change (if positive), or the minimum amount they would be willing to accept in compensation, or the maximum they would be willing to pay to prevent the change (if negative). In using the contingent valuation method, one must be aware of the underlying behavioral assumptions regarding individual's behaviour. First, it is assumed that respondents know or can determine their preferences over the environmental good(s) and all other goods and/or services. They are assumed to be familiar enough with the good to evaluate it. Secondly, it is assumed that respondents will not act strategically; they will reveal their true willingness to pay assessment and not attempt to bias survey results in such a manner as to obtain a desired outcome.

Initial attempts to use the contingent valuation method were crude, and hence easily criticized, although survey techniques and questionnaire design have been refined in the last few years. Survey questions typically ask respondents to value specific outputs (e.g., a fishing day in a particular location) rather than to value changes in environmental quality. Successful surveys have three key components. First they provide a detailed description of the commodity that consumers are to value. Secondly, the method by which payment for the commodity - the so-called payment vehicle - is to be made, is described. For example, if the output is fishing days at a local lake, the question would ask if the respondent would be willing to pay a given fee in order to ensure improved water quality. Thirdly, the successful survey must use a method for eliciting values. (For example, a yes/no type response to the question of whether one would be willing to pay a stated amount does not reveal the actual willingness to pay for that respondent. The stated amount might be a maximum or a minimum.) Fourthly, surveys generally ask respondents to provide additional socioeconomic and demographic data, e.g., income, age, sex.

The actual survey instrument is very important. Over time, researchers have discovered that close-ended questions (those requiring a yes/no response) are easier for respondents to answer than open-ended questions (those that require respondents to express opinions). Answering a yes/no question and other close-ended questions, such as decisions that resemble conventional market choices, elicit better responses. Respondents are unwilling to go through the necessary mental reasoning that open-ended questions demand to provide rational values for goods. However, researchers have developed a method for eliciting better responses from open-ended questions. This is called convergent direct questionning. It involves a sequence of questions that simultaneously move the respondent up from a low money value and down from a high money value to converge on a more precise value for willingness to pay.

Once the values are obtained from the survey, the researcher typically performs a regression of the stated willingness to pay (or willingness to accept) on a variety of demographic or attitudinal variables included in the survey data. Then, the researcher obtains a predicted value for the willingness to pay based upon the subset of individuals with a given set of identical demographic characteristics. Willingness to pay values are aggregated over all subsets to obtain the entire measure of population benefits.

Strengths of the Contingent Valuation Methodology

In measuring benefits attributed to the environment, one must distinguish between the concept of total economic value and that of total user value. The total user value is the sum of the actual use value (measure by the indirect methods) and the option value, where the option value has three components. The first is the option value in use by the individual in question, the second is the option value in use by his/her descendants, and the third is the option value in use by other individuals now and in the future. Each of these option values can, in principle, be measured via the contingent valuation method, but not by indirect methods. The total economic value is the sum of the total user value and the existence value. Economic theory assumes that existence value is equivalent to intrinsic value. This may seem like a trivial point, but it underlies an assumption that value resides 'in' something but is captured by humans through their preferences in the form of non-use value. This is a subtle distinction from the belief that value exists 'in' something independent of human existence. Again, existence value can be measured in principle by the contingent valuation method, but not by indirect methods.

Thus, the contingent valuation method can be used to provide values for a wide range of public and open access environmental resources. It can also be used to provide *ex ante* values; for example, respondents can be asked to value a proposed change to a particular recreational site.

Weaknesses of the Contingent Valuation Methodology

Some researchers are concerned about strategic biasing of reported willingness to pay/accept values. This occurs when respondents either inflate their willingness to pay so that personally desirable projects will be implemented or deflate their willingness to pay in order to stop an undesirable project from going forward. However, strategic behaviour requires much effort and knowledge of the survey and the weight of accumulated evidence suggests that this fear is unfounded, especially when surveys are designed to reduce the possibility of the bias occurring (Freeman, 1979; Cummings et. a. 1984). Experimental economics has developed iterative bidding processes that reduce or eliminate strategic problems (Coursey, Hovis, and Schulze, 1987)

Other types of biases may be introduced into the willingness to pay values via other aspects associated with the survey design; for example, the choice of starting bids, payment vehicles (e.g., will taxes be used or will the government rely on voluntary contributions), information provided (Samples, Dixon, and Gowen, 1986), question framing, etc., (Cummings, Cox Jr., and Freeman, 1986). Again, as researchers gain more experience with designing surveys, the possibility of these biases is reduced. However, a bias may still result in situations in which the respondent does not have a great deal of familiarity with the item and, therefore, does not know his/her true willingness to pay value. Researchers have responded to this difficulty by providing the respondent with information scenarios designed to put the respondent into the picture, so to speak. (In this regard, researchers have often resorted to using pictures to provide information.)

The role played by property rights is often overlooked by researchers designing contingent valuation studies. They must be aware of the necessary distinction between a contingent valuation versus an equivalent valuation measure of the willingness to pay/accept. On the one hand, respondents may perceive the environmental amenity as a "right". Furthermore, preservation of this right to an amenity may be perceived as been the responsibility of government. In this case, status quo utility is the reference point; therefore, contingent valuation is the appropriate measure to use. If the change in the environmental amenity is positive (an improvement in environmental quality), then the willingness to pay to remain at the status quo level of utility is maximum amount of income that could be taken away from the consumer while leaving him/her just as well off as before the increase in environmental quality. On the other hand, suppose the environmental change is negative. In this case, the willingness to accept compensation for the worsening environment is the sum of income sufficient to return the consumer to the status quo level of utility.

If, however, respondents perceive the amenity as being a privilege with an associated cost, then equivalent variation is the appropriate measure. This takes the new level of utility, post change, as the reference point. Again, suppose the change in the environment is positive. Then, the willingness to accept compensation is the amount of income the person would be willing to receive in lieu of the environmental improvement that would put him/her on the new higher level of utility as would the environmental change. If the change in the environment is negative, then the consumer's willingness to pay is the amount of income that the consumer would be willing to give up to prevent the environment change, but would keep the consumer on the new lower utility level.

One difficulty arising from this dichotomy in the notions of property rights is that a group of respondents may refuse to cooperate with the survey based on the choice of initial 'rights reference point'. Respondents may even attempt to strategically model their responses (see above). These same individuals may have well defined willingness to pay or accept values for the amenity but the researcher will be unable to recover them with contingent valuation methods. To avoid this problem, it is useful to utilize both contingent valuation and an equivalent valuation estimates of willingness to pay/accept since respondents may have differing perceptions of property rights.

There is a second difficulty with the notions of willingness to pay and willingness to accept. In experiments with private goods, comparing these measures has shown no statistical difference between hypothetical (from contingent valuation surveys) versus actual (market) values (Bishop and Heberlein, 1986). However similar experiments with public goods have shown that the willingness to pay and willingness to accept values do differ statistically. This discrepancy is not confined to the contingent valuation method, although it is often attributed as a shortcoming of the hypothetical nature of the method.

One explanation of the gap is that households are used to buying rather than selling. This is referred to as a 'pay-sell' cognitive dissonance. This dissonance can also be seen as the 'wishful thinking' of individuals as to how much they ought to be paid to be compensated. Cognitive dissonance does not largely apply on the willingness to pay side, but rather may be induced by the prospect of receiving money on the willingness to accept side. In experiments where individuals were allowed to bid for the same commodity, willingness to accept approached willingness to pay after several rounds of transactions. The divergence obtained in these estimates may result from lack of market experience.

On the other hand, recent work by Hannemann (1991) shows that the amount by which willingness to accept measures exceed willingness to pay measures varies directly with the income elasticity of the demand for the environmental good and inversely with the elasticity of substitution between the environmental good and private goods. For a large number of environmental goods, it is reasonable to assume a positive income elasticity of the environmental good and an elasticity of substitution between the environmental good and private goods which is close to zero. In these cases, we would expect to see a large gap between the willingness to accept compensation for the loss of a natural site and the willingness to pay to maintain the nature site.

C. TAXONOMY OF BENEFIT TYPES AND METHODOLOGIES

There are three categories of benefits that can be identified as follows:

1) Recreational or Amenity Benefits

2) Health Benefits

3) Industry Benefits

The following are some examples of each of the three categories of benefits that can be calculated using one or more of the above methods. For recreational benefits, it is possible to value the benefits of lakes, national parks, fishing, boating, canoeing, hunting.

Both the use and non-use (existence or option) values can be calculated. Use values can come from indirect benefit methods, most specifically, the Travel Cost Method, while non-use values must come from the Contingent Valuation Method. For health benefits, we can value the benefits of improved drinking water, reduced morbidity associated with air quality improvements, reduced fatalities, etc. Generally, the Averting Behaviour, Hedonic Market, and Contingent Valuation Approaches are used to obtain these values. The final category of benefits accrues to industry. Examples of industries that can benefit from improvements in the quality of the environment include forestry, agriculture, and some manufacturing industries that use the environment as key inputs into the production process. The benefits to these firms from a better quality environment occur in the form of increased profits (or lower costs of production). As indicated in Chapter 3 the producer surplus concept can be used to measure these benefits. This measurement can be done via observations in the market. The discussion of the measurement of these benefits is left to Chapter 5.

D. OVERVIEW OF COST MEASUREMENT

Both consumers and firms can be adversely affected by a change in the level of environmental quality. Insofar as consumers are concerned the methods described above to generate measures of benefits (gains in consumer surplus) from environmental improvements can just as easily be used to calculate the costs (lost consumer surplus) of a worsening of the environment. As discussed above, this measurement is made difficult by the fact that there are no direct markets for the purchase of environmental amenities, so researchers need to use either indirect methods or direct questionning methods to elicit values.

Firms, too, can experience costs (or benefits) associated with an improvement/worsening of the environment. In this case, researchers measure either producer surplus lost or gained. This is a much more straightforward process than the measurement of consumer surplus. This is because firms are subject to production functions (which include the environment as an input) describing the physical process of turning inputs into outputs. By and large, markets exist both for inputs and outputs, so that market prices are readily available for these items. The firm's cost function incorporates both the technical constraints of the production function and the market constraints of the prices for inputs and outputs. Economists tend to work with the first derivative with respect to output of this cost function, the marginal cost function, and for perfectly competitive firms, this is the firm's supply function. It is this supply function that is used to evaluate producer surplus. Changes in the marginal cost function (supply function) mean changes in the producer surplus. These are already measured in dollars of lost profits. Thus, welfare measurement in the context of the firm is a simpler exercise.

Chapter 5 integrates both the theory and empirical measurement of the costs associated with environmental changes.

Endnotes

1. Note that these preferences may not be in the best interests of society, nor do these preferences necessarily obtain the same result as would a majority rule approach.

2. Sunsetting of chemicals is defined to be the virtual elimination of certain chemicals from production and use.

3. Alternatively, it can be used to value the damages or costs associated with a worsening of the environment.

4. This is an alternative approach for the calculation of compensating variation to the expenditure function described in Chapter 3.

5. In the previous example, we assume that this cost is constant per unit.

6. The converse is true. Namely, when the environment deteriorates and the marginal cost of health increases, then this measure overstates the value of the welfare decrease.

7. This is in contrast to the models already discussed that assume a continuous damage function.

8. A further problem arises when the afflicted individual suffers minor ailments from exposure to pollution, but is reluctant to undertake averting expenditures of any kind. Since the researcher would have no record of expenditures by this person, he or she would be assumed to have a zero willingness to pay to avoid the health effects. However, it is likely that the true willingness to pay would be greater than zero.

9. If the hedonic price function is linear in Q_i, then identification of the inverse demand function is not possible. Furthermore, if all household have identical incomes and utility functions, then equation (10) is the inverse demand function.

10. Normally, one thinks of a demand function with quantity demanded being endogenously determined by the price and other factors. The inverse demand function turns this around. It says, that the price is endogenously determined by quantity and other factors.

11. This model does not assume that the value of one's leisure time is the same as the value of one's working time. However, simpler models do make this restrictive assumption. This is an important assumption, because the valuation of benefits has been shown to be very sensitive to the valuation of one's leisure time.

12. This is needed to generate benefit estimates, of course. So, often a regular model is estimated to predict total number of trips.

13. Either the traditional utility function specification or the household production function approach can be incorporated into the utility function in this model.

14. The random utility model deals with a second problem that researchers have had with the traditional travel cost models. These latter could not deal with a non-visit to a site. In the random utility model, non-visits are incorporated.

15. The exception here occurs when there is overcrowding of a site, so people perceive quality variation in the sense of having too many people on site. But, even then, all people may perceive the same reduction in quality.

5. EMPIRICAL ESTIMATES OF THE BENEFITS AND COSTS OF ENVIRONMENTAL REGULATIONS

A. INTRODUCTION

This report has thus far been concerned with the theoretical and methodological issues associated with using economics to value the benefits and costs that arise from environmental regulations. It has been shown that economic theory provides a comprehensive framework within which a regulation's impacts may be assessed and, in principle, measured. It should come as little surprise, however, that data availability and measurement problems frequently limit the extent to which economic methods of valuation can be brought to bear. This chapter and the next chapter shift the focus away from theoretical issues and concentrate on the issues arising from applying the economic valuation methods to environmental resources.

This chapter has two purposes. First, it presents empirical estimates of the various benefits and costs which have been associated with government policies aimed at protecting and\or restoring environmental quality. Second, it reviews and critically assesses the empirical techniques used to obtain these estimates.

This chapter is also meant to provide a context in which chapter six can be read, and the detailed case studies evaluated. That is, this chapter presents cost and benefit estimates from several empirical methodologies and a wide variety of circumstances:

different government policies, different environmental contaminants, different ecosystems, etc. Conversely, chapter six considers three specific case studies (DDT, lead and dioxin) and examines the role of economics in government decision-making and, where it is possible, provides estimates of costs and benefits that arise from government efforts to regulate these environmental contaminants. Thus, chapters five and six are designed to complement one another: chapter five gives an overview of the range of estimates while six provides a more detailed examination of a small set of specific cases.

This chapter is organized according to the categories of benefits and costs that have already been identified. For benefits, the categories are the following: recreation, health, and industry.[1] For costs, the categories as follows: compliance expenditures, productivity losses, GNP losses and aggregate welfare losses.

In each category, the discussion is organized as follows: first, the nature of the benefit or cost is defined. Next, the empirical techniques are described and estimates are reported. In addition, a table in each section presents a summary of these empirical estimates. Finally, an assessment of the empirical techniques is made. The studies which are examined here are published research reports or unpublished reports made to government.

Before proceeding, there are three unrelated notes to be made. First, a discussion of the value of applying cost-benefit techniques to issues concerning changes in environmental quality is postponed until chapter seven. Second, the discussion that follows points out that there are a number of effects of changes to environmental regulations that are difficult to evaluate using economic techniques. However, any inability of economics to quantify a particular type of cost or benefit should not be taken as proof that the benefit or cost does not exist. This warning is especially important in the case of benefits where the scientific understanding of the impact of contaminants in complex ecosystems is still limited. Third, all of the cost and benefit estimates presented in this chapter have been converted to 1992 Canadian dollars[2].

B. ESTIMATES OF THE BENEFITS OF ENVIRONMENTAL REGULATIONS

B.1 Recreation

B.1.1 Definitions

Empirical studies examined in this section measure the demand for recreation and attempt to determine the relationship between environmental quality and the value people assign to recreational experiences. Estimates of the extent to which the value of recreational activities is increased by improvements in environmental quality (such as reduced coliform counts near municipal beaches or reduced SO_2 deposition in National Parks) provide regulators with data valuable to decision making.

There are several surveys of the empirical estimates of recreational benefits and their relationship to environmental quality (Freeman, 1985; McConnell, 1985; Bockstael, McConnell and Strand, 1991; Cropper and Oates, 1992). In addition, there are a number of reports to Canadian governments which either generate estimates of recreational benefits or employ the estimates of others in order to assess proposed environmental regulations (cf. Apogee Research et. al., 1990; DPA, 1990). In these surveys and government reports, estimates of recreational benefits are usually divided into the several categories of activities and that practice is followed here. The categories include fishing and swimming, camping, hiking and hunting and nonuse benefits. The last category, nonuse recreational benefits, differs from the other categories because it refers to individuals' willingness to pay to preserve rather than use ecosystems. An individual's motivation for this WTP may be to retain the option of use in the future for her\himself or for others or it may be that simply contributing to the preservation of an ecosystem yields benefits to the individual. Table 5.1 provides a summary of the empirical estimates of the use and nonuse recreational values presented in this section.

For the most part, individuals' valuation of recreational activities and the way in which these values vary with changes in environmental quality must be inferred or imputed by economists. This is because there are few instances where a competitive market place generates an equilibrium price for recreational activities that could be used as estimates for consumers' valuations.

There are two principle techniques used to elicit individuals' valuation of recreational activities. The first is the travel cost method. It is an example of the class of indirect methods where valuation is inferred from individuals' observed actions. This technique is based on the assumption that the costs incurred to engage in a recreational activity (out of pocket expenses plus the time taken to get to the site) can be used to approximate an individual's WTP for that experience. In the early 1980's, this method was combined with hedonic models (and, therefore, is sometimes referred to as the hedonic travel cost method or HTCM) so that researchers could determine the marginal contribution of specific site attributes (such as environmental quality) to the valuation of the entire recreational activity.

The second empirical technique is the contingent valuation method (CVM). This is an example of the class of direct valuation techniques where individuals' WTP for recreational benefits or improvements in environmental quality are ascertained by asking them through surveys. An important characteristic of this technique is that it is better suited to derive WTP estimates for hypothetical situations and for nonuse values (cf. the discussion in Randall, 1991). These techniques for estimating recreational benefits are critically assessed in McConnell (1985); Bockstael, McConnell and Strand (1991) and Smith (1993).

Table 5.1 : Estimates of Recreational Benefits

1. Sports Fishing

Date of Study	Researchers	Method[1]	Benefit Estimate[2]
1978	Charbonneau and Hay	HTCM	WTP of \$9.05 - \$13.58 per person per day
1982	Vaughan and Russell	HTCM	WTP of \$7.88 - \$15.76 per person per day
1982	Vaughan and Russell	TCM	WTP of \$8.48 - \$16.97 per person per day
1983	Smith, Desvousges, and McGivney	TCM	WTP of \$0 - \$18.95 per household per season Average CS is \$9.96 per household per season
1988	Carson and Mitchell	CVM	WTP of \$135 per person per year

2. Swimming

Date of Study	Researchers	Method	Benefit Estimate
1983	Smith, Desvousges, and McGivney	TCM	WTP of \$0 - \$40.32 per household per season Average CS is \$20.91 per household per season
1990	Bjonback	CVM TCM	1. Linear 2SLS : WTP \$8.22 per person per recreation day CS \$42.85 "

			2. Linear OLS : WTP $5.8 per person per recreation day CS $42.11 3. Log Linear 2SLS : WTP $0.11 CS $28.62

3. Camping, Hiking, and Hunting

Date of Study	Researchers	Method	Benefit Estimate
1979	Bishop and Heberlein	CVM	$59 - $177 per license to hunt
1991	Englin and Mendolsohn	HTCM	Marginal Value per trip per person $4.82 for trail with old growth $13.11 for camp site $11.57 for view attribute Average CS per trip $132.03: trail with old growth $332.04: camp site $468.11: view attribute

4. Non-Use Recreational Benefits

Date of Study	Researchers	Method	Benefit Estimate
1987	Boyle and Bishop	CVM	WTP $10.22 - $38.72 per person for preservation of whooping crane
1987	Alberta Forestry Lands and Wildlife (Bow River)	CVM	Average $29 per year per person

1988	Bowker and Stoll	CVM	Median $0 - $112 per household per year Average $37.32 per household per year
1991	Stevens et. al.	CVM	Median WTP per person per year to preserve : coyote $4.80 Atlantic Salmon $9.07 Bald Eagle $22.07
1991	Whitehead and Blomquist (wetlands preservation)	CVM	WTP $6.51 - $21.25 per household per year

Notes:

[1] HTCM is the hedonic travel cost method, TCM is the travel cost method and CVM is the contingent valuation method.

[2] This is the either the willingness to pay (WTP) for an improvement in water quality that supports a recreational fishery or the consumer surplus (CS) associated with the improved water quality.

B.1.2 Sport Fishing and Swimming

Environmental quality is an important factor in explaining the frequency of recreational water-based activities such as swimming, boating and fishing and the level of benefits that individuals derive from these activities. For example, the level of water quality can influence fish populations and, as a result, the popularity of sport fishing in several ways. Reduced water quality can reduce the population of an individual species and also allow new species to thrive. For example, reducing the oxygen content of some rivers decreases brook trout populations while allowing carp and other 'rough' fish to thrive. Impaired water quality can also render fish unsuitable for human consumption.

Early applications of travel cost and CVM techniques are concerned with inferring the value of a particular recreational experience. Thus, willingness to pay (WTP) estimates are calculated for a specific site and for a particular set of characteristics such as water quality. Apogee et. al. (1990) surveys a number of early research efforts that employ both indirect and direct techniques. The study concludes that estimated consumer surplus (or net benefit) from a day of recreational fishing ranges from $6 to $108. While this is a fairly broad range, most estimates are clustered in the range of $27-$54/day.

More recently, these valuation techniques are used to assess the change in users' valuation when some important characteristic of a facility, such as water quality, changes. For example, Carson and Mitchell (1988) employ a CVM and report that U.S. households are WTP an average of $135/year to ensure an increase in water quality from boatable to fishable. Vaughan and Russell (1982) estimate the WTP for the same quality improvement using a travel cost method and conclude that mean WTP ranges between $8.48 and $16.97 per person per day.

In an influential study, Smith, Desvousges, and McGivney (1983) address the change in valuation of sport fishing associated with changes in the water quality of specific sites by investigating the role of water quality in approximately twenty individual recreation demand models. Each of the demand models corresponds to a specific recreational activity or recreational site. The authors employ a varying parameter model in order to determine whether differences in environmental quality play a role in explaining differences in estimated WTP across the individual demand models. In each study, the travel cost method is employed to estimate the demand for recreational fishing. This method produces demand parameters that are functions of site characteristics or attributes. Next, a generalized least squares (GLS) estimator is developed to evaluate the role of site attributes and water quality in particular. The principle finding of the study is that increases in water quality are found to increase the demand for recreation. Water quality improvements are shown to also increase the valuation of recreational activities. For example, the estimated model is applied to data involving residents of Pennsylvania within the Monogahela River Valley to generate empirical estimates of WTP. The results of this study produce WTP estimates for a water quality improvement from boatable to game fishing conditions on the range of $0 to $18.95 per household per season with an average consumer surplus of $9.96 per household per season.

The regression equation underlying the HTCM is not problem free as diagnostic testing indicated the presence of truncation bias in one-third of the observations. In addition, these results for water quality improvement in the Monongahela appear to be relatively small when compared to other studies. Charbonneau and Hay's (1978) water quality benefit estimates associated with fishing range from $9.05 to $13.58 per person per day. Vaughan and Russell (1982) water quality study estimates sport fishing benefits in the interval of \$7.88 to \$15.76 per person per day.

Smith, Desvousges, and McGivney (1983) also provides estimates for the benefit of improved water quality to allow swimming. The same estimation procedure is used, but the focus is on the site attribute pertaining to swimming. Using the same data from the Monongahela River Valley, the increment from boatable to swimmable water quality results in estimates in the range of \$0 to \$40.32 per household per season with an average consumer surplus of \$20.91 per household per season.

Bjonback (1992) addresses the valuation of water quality changes at a recreational facility with multiple uses. This study assesses the WTP to experience an improvement in lake levels at Lake Diefenbaker in western Saskatchewan. Because of its configuration, the depth of the lake has a strong influence on its attractiveness (as the lake becomes more shallow it promotes weed growth and may exhibit foul odours). A wide range of water-based activities are offered at this provincial park and the economic implications of a reduction in the quality of this key recreational site are explored and estimated.

The study uses data from a recent drought that had a tremendous impact on the lake and its users. Park users are surveyed during the peak recreational season and are asked to identify what the impact of the drought has had on their outdoor recreational activities and to identify what their WTP to improve water levels to meet their recreational requirements.

In order to obtain WTP estimates, both contingent value method (CVM) and travel cost method (TCM) are used. A travel cost model is built using a simultaneous equations technique that estimates WTP and incorporates a parameter that accounts for the value of an improvement in the recreation experience. The report includes both two-stage least squares and ordinary least squares OLS of linear and logarithmic forms. Overall, diagnostic tests indicate that the travel cost system of equations performs well in explaining preferences for site features and in explaining visitation to the site.

Bjonback finds that his estimates for consumer surplus and WTP vary according to the functional form and estimation technique used. For example, the linear two stage model yields a consumer surplus of $42.85 and an estimated WTP for improved water quality of $8.22 per recreation day. In contrast, the linear OLS model produces a consumer surplus of $42.11 and a WTP of $5.18 per recreation day. Finally, The log-linear two stage model's consumer surplus and WTP per day are $28.62 and $0.11, respectively. Because the estimates are found to be sensitive to model specification and estimation method, the author proposes a broader study that incorporates more recreational sites. In addition, this sensitivity of estimates to functional form and estimation technique has been noted frequently in the literature. This finding suggests that care must be taken in applying estimates from a single study to assessing policy changes (cf. the discussion in McConnell, 1985).

B.1.3 Camping, Hiking and Hunting

Economists have devoted most of their attention to water-based recreational activities. However, there are a number of studies concerned with estimating the valuation of 'land-based' activities such as wilderness camping and hunting. It is to be expected that the demand for these activities and the value assigned to them are sensitive to environmental factors. In the United States, for example, air pollution has been found to reduce visibility in some National Parks and to reduce forest growth rates (Cropper and Oates, 1992).

Englin and Mendelsohn (1991) measures the recreational value of wilderness site visits. A hedonic travel cost (HTC) model is applied in order to provide estimates of the value of both marginal and non-marginal changes to site characteristics. Linear demand functions for site qualities conditional upon the event of an excursion are developed from conditional quadratic utility functions.

The empirical estimates of the values of wilderness users are obtained through a two stage process. First, a hedonic travel cost function is estimated using ordinary least squares. Secondly, demand functions for site characteristics are estimated using OLS. The site attributes that are found to be statistically significant are the following: campground, dirt road, excellent view, clear-cuts, closed spruce forests, open parks of alpine and Engelmann firs, silver and noble firs, Douglas fir, hemlock, rock and ice, and old growth large trees.

The data used in this study consisted of overnight camping permits collected in the summer of 1982 at four USFS Wilderness areas in Washington state. Entrants are asked to identify their origin zip code and the trail that they intend to hike. Additional information such as party size and the number of visits to the trail in the past ten years is obtained through the permits.

Demand functions are estimated for individual characteristics. This analysis provides empirical estimates of forest attributes such as old-growth, campgrounds, and views. Using the available data the average wilderness-user's marginal value per trip of these forest attributes are: $4.82 for a trail with a mile of old-growth, $13.11 for a camp site, and \$11.57 for a view attribute. In addition to the marginal values, average consumer surplus per trip estimates are provided for each of the attributes; \$132.03 for a trail with old growth, \$332.04 for a camping site, and \$468.11 for a scenic view.

A very interesting estimate for the total value of these four wilderness areas is given by the authors. Assuming the sample obtained is representative of the region's population, approximately 125,000 trips are being made into these wilderness areas annually. If the per capita WTP and CS estimates are applied to this aggregate number of trips, the estimated consumer surplus of the old-growth in these areas is worth about \$16.6 million and the loss of all the campgrounds in the forest would represent a loss of \$41.4 million per year.

Bishop and Heberlein (1979) employs a CVM survey to study the value hunters in Northern Wisconsin assign to the right to hunt geese. The authors have access to the list of individuals who applied for a hunting licence. In a mail survey, successful applications are offered randomly sampled cash payments in exchange for their licence and unsuccessful applicants are questioned as to their WTP to obtain a licence.

The study's results indicate that the WTP expressed by unsuccessful applicants for the right to hunt tends to be less than the amount expressed by successful applicants as the minimum payment required to relinquish a licence. The average WTP and WTA figures are \$59 and \$177, respectively. As discussed in chapter 4, part of this difference is due to the presence of an 'income effect' in consumers' valuation of transactions. However, once the portion of this difference due to income effects is accounted for, some difference remains. This suggests that hypothetical responses may be inaccurate proxies for valuations reflected by actual market transactions (such as trading licences for cash).

B.1.4 Nonuse Recreational Benefits

During the 1980s a small number of studies applied CVM in order to elicit individuals' WTP for preserving individual species or ecosystems. Unlike other categories of benefit estimation, applied researchers concerned with option, bequest and existence values have been constrained by the lack of consensus regarding the theoretical underpinnings of their models. Randall (1991) surveys the theoretical and applied literature, and demonstrates that uncertainties in the former inhibit the development of the latter.

Bowker and Stoll's (1988) study of the existence value assigned to preserving the Whooping Crane is an example of the dichotomous choice method of CVM. Respondents are asked whether they would be willing to contribute voluntarily a prespecified amount to a non-government trust fund dedicated to preserving the Whooping Crane. Respondents may answer only yes or no and the dollar amount of the contribution is varied randomly across surveys. This format reduces the amount of information regarding a respondent's preferences that is revealed, but also avoids some of the difficulties associated with confronting people with hypothetical situations. Statistical techniques which accommodate dependent variables which are discrete and\or have truncated distributions are used to derive WTP estimates from the survey data. Estimates of median WTP per household per year range from $0 to $112 . The average of these estimates is $37.32

Another CVM study attempts to assess whether people who either contribute to environmental protection groups or who are active in outdoor recreation differ in their existence values from people who do neither. Boyle and Bishop (1987) use a dichotomous choice style questionnaire to obtain WTP estimates for the preservation of the Bald Eagle and the Striped Shiner. With respect to the WTP for Bald Eagle preservation, the results accord with one's intuition. Specifically, WTP estimates are highest for outdoor recreationists who have previously contributed to environmental protection (median WTP of $38.72). Inactive people who have contributed in the past exhibited a median WTP of $21.18, and inactive noncontributors expressed a median WTP of only $10.22. These differences are statistically significant and the pattern of responses is roughly the same for the Striped Shiner.

A recent study which highlights the strengths and weaknesses of this approach is Stevens et. al. (1991). The authors conduct several CVM surveys in order to elicit existence values from New England residents for several endangered species. The valuation questions are contained in a mail survey and are of the dichotomous choice form used in previous studies. Both Logit and Tobit estimation procedures are used to estimate WTP.

The results of the study are largely consistent with those of previous studies but there are some unexpected and puzzling results as well. The median WTP (per person per year) range from $4.80 for the coyote, to $9.07 for Atlantic Salmon and $22.07 for Bald Eagles. Despite these results, a majority of respondents (approximately 60%) refuse to pay anything to preserve any species. Approximately 25% of those respondents who refuse do so on ethical grounds, arguing that it is morally wrong to determine whether another species survived based on society's collective WTP to preserve it. On the other hand, for those who do report a positive WTP, existence values are important. Survey results indicate that only 7% of the median WTP is attributable to current use or private option values, 34% is due to bequest values and 48% is related to existence motives. Finally, there are a number of potential inconsistencies in the responses. An example of this is a respondent answering that preservation of the Bald Eagle is "very important", but then reporting a WTP of zero. One possible explanation for this type of result is that people are using decision-making rules which are not fully consistent with the utility-maximizing behaviour assumed by economists.

Unfortunately, very few studies of Canadian non-use values are available. Alberta Forestry, Lands and Wildlife (1987) summarizes an early effort to determine the WTP of residents of the Bow River Valley area to preserve the scenic and recreational qualities of the Bow River. Forty one percent of the respondents to the CVM mail survey indicate a positive willingness to pay an annual preservation fee. The average of their offers is $29 per year.

Apogee Research et. al. (1990) considers the use and non-use values associated with remedial action in Ontario's Areas of Concern. Local surveys and sources are relied on to provide estimates of the increases in swimming, fishing and boating that are expected to occur after water quality improvements. However, point estimates of the economic value of these increases are derived from the existing literature, most of which uses American data. Using this literature, the authors make the assumption that the average WTP (use and non-use) for water quality improvements is $135/household/year in Ontario and that half of this value can be assigned to non-use motivations. Given the current population of Ontario, this estimate implies that the non-use values alone associated with the water quality improvements under the Remedial Action Plan can be expected to exceed $230 million each year.

The empirical results of the CVM studies examining existence values indicate fairly clearly that it is an important component of total value. This is an valuable finding because it supports the argument that there are significant recreation-related benefits to society from pollution control, other than those accruing to the current generation of people pursuing recreational pastimes. If these results are accurate, and the samples from which they are drawn are representative, then the collective value assigned to the preservation of damaged ecosystems or of endangered species might exceed the values assigned by people currently 'using' these environmental goods. Ignoring non-use values in any CBA of pollution control will seriously underestimate total benefits.[3]

Offsetting the potential significance of these findings are a series of methodological and empirical problems associated with using CVM's to elicit existence values. The methodological or theoretical problems are dealt with in other parts of this report. However, it is important to point out that successful estimation of individuals' WTP for preserving environmental goods must await the resolution of the theoretical problems surrounding the definition of non-use values.

A general problem with conducting CVM studies of existence value is that they concern a topic with which most people have little or no experience. Furthermore, the issue of exactly how these things are defined is critical to respondents' valuations. For example, when discussing preserving a species, is preservation guaranteed or is there some probability of success? Is an ecosystem defined by its area or by its characteristics? These concerns imply that the way in which information is presented in the survey is crucial. Whitehead and Blomquist (1991) find, for example, that as more information is provided about the availability of substitutes for an endangered wetland, respondents' WTP fell. Depending on the amount of information provided and several other factors, estimated WTP to preserve the wetland range from $6.51 to $21.25 per household per year and the difference between the 'no information' WTP and the 'fully informed' WTP is $7.44 per year.

Another area of concern is related to the statistical techniques applied to the data collected by CVM surveys (cf. Bowker and Stoll, 1988). WTP estimates tend to be sensitive to estimation procedure (for example, whether Tobit, Logit or Probit methods are used). In addition, it is common for researchers to omit respondents providing extremely high WTP's on the basis that these responses violate respondents' budget constraints or that they are rarely reflected in observed market transactions. Deleting these outliers (and, thus, truncating the distribution of observed responses) can obviously have a large impact on the resulting mean WTP estimates. However, there are no clear and agreed upon rules for deleting these outliers and the decision to truncate remains controversial (Cambridge Economics, 1992).

B.1.5 Assessment of Recreation Benefit Estimates

Judging by the amount of research in this area and the number of surveys of this literature, the estimation of the benefits from recreation has received a large amount of attention from economists. Much of the impetus for this effort stems from the observation that there are usually nonexistent or poorly defined markets for recreational services or outputs. Thus, economists have been challenged to construct methods to infer or impute consumption values that would normally be readily available in a well-functioning competitive market.

The travel cost method (and its recent variant, the hedonic travel cost method) has been the 'workhorse' of past efforts to estimate recreation benefits. This method is best suited to estimate the net benefit associated with travel to a single or small number of sites. Estimating demands for several closely situated sites presents significant data and estimation problems (cf. the discussion in Bockstael, McConnell and Strand, 1991).

In addition, the HCTM once held the promise of being able to provide estimates of the relationship between recreational benefits and environmental quality. In particular, it was thought that these studies could indicate how users' valuations altered with small changes in environmental quality. Unfortunately, HTCMs have lost some of their popularity due to the difficulties associated with their implementation. These problems include the uncertainties surrounding the interpretation of the reduced-form model's estimated coefficients and the difficulties in constructing data sets with sufficient variation in the environmental and economic variables.

Contingent valuation studies of recreational benefits are increasing in popularity amongst economists. Despite a commonly held distrust of direct survey techniques, economists have come to recognize the advantages of the contingent valuation approach. These advantages include the possibility of estimating nonuse values, the ability to relate changes in valuation to changes in perceived environmental quality and the ability to influence the amount and form of information possessed by the respondent. These considerations must be balanced with the problems arising from presenting respondents with hypothetical questions and the sensitivity of responses to the framing of questions.

B.2 Health

B.2.1 Definitions

Probably the most prominent benefit from reduced pollution is decreased risk of illness and premature death for humans. These risks arise because a large number of manufactured substances that are emitted to the natural environment are known or suspected of having caused negative health effects upon exposure to humans. For example, various forms of air pollution (ground level ozone, suspended particulate and sulphates) are correlated with higher incidences of respiratory illnesses (Ostro, 1983). In addition, low concentrations of substances such as PCB, dioxins, mercury, which are found in drinking water supplies, may be associated with increased incidence of serious chronic illnesses (certain forms of cancer and adverse effects on neurological, immunological and reproductive systems; Flint and Vena, 1991).

Unfortunately, this class of benefits is also the most difficult to evaluate in economic terms. This is due, in part, to a variety of 'non-economic' causes, including the uncertainty surrounding the 'dose-response' relationships for many toxic substances, the latency periods associated with some diseases, the extent to which responses can be reduced through 'defensive actions', and the compounding effects of exposures to multiple substances.

Even if these epidemiological difficulties did not exist, there would remain the problem of assigning dollar values to avoided cancers, asthma attacks, eye irritations and the pain and suffering that accompany these illnesses. Some costs, such as hospital visits, medication and lost wages, are relatively straight forward to measure. Unfortunately, these are not all of the costs to the individual of being ill as they neglect the lost leisure time, the pain and suffering associated with illness and the anxiety that comes from knowledge of an increased probability of becoming ill.

The techniques used by economists to produce estimates of the total benefits from avoiding illness and premature death, and the results of these efforts, are reviewed in this section. The guiding principle for the most recently developed techniques is to use information supplied by individuals regarding the importance they attach to these risks. Specifically, economic estimates are based on what individuals indicate (through their actions or directly in surveys) to be their willingness to pay to avoid the problem. Table 5.2 summarizes the empirical estimates of health benefits presented in this section.

Table 5.2 : Estimates of Health Benefits

A). Value of Avoided Mortalities

1. Contingent Wage Studies

Date	Researcher	Benefit Estimate
1988	ABT Associates	$2.8 million VSL
1991	Cropper and Freeman (summary of early work)	$1 million - less than $5 million VSL
	(more recent work)	$1.5 - $3.0 million
1992	Martinello and Meng (Canadian data)	$2.4 million - $7.7 million

2. Averting Behaviour Studies

Date	Researcher	Benefit Estimate
1990	Murdoch and	$2.04 million VSL (sunscreen use in face of ozone depletion)
1991/92	Cropper and Freeman	Less than $1.0 million

3. Direct Method (Contingent Valuation Method)

Date	Researcher	Benefit Estimate
1986	Mitchell and Carson	$326,000 - $698,000 (for reduction of concentration of THM in drinking water)

1991	Cropper and Freeman (survey)	$3.53 million - $5.29 million
1991	Viscusi, Magat, and Huber	$0.8 million - $3.72 million (reduction in risk of bronchitis)

B). Value of Reduced Morbidity

1. Contingent Wage Studies

Date	Researcher	Benefit Estimate
1992	Martinello and Meng (Canadian)	Value of reduction in risk of Minor Injury : $12,277 - $16,369 Severe Injury : $181,600 - $217,400

2. Averting Behaviour Studies

Date	Researcher	Benefit Estimate
1990	Murdoch and Thaler	Present Value of Aggregate Expenditures on sunscreen attributable to ozone depletion $16.03 billion (United States)

3. Cost of Injury

Date	Researcher	Benefit Estimate
1991	Cropper and Freeman	$3.29 per day of drowsiness $26.69 per day of itchy eyes
1991	Schecter and Kim	$26.50 per year for 50 % increase in perceived air quality

4. Direct Method (Contingent Valuation Method)

Date	Researcher	Benefit Estimate
1991	Cropper and Freeman	Median WTP per day for cessation of coughing : $1.83 - $20.15 Mean WTP per day for cessation of coughing : $45 - $650.50
1991 Huber	Viscusi, Magat,	WTP for a reduction in the risk and of chronic bronchitis : $551,600 - $772,500
1991	Schechter and Kim	Median WTP for 50 % increase in perceived air quality : $39 - $105.00 per year

B.2.2 Estimates of the Value of Avoided Mortalities

For certain toxic substances in the environment, exposure is thought to increase the likelihood of premature death. For example, benzene is emitted from certain industrial processes and is a known carcinogen. Similarly, lead is emitted from a number of sources and is thought to be related to an increased incidence of heart attacks and strokes. If a government regulation lowers the ambient concentration of such substances and thus lowers the probability of death, then the task of the economist is to estimate the dollar value of the reduction in risk of death. As is generally the case, the guiding principle for the economist is the affected population's expressed or observed willingness to pay to acquire the reduction in risk.

An individual's WTP for a small reduction in risk of death can be estimated using indirect or direct methods. Indirect methods infer the person's WTP from his/her observed consumption and labour supply decisions. Direct methods obtain the same information by questioning the individual. In both cases the economist tries to determine the individual's willingness to trade (or marginal rate of substitution) between income and risk of death.

Before proceeding to examine these two methods, it is useful to define a concept which appears frequently in the empirical literature. As explained by Cropper and Freeman (1991, p. 182), "Since risks of death from environmental contaminants are thought to be small, it is customary to express risks in terms of the number of *statistical lives* lost due to the contaminant." If exposure to some substance increases the probability of death for person i by ΔD_i and there are N people in the exposed population, the number of statistical lives lost is the following:

$$SL = \sum_{i=1}^{N} \Delta D_i \tag{1}$$

This expression implicitly assumes that everyone in the effected population is the same age. If each person's willingness to pay for these risk changes is WTP_i, then the value of a statistical life (VSL) is sum of the affected populations' willingness to pay for the risk changes divided by the number of statistical lives, or

$$VSL = \frac{\sum_{i=1}^{N} WTP_i \; \Delta D_i}{\sum_{i=1}^{N} \Delta D_i} \tag{2}$$

For example, suppose reducing the ambient concentration of a toxic substance reduces the risk of death by 10^{-5} for each 200,000 people. Then, the reduction saves 2 statistical lives (10^{-5} x 200,000). If each person is willing to pay $20 for the reduction, then the VSL implied by the reduction is $2,000,000 ($20 x 200,000/2). Thus, if a researcher can determine the average WTP or distribution of WTP's for small reductions in risk, then the implied VSL for the effected population can be computed and used in evaluating the benefits associated with reduced health risks.

The indirect method of inferring WTP for risk reductions uses observations from two different types of decisions made by individuals. The first involves compensating wage studies. Here, individuals are observed choosing over jobs which differ in their riskiness and wage levels. Under a certain set of assumptions, the wage premium that must be paid to the average worker in order to have him\her accept a slightly greater probability of death, may be computed from observed (wage, riskiness of occupation) pairs. These wage premia then form the basis for VSL estimates. The second approach uses averting behaviour studies. In this case, the expenditures undertaken by individuals in order to reduce their risk of death are used to compute WTP and, thus, VSL figures. Each of these indirect methods is reviewed below.

The compensating, or hedonic wage, studies examine the relationship between workplace risks of fatalities and wages. It is assumed that workers are fully aware of the relative riskiness of alternative occupations and demand higher wages to compensate them for assuming greater risks of injury or death. In order to determine the magnitude of the increase in wages required for differing risk levels, a regression equation is specified. The dependent variable is the wage rate and the explanatory variables include workers' characteristics (age, education, gender, etc), risk of fatality and other variables thought to explain wage variations (Martinello and Meng, 1992 is a recent reference).

Once the coefficients of the equation are estimated, they are used to compute the markets' equilibrium wage premia for alternative risk levels. These premia are then combined with the risk data to compute VSL estimates.

Cropper and Freeman (1991) summarizes the results of a number of compensating wage studies. The studies are divided, somewhat arbitrarily, into "early" and "recent" studies. The early studies are further divided into "low-range" and "high-range" studies. The early "low-range" studies underestimate VSL because they use actuarial risk data rather than the actual number of recorded fatalities. These studies tend to yield VSL estimates of less than one million dollars. In contrast, Cropper and Freeman argue that the early "high-range" estimates systematically overestimate VSL by not controlling for type of occupation.

These studies provide VSL estimates which often exceed $5 million.

The set of "recent" studies use more accurate risk data and improved statistical techniques. As a result of these changes, the range of VSL estimates has been reduced. Most studies now are in the range of $1.5 million to $3.0 million. Indicative of this degree of consensus is the fact that the "official" VSL estimate used by the USEPA in many of its studies is $2.8 million (Abt Associates, 1988).

Representative of the recent compensating wage studies is Martinello and Meng (1992). The authors use Canadian data on industrial wages and workplace fatalities and injuries to identify the risk premium in alternative occupations. Wages are assumed to depend on workers' characteristics, risk of injury, risk of death and a variety of other factors (industry type, region of Canada, etc). This relationship is characterized statistically using OLS and several different equation specifications. Using the estimated risk premia and the alternative equation specifications, the authors calculate the a range of values for VSL ($2,462,000-$7,679,000). With respect to the range of values, the authors' preferred point estimate is $6,926,000. This study's relatively high VSL estimate may be due to the fact that it is one of the few studies to separate out risk of injury from risk of death. Previous studies may have underestimated the VSL by combining these two types of risks.

Despite the relatively large number of studies examining VSL's and the recent improvements in their methodologies, there remain a number of concerns in regard to using these estimates in the context of environmental decision-making. First, these studies assume that workers are fully informed regarding the risks they face. It may be debatable whether this is a reasonable assumption in the context of workplace risks, but there is little reason to believe that people in the general population have an accurate understanding of the risks posed by toxic substances in the environment. Given the lack of consensus among scientific experts, this is hardly surprising (cf. the discussions concerning exposure to DDT, lead or dioxin in the next chapter).

Second, the observed wage differentials reflect workers' compensation for risks voluntarily undertaken. This is some evidence that people differentiate between voluntarily and involuntarily assumed risks when making decisions (Mendeloff, 1988). Since most exposure to toxic substances is involuntary, this means that workplace-based measures of VSL may not accurately reflect valuations in reductions of risks from them. Third, the character of fatalities that occur in the workplace differs from those that are thought to occur from exposure to toxic substances in the environment. The former (e.g., being crushed to death) tend to be sudden and have relatively little pain and suffering associated with them while the latter (e.g., skin cancer) tend to be lingering and imply substantial amounts of pain and suffering.

The second indirect method of inferring VSL uses information regarding individuals' expenditures to avoid or reduce pollution-related health damages. This approach stems from the observation that, for a number of types of risks, the probability of harm can be influenced by an individual's actions. Thus, wearing seatbelts reduces the risk of dying in an automobile accident and installing smoke detectors reduces the risk of dying in a household fire. In principle, the expenditures require to take these averting actions can be used to infer an individual's WTP for a small reduction in the risk of fatality and, thus, his\her VSL.

Cropper and Freeman (1991) and Cropper and Oates (1992) survey the literature concerned with estimating VSLs from averting behaviour. They conclude that these studies produce VSL estimates that are generally less than $1.0 million. An exception to the finding that averting behaviour studies generally find low VSL estimates is Murdoch and Thaler (1990). The authors estimate the increase in expenditures on sunscreens as one means of measuring averting behaviour in the face of ozone depletion. Their estimate of VSL is $2.04 million.

There are several difficulties with using averting behaviour to infer VSL. First, it is difficult in some cases to measure the cost of the activity. For example, the cost of using a seatbelt depends on one's like or dislike of seatbelts and the prevailing enforcement of mandatory seatbelt laws. Second, some averting activities produce more than one output. For example, running an air conditioner improves indoor air quality (and thus may reduce the onset of an asthma attack), but it also reduces indoor temperatures. It is difficult to determine the relative importance of these two outputs and thus infer what proportion of the capital and operating costs of the air conditioner can be said to be related to pollution averting behaviour.

The third indirect method of inferring the value of avoiding premature death uses information regarding an individual's expected lifetime earnings and is termed the 'human capital approach'. The advantage of using earnings as the basis for the benefit to society of avoiding a premature death, is that the data are relatively straightforward to collect. The obvious difficulty, however, is that the 'value' of an individual's avoided death is a function only of his\her contribution to the market economy. Thus, the value of avoiding deaths for children, unemployed people and people not in the labour force is very difficult to determine. Finally, these estimates are not directly based on the individual's own WTP to acquire small reductions in the risk of premature death. Because of these methodological concerns, the human capital approach is now used infrequently. The basis for this method and its limitations are reviewed in Victor (1985), Mendeloff (1988) and Cropper and Freeman (1991).

In contrast to the indirect methods, the direct method of estimating individuals' WTP for risk reductions relies on surveys which elicit their valuations. This method, known as the contingent valuation method (CVM), is most commonly used in recreational benefits studies but is becoming increasingly popular as a means of obtaining VSL estimates. These studies offer the possibility of providing WTP and VSL measures that can be applied to environmental decision-making with more confidence than those from compensating wage studies for several reasons. First, they are not restricted to using employed workers' valuation of risks and, thus, can draw samples that are representative of the entire population. Secondly, the type of risk to be considered and the amount of information that the individual possesses can be controlled in the survey's design.

Cropper and Freeman (1991) survey the limited number of CVM efforts and conclude that they yield VSL estimates that are usually comparable to those found in compensating wage studies. Specifically, the CVM-based estimates are in the range of $3.53 million to $5.29 million. For example, Viscusi, Magat and Huber (1991) find VSL estimates ranging from $0.8 million to $3.72 million depending on how the risks of bronchitis are presented to respondents.

Mitchell and Carson (1986) provide an interesting exception to the congruence between CVM and compensating wage studies' results. In this study, in-home surveys are conducted in order to determine individual's WTP for reductions in the concentration of THM (trihalomethane) in drinking water supplies. Depending on how the risk reduction is specified, the WTP responses imply VSL estimates of between $326,000 and $698,000. These estimates are an order of magnitude below the results of the compensating wage studies. One possible explanation for this discrepancy stems from the fact that the average age of respondents is 43 years and the latency period for cancers caused by THM exposure is on the order of 20-30 years. These observations suggest that respondents may be discounting the significance of a disease that might happen far off into the future and one that would have a relatively small effect on the expected number of years lived.

This study is important because it suggests that the characteristics of the risk being considered are important for establishing how people value them. In particular, it may be the case that the 'pain and suffering' feature of cancers induced by exposure to environmental contaminants may be partially or wholly offset by individual's discounting of diseases that are perceived to occur only in the distant future.

While CVM studies hold the promise of more accurate valuations of risks from environmental contaminants, they are not without their problems. Perhaps the most significant problem concerns the ability of respondents to understand and work with changes in the very small probabilities associated with exposure to toxic substances in the environment. The problem is specific to efforts to elicit VSL estimates and complements the problems of CVM such as the possibility of strategic behaviour that have already been identified.

B.2.3 Estimates of the Value of Reduced Morbidity

Exposure to toxic substances in the environment is more likely to cause acute or even chronic illnesses than death. For example, the types of illnesses that can be induced from air pollution include eye irritation, coughing, asthma attacks and bronchitis. In addition, exposure to water-borne contaminants can lead to developmental problems in children and impaired operation of the immunological system. The techniques used to obtain estimates of WTP for small reductions in the risk of contracting these illnesses are the same as those used to obtain estimates of WTP for small reductions in the risk of death.

Compensating wage studies are used infrequently to value acute or chronic illnesses (Mendeloff, 1988). A recent exception is Martinello and Meng (1992). The authors include the risks of minor and severe injuries in their hedonic wage determination equation. Minor injuries include hernias, hearing loss and burns while severe injuries include concussion, poisoning and amputation. The authors use the wage equation's estimated coefficients to derive wage premia and thus, workers' valuation, for the two types of injury. The imputed value for a minor injury varies between \$12,277 and \$16,369 and for a severe injury between \$181,600 and \$217,400.

Another indirect method of inferring the value assigned to avoid illness employs data on expenditures by individuals to avoid those illnesses. For example, these 'averting behaviour' studies use the purchase of bottled water as an indication of the WTP to avoid water-borne contaminants and the use of air conditioners to avoid air borne contaminants.

Murdoch and Thayer (1990) is an example of the averting behaviour technique. In this case, it is assumed that the only defensive action taken by people in response to the increased risk of skin cancer (due to ozone depletion) is increased use of sunscreen products. Based on a projection of the increase in Uvb exposure and on estimates of sunscreen's efficacy in avoiding skin damage, the authors estimate that use of sunscreen will grow by an annual rate of 0.672% (holding fixed income, price and other relevant factors). The present value of the increase in aggregate expenditures on sunscreen that is attributable to ozone depletion is estimated for the U.S. to be $16.03 billion. The authors point out that this estimate is obviously an overestimate of the savings to be enjoyed by slowing ozone depletion because there are less costly ways of avoiding skin cancer (e.g., reducing sun exposure, wearing a hat, etc.). Nonetheless, the estimate of $16.03 billion is only half of the USEPA's estimate of the aggregate costs of illness (defined more fully below) associated with ozone depletion.

The third indirect method used to evaluate avoided illnesses is based on the monetary expenses incurred due to illness. Thus, the 'cost of illness' (COI) approach includes expenditures on visits to doctors' offices, hospital stays, medication as well as lost wages, as the savings to society enjoyed by avoiding an illness. This technique has the advantage of being relatively straightforward to measure. Unfortunately, this advantage is offset by the fact that the approach considers only the pecuniary aspects of illness and ignores the costs of pain and suffering. In addition, it is not based on the effected individuals' WTP.

Cropper and Freeman (1991) survey the results of efforts to measure the COI for a variety of illnesses related to exposure to environmental contaminants. COI estimates range from $3.29 per day of drowsiness to $26.67 per day for itchy eyes . However, WTP to avoid these symptoms usually exceeds the COI estimate. For the same illnesses, the estimated mean WTP's are $260.00 and $88.81, respectively.

Direct survey techniques are also used to elicit individual's WTP to secure reductions in the risk of illnesses related to exposure to environmental contaminants. For example, estimates for the median WTP per day of a cessation of coughing from a variety of studies range from $1.83 to $20.15 (Cropper and Freeman, 1991). Interestingly, the mean WTP for the same problem and the same set of studies ranges from $45.00 to $650.50 per day.

Viscusi, Magat and Huber (1991) report on a variant of the CVM survey approach to value avoidance of chronic bronchitis (a disease whose condition is aggravated by air pollution). There are several novel features to this work. First, the study examines the valuation of a chronic illness as opposed to the more commonly studied acute illnesses (e.g., asthma attacks). Second, the authors treat contracting bronchitis as a stochastic event rather than a certain one. Finally, the authors use two types of trade-offs in order to estimate WTP: first, the risk of bronchitis versus the risk of automobile accidents and, secondly, the risk of bronchitis versus changes in income. The first trade-off is thought to be somewhat easier for respondents to conceptualize and acts as a 'double-check' on the estimates from the second, more conventional trade-off. The results of the study indicate that the median respondent treats the risk of contracting chronic bronchitis as being equivalent to (0.32)x(the risk of a fatal automobile accident). In addition, the median valuation of a case of bronchitis is $772,500 using the bronchitis-auto trade-off and $551,600 using the bronchitis-income trade-off.

Schecter and Kim (1991) also employ a two-part approach to valuing illnesses related to air pollution. However, in this case the authors compare the results of a CVM survey with the results of indirect COI valuation based on avoided medical costs. The survey asks respondents for their WTP for a 50% improvement in perceived air quality conditions. Median WTP for the improvement ranged from $39.00 to $105.00/year depending on the perceived level of air quality. In contrast, the median COI estimate is

$26.50/year.

The CVM approach has several potential advantages. It obtain's WTP estimates directly from individuals and elicits detailed information on the severity and characteristics of the illness being considered as well as socio-economic data for the respondent. In addition, because the survey uses individual responses, researchers can determine the shape of the distribution of WTP estimates. This feature has allowed researchers to establish that most WTP distributions are skewed with mean responses exceeding the median because of the presence of a small number of very large WTP responses.

The CVM approach to valuing changes in morbidity also has its disadvantages. Cropper and Freeman (1991) report that estimates of WTP to avoid symptoms that violate respondent's budget constraints are not uncommon. In addition, there are problems associated with defining illnesses. For example, it is difficult to ensure that all respondents mean the same thing when comparing WTP for 'mild' versus 'severe' symptoms. Finally, a general problem is that CVM surveys usually provide WTP estimates for symptoms rather than for changes in environmental quality. It is left to the researcher to establish the link between the incidence of symptoms and environmental quality. This last task is frequently made quite difficult by physical scientists' incomplete understanding of the relationship between the presence of contaminants in the environment and the prevalence of disease in the affected population.

A number of the issues that are raised in this review of health benefits arise when an analysis is attempted of a specific policy proposal. For example, DPA (1990) is a recent application of health benefit estimates to the assessment of an environmental policy. The authors combine a sophisticated atmospheric dispersion model, dose-response curves and estimates of the economic value for a variety of damages to assess proposed revisions to Ontario's air pollution legislation. The analysis covers 96 airborne contaminants emitted by over 3500 establishments.

Health benefits are defined to include avoided mortalities, reduced consumption of medical services and reduced restricted activity days. The valuation of these benefits is based on point estimates of VSL, cost of restricted activity days, hospitalization costs and the cost of emergency room visits taken from published U.S. studies. For example, the authors use two estimates of VSL: a "low" estimate of $2.2 million and a "central" estimate of $6.95 million.

Benefits from emission abatements include avoided damages to human health, plants, other wildlife and physical structures. The total value of these benefits is estimated to range from $1.5 to $5.0 billion annually .The proportion of this total that is related to avoided human health damages is large. For example, the value of reduced mortalities and morbidity due to reduced SO_2 emissions comprise the largest single component of total benefits (the SO_2-related health benefits are estimated at between $0.37 and $3.2 billion annually and represent between 133 and 372 avoided deaths each year).

B.2.4 Assessment of Health Benefit Estimates

As the above discussion indicates, estimating the economic value associated with avoiding serious illness or premature death is fraught with difficulties. Indirect methods, such as compensating wage studies, exploit observed market behaviour to obtain estimates. However, they suffer from the use of objective risk data rather than workers' subjective evaluations and from the difficulty in transferring benefit estimates from an occupational setting to an environmental context. Direct methods allow researchers to learn more about respondents' decision making processes but are also constrained by respondents' ability to comprehend hypothetical questions concerning small changes in statistical risks.

Environmental economists have had more success deriving VSL estimates than calculating the values of avoided illnesses. This is due, in part, to the fact that the former had already been examined by labour economists. Another factor may be definitional. In order for people to demonstrate or reveal their WTP to avoid illness, they must have a clear idea of what is being avoided. For the researcher, this poses the difficulty of defining what is meant by an illness. For example, do people avoid the illness itself or its symptoms? Furthermore, how is the severity of an illness to be measured or indexed objectively?

Despite the host of problems identified in this section, the valuation of avoided illnesses and premature deaths is one of the most significant contributions made by economists to the appraisal of environmental regulations. The evaluation techniques reviewed here are all consistent with fundamental economic theory and are based on increasingly sophisticated data sets and statistical methods.

B.3 Benefits to Industry

B.3.1 Definitions

Measuring the benefits accruing to firms from improvements to environmental quality has not received the same attention as have the benefits associated with recreation or human health. This is probably because in many if not most instances of environmental contamination, firms are associated with the generation of pollution rather than being amongst the group of agents harmed. In addition, with the exception of a fairly small group of industries, most firms do not rely heavily upon environmental quality as an input to their production processes. The obvious exceptions to this rule include agriculture, forestry, tourism and some sectors of the manufacturing industry.

The modelling technique used to consider the benefits accruing to firms begins by assuming that the environment supplies some input to the firm's production process. This input can be fresh water for a beer maker, a school of fish for a charter boat operator or clean air for a forestry company. In all of these cases, the task of the economist is to estimate the impact on the firm of a reduction in either the quantity or quality (i.e., productivity) of the environmental input. Table 5.3 summarizes the empirical estimates of benefits to industry that are presented in this section.

Table 5.3 Estimates of Benefits to Industry

1. Agriculture

Date	Researcher	Benefit Estimate
1991	Adams and Crocker	The value of agricultural output would fall by \$438 million - \$3.73 billion per year if there were to be an increase in air pollution.
1992	Infometrica	Estimated damages of acid rain in Ontario : \$3.2 million - \$23.5 million annually.
1992	OECD	If double CO_2 loadings, loss in value of U.S. agriculture : \$11.4 billion - \$64.9 billion.

2. Forestry

Date	Researcher	Benefit Estimate
1992	Infometrica	Annual value of lost production attributable to acid rain : \$3 million.
1992	OECD	If double CO_2 loadings, annual loss in value of US output : \$5.12 billion.

3. Tourism

Date	Researcher	Benefit Estimate
1990	Apogee	An increase in water quality will lead to 1200 new full time jobs and increased income of \$44.6 million annually.

1992	Infometrica	Losses in revenue due to acid rain in recreational fishing - $42 million annually.
1992	Infometrica	Revenue loss of $2.6 million annually for recreational operators attributable to phosphates in the Great Lakes.

4. Commercial Fishing

Date	Researcher	Benefit Estimate
1992	Infometrica	Revenue Losses to commercial fishery attributable to acid rain : $0.5 million annually.

B.3.2 Benefit Estimates for Specific Industries

The industries for which benefit estimates are available are those which are believed to be the most heavily reliant on environmental inputs. In the cases of agriculture and forestry the principle concern stems from current and future atmospheric temperature changes due to the depletion of stratospheric ozone and the accumulation of 'greenhouse gases' such as CO_2. On the other hand, in the case of most tourism-related industries, commercial fisheries and some manufacturing industries, the major concern stems from a variety of types of water pollution.

The studies assessing the impact of changes in air pollution on agriculture and forestry share a common methodology. The first step involves estimating the physical relationship between environmental quality and the productivity of farms or woodlots. The second step predicts the impact on firms' costs due to the change in productivity. This step assumes firms' seek to minimize the cost of the lost productivity of the environmental input by substituting the use of other inputs if this is feasible and profitable. The third stage uses a model of the output market structure to translate changes in the costs of production into output price changes and uses the latter to predict total welfare changes for firms and consumers.

The task of determining the economic value of the damage to the agricultural industry from degradation of the environment is complicated by two factors. The first of these is the uncertainty in the scientific community regarding the implications of global warming on specific crops and growing regions (cf. the discussion in OECD, 1992).

The second complicating factor is something that is shared with all efforts to determine industrial benefits from reductions in pollution. In order to determine the reduction in welfare of a firm (and its consumers), it is necessary to know how a reduction in the productivity of the environmental input will effect the firm's cost of production and what this will in turn imply for the price of the firm's output. In order to estimate these two responses, it is necessary to know something about the role of the input in the firm's production function (specifically whether there are substitutes for the input) and the structure of the market for the firm's output (specifically whether there are substitutes for the firm's output).

To see the role played by the availability of input and output substitutes, consider the extreme case where a firm possesses a perfect substitute for the environmental input (increased irrigation can offset the effects of higher temperatures) and consumers possess a perfect substitute for the firm's output (suppose rice flour can be used instead of wheat flour). In this case, the firm and consumers can avoid much of the reduction in welfare associated with higher temperatures through greater use of substitutes. To the extent that these mitigating activities are feasible but are not incorporated in the analysis, the industry benefits from pollution reduction will be overestimated.

Most efforts to estimate the damages to agriculture and forestry incorporate these substitution possibilities to some degree. For example, Adams and Crocker (1991) survey the set of studies concerned with the impact of air pollution on U.S. agriculture and find reductions in the aggregate value of agricultural output that range from $438 million to $3.73 billion. This aggregate estimate masks the fact that in a number of specific cases farmers actually benefit from pollution because crop reductions lead to higher prices and, thus, to higher revenues. Similarly, studies of the impact of acid rain on Ontario's agricultural industry put the estimated value of damages in the range of $3.2 million to

\$23.5 million annually (Infometrica, 1992). More recently, the EPA considers the potential impact of global warming on U.S. agriculture (reported in OECD, 1992). A simulation which assumes a doubling of CO_2 loadings leads to future agricultural losses with an annual value in the range of \$11.4 billion to \$64.9 billion.

The forestry industry also is also susceptible to air pollution. The EPA study referenced above also considers the impact of a doubling of CO_2 loadings on the output of the U.S. forestry industry. The results are quite severe: simulations indicate that up to 40% of American commercial timberland could be lost with a resulting reduction in the annual value of output of \$5.12 billion. Studies of the impact of acidic deposition on Ontario's forests estimate the annual value of lost production to be approximately \$3 million (Infometrica, 1992).

The third industry whose productivity and profitability is closely tied to environmental quality is the tourism industry. Modelling the impact of pollution changes on this differs somewhat from the method used for agriculture and forestry. In the case of tourism the analysis is in part based on changes in firms' productivity and in part on changes in the demand for the firms' output. For example, a reduction in water quality may reduce fish populations (making it more difficult for chartered fishing guides to ensure that anglers are successful in their efforts to catch fish) and it may reduce anglers' willingness to hire guides and charter boats. Thus, changes in environmental quality can have both supply-side and demand-side effects on this industry.

Despite the extensive research done on the consumer benefits from improved recreational opportunities, there is relatively little information regarding the impact of pollution on firms in the tourism industry. Infometrica (1992) reports on several studies examining the Ontario tourism industry. The presence of phosphates in the Great Lakes, for example, is responsible for fish kills and has lead to revenue losses of approximately \$2.6 million annually for recreational operators. In another study the impacts of acidic deposition on the spot fishing industry are considered. Losses in revenue due to reduced fish populations amount to \$42 million annually. Finally, Apogee Research et. al. (1990) forecast increased demand for recreational boating, swimming and fishing as a result of water quality improvements under the Remedial Action Program. These increases in demand are predicted to generate 1,200 new full time jobs in Ontario and create approximately \$44.6 million in new incomes annually.

The fourth industry to be considered is the commercial fishery. As in the case of tourism, water quality reductions can have both supply and demand-side effects for this industry. For example, the presence of heavy metals in the Great Lakes is suspected of reducing the populations of some fish which are caught commercially. The presence of heavy metals also reduces the demand for fish caught on the Great Lakes.

There is also surprisingly little research on the relationship between the profitability of Canadian commercial fisheries and water quality. For example, two recent studies of proposed Ontario environmental regulations (DPA, 1990; Apogee Research et. al., 1990) do not estimate the impact of improved air and water quality on the commercial fishery. One potential reason for this omission is that the Ontario commercial fishery is quite small and, thus, reductions in its level of activity will have a small impact on the Ontario economy. For example, Infometrica (1992) reports on a study of the impact of acidic deposition on the Ontario commercial fishery. The study concludes that revenue losses will amount to \$0.5 million annually.

B.3.3 Other types of Industrial Benefits

There are a variety of other types of benefits that can accrue to firms and industries as a result of government efforts to reduce pollution levels. These benefits range from cost savings that result from the introduction of pollution abatement and control (PAC) equipment to increases in the demand for the outputs of environmental consultants and equipment manufacturers that occur from new government regulations. Each of these impacts is discussed briefly here.

Firms are often required to install PAC equipment and reform their operations to meet environmental regulations. These actions can sometimes bring benefits to the firms in the form of cost savings (for example, improved control systems may reduce energy use as well as cut emissions of a pollutant) or in the form of increased revenues (for example, the firm may find trace elements of marketable products in its effluents).

Recent surveys conducted in Canada and the U.S. suggest that these effects are relatively small. Statistics Canada (1992) reports that PAC activities result in new revenues of $80.1 million and cost savings of $73.7 million for Canadian firms. Each of these categories constitute approximately 10% of PAC operating and maintenance costs. Rutledge and Leonard (1992) report "costs recovered" from PAC expenditures in the U.S. to be in the range of $2.43 billion to $2.9 billion for the period 1985-1990. This represents approximately 5% of PAC operating costs [4].

It should not come as a surprise that PAC expenditures of the magnitude reported above provide positive stimulus for the suppliers of this type of equipment. A report prepared for the Ontario Ministry of the Environment (Woods Gordon Management, 1989) indicates that the industry devoted to the production of environmental protection equipment and services reported sales in excess of $2 billion in 1989 and employed 28,000 people. Glenn (1992) points out that Canadian sales of air pollution control equipment exceed $300 million in 1985 and these sales support employment of more than 17,000 people. The research done by Apogee Research (1990) supports these findings. This report forecasts the economic impacts of meeting the water quality objectives contained in the Remedial Action Plan for the Great Lakes. The authors predict that 4,500 new full time jobs will be created in Ontario as a result of pollution abatement and control expenditures alone.

It is necessary to use caution in interpreting these employment and expenditure figures as benefits to environmental protection regulations. Suppose a new environmental regulation leads firms to increase their expenditures on PAC capital. The fact that some of society's productive resources (including labour and capital) are diverted away from other uses to the production of this capital equipment is indicative of a cost being imposed rather than a benefit being earned. That is to say, there is an opportunity cost to PAC expenditures because they necessarily reduce the quantity of resources available to produce other goods and services. Unfortunately, a discussion of the evaluation of the employment effects of government policies is beyond the scope of this report. Suffice to say that the opportunity costs of diverted resources (the value of the outputs they would have produced) must be subtracted from the value of any output those resources produce (the reductions in pollution emissions) in order to assess the net benefits of the government policy.

Finally, firms in industries such as pulp and paper, food and beverages, and petrochemicals use water as an important input to production. These firms differ, however, in the quality of water that is acceptable as an input. Food and beverage producers as well as firms producing optical lenses, photographic film and electronic components usually require very clean water and employ extensive in-house water treatment facilities. Other firms such as pulp and paper producers and metal producers do not have particulary stringent water quality requirements. For firms that require very clean water, there is the possibility that water quality improvements in the Great Lakes will yield direct monetary benefits in the form of lower water treatment costs.

There is, unfortunately, a surprising dearth of information regarding water use in Canadian industries and the expenditure levels that support that use. The limited information that is available, however, suggests that water use expenditures, in general, and water treatment expenditures, in particular, are not a significant component of total costs for most manufacturing firms. The basis for this conclusion is the data collected in Environment Canada's Industrial Water Use Survey (IWUS). The IWUS is a cross-sectional survey conducted every five years (since 1980) of the largest water using manufacturing firms in Canada (Tate and Scharf, 1985). For 1985 (the most recent year for which data are available) the survey indicates that no major water using industry in Canada spent more that 1% of its total costs on all water-related costs (the survey collects data on four separate facets of water use: intake, pre-use treatment, recirculation, and post-use treatment and discharge). Furthermore, pre-use treatment costs accounted typically for less than 25% of total water-use expenditures. Thus, while there is the possibility that improvements in the quality of the Great Lakes water may benefit a small number of intensive users of high-quality water, it is not expected to lower costs significantly for the majority of firms.

B.3.4 Assessment of Industry Benefit Estimates

The measurement of the magnitude of benefits accruing to firms from improvements in environmental quality is an area of research which has received relatively little analysis. The most important exception to this conclusion is the extensive literature concerned with agricultural productivity and global warming.

One important factor limiting this type of research is the current lack of understanding of the role of environmental quality as an input in industrial production processes. It is well known that many firms use the assimilative capacity of the environment as an input but much less is known about the contribution of improvements in environmental quality to reducing production costs.

B.4 Benefit Transfer

The benefit estimates which have been presented thus far are valuable because they indicate the order of magnitude of benefits accruing from environmental quality changes. It could be wondered, however, whether the benefit estimates generated from a wide variety of circumstances and jurisdictions can be applied usefully to the Great Lakes environment. This question is a specific example of a more general methodological issue concerned with the 'transfer' of empirical estimates of benefits from one situation or case study to another.

It is clear from a review of the empirical literature that the use of benefit estimates originally estimated in another context is common practice. For example, it is standard practice for consultants producing reports to government to use values for benefits and costs which are taken from the existing literature rather than being generated by original research (Apogee Research, 1990; Abt Associates, 1988). However, despite the frequency of its application, there has been little analysis of the appropriateness of this methodology or of the best way to evaluate the transfer of benefit estimates from one study to another.

In the Foreword to the proceedings of a recent conference devoted to examining the topic of benefit transfer, Bingham provides a succinct definition of benefit transfer and a defence of its use:

> Benefits transfer is the use of information from existing nonmarket valuation studies to develop estimates for another valuation problem. It can reduce both the calendar time and resources needed to develop original estimates of values for environmental commodities. These estimates are used to evaluate the attractiveness of potential government policies, to assess the value of policies implemented in the past and to identify the compensation required...when toxic substances, such as oil or PCBs, are released to the environment. (Bingham, et. al., 1992, p. iii)

The examples of the benefit transfer methodology that are referenced in the previous paragraph are representative of the simplest way to do benefit transfer. In these instances, a point estimate for some value (for example, the value of a day's recreational fishing or the value of a statistical life) is culled from the literature and used without adjustment in the case under study. Some effort may be applied by the researcher to choose the 'best' or 'most reasonable' estimate from the available range of estimates but the criteria employed to make this determination are frequently not reported.

More recently, researchers have grown more sophisticated in their attempts to transfer benefit estimates (cf. the discussion in Bingham et. al., 1992). In these cases, estimates are adjusted before being applied to new situations. This adjustment procedure is meant to take into account differences in socioeconomic and demographic characteristics as well as institutional features between the original case study and the new case study. Kask (Bingham et. al., 1992) examines the benefits of altering water quality regulations and employs this type of methodology. The initial results of the Kask study are reviewed in chapter 6 in the dioxin case study.

The advantages of transferring benefits are summarized in the quotation from Bingham. These include savings of time and scarce research funds. These savings may in turn mean that government can act more rapidly to confront environmental problems. There are, however, potential problems associated with transferring benefit estimates from existing studies. First, some researchers may neglect to assess the quality of the original research that generated the point estimate that they plan to use. This is particularly worrisome as repeated use of any specific estimate tends to lends legitimacy to the estimate by sheer weight of repetition. Second, it is not clear what is the best method for adjusting transferred benefit estimates. Unfortunately, economic and statistical theories usually provide little guidance in this area and the preferred means of adjustment is often left to the judgement of the researcher.

C. ESTIMATES OF THE COSTS OF ENVIRONMENTAL REGULATIONS

A government policy is said to impose a cost on the private sector of the economy if it demonstrably leads to a lowering of some private agent's welfare. For example, the introduction of the Free Trade Agreement caused some households' income to fall and also caused some retail prices to rise. These impacts were counted as costs associated with the FTA because economic theory predicts that they lower the welfare of households.

When assessing a government policy, however, it is usually not enough to be able to say that welfare levels fall for some households. It is usually also necessary to attempt to say by how much welfare levels changed, so that their magnitude may be compared to whatever benefits are associated with the policy. The general principle guiding the measurement of the magnitude of costs for any one household is to ask, "what is the minimum amount of compensation that would be hypothetically required to leave the household's level of welfare at the same level as it was prior to the policy change?". The total cost associated with a policy is the sum of these hypothetical compensations across all of the affected households.

An ideal measure of the costs of environmental regulations would be based on private agents' responses to the constraints imposed by the regulations. In addition, it would employ the compensation measures described above to measure costs. The data and computational requirements for such an exercise are daunting and it is only fairly recently that an empirical technique (computable general equilibrium, or CGE, models) has been developed which is capable of carrying it out. A detailed discussion if this type of model is in section C.4 of this chapter.

There are several alternative ways of considering the costs arising from environmental regulation. Each of these requires less data than a CGE model but is, in some way, less satisfactory a way of measuring costs. These methods include examining pollution abatement and control (PAC) expenditures by firms, measuring productivity losses by firms, and measuring the macroeconomic impacts of regulations. In the next several sections, each of these techniques is presented and assessed. Table 5.4 summarizes the cost estimates that are presented in this section.

Table 5.4 : Estimates of Costs of Environmental Legislation

1. Pollution Abatement and Control Expenditures

Date of Study	Researchers	Cost Estimate
1985	Federation of Canadian Municipalities	Sewage collection and treatment costs Average per capita spending : 1983 $21.75 1985 $19.57
1989	Sanger, Heeney and Victor	4% reduction in nitrogen oxide has a Marginal Abatement Cost (MAC) of $63 per tonne. 22% reduction - MAC of $2,360 per tonne 50% reduction - MAC of $102,262 per tonne
1990	Apogee et. al.	To meet edible fish water quality Annual increase in capital and operating expenditures for Bay of Quinte : $6.56 million To meet self-sustaining sport fishery for Hamilton Harbour : $9.49 million annual increase in capital and operating expenditures, and for Toronto : $125 million - $142.35 million.
1990	Hickling	For Ontario newsprint mills to meet proposed air quality regulations : $1.42 - $2.20 per tonne of output. For petroleum firms : $0.65 - $1.69 per tonne of output.
1991	VHB Research and CHAM Hill Engineering	Petroleum refineries annualized capital and operating expenditures per kg of effluent $14.18 - $337.04.
1992	Rutledge and Leonard	Annual US government PAC expenditures : $27.5 billion - $30.9 billion.

1992	Rutledge and Leonard	Annual US firms PAC - $73.02 billion.
1992	Statistics Canada	Canadian firms (from survey) increase in PAC capital and operating expenditures : $889 million.

2. Productivity Losses

Date of Study	Researchers	Cost Estimate
1985	Christainsen and Tietenberg	8% - 12% of productivity decline in US since mid-70's due to impact of environmental regulations.
1990	Barbera and McConnell	US iron and steel 10 % of loss in productivity. US pulp and paper 30 % of loss in productivity.

3. Macro-economic measures of Costs

Date of Study	Researchers	Cost Estimate
1985	Christainsen and Tietenberg	Environmental legislation in the US has raised the level of unemployment by .21 % in late 1970's.
1990	Sonnen (Canada)	By 2005 a reduction in per capita real disposable income of $710 - $1,830 due to the imposition of 7 specific environmental regulations through MISA. In addition, a higher level of unemployment, higher prices, and lower capital stock.
1990	Jorgenson and Wilcoxen (United States)	Between 1974-1985 the economy would have grown .191% faster than observed had there been no environmental regulations. By 1985, GNP would have been 2.6% higher than what was actually observed.

4. Aggregate Welfare Losses

Date of Study	Researchers	Cost Estimate
1990	Hazilla and Kopp	Costs of Clean Water and Air Acts. Full social cost for 1981-1990 : $1,164 billion. US GNP and capital stock had fallen by 6 % by 1990 because of regulations.
1992	Jorgenson, Slesnick, and Wilcoxen	Carbon tax to keep greenhouse gases to 1990 levels would reduce GNP and capital growth and result in a lifetime loss of aggregate welfare of less than 1 %.
1993 (?)	Nordhaus	Carbon tax of $70 per ton would lower greenhouse gas emissions by 30 % and increase net welfare by $238 billion for the US economy.

5. Remediation of In-Situ Toxins

Date of Study	Researchers	Cost Estimate
1990	Apogee Research	Canadian AOC annualized costs for sediment removal and treatment : Hamilton Harbour $230,000 Thunder Bay - $3.55 million Port Hope Harbour - $100,000 - $150,000
1992	Leger	US AOC remedial Total Costs

C.1 Pollution Abatement and Control Expenditures

In the last twenty years a large number of firms have had to alter their operations in order to comply with regulations aimed at preserving environmental quality. These alterations can include redesigning a product, installing capital equipment to reduce waste flows, changing production processes to reduce the creation of waste residuals and substituting away from pollution-causing inputs. The common feature of all of these alterations is that they cost money to implement. As a result of these effects, the most commonly cited measure of the cost of pollution control legislation is the level of pollution abatement and control (PAC) expenditures undertaken by firms.

Several studies document the aggregate PAC expenditures undertaken by industry in Canada and the United States. Statistics Canada (1992) surveys Canadian firms in order to determine their PAC capital and operating expenditures. If allowance is made for nonrespondents, then Statistics Canada finds that annual PAC capital expenditures are $1.45 billion. This represents approximately 6% of total private capital expenditures for 1989 and less than 1% of Canadian GNP. In the same year, PAC operating and maintenance expenditures are $889 million. These expenditures are not evenly distributed across industries. Pulp and Paper accounts for 32% of PAC capital expenditures and Primary Metals accounts for 25%.

Rutledge and Leonard (1992) review PAC expenditures by firms, households and government in the U.S. over the period 1972-1990. Over that period, real PAC expenditures rose at annual rate of 3.2%. In addition, Oleweiler (1992) points out that the size of total PAC expenditures relative to U.S. GNP over this period remain fairly constant (1.4% in 1972, 1.9% in 1981 and 1.6% in 1990). These figures differ from the Canadian experience because they include all PAC expenditures by all sectors of the economy, not just capital expenditures by firms.

Rutledge and Leonard also indicate that in 1990, total PAC expenditures are $115.9 billion. Of this total, firms are responsible for 63%, households for 11% and Government for 26%. Another way to examine the total PAC expenditures is to divide them according to their purpose. Of the total for 1990, 94.6% is used for pollution abatement, 2% for regulation and monitoring and 3.3% for research and development.

While these figures indicate that PAC expenditures have been a relatively small and constant proportion of national income, they tend to mask a significant degree of variation in PAC expenditures across industries and individual firms.

Hickling Management (1990) estimates the expenditures necessary for Ontario firms to meet a set of proposed air quality regulations in Ontario. These expenditures include annualized capital costs, operating and maintenance costs and monitoring costs and are estimated using a combination of engineering and econometric techniques. PAC costs are usually small relative to output prices but there are significant variations across industries. For example, PAC costs for newsprint mills range from $1.42 to $2.20/tonne of output which is well less than 1% of selling price. However, petroleum refineries are forecasted to have PAC costs of $0.65 to $1.69/tonne of output which the authors calculate to be between 3.3% and 8.7% of profit per unit.

Another important source of forecasted PAC expenditures comes from a series of studies undertaken to assess the impact of Ontario's MISA (Municipal and Industrial Strategy for Abatement) program. VHB Research and CH2M Hill Engineering (1991) provide a detailed study of sector-specific waste water flows, abatement technology alternatives and PAC expenditures. For each sector, engineering data are used to identify the "minimum technically achievable loading", and to estimate the cost of reaching this emission target. PAC expenditures vary across industries and even across firms within an industry. In Petroleum refining, for example, annualized capital and operating PAC expenditures per kilogram of effluent are $14.18 for Shell, $50.60 for Petro-Can and $337.04 for Suncor.

There is a limited amount of information regarding government expenditures to control or eliminate its own pollution. Rutledge and Leonard (1992) indicate that the annual PAC spending by all levels of U.S. government averages between $27.5 and $30.9 billion during the last half of the 1980's. Approximately half of this is accounted for by spending on sewage treatment facilities.

Unfortunately, PAC expenditures by Canadian governments are not separately recorded in government accounts or by Statistics Canada. There are however, some indications of the magnitude of these expenditures. For example, the results of a survey of Canadian municipalities' spending on environmental quality control are summarized in Federation of Canadian Municipalities (1985). With respect to sewage collection and treatment, the survey indictates that past increases in costs have been quite small. In 1968, average per capita spending is $19.57 while in 1983, it is $21.75 (recall, all figures are being expressed in 1992 dollars). It is not surprising, then, that the same municipalities estimate that the average per capita cost needed to bring their sewage systems up to an "acceptable" condition is close to $290.

More recent and detailed data regarding forecasted government PAC expenditures is available from Apogee research <u>et. al.</u> (1990). This report uses engineering data to estimate the costs of meeting the alternative water quality objectives in the Great Lakes'

Remedial Action Program. In particular, the spending by local and provincial governments needed to reduce non-point urban runoff and to upgrade sewage treatment plants is estimated for each Area of Concern. For example, to meet the 'edible fish' water quality objective (the second most stringent objective), the increase in annualized capital and operating expenditures for the Bay of Quinte is forecasted at $6.56 million. In order to meet the 'self-sustaining sport fishery' objective (the most stringent), annualized expenditures for Hamilton Harbour and Metro Toronto are projected to be $9.49 million and between $125 and $142.35 million, respectively.

The PAC expenditure figures cited thus far are point estimates. That is, they are estimates of the level of expenditures required to meet a specific regulatory goal. As such, they do not provide information regarding the relationship between expenditures and differing degrees of pollution abatement. This type of information is required if a government proposes to loosen or tighten a regulation by a significant amount. In cases such as this, it is the marginal change in costs (as opposed to the average cost) that is relevant and the larger the policy change (e.g. a 50% decrease in allowable emissions), the greater the likelihood that the two will differ. This point is relevant for the policy of virtual elimination and the discussion is taken up in the conclusions to this chapter.

One of the few recent empirical studies to provide this type of information is Sanger, Heeney and Victor (1989). The authors summarize the results of a study which examines the costs for Ontario firms to reduce their emissions of photochemical oxidants. This class of chemicals includes ground-level ozone and nitrogen dioxide and is responsible for damages to vegetation, materials and human health. Engineering studies of 41 industrial establishments are used to estimate the capital and operating costs required to achieve differing degrees of abatement. Two important results emerge. First, under the assumption that the largest emission reductions are imposed on the firms with the lowest abatement costs, then the total abatement cost function is convex; that is, marginal abatement costs (MAC) are increasing in the level of abatement. For example, the MAC at a 4% reduction of nitrogen oxide emissions in Ontario is $63/tonne, while the MAC measured at a 22% reduction is $2,360 and the MAC at 50% reduction is $42,262. Based on this result, the second finding is that a policy which dictates uniform emission reductions for all industrial establishments will lead to industrial compliance costs that are millions of dollars greater than they would be if government followed a cost minimizing abatement strategy.

There are a number of features of the group of PAC cost studies that should be noted. First, the jointness of capital expenditures may lead PAC expenditures to be overestimated. This is because a new piece of capital may simultaneously provide several benefits to a firm. For example, the installation of high efficiency burners may lower fuel costs, improve product quality and reduce the production of waste residuals. In this case it is difficult to determine what portion of the burners' cost should be said to be PAC related. If the entire cost is identified as related to abatement, then this will overestimate PAC expenditures.

A second issue concerns whether the PAC expenditures are an accurate representation of the costs to society of pollution control legislation. There are two separate reasons to believe that they will not be accurate. In the short run, using PAC expenditures as a proxy for the cost of pollution legislation is likely to lead to an overestimate of the legislation's costs. This is because such a measure does not fully reflect the actions that economic agents can be expected to take to mitigate against these higher costs. Firms will try to shift the burden of their PAC expenditures forward onto consumers through higher output price increases and backwards onto input suppliers through lower input price. Consumers and input suppliers in turn will to act to reduce the impact of the firms' actions by altering their consumption and input supply decisions. For example, if a carbon tax were imposed on all fossil fuels, then some consumers of coal (which has a high carbon content and, thus, would bear relatively large taxes) would be predicted to switch to natural gas (which contains little carbon).

A long run perspective also suggests that PAC expenditures are an inaccurate guide to the full social costs of environmental legislation. In this case, however, it is argued that they are inaccurate because they underestimate those costs. Environmental regulations often induce firms to install capital equipment and to divert some of their research and development resources in order to comply with the provision of the regulations. Because these efforts produce nonmarketed outputs (improvements in environmental quality) it will appear (incorrectly) that productive resources are being diverted to unproductive uses. The diversion of capital and R and D resources is particularly significant because economists have identified the rate of growth in a nation's capital stock and stock of knowledge as key determinants of its economic rate of growth (Jorgenson, Slesnick and Wilcoxen, 1992). Any analysis that ignores this effect that firms' compliance efforts have on an economy's growth potential, will underestimate the long run costs of environmental regulations.

Empirical estimates from economic models which address the concerns associated with, first, modelling agents' reactions to regulations and, second, the long-run consequences of regulations are presented in section C.4.

C.2 Productivity Losses

Another way to measure the cost of environmental regulations relies on establishing their influence on a firm's or industry's productivity. As explained in the previous chapter, productivity can be measured using either single or multi-factor approaches. A firm's productivity is computed by taking the ratio of marketed output to the quantity of either a single input or an index of all inputs. To the extent that regulations lead firms to devote resources to producing nonmarketed outputs such as environmental quality, then the regulations will be seen to be effecting productivity adversely.

The empirical technique most commonly used to measure this relationship involves a two-stage procedure. The first stage establishes a measure of an industry's productivity performance using time-series data on output and input quantities and expenditures on inputs. The second stage employs a regression equation in which the dependent variable is the measured productivity time-series, and the independent variables include those which are believed to explain variations in the observed productivity levels. These variables can include the following: the capital stock growth rate, expenditures on R and D and the intensity of environmental (or other types) regulations (Christainsen and Haveman, 1981). Finally, the estimated coefficients from the regression are used to determine the share of the total observed change in productivity levels that can be associated with environmental regulations.

Christainsen and Tietenberg (1985) review a number of studies and find that the variable measuring the degree of environmental regulation usually has a negative coefficient. This indicates that increases in these regulations are correlated with decreases in measured productivity. Furthermore, the reviewed studies suggest that approximately

8-12% of the entire productivity decline observed in the U.S. since the mid 1970's may be ascribed to the impacts of environmental regulations.

The use of aggregate data in these productivity studies can mask significant variations across industries. If there is a negative correlation between regulation and productivity growth, then more heavily regulated sectors will show the largest impacts. This is indeed what is found by Barbera and McConnell (1990). The authors use estimated translog cost functions using industry-level data from the U.S. to derive estimates of the impact of environmental regulations on costs (and, therefore, productivity). The proportion of total productivity decline for which environmental regulations are responsible varies from 10% for Iron and Steel to 30% for Pulp and Paper.

Despite the enormous amount of research effort put into trying to understand the causes of the decline in North American productivity since 1975, Christainsen and Tietenberg (1985) point out that there are still a number of problems associated with these empirical techniques. Furthermore, these problems are particularly relevant for the question of whether environmental regulations lead to lower productivity levels.

Perhaps the most obvious criticism concerns the definition of output used in these studies. It can be argued that inputs employed for PAC related reasons do indeed have positive productivities (since they provide waste reduction as a measurable output) but that current accounting procedures fail to measure the outputs they produce. It comes as no surprise then, that conventional productivity measures are negatively correlated with PAC expenditures.

C.3 Macroeconomic Measures of Costs

The last two sets of measures of the costs of environmental regulations to be discussed in this chapter are broader in scope than the first two measures. They attempt to measure the costs of environmental regulations on the entire economy from two different modelling perspectives. The first is the macroeconomic approach. This approach employs economists' understanding of the structure of the macro-economy, and integrates financial and real sectors of the economy. The second is the computable general equilibrium approach and it is discussed in detail in the next section.

A number of studies assess the impact of environmental regulations on important macroeconomic variables such as GNP and unemployment. Jorgenson and Wilcoxen (1990) use a detailed model of the U.S. economy to estimate what GNP growth would have been like in the absence of environmental regulations. They find that for the period 1974-1985, the 'no regulation' economy would have grown at an annual rate of 0.191 percentage points faster than what was historically observed. By 1985, this would have implied a level of GNP 2.6% higher than what was observed.

Christensen and Tietenberg (1985) survey the studies which address the relationship between environmental regulations and the level of unemployment in the U.S. The results of the studies are mixed. Hollenbeck (1979) finds that these regulations raised unemployment by 0.21 percentage points during the late 1970's. Conversely, a Data Resources International study finds that regulations contribute to a lower level of unemployment. This finding, however, is a combination of the regulations' contributing to a lower level of GNP while promoting employment growth in a small number of service industries (Christensan and Tietenburg, 1985, p.384).

Christainsen and Tietenberg (1985) also report on the limited amount of research on the relationship between environmental regulations and industrial research and development. There is some evidence that these regulations lead firms to redirect their R and D efforts towards the production of 'non-market' goods (i.e., environmental protection). As a result, they also induce firms to reduce their efforts in creating new processes and technologies to improve their marketed output. By the same token, Christainsen and Tietenberg report on a minority of firms that indicate a degree of complementarity between environmental R and D and product-related R and D. In these cases, innovations in reducing effluent flows and the creation of waste materials also reduces the cost of producing the marketed output. Unfortunately, there is little empirical evidence that establishes the magnitude of these effects.

Sonnen (1989) is one of the few studies of the relationship between environmental regulation and the macroeconomic performance of the Canadian economy. Using a computer model of Canada's macro economy, Sonnen forecasts the impact of adopting several specific environmental regulations. These regulations would extend initiatives undertaken in Ontario such as the MISA program to the rest of the country. Three simulations are run and each is compared to a 'no new regulations' base-case. Simulation A assumes Canada alone adopts the new regulations and that there is no feedback from improved environmental quality to performance of the economy. Simulation B also assumes Canada alone adopts the regulations but that certain industries (fisheries, agriculture and forestry) enjoy productivity growth because of the improvements to the environment. Finally, simulation C assumes that Canada acts in concert with the rest of the world in enacting environmental regulations (but there are no feedbacks of the sort in simulation B).

The results of imposing the regulations are fairly uniform across all three scenarios. Real GNP initially rises due to the boost provided by increased government and private sector spending but eventually falls as induced price level increases outstrip growth in nominal GNP. Tax increases also contribute to the decline in output growth by curtailing consumer spending. By 2005, per capita real disposable income is $930 lower in simulation A, $710 lower in simulation B and $1,830 lower in simulation C (all results are relative to the 'no new regulations base-case'). In all three simulations, the level of unemployment and price level rise while the growth rate in the private capital stock falls. The result in simulation C is so poor because Canada not only experiences the costs of its own regulations but experiences reductions in export sales because the rest of the world also experiences reductions in economic activities from their environmental regulations. These results must be interpreted with caution, however, because the modelling approach emphasizes the costs of environmental regulations and has a limited ability to incorporate the benefits such as avoided health damage.

C.4 Aggregate Welfare Losses

The last measure of the costs of environmental regulations is the method most consistent with economic theory and also the most demanding in terms of data requirements. The econometrically-based, dynamic, computable general equilibrium model represents the state of the art in terms of modelling the long-run costs to society associated with regulations. Given its sophistication and the likelihood that it will be the dominant modelling method for the foreseeable future, it is worth spending some time to describe its structure.

First, it is necessary to distinguish between a partial equilibrium model and a general equilibrium (GE) model. The former models the workings of a single industry and ignores what effects changes in that industry might have on other industries or sectors of the economy. The latter attempts to model the simultaneous workings of several markets or an entire economy. In this case, modelling the linkages or interactions among the markets is a key element of the model. A problem with GE models is that they tend to have few definite comparative statics properties. That is, it is difficult to use these models to make predicts regarding the impact on endogenous variables of small changes in an exogenous variable. For example, if a tax were imposed on the use of labour in market X, a theoretical GE model cannot usually determine what this will imply for the incomes of suppliers of capital to market Y.

One solution to the problem of GE model's lack of comparative statics results is to impose enough structure on them to ensure they yield definitive results. This is called making them computable (CGE). This involves assigning specific values to some of the structural parameters such as price, income and scale elasticities. If the values of these parameters are estimated using market data and statistical techniques then the model becomes an econometrically based CGE model.

The final feature that is relevant is the dynamic nature of some CGE models. A static model considers the workings of a market or set of markets at a point in time. Furthermore, static models are not concerned with the way in which a new equilibrium is reached after some shock such as a change in regulations. In contrast, a dynamic model is concerned with the operations of a market or set of markets over a period of time. It is also very concerned with how markets adjust over time. An important feature of dynamic models is that the value of a variable may be a function of not only the concurrent values of other variables, but also on past values of itself and other variables. A good example is the nation's private capital stock. In any year, the size of the stock is a function of that year's investments (i.e., additions to the stock), and that year's depreciation (i.e., reductions in the stock), but also the size of the stock in the previous period.

The above example is valuable for understanding why dynamic CGE models are so valuable in modelling the costs associated with environmental regulations. These models are capable of measuring two types of costs, or more generally, welfare losses. First, they measure static losses; that is, they numerically simulate the impact of a new regulation on an economy's markets for a specific time period (a year, for example) and then are able to compute the loss in that year's aggregate consumer surplus. Second, they measure dynamic welfare losses. These models are able to establish the effects of the same regulation over several time periods and to compute the (discounted) sum of welfare losses over that time period.

It might be thought that the dynamic model could be replicated by 're-running' a static model for several time periods and then simply summing the results. To see why this is not the case, suppose an environmental regulation was thought to lead to increased PAC capital expenditures at the expense of other productive investments. The static model would include the foregone profits that would have been earned from the diverted investments as part of the opportunity costs of the regulation. A series of these static models would generate the same kind of estimates: each year, the cost would include these foregone profits.

A dynamic model, on the other hand, would recognize that a lower level of investment in one year not only lowers profits in the same year but also lowers profits in future years by slowing the rate of growth of the capital stock. In any one year, then, the dynamic model would identify two separate costs associated with reduced investment: the cost of lower current period profits and the cost of lower wealth (measured as the present value of the lost future profits).

Representative of this class of models is Hazilla and Kopp (1990). The authors demonstrate that a dynamic CGE model provides estimates of regulations' social costs which differ significantly from firms' PAC related expenditures. The difference arises because the former is able to model not only firms' shifting of PAC expenditures through price changes, but also their long run impacts on the capital stock. In addition, the model allows consumers to react to the adverse effects of regulations (lowered wages and higher retail prices) by substituting leisure for consumption.

The production side of Hazilla and Kopp's model is disaggregated to 36 sectors, each of which is represented by an estimated cost function. The consumption side of the economy is characterized by an aggregate household whose preferences are represented by an estimated indirect utility function. The social costs of environmental regulations are measured by the household's compensating variation.

The authors consider the impacts of two U.S. environmental laws: the Clean Water Act and the Clean Air Act. According to EPA estimates, the cumulative PAC expenditures arising from the regulations under these two Acts were $772 billion for the period 1981-1990. In contrast, the model's estimate of the full social cost of these regulations is $1,164 billion . Furthermore, the model indicates that the level of U.S. GNP and aggregate capital stock are both approximately 6% lower by 1990 than they would have been in the absence of the regulations. These results would suggest that reliance on firms' reported PAC expenditures may lead to an underestimation of the long run cost of environmental regulations by as much as 50%.

Dynamic CGE models have also recently been used to forecast the impacts of alternative policy instruments aimed at curbing the emission of greenhouse gases. Jorgenson, Slesnick and Wilcoxen (1992) forecast the impact of imposing a carbon tax on the U.S. economy. The authors assume that PAC capital does not produce any marketable output and that the revenues from the carbon tax are recirculated back to the private sector through reductions in personal and corporate income taxes. Imposing a carbon tax sufficient to maintain greenhouse gas emissions to 1990 levels reduces GNP and capital growth rates but lowers aggregate lifetime welfare by less than 1%.

Nordhaus (1993) also uses a econometrically based dynamic CGE model to assess carbon taxes. A distinctive feature of this model is that reductions in greenhouse gas emissions are assumed to benefit certain industries (e.g., agriculture). According to the Nordhaus model, a carbon tax of $70/ton whose revenues are recirculated through the existing tax system lowers greenhouse gas emissions by 30% (compared to a 'no tax' basecase). The tax also yields increases in net welfare (as measured by aggregate compensating variation) of greater than $238 billion for the U.S. economy.

The dynamic CGE models represent the 'state of the art' in measuring the social costs of environmental regulations. Their structure incorporates the interactions between markets and reflects consumers and firms' optimizing responses to regulatory changes. They are able to measure long run effects of regulations and to express these effects in fundamental terms: the extent to which households' welfare rises or falls as a result of regulations.

Nonetheless, the results of these models are still influenced by the assumptions adopted by their builders. For example, Hazilla and Kopp (1990) represent the production side of the economy with a series of aggregate cost functions estimated using time series data for the period 1958-1974. This implies that there is no allowance for cost-reducing technological change in response to environmental regulations. Furthermore, all of the CGE studies share the assumption that PAC expenditures are unproductive (in terms of marketed output) and most assume that the 'no regulation' scenario is characterized by higher levels of pollution but not by higher costs associated with worsened health or greater need for water treatment.

C.5 Remediation Of In Situ Toxins

The preceding analysis is concerned with the costs to society of reducing the release of man-made (change) contaminants associated with current economic activity. This analysis does not address the presence of pollutants currently suspended in soil, landfill sites or lake and river sediments. These sources also contribute to the current levels of air and water contamination but are present because of past economic activity.

From an economic perspective the most significant feature of this distinction between currently active and inactive sources is that the behavioral consequences of remedial efforts are much more important in the former. For example, a government regulation which compels current manufacturers of automobile batteries to reduce water use may have effects of the industry's current and future employment, research and development and investment. In contrast, government efforts to clean up abandoned waste sites will have relatively little effect on current economic activity [5].

The presence of contaminated soils and sediments is a serious problem for the RAP program on the Great Lakes. In Canada, 12 of 15 Areas of Concern (AOC) are known to have this problem while in the U.S., 28 of 30 AOC's exhibit contaminated lake or river sediments. In addition, in North America there are literally thousands of closed or abandoned landfill sites, waste treatment facilities, factory sites, military bases and gasoline stations. Many of these contain contaminated soil or sediments.

The USEPA estimates that the total annualized costs of compliance with its Superfund legislation is approximately $3.0 billion (Cropper and Oates, 1992)[6]. In addition, the USEPA estimates that the average cost to remediate each Superfund site is $38.8 million. Unfortunately, similar figures for Canadian sites are unavailable.

Research undertaken for individual RAP programs provides some information of remediation costs in the Great Lakes (Apogee Research et. al, 1990; Leger, 1992). For Canadian AOC's, annualized costs associated with sediment removal and treatment range from $230,000 for Hamilton Harbour (representing 1% of total annualized remediation costs) to $3.55 million for Thunder Bay (approximately 40% of total costs). In the case of Port Hope Harbour, annualized sediment removal and treatment costs are between $100,000 and $150,000 but this represents 100% of total costs. Unfortunately, of the 12 Canadian sites known to possess contaminated sediments, cost estimates are available for only five. The remainder are rivers where the technology needed to safely remove contaminated sediments has not yet been identified.

Data for U.S. AOC's are equally sparse. Most of the 28 sites possessing contaminated sediments have not been costed fully. Current remedial total cost estimates range from \$1.2 million to simply determine the extent of the problem in the Rouge River to \$51.6 million to dredge and dispose of 900,000 yd^3 from the Grand Calumet River to \$144 million to clean up the Ashtabula River and Harbour. An idea of the degree of uncertainty surrounding these cost estimates can be obtained by examining the estimated costs of remedial action for the Fox River\Green Bay AOC. The current estimated cost for sediment removal and disposal is between \$2.9 and \$705.6 million.

Estimating the cost of remediating contaminated sediments is a difficult task. Costs are generally a function of the quantity of material to be removed and treated but can vary significantly across sites. In addition, estimation of treatment costs is confounded by uncertainty regarding the necessary degree of treatment and whether there exists 'safe' threshold concentrations for pollutants in sediments.

D. ESTIMATES OF THE DISTRIBUTIONAL CONSEQUENCES OF ENVIRONMENTAL REGULATIONS

There are a number of ways in which an environmental regulation can induce changes to the distribution of households' income. For example, a firm which is required to install PAC capital may try to shift some of its costs onto workers by lowering wages. To the extent that the firm is successful in doing this, the workers' incomes will decline relative to others in society. Alternatively, the firm may try to shift the burden of its added costs onto consumers in the form of higher prices. If the firm is successful in doing this, then consumers of the firm's output will see their incomes fall relative to others in society.

However, whether any observed change in the distribution of income represents a cost or a benefit to society depends fundamentally on two factors.

The first factor is the empirical problem of establishing the estimated impact of the regulation on the distribution of income. This can be done by calculating the PAC expenditures by the industry and then using knowledge regarding the structure of input and output markets to predict how the incidence or burden of the PAC expenditures will be shifted through price changes. Once the final incidence of the PAC expenditures is determined, then the implied relative changes in the incomes of the industry's owners, input suppliers and customers can be estimated.

The second factor concerns the normative issue of society's attitude towards income inequality. Suppose there is a degree of consensus that government policies should contribute to a greater degree of income equality than currently exists. Then, if an environmental policy increased the degree of income inequality this would be considered a negative feature of the policy. This negative feature can be interpreted as a cost to society because it moves society farther away from its preferred distribution of income. Specifically, the extent to which society is willing to sacrifice consumption of some goods, in order to avoid this effect, can be taken as a measure of the social cost of the policy's impact on income redistribution.

Thus, in order to assess the distributional significance of any environmental regulation, two things must be done. First, the regulation's impact on the distribution must be documented. Second, how society evaluates the documented change must be determined.

Christiansen and Tietenberg (1985) and Cropper and Oates (1992) survey the empirical literature concerned with the distributional consequences of environmental regulations in the U.S. The costs of complying with both water and air pollution regulations are seen to be regressively distributed. That is, they impose a disproportionate burden on lower income groups. For example, Dorfman (1975) reports that regulations on stationary sources of air pollution lead to cost increases which are partially passed on to consumers in the form of higher prices. These price increases mimic the effect of a general sales tax with differentiated rates. Because the poor spend a larger share of there income on commodities, they are harmed relatively more than the rich by such a tax-like set of price increases.

While several studies of U.S. environmental regulations find that the cost of regulations are regressively distributed, this is not the entire story. What is really important is the distribution of net benefits and not just costs. Unfortunately, the variety of difficulties associated with benefit estimation make the calculation of their distribution quite difficult. Christiansen and Teitenberg (1985), however, report on research efforts that suggest that benefits are distributed uniformly or possibly progressively. For example, air quality regulations tend to have the largest impact in large urban centres where a disproportionate share of lower income people live. This finding would tend to offset the regressivity of the distribution of the costs of regulations.

Another <u>caveat</u> concerns the use of aggregate data to reach conclusions regarding the distribution of costs. These data tend to mask significant effects that can occur for specific regions and industries. For example, as described above, Hickling Management (1990) examines the PAC costs associated with Ontario's proposed revisions to its air quality regulations. The authors find that the impact on costs and output prices is on average relatively small, but this average conceals a significant range of impacts across industries.

The finding that the costs of environmental regulation tend to be regressively distributed represents a cost to society, only to the extent that society objects to the distribution of income becoming more unequal. This is a normative issue which is logically quite distinct from the empirical finding of regressivity.

Economists have extended cost-benefit techniques in order to reflect society's attitudes towards changes to the distribution of income (cf. Dupont and Phipps, 1991). In its simplest form, this extended cost benefit analysis allows differential weights to be assigned to welfare or income changes accruing to different income groups. Thus, if the analysis is meant to reflect a degree of social aversion to income inequality, then income or welfare changes for the poor are more heavily weighted.

An example of this extended analysis is found in Jorgenson, Slesnick and Wilcoxen (1992). This research employs a highly detailed, dynamic CGE model to forecast the aggregate welfare effects of imposing carbon taxes in the U.S. In calculating the aggregate welfare effects, the authors employ a technique which incorporates a parameter whose value reflects the degree of concern for income inequality in society by providing differential weightings to welfare changes at different income levels. The greater the value of the 'income inequality aversion' parameter, the greater social abhorrence of income inequality. Thus, including this parameter allows the authors to determine whether their findings regarding aggregate welfare outcomes are sensitive to the assumed ethical position taken by society with respect to the distribution of income.

Jorgenson, Slesnick and Wilcoxen find that the carbon tax has a small negative impact on aggregate welfare. They estimate that the decrease in average lifetime welfare ranges between -0.15% and -0.20% depending upon the degree of social aversion to income inequality that is assumed. The carbon tax is regressive in all cases and the larger welfare loss corresponds to the assumption that society is indifferent to income redistribution. These empirical results indicate that incorporating an inequality parameter can have large relative effects on the conclusions of the analysis. However, the fact that pollution control costs are still a small share of national income implies that incorporating such a parameter is unlikely to alter the conclusions of the analysis significantly.

Endnotes

1. There are several other types of benefits but these are either small in magnitude or have not been studied extensively. These include materials damage (surveyed by Adams and Crocker, 1991) and reduced aesthetic benefits (surveyed in Graves, 1991)

2. The dollar estimates of other studies are converted into Canadian dollars using a two step procedure. First, if necessary, the estimate is converted into Canadian dollars using the exchange rate prevailing at the time the estimate was generated. Second, the estimate's Canadian dollar equivalent is inflated to 1992 dollars using Statistics Canada's Consumer Price Index for non-durable goods

3. OECD (1992) reports on the 1991 decision by the U.S. government to set aside 11.6 million acres of timberland to preserve the habitat of an endangered species that was previously probably unknown to most households (the spotted owl). The government's decision implies that the value of the spotted owl's preservation exceeds the commercial value of timber which is roughly $204 million.

4. Huisingh et al (1986) presents several case studies where firms are reported to actually earn positive net benefits from PAC activities mandated by the U.S. government under Conservation and Recovery Act. Caution must be exercised with these claims as they would appear to violate economic intuition. Specifically, if the changes had been profitable, then it be presumed that firms didn't require the government to order them to undertake them.

5. An exception occurs when the government finances the cleanup through taxation on current consumption or production or if alters liability rules concerning responsibility for future clean ups. Both are relevant for Superfund in U.S. (cf. the discussion in Kopp and Smith, 1993)

6. This may sound like a large expenditure but according to the USEPA (as reported in Cropper and Oates, 1992), it represents only 1-2% of total annualized compliance costs for all U.S. environmental legislation.

6. CASE STUDIES

A. INTRODUCTION

The previous chapter provides empirical estimates of the costs and benefits associated changes in environmental quality. In some cases, these costs and benefits are calculated for a specific government initiative. In many instances, however, the estimates were not derived to assess a particular regulation or piece of legislation. Thus, the range of empirical results presented in chapter five provides a context in which the case studies presented in this chapter may be assessed.

The purpose of this chapter is to present several case studies of environmental decision-making. In each case, the physical properties of the toxic substance and the chronology leading to the government's decision, are presented as background data. Next, the method by which the government came to a decision is reported. Particular attention is made to any economic analysis that was done at the time of the decision. Finally, the economic features of the government's decision are critically assessed. Specifically, we try to determine whether the benefits of the decision exceeded the costs in order to ascertain if the decision represented an improvement in efficiency.

The substances that form the bases for the case studies are DDT (an example of a manufactured toxin), lead (an example of a heavy metal) and dioxin (an example of a toxic byproduct of manufacturing processes). The three substances share the following characteristics: they have presented or currently present a threat to the Great Lakes ecosystem; they are persistent, bioaccumulative and toxic; and they are all candidates for phase-out or virtual elimination.

The case studies are presented in approximate chronological order. Most of the analysis and decision-making regarding DDT occurred in the late 1960's and early 1970's. The concern over ambient lead levels has a history predating World War II but the major legislation initiatives did not appear until the 1980's. Dioxin has been the subject of government inquiry and regulation primarily in the 1980's and into the present. In keeping with the presentation of empirical estimates in the preceding chapters, all dollar figures have been converted to 1992 Canadian dollars. The method for doing this is set out in the introduction to chapter five.

Before presenting the details of these case studies, it may be worthwhile to anticipate the conclusions reached in each. With respect to DDT, there was clear evidence of toxicity in low doses to humans and other species. In addition, it was found that the costs associated with banning DDT were relatively small. Conversely, while there was little doubt about the presence and qualitative nature of the benefits of the decision to ban, there is not enough information to calculate the magnitude of the benefits with any degree of certainty. This lack of a complete economic analysis preceding the ban is unfortunate because it means that decision makers did not utilize all of the analytic tools available to them and because such an analysis would have, in all likelihood, supported a ban of DDT.

The case of lead differs somewhat from that of DDT. There was, on the one hand, a substantial body of scientific evidence demonstrating that exposure to lead could imply serious and adverse health effects in humans and other species. In addition, there was evidence that these health effects could be observed at exposure levels frequently observed in urban areas in North America. In contrast to the case of DDT, however, a careful cost-benefit analysis was conducted by the United States EPA regarding the proposal to eliminate lead additives from gasoline. This analysis showed that the estimated dollar values associated with avoided health damages as well as avoided materials damages (due to the use of leaded gasoline in cars designed for unleaded gas) exceeded the costs associated with re-tooling petroleum refineries in order to produce more unleaded gasoline. What made this analysis so convincing was the fact that it was quite conservative in its estimates of the benefits because it did not 'monetize' several important categories of health benefits due to lack of data.

The dioxin case study points out how large the costs and benefits associated with environmental regulations can be. The dioxin family of chemicals is difficult to regulate because the various forms of dioxin are introduced into the environment as unwanted by-products of certain activities such as incinerating garbage and bleaching pulp. This means that the changes to industrial processes needed to ensure that no dioxins are released (or no measurable amounts are emitted) may be very expensive. Nonetheless, the demonstrated toxicity of certain forms of dioxin means that it is quite reasonable to fear very serious health damages to humans and wildlife due to exposure to dioxin.

An important issue that is addressed in this case study, however, is the difference between total costs and benefits and marginal costs and benefits. Concern for the former is relevant when considering a proposed project or policy which is indivisible. For example, an analyst might be required to determined whether the total benefits outweigh the total costs of Canada adopting the North America Free Trade Agreement. In the case of a divisible policy proposal, then marginal analysis may be used. For example, the analyst may be required to determined if the marginal benefits exceed the marginal costs of reducing the allowable ambient concentration of a particular contaminant from 0.1 ppm to 0.01 ppm.

The analysis presented in the study of dioxin suggests that there are real and substantial benefits associated with virtually eliminating all sources of dioxins. These include (but are not limited to) avoided health damages and avoiding losses to both use and nonuse recreational benefits. When considering virtual elimination, however, the current state of knowledge regarding the health effects of dioxins implies that it remains unclear whether benefits exceed costs at the margin of virtual elimination.

B. MANUFACTURED TOXIC SUBSTANCE: DDT

B.1 Characteristics of DDT

DDT, the commonly used acronym for Dichloro-diphenyl-trichloroethane[1], was one of the most widely used chemicals for controlling agricultural pests and malaria-carrying insects. DDT is a white, tasteless solid with a faint, sweet odour. DDT is highly soluble in fat (including human body fat) but is insoluble in water. DDT is also present as a contaminant in other organochlorine insecticides. The following review of the toxicology of DDT is based upon USEPA (1975), World Health Organization (1979), U.S. Public Health Service (1989) and Geiser and Rossi (1993) as well as the individual references cited below.

DDT is a neurotoxin in mammals and insects. Effects of DDT poisoning in humans include nausea, tremors and convulsions, numbness of the face and extremities, disorientation and fatigue. DDT is suspected of long term mutagenicity and reproductive failure. Death due to respiratory or heart failure occurs within 1-3 days of high levels of direct (oral) exposure. Humans are exposed to DDT, and its metabolites (DDE,DDD), by consuming foods containing the compound. In the United States, in 1981, the average amount of daily DDT and DDE intake was 2.24 micrograms per day, primarily from vegetables. The level of DDT in the environment is steadily but slowly decreasing over time due to the legislative restrictions placed on its use and production. Once consumed, DDT stores most readily in body fat tissue where it may remain for years. DDT, DDE, and DDD have been found in human blood, placental tissue, and umbilical cord blood placing it under suspicion for reproductive toxicity in humans. DDT primarily leaves the body as DDA in urine. Breast milk is also another minor excretion route. In any case, stored DDT leaves the body slowly.

Human Health Effects

The health effects of DDT today, as when it was banned in 1972, are still relatively unclear. Low levels of orally or lung induced DDT (10-16 mg DDT/kg body weight) are known to have effects on the human central nervous system (CNS) where they are thought to be temporary, with no apparent long-term pathological effects. Studies indicate that the CNS is the primary target organ for DDT toxicity in humans. Respiratory effects in humans of low level exposure to DDT include moderate irritation and minor rashes (Neal et al. 1944). Tests on animals suggest that DDT may have negative effects on reproduction; also long-term exposure may affect the liver. Epidemiological studies of DDT-exposed workers indicate that exposure to insecticides (orally or inhaled) such as DDT induces hepatic enzyme activities. However, there is no conclusive evidence of irreversible human liver damage (Kolmodin et al. 1969 & Poland et al. 1970). Animals exposed to DDT experienced reproductive and developmental effects. Similar effects in humans have not been found, apart from an increase in chromosomal aberrations in men occupationally exposed to insecticides such as DDT (Rabello et al. 1975 & Yoder et al. 1973).

The USEPA has Minimal Risk Levels (MRL's) based on federal regulations as a basis for comparing levels found in air, water, and food. Persons receiving less than the MRL are not expected to experience harmful (non-cancer) effects. Furthermore,

> "The Occupational Safety and Health Adminstration (OSHA) states that workers may not be exposed to quantities of DDT greater than 1 milligram per cubic meter (mg/m^3)of air for an 8-hour day. EPA estimates that an ambient criteria level of 0.024 nanogram per litre (ng/L), consuming 2 litres of drinking water and eating 6.5 grams (g) of fish and shellfish per day would be associated with an increased lifetime cancer risk of one extra cancer for every one million persons exposed."[2]

The EPA classifies DDT, DDE, and DDD as probable human carcinogens. The MRL criterion does not provide any information about cancer risk. Unger et al. (1982) report a correlation between DDE levels in adipose tissue of deceased cancer and non-cancer patients, after accounting for weight, height, occupation, and location of residence factors. DDT administered to animals in combination with known carcinogens results in greater or lesser tumour occurrence depending on the combination. DDT may be a promoter of tumours rather than a direct cause. Similar effects on humans are unknown. In general though, studies of the carcinogenicity of DDT are limited and inconclusive. However, the probability of the human carcinogenicity of DDT is strongly suspected due to its effects in animals.

B.2 Regulatory Chronology

DDT was first synthesized in 1874 but its effectiveness as an insecticide was not discovered until 1939. From 1939 to 1945 production of DDT was limited to military use. DDT was used successfully in World War II for the protection of military personnel against malaria, typhus, and certain other vector-borne diseases. It was not until 1945 that DDT was released for commercial sale in the United States and its use became widespread in that country. This chronology of the introduction and subsequent banning of DDT in 1972 is drawn from Carson (1971), USEPA (1975) and Dunlap (1981).

DDT received uncritical acceptance from a society that believed technology and science could liberate mankind from every imaginable obstacle that nature could throw its way. It is this frame of mind that explains the passionate defence of DDT conducted by industry and agriculture and the initial apparent indifference of the public when serious questions as to DDT's toxicity were raised. The enthusiasm over DDT was such that total 1944 production of DDT in the US was 4,366 tonnes, increasing to 15,079 the following year.

The fifties saw the spraying of DDT on millions of acres of the United States. Against the gypsy moth alone the USDA sprayed over 4,000,000 acres between 1954 and 1958. However, serious questions were being asked about DDT as early as 1944. In 1948 the Committee on Medical Research recognized that DDT accumulates in animals, particularly those higher up the food chain such as humans. Specifically, the committee reported evidence on DDT's accumulation in breast milk, progressive damage caused by chronic doses, and stressed the still unknown consequences of long term exposure.

In 1950 and 1951 the first serious debate on the safety of DDT took place when the

House Select Committee to Investigate the Use of Chemicals in Food Products (the 'Delaney Committee') held hearings on the safety of food additives and other chemicals, and on the legislation regulating their use. New legislation on insecticides was recommended (the 'Miller Amendment' to the 1938 Food, Drug, and Cosmetics Act), resulting in a bill which placed on the applicant the burden of proof of safety. Unfortunately for opponents of DDT, the Delaney Committee remained in closed professional circles and the public, still confident in the unmixed blessings of science, took little interest in the affair.

However, by the late fifties and early sixties attitudes were slowly changing. In the late fifties, a programme of DDT spraying carried out by the Bureau of Entomology and Plant Quarantine to eliminate Dutch Elm disease led to high mortality rates among nesting birds. This was a clear and publicly visible demonstration of DDT's *direct* toxicity that elicited substantial public outcry (Wallace, Hickell and Bernard, 1961).

In 1962, Rachel Carson's Silent Spring (a metaphor for the often bizarre absence of songbirds in towns sprayed with DDT) contributed enormously to the establishment of the environmental movement in North America. It was a book that appealed to ordinary people, confirming their concern over the routine and widespread use of DDT. The Environmental Defense Fund (EDF) was organized in 1967 by academics and professionals concerned with the growing case against DDT. The EDF sought a complete ban on DDT use in the United States (it later sought a ban on production). The EDF's approach was legal action backed by scientific evidence before the 'court of public opinion'. New concrete evidence of the reproductive failure of birds and other creatures higher up the food chain was a key element in the EDF's case against DDT. In particular, research done on the Peregrine Falcon (or

Duck Hawk), in the United States and the United Kingdom, provided such evidence on DDT's persistence, toxicity, and effects on reproductivity. Furthermore, the development of the gas chromatography allowed for the analysis and detection of low levels of DDT. Prior to this development, DDT was being confused with PCBs, inhibiting attempts to verify the presence and/or level of DDT contamination.

In 1969, Congress passed the National Environmental Policy Act, declaring that the preservation of the environment was the prime directive for all government agencies when considering major development projects. Also in 1969, the National Cancer Institute contended that DDT was under serious suspicion as a carcinogen. The EDF and the USDA engaged in numerous legal skirmishes in the years 1968-72, primarily in the form of petitions by the EDF asking for the ban on DDT due to its carcinogenic properties. In 1971, the United States District Court for the District of Columbia ordered the EPA to cancel all remaining uses of DDT. It was up to the recently formed Environmental Protection Agency (EPA), which had the responsibility for pesticide regulation formerly held by the USDA, to make a final decision on DDT. In June 1972, after the Consolidated DDT Hearing of August 1971, DDT was officially banned in the United States. Exceptions were made for public health emergencies, green peppers, onions, and sweet potatoes in storage. Today, DDT can no longer be used as a pesticide in the United States though it is still produced in limited quantities for export. Presently, DDT is still used in third world countries for disease control and as a pesticide.

The government's decision, however, was not the only factor explaining the reduction in DDT use during this period. In fact, the decline in DDT use appears to be more closely tied with its declining effectiveness as an insecticide. Evidence presented at a 1970

EPA hearing indicated that cotton insect pests were becoming resistant to DDT. Expert witnesses indicated that by 1975 the bollworm would have established a very strong resistance to DDT. Conceivably, if the EPA had not banned DDT in 1972, factors such as the development of DDT-resistant pests, and effective chemical and non-chemical alternatives, would have contributed to the further decline in the of DDT throughout the seventies.

In 1974, due to the growing resistance among pests, the EPA published a summary of alternatives to DDT (USEPA, 1974). Around the time of the ban, DDT was being used primarily by cotton growers on a regularly scheduled basis, with little regard for whether an infestation actually existed. This indiscriminate use of insecticides was a major factor in the development of resistant strains of cotton pests. Also around the time of the DDT ban, the USDA, the EPA, the individual states, and industry were co-operating in an Integrated Pest Management (IPM) programme to promote improved use of insecticides and education in both chemical and non-chemical control of cotton pests. Large reductions in insecticide application were realized. For example, in the Texas High Plains Reproductive Diapause Boll Weevil Control programme, cotton growers experienced a $37.5 million decrease in production costs while 76,000 additional bales of cotton were grown. The IPM programme demonstrated two things. First, economic dependence of insecticides like DDT was not as heavy as previously thought. Second, DDT was a very minor component of total cotton growing costs, a component that could be even further reduced through astute crop management.

The history of pesticide regulation and DDT use in Canada is almost identical to that of the U.S., apart from the different dates of various regulations. The 1939 Pest Control

Products Act was revised in 1969 to allow the Canadian federal government to regulate modern synthetic organic pesticides. The regulations under this Act which restricted DDT use were in part responsible for the observed decline in DDT's commercial popularity. Usage of DDT peaked in 1960-62, during an outbreak of spruce budworm infestation. Following on scientific reports of DDT's toxicity in the U.S., DDT was banned in Canada in 1972.

In the same year as the ban on DDT, The Great Lakes Water Quality Agreement was signed between Canada and the United States. Under this agreement, the International Joint Commission set allowable limits for pesticide levels in the Great Lakes. In the intervening years, Canadian researchers have established the persistence and toxicity of DDT using Canadian cases and data. Environment Canada (1991) reports that DDT levels have gradually declined in the Great Lakes but are still present in measurable concentrations.

B.3 Assessment of Regulatory Decision Making

DDT was eventually banned due to the fact that it was a persistent, bioaccumulative toxin with diminishing usefulness in its principle application. At the time of the ban there was some information available of the health risks posed by DDT exposure and of the cost that a ban would impose on users of DDT. However, despite the availability of the necessary economic tools, no significant ex ante cost-benefit analysis was conducted on the effects of a ban. Most of the economic information available today concerns the economic effects on the U.S. cotton industry which was by far the largest user of DDT in North America.

Declining DDT use and the Cotton Economy

Production of DDT in the United States peaked in 1963 at approximately 188 million pounds. Domestic consumption however reached a peak four years earlier at approximately 79 million pounds and declined to 22 million by 1971. In fact, in the years prior to its cancellation, total DDT use in agriculture was rapidly declining most particularly in non-cotton applications. There appeared to be resistance to using alternative insecticides among cotton growers; for the years 1964, 1966, and 1971, 74%, 73%, and 94%, respectively, of total DDT use was for cotton protection. Historically, the cotton industry used more insecticides of all types per acre than any other domestic agricultural crop industry. Until the late sixties, roughly 66% of the cotton acreage was insecticide treated, with DDT being the chemical of choice. As of 1971, 50% of all agricultural insecticide use was by the cotton industry. By 1971-72 it was used on only about 17% of U.S. cotton farms, and equalled about 25% of total cotton insecticide use.

Around the time of the ban on DDT, the economic success of the typical cotton grower relied less on insecticides than it did on weather, cotton prices, competition from other crops, and government acreage allotment policies. Exogenous factors in the years immediately proceeding the ban, such as poor weather, competition from man-made fibres, and declining demand, resulted in reduced profits for growers and had as important an impact as the ban on DDT. Despite the DDT ban, the cotton industry was able to maintain its production levels to meet domestic and export markets. The record high prices for cotton in 1973 were not attributable to the cancellation of DDT, but rather to some of the factors mentioned above.

Costs and Benefits associated with the ban on DDT

At the time that a ban was being considered, the issue of increased production costs was raised frequently. For example, estimates presented at the 1970 EPA DDT ban hearing indicated that the banning of DDT would double annual cotton insecticide costs to $231.5 million; another estimate indicated that the replacement of DDT with organophosphates would increase cotton production costs by $63.7 million (USEPA, 1975). Estimates presented at the hearing would have translated into a 5% increase in production costs in 1970 terms.

Although growers did see increased insecticide costs of approximately $169 million (or a 32% increase) between 1972 and 1974 after the ban on DDT, this increase constituted a minor part in the total increase in costs. This is because the average cost share for pesticides and insecticides was only 4% in 1972 for American cotton farmers, although in some regions the cost share was as large as 14%. This increase was due primarily to a shift to more expensive insecticides.

The South Atlantic Region (SAR) and the East South Central Region (ESCR) of the U.S. were the hardest hit, as they were the main DDT using areas. In the SAR, annual total expenditures increased from $60.3 ($33.8 million on DDT) million in 1971-72, to $112.9 million in 1973-74. Total insecticide costs more than doubled in those periods from $65.87 to $137.50 per treated acre. Total annual expenditures on insecticides associated with the cancellation of DDT amounted to $25.5 million. A similar situation prevailed in the ESCR. Average annual expenditures for cotton insecticides increased from $128.7 million ($35.25 million on DDT) during the precancellation period to $221.7 million after. Insecticide costs per treated acre increased from $40.98 to $62.47. Total annual insecticide and application costs associated with the DDT cancellation in the ESCR was estimated to be $1.75 million.

Thus, for the two regions that were still using DDT in 1972, $29.23 million is estimated to be the increase in total costs following the DDT ban. It is unclear, however, if this cost can be entirely attributed to DDT cancellation. The figures presented do not take into account the declining trend in DDT use (ie., the unlikely assumption was made that DDT use remained unchanged 1971 through 1974). Nonetheless, the $29.23 million is a negligible cost to the average cotton consumer, working out to approximately 6.5 cents per capita per annum in 1975.

Unfortunately, the same degree of information regarding the benefits from banning DDT is not available. It is clear that the ban implied a rapid decline in the rate that DDT was being introduced into the environment. This in turn implied that DDT readings in wildlife and humans would begin to decline in this period (Environment Canada, 1991). Given the demonstrated toxicity of DDT, it certainly seems reasonable to conclude that the ban implied avoiding serious health effects for certain species of wildlife (peregrine falcons, for example) and ultimately for humans. There are, however, no epidemiological studies available that calculate the number and severity of avoided illnesses.

In addition to the avoided health damages, the ban on DDT yielded other benefits. At the time of the ban, the presence of DDT in the ecosystem was already having effects on the health of certain bird and fish populations in the Great Lakes region. Aside from the obvious fact that these impacts were unwanted because of the harm imposed on those species, these impacts indirectly effected the welfare of humans who derived pleasure from watching, hunting or catching these animals or who derived pleasure simply from knowing of their existence. It was demonstrated in chapter five that individuals' expressed willingness to pay for these use and nonuse categories of recreational benefits can often be quite substantial. Unfortunately, just as in the case of the health benefits, the necessary data are not available to calculate the magnitude of the dollar value of the recreational benefits arising from the ban on DDT.

It is evident from this discussion that economic analysis in the DDT decision was quite limited. Forecasts of potential cost increases to cotton growers were inaccurate, and did not take into account DDT's declining popularity. There was also little consideration of regulatory measures other than a ban. In addition, there was some effort to establish the avoided health damages to humans and wildlife, but these analyses were not used to place an economic value on this benefit. The omission of a carefully formulated socioeconomic assessment before the decision to ban DDT was a flaw in the decision-making process. The economic tools to conduct a cost-benefit analysis were available. In all likelihood, given the toxicity of DDT, its indiscriminate use by farmers and the growing problem of pest resistance, an objective CBA would have added considerable weight to the argument to have DDT banned.

It is important to point out here that our concern is with the process that led to the ban on DDT rather than the ban itself. In retrospect, it appears that the ban was an appropriate decision: this was a toxic substance who indiscriminate use was yielding declining benefits to farmers and was increasing likely to harm to wildlife and humans. In addition, it should be acknowledged that all regulatory decisions, and especially those concerning environmental matters, must be made with limited information and that in the case of DDT, regulators did attempt to determine the costs of a ban. Nonetheless, the lack of data regarding avoided damages meant that the very real benefits accruing from the ban could not be 'monetized' and compared to the costs.

C. HEAVY METALS: LEAD

C.1 Characteristics of Lead

The physical properties of lead, and what those properties imply for the toxicity of lead, are reviewed in the following references: Hare (1986), Lansdown and Yule (1986), World Health Organization (1989) and Environment Canada (1990). The findings of those reports are very briefly summarized here.

Lead is a soft, grey metal. Relative to other metals, it has a low melting point, high density, malleability and is resistant to corrosion. It is chemically inert, except when in organic forms. It is persistent when deposited in the environment but does not bioaccumulate in higher trophic levels.

Lead is a naturally occurring substance but most human exposure in the Great Lakes environment is to manufactured sources of lead. Lead is emitted in the following industrial processes: mining, smelting and refining of lead ore; secondary lead smelting (primarily of old automobile batteries), manufacturing processes which use lead as an input (in the production of automobile batteries, paint, building materials, solder, corrosion-resistant containers, gunshot, cosmetics and glass crystal). These sources emit lead and compounds containing lead into the air and water. Until the phasing out of leaded gasoline, two major sources of lead emissions were the production of lead-based additives to gasoline and the combustion of gasoline containing these additives.

In 1978, total lead emissions exceeded 14,000 tonnes in Canada (Environment Canada, 1990) and gasoline combustion was responsible for approximately two-thirds of this total. By the early 1990's total emissions have fallen to 4,000-5,000 tonnes and gasoline combustion has been almost eliminated as a source of lead emissions.

Humans ingest lead in the air they breathe, the food they eat, and the water they drink. The approximate proportions of these three sources for a representative resident of the Great Lakes region are 20%, 75% and 5% respectively (Hare, 1986). For children, the 'mouthing' of lead-contaminated soil and dirty toys and hands is an additional, and often important, source of exposure.

Lead is a toxic substance and has no known beneficial effects on human health. There is an extensive literature concerned with the toxicity of lead and the impacts on human health resulting from its ingestion and retention (Hare, 1986; World Health Organization, 1989). It is well understood that short-term exposure to very high concentrations of lead can cause poisoning, anaemia, kidney failure and even death.

What is more relevant for this case study is the set of health effects caused by or thought to be related to long-term exposure to low concentrations of lead, as are characteristic of levels found in the air, soil and water of industrialized countries. The health effects of long-term lead exposure are thought to include the following: tissue and organ damage, damage to the neurological and reproductive systems, impairment of cognitive development in children, and high blood pressure. Lead is also readily transferred from a mother's blood to her fetus, and there is concern regarding fetal exposure to lead.

The dose-response relationship for humans' exposure to lead is clearly positive. That is to say, exposure to higher concentrations, or longer exposure at a given concentration, imply greater adverse health effects. What is less clear is whether there is a 'safe' threshold of exposure below which there are no discernible adverse physiological or behavioral effects. The most common way of expressing the quantity of lead in the body is the concentration of lead in blood (measured in $\mu g/l$ or micrograms of lead per litre of blood). The last 40 years has seen a continual lowering of what is thought to be the no-effects blood lead level.

In the 1950's the level was thought to be approximately 50-60 $\mu g/l$. By the late 1970's, new evidence and improved testing methods lead scientists to lower the threshold to approximately 25-30 $\mu g/l$. By the late 1980's, further refinement caused the threshold to be revised downward to 10-15 $\mu g/l$. Today, there is a active debate as to whether there is any positive threshold below which no effects are observed. For example, Hare (1986) contends that the existing evidence supports a threshold at approximately 20 $\mu g/l$. However, Hare's report contains a statement by a group of dissenting scientists who argue that there is no empirical basis for a positive threshold.

C.2 Regulatory Chronology

Lead has been used by humans for over 4000 years and the dangers of human exposure to lead have been recognized since at least the eighteenth century (Lansdown and Yule, 1986). In 1914, for example, British scientists reported that lead is a neurotoxin (meaning that it could impair the functioning of the neurological system). Most of the pre-twentieth century exposure to lead was due to occupational exposure or through the use of pewter and other food storage implements which contained lead.

In 1926, lead was first added to gasoline as an inexpensive means of increasing its octane rating (Hare, 1986). By the post-war period, the widespread use of cars and trucks meant that gasoline combustion became the single most important source of lead in the environment in North America. In the early 1970's unleaded gasoline was introduced in North America. This type of gasoline was needed because lead seriously reduces the efficiency of catalytic converters, which were being introduced at this time.

In the mid 1970's both the Canadian and U.S. governments acted to define what lead content was allowed in 'unleaded' gasoline. In Canada, the limit was set at 0.77 g/l. These regulations stayed in place until approximately a decade later when both governments initiated 'phase-down' programs for lead in gasoline. In Canada, the allowable lead content was restricted to 0.45 g/l in 1985, then reduced to 0.29 g/l in 1987 and effectively banned in 1990 (Environment Canada, 1991). The chronology of the phase-down of lead in gasoline in the U.S. is roughly the same (cf., Schwartz et.al., 1984).

C.3 Assessment of Regulatory Decision-Making

The purpose of this sub-section is to review the economic analysis that supported the regulatory decisions regarding lead in the environment. Two specific sources of lead exposure are considered separately. These two sources are lead from gasoline and lead from drinking water. The empirical evidence presented for this case study is more extensive than that available for the other two cases. As a result, the evidence is summarized in Table 6.1.

Table 6.1 : Costs and Benefits of Banning Lead

A) Removal of Lead from Gasoline

Costs

Item	$ Value
Total annualized capital and operating costs of U.S. refineries (Schwartz et. al., 1984)	$894 million - $1,228 million
Increased PAC costs for Canadian refineries :	
capital expenditures	$721 million - $1.87 billion
operating expenditures	$396 million - $1.04 billion
Increased retail price of gasoline (Hare 1986)	$1.44 - $3.0 cents per litre

Benefits

Item	$ Value
(All Schwartz et. al, 1984) (All for U.S. only)	
Avoided maintenance costs and reduced fuel consumption	$1.2 billion - $1.3 billion
Total health related benefits associated with eliminating misfueling (not include estimate of reduction in chronic illness from reduction in ozone levels)	$718.3 million
Reduced damage to catalytic converter	$49.8 million

B) **Removal of Lead from Drinking Water**

Costs

Item	$ Value
USEPA (1986) Cost/Benefit of Removing Lead From Boston Drinking Water	
Least Cost Method of Removing Lead Annual Capital and Operating Expenditure Costs	$1.78 million - $2.87 million

Benefits

USEPA (1986) Cost/Benefit of Removing Lead From Boston Drinking Water	
Annualized Benefits	$9.9 million - $11.3 million
Levin (1986) Reduced Medical Care (Annual Savings)	$13.8 million
Savings from Less Cognitive Impairment a) Reduced Need for Remedial Action	$40.9 million
b) Lost Future Income due to lower IQ (Present Value)	$273 million over lifetime
Avoidance of conditions Associated with High Blood Pressure in Adults (white males only : 40 - 59 years)	$759.4 million annually

C.3.1 The Benefits and Costs of Removing Lead from Gasoline

The phase-out of the use of lead in gasoline is aimed at minimizing two major types of damage. The first is the damage to cars and light trucks due to the presence of lead in gasoline. The second, and the more significant, is the set of adverse effects on human health that occur from prolonged inhalation and ingestion of lead and other pollutants emitted during the combustion of gasoline containing lead. Estimates of the magnitude of these problems and the economic value which can be assigned to them is available only from the U.S. (Schwartz et. al., 1984). That information is reviewed here.

In order to define the benefits and costs arising from reducing lead in gasoline, Schwartz et. al., (1984) considers two regulatory scenarios. First, the U.S. government mandates a move from what was the current situation in 1982 (1.1 g/U.S. gallon) to 'low-lead' gasoline (defined as 1.0 g/U.S gallon) and, second, the government dictates a move from the current situation to 'no-lead' gasoline. The data regarding the benefits and costs of removing lead in the U.S. will be presented using these two scenarios.

The presence of lead in gasoline increases its octane rating but it also promotes engine wear and the fouling of certain components of the ignition system. Thus, reducing lead content is expected to imply less corrosion and better fuel efficiency. These effects can be translated into economic benefits by estimating the dollar value of avoided maintenance costs and reduced fuel consumption. Aggregate benefits from reduced wear and improved efficiency are $1,173 million under the 'low-lead' scenario and $1,342 million in the 'no-lead' scenario.

Another benefit stems from the reduction in 'misfuelling' that is expected to take place with the decrease of gasoline's lead content. The authors employ auto industry data to estimate that, by 1984, approximately 12% of all cars are wrongly using leaded gasoline in cars designed for unleaded gas. This is done to obtain higher octane levels. Unfortunately, this action seriously diminishes the efficiency of the car's catalytic converter and causes an increase in the amount of contaminants escaping from the car's exhaust system. The latter contributes to ground-level ozone and, thus, can have serious health effects.

If misfuelling is curtailed or eliminated then these human health damages will be avoided. The study employs estimated atmospheric dispersion models and dose response relationships to predict the impact on human health of a reduction in automobile emissions. It predicts that reduced emissions would lead to an average 1.5% decrease in nation-wide ozone readings. This in turn implies 227,000 fewer asthma attacks and between 6.4 and 11.0 million fewer MRAD's (minor restricted activity days). Point estimates of the willingness to pay to avoid these effects are taken from the existing benefit estimation literature and assigned to the avoided health effects. For example, the value of a MRAD is assumed to vary between $12.40/day and $35.50/day. The total health related benefit is valued at $718.3 million under both scenarios. The authors point out, however, that a reduction in chronic illnesses can be expected to result from the reduction in ozone levels but these are not 'monetized' because of data limitations. As a result, the estimate of $718.3 million underestimates the true value of health-related benefits from eliminating misfuelling.

Reducing the reason for misfuelling will also reduce the damage being done to cars' catalytic converter. The study uses engineering reports to estimate that the total value of avoided damages is $49.8 million.

The last category of benefits is the avoided human health damages that are directly related to inhalation and ingestion of lead. As indicated above, lead is a toxic substance. In extreme cases, short-term exposure can lead to serious harm and even death. In the case of long-term exposure to the lead found in the air, drinking water supplies, plants and soils, serious health effects are also suspected. Long-term exposure can lead to elevated lead blood levels which are, in turn, correlated with a variety of negative health effects including kidney disease, anaemia, neurological disorders and impaired cognitive performance. Young children tend to be more susceptible to these problems because they metabolize lead differently from adults and because they may ingest extra lead by eating dirt.

The study conducted by Schwartz et. al. (1984) restricts its attention to children with blood lead levels in excess of 30 μg/l (micro grams per litre) in order to estimate the health benefits of removing lead from gasoline. For this group of children, the authors concentrate on impaired cognitive development as the primary health effect. Thus, they do not evaluate non-neurological impacts for children, all health effects for adults and all health effects for children younger than 6 months (including fetuses). The authors indicate that these impacts are ignored not because of any complacency regarding the magnitude of their effects (or the economic values associated with these effects) but rather because of a lack of data. In any case, their omission clearly indicates that the overall health benefits from removing lead in gasoline are underestimated.

The study uses epidemiological data and estimated dose-response relationships to determine the number of children whose blood level readings would fall below the 30 μg/l threshold in each of the two scenarios. In the low-lead scenario, it is estimated that the number is 43,000 cases while in the no-lead scenario, it is 45,000 [3]. There are two economic values that can be assigned to each case of reduced blood lead. Point estimates of the economic magnitude of these two values are taken from the existing literature and used to calculate aggregate health benefits. The first is avoided medical care and the aggregate value of this saving is $72.9 million in the low-lead case and $76.5 million in the no-lead case. The second is the avoided remedial schooling that would be needed for children with impaired cognitive development. The value of this saving is estimated to be $327 million in the low-lead case and $343 million in the no-lead case. In contrast to the analysis of the benefits from removing lead from drinking water that is reported below, no effort is made to estimate the value of future income losses due to the lowered IQ levels related to lead exposure.

There are two conclusions which can be reached regarding the benefits of removing lead from gasoline. The magnitude of the benefits is quite remarkable. Second, the empirical estimates presented here indicate that marginal benefits decline with increasing degrees of pollution abatement. That is to say the lead reduction in the first scenario brings about larger benefits per unit of lead removed than does the second scenario.

In contrast to estimating the benefits of lead removal, it is fairly straightforward to determine the industry expenditures necessary for the removal of lead from gasoline. Petroleum refineries have a number of options for maintaining the octane levels of gasoline without using lead. In economic terms, the options require a variety of alterations by the refineries: input substitution, process change and capital investment.

Schwartz et al. (1984) considers the PAC costs for U.S. refineries under the two scenarios described above. The authors employ a linear programming model to determine the least cost means for each refinery to make up the octane loss from reducing lead use.

The authors do not attempt to determine how the increase in costs will affect retail prices because of a lack of data regarding price mark-ups and market structure at the retail level. The total annualized capital and operating costs of meeting the 'low-lead' alternative is \$894 million. The same cost for the 'no-lead' alternative is \$1,228 million.

This is an interesting finding as it implies that the cost of meeting the first 1.0 g/gallon reduction is approximately \$89.4 million per tenth of a gram of lead while the incremental cost of removing the last one-tenth of a gram of lead costs \$188 million. This is evidence for increasing marginal costs of pollution control: the marginal abatement cost for the last unit is 3 to 4 times the average over the preceding units.

Hare (1986) reports predicted PAC expenditures for Canadian refineries under a similar set of scenarios. The first scenario considers the impact on production costs and retail prices of a move from 0.45 g/litre to 0.29 g/litre and the second considers a move from 0.45 g/litre to 0.026 g/litre (or "virtual elimination" as Hare defines it). The cost estimates are based on data supplied by the petroleum refineries. The cost changes are substituted into a computer model of the retail gasoline market in order to predict their impact on retail prices.

Hare reports that the first scenario requires an increase in capital expenditures of \$721 million and an increase in annual operating costs of \$396 million. This is predicted to lead to an increase in retail price of approximately 1.44 cents/litre. The alternative scenario leads to an increase in capital investment of \$1.87 billion, an increase in operating costs of \$1,038 million and a predicted price increase of 3.0 cents/litre. These empirical results reinforce the earlier finding of increasing marginal abatement costs. Nonetheless, it should be recalled from the discussion in the previous chapter that PAC expenditures estimates may misrepresent the social costs of regulation. Using PAC expenditures neglects the possibility of cost reductions through induced research and development efforts and technological change. It particular, Hare's analysis recognizes the possibility that Canadian firms should be able to adopt some cost saving technologies from American refineries that had already adapted to the no-lead regulations. This possibility, however, is not incorporated into the numerical analysis of the costs of the regulation.

C.3.2. The Benefits of Removing Lead from Drinking Water

Lead can also be ingested if it is contained in drinking water supplies. For this reason, the USEPA proposed in 1985 to reduce the Maximum Concentration Level Goal for lead from 50 μg/l to 20 μg/l. The EPA believed that approximately 38-40 million people (or 15-20% of the U.S. population) were consuming drinking water with lead levels in excess of 20 μg/l. As part of its decision-making process, the EPA examined the benefits of the proposed regulatory change. The results of that analysis as reported in Levin (1986) and USEPA (1986), are summarized here. Benefits are reported as annual values and 1988 is chosen as the study year. Unfortunately, the EPA did not estimate the national costs of meeting the tighter drinking water standard. In an Appendix to its report, however, the costs and benefits of the standard are considered for Boston. The report argues that Boston is representative of large metropolitan areas with relatively old water supply systems. The results of the Boston case study are reported at the end of this section.

The benefits from reducing lead in drinking water supplies include reduced damage to human health and to water supply systems. In this study, health benefits include reduced exposure of children and also reduced incidence of high blood pressure among adults. The first of these two benefits is treated somewhat differently from the EPA study of removing lead from gasoline. There are again two sub-categories of children's health benefits: reduced medical care and reduced incidence of impaired cognitive development. The study raises the issue of the health effects arising from fetal exposure to lead but does not monetize this effect due to data limitations.

The first of these sub-categories is measured in the same manner as the lead in gasoline study. This analysis results in a point estimate of an annual saving of $1,711 for every child with blood lead levels exceeding 25 μg/l or an aggregate savings of $13.8 million.

The second sub-category of children's health benefits is treated more broadly than in the lead from gasoline study. Here, there are two alternative estimates of the value of the avoided impairment of cognitive development. The first follows the gasoline lead study and computes the needed remedial education required to overcome the cognitive deficiencies arising from lead exposure. This leads to a point estimate of annual savings of $5,043 per child with blood lead of 25 μg/l or total saving of $40.9 million.

The second method estimates the lost future income resulting from the lowered IQ's associated with chronic lead exposure. The basis for this approach is a three-part linkage established from the existing body of health and economic research. The parts of the linkage are the following: between lead exposure and IQ levels, between IQ levels and school achievement and between school achievement and lifetime earnings. For example, the study estimates that the present value of reducing a child's blood lead reading below 50 μg/l is $5,133. The incremental value for reductions to 30 μg/l is $4,683 and to 15 μg/l is $1,873. In passing, it is worthwhile pointing out that these findings are further evidence of diminishing marginal benefits. The total savings from avoiding these lifetime income losses is $273 million.[4]

The fact that the two methods used to estimate the benefits from avoiding cognitive impairment come to such differing conclusions is not surprising, as they measure different things. As discussed in the previous chapter, the first method is an example of the Cost of Illness approach to valuing health benefits. It is well understood that this method considers only out of pockets costs and, thus, underestimates the full valuing of avoiding serious illnesses.

The second category of health benefits relates to the possibility of avoiding certain conditions associated with high blood pressure (strokes, hypertension, heart attacks) that are thought to be linked to blood lead levels in adults. The study restricts its attention to white males, aged 40-59 in its analysis of these links. Using epidemiological data, it estimates that reducing lead levels in drinking water will lead annually to 118,400 fewer cases of hypertension, 370 fewer heart attacks and 75 fewer strokes. In order to evaluate these effects, the study employs a variety of methods including Cost of Illness estimates (avoided medical costs and incomes losses) for the nonfatal conditions and VSL estimates from the compensating wage literature for fatal conditions. For example, a stroke is valued at $86,000 and a non-fatal heart attack at $117,000. The total savings from avoiding these health effects is estimated to be $759.4 million annually.

The final benefit category concerns avoiding damage to the water supply networks. Drinking water that is soft and acidic corrodes water supply pipes, distribution mains and storage facilities and leaches metals from the surfaces of these structures. Water utilities treat water to reduce its corrosiveness in part to avoid having lead in their water. Thus, reducing lead is expected to reduce the need for anti-corrosion expenditures. Using engineering reports and a survey of U.S. water utilities, the study estimates the average per capita annual cost of anti-corrosion efforts to be $15.31. The study also estimates the number of people served by water that is corrosive to be approximately 50 million. Thus, the total annual benefit from reduced materials damage is $759.4 million.

An example of how this analysis can be applied is found in the EPA's case study of the costs and benefits of removing lead from the Boston water supply (USEPA, 1986, Appendix A). The analysis is conducted for a hypothetical move to a water quality standard of 0.01 μg/l of lead. The calculation of benefits is an application of the techniques discussed above and need not be repeated. Annualized benefits are in the range of $9.9 million to $11.3 million over the study period of 1988 to 1992.

The Boston case study estimates the costs of meeting the lowered lead standard by identifying the alternative means of reducing lead and the costs associated with each of these. The least cost method involves two stages. First, intake water is treated chemically in order to reduce its Ph level (ie., make it less acidic). This stage not only requires adding chemicals to the raw water supply but it also requires the installation of new pumping capacity to ensure even mixing of the less acidic water. The second stage also involves the use of additional chemicals but this time to control corrosion. Engineering cost estimates are available for these water treatment technologies. The annualized capital and operating expenditures required to meet the new lead standard range from $1.78 million to $2.87 million over the study period of 1988 to 1992.

The preceding analysis of the costs and benefits associated with removing these two important sources of lead exposure makes it clear that benefits exceed costs by a wide margin. Based on this evidence, the removal of lead from gasoline (and to a lesser degree from drinking water) constitutes one of the 'success stories' of environmental policy in North America in that policy makers utilized the evidence provided by the physical and economic sciences which demonstrated that society would be better off after a ban. As such, it is worthwhile considering the factors that contributed to this result. It may be that identifying these factors will help develop more 'success stories' in the future.

One significant factor is the degree of consensus amongst the scientific community regarding the health effects of lead (although, Hare's report and the dissenting letter contained in it are evidence of how incomplete this consensus is). What is particularly important is that there is evidence of health effects at concentration levels not far removed from those found in North American air and water. A related factor is the years of painstaking research that is referenced in the health literature concerned with lead. A third factor is that there are a relatively small number of sources and a small number of industries responsible for lead emissions. Finally, in the case of removing lead from gasoline there are alternative technologies which are well understood.

D. TOXIC BY-PRODUCTS: DIOXIN

D.1 Characteristics of Dioxin

Dioxin refers to a family of structurally related chlorinated aromatic compounds, known as chlorinated dibenzo-p-dioxins and chlorinated dibenzofurans (Environment Canada, 1991). They do not occur naturally, nor are they produced intentionally. Except for small amounts used for research, these compounds serve no useful purpose and are highly persistent, with a high potential for bioaccumulation and biomagnification. For example, the half life of 2,3,7,8-tetrachlorodibenzo-dioxin (TCDD) is estimated to be 5.8 years (Poiger and Schlatter, 1986), and between 5 and 8 years (Ryan and Norstrom, In Press). Dioxin is found in all components of the ecosystem, including air, water, soil, sediments, animals, and foods. All animals and humans in the Great Lakes basin are exposed to these substances.

There are 75 PCDD's (polychlorinated dibenzo-p-dioxins) known to be present in the environment. Except for a selected few, most forms of dioxin appear to possess little or no toxicity. The most toxic and hazardous dioxin, 2,3,7,8 -TCDD, receives more attention than any other form of dioxin. It is found in relatively large amounts in some widely used chemical products.

Dioxin came into existence as a by-product in the manufacturing processes of products such as biocides, paper, disinfectants, and preservatives, and as a result of incinerating garbage. Dioxin is usually created when a substance containing chlorine is heated, or when a substance is heated in the presence of chlorine (Muir et. al. 1993). Dioxin enters the environment as complex mixtures from major sources, such as commercial chemicals (pesticides and wood preservatives containing chlorophenols), incineration of chlorine-containing substances, and pulp and paper mills that use chlorine bleaching (Boddington et al. 1990).

A wide variety of commercial chemicals contain dioxin as by- products. None of these are currently manufactured in Canada. However, pentachlorophenol (which includes tetrachlorophenol) is still widely used in Canada to preserve and protect wood. This compound was one of the largest potential sources of dioxin to the environment before 1981. But since then, the amounts have been greatly reduced. In addition, dioxins are found in trace concentrations in a variety of paper products including disposable diapers, disposable surgical gowns, food containers and coffee filters.

Dioxins are present in soil, water, and air. Dioxins are found in low concentrations in the sediments of Lakes Erie, Ontario, Huron, and Michigan (Czuczwa and Hites, 1986), and the Niagara River (Hallet and Brokksbank, 1986). The air in urban areas is contaminated with dioxins caused by atmospheric emissions from incinerators or from nearby contaminated sites. There are also several 'hotspots' in North America with elevated loadings of dioxins. For example, soil contaminated areas are present in Missouri where 2,3,7,8-TCDD laden oil has been sprayed on the roads and horse arenas. Several military bases in the U.S. are also contaminated where Agent Orange (a herbicide containing dioxins that was used by the U.S. in the Vietnam War from 1962 to 1970) has been stored or heavily sprayed.

There is some evidence that the amount of dioxin being released into the environment reached a peak in the mid-1970's and has declined since. Sediment cores analyzed by Czuczwa and Hites (1986) shows such a trend. Concentrations of 2,3,7,8-TCDD in bottom sediments (tested in 1982) are .004 ppb in western Lake Ontario (Onuska et al, 1983). Other recent data for air, water, and soil report on dioxin and furan levels in Canada (Birmingham et al, 1989). Dioxin and furan levels in North American air ranged from 0.4 to 36.7 picograms per cubic metre. Soil levels varied from 50 picograms per gram to 14,100 picograms total dioxins and furans per gram of soil.

Norstrom reports the average ambient level of total PCDD's in the Great Lakes sediments to be 2 ppb (Canadian Council of Ministers of the Environment, 1987). Combustion and chlorophenol are given as the primary sources of the contamination. Environment Canada surveys find detectable levels of 2,3,7,8-TCDD in organisms. 2,3,7,8-TCDD ranges up to 37 nanograms per kilogram in mammals; up to 1996 nanograms per kilogram in birds; up to 474 nanograms per kilogram in reptiles; and up to 35 nanograms per kilogram in amphibians (Government of Canada, 1988-89).

Dioxin can remain in the environment for a long time. Many factors influence the persistence of dioxin in the environment including the type of dioxin, what it is mixed with, and what kind of environment it is exposed to. Depending upon these factors 2,3,7,8-TCDD can be broken down in a matter of days or degrade slowly, with an environmental half-life of one year. This compound degrades much more slowly when in land or lake sediment, with a half-life of approximately 10 years (Paddock, 1989). This problem of dioxin in the environment raises serious concerns of the health consequences for humans and wildlife.

Health Effects of Dioxin in Wildlife

More than 100 species of invertebrates, fish, reptiles, amphibians, birds, and mammals across Canada contain detectable levels of dioxin (Norstrom et al., 1982,1986; Stalling et al., 1983, 1985, 1986; Braune and Norstorm, 1989; Kubiak et al., 1989; Elliott et al., 1988, 1989). For example, fish in the Great Lakes contain mainly 2,3,7,8-TCDD at levels not exceeding 50 ppt (parts per trillion) (Canadian Council of Resource and Environment Ministers, 1987). Concentrations are higher near 'hot spots' with known point sources of dioxin emissions.

The toxicity of dioxin is highly variable; it has a wide range of effects, involving many different organs and body systems. The toxicity of these affected areas usually vary with the species of animal. The toxicity level of different forms of dioxin varies within a given species of animals, as well as between different species of animals. Birds and fish species seem to be more sensitive than most mammals to acute exposure to dioxin. All toxic forms of dioxin produce similar effects in a particular species of animal, and mammals exhibit several common effects after receiving a lethal dioxin dose. The health effects of dioxin on mammals are reviewed in Gupta et. al., 1973; Poland and Knutson, 1982; USEPA, 1985.

Dioxins contribute to organ damage, cancer, reproductive impairment and teragenic effects in animals (Paustenbach, 1986). In addition, 2,3,7,8-TCDD appears to inhibit the immune systems of mammals when administered less than lethal doses. The impact of dioxin on organisms in the environment is still not fully known. It is difficult to determine the full extent to which dioxin contamination affects the environment because of the parallel presence of a large number of other chlorinated organic compounds. In addition, the extrapolation of health effects found in laboratory settings to the concentration levels typically found in environmental contexts is controversial. For example, some studies find little or no health effects of dioxin on organisms exposed to 2,3,7,8-TCDD (Young et al., 1987).

Health Effects in Humans

Dioxin exposure is suspected of causing a variety of human health risks, fertility problems including birth defects, cancers of the soft tissue, and skin problems. A major problem for researchers is to extend their understanding of the effects of accidental exposure to high concentrations of dioxins to predict the health effects of exposure at levels commonly found in the Great Lakes area.

Generally, dioxin absorption by ingestion (through eating contaminated food) is greater than that of external application. Dioxins bioaccumulate through the food chain and are found in food consumed by humans. Recent studies suggest that the half of the dioxin present in humans results from eating beef. Dioxin exposure through fish and dairy products account for about 15% each. The remainder stems from dioxins in drinking water and ambient air. It should be noted that exposure levels and the relative significance of alternative sources of exposure are not uniform for all residents of the Great Lakes. Recreational fishers and native Canadians account for a small proportion of the total population but consume higher than average quantities of fish contaminated with dioxin. As a result, these people have elevated risks of adverse health effects stemming from high dioxin concentrations in their bodies (Flint and Vena, 1991).

Short-term Health Effects on Humans

The acute or short term effects are based on knowledge from people accidentally exposed to high concentrations of dioxins. Directly observed or reported effects include nausea and vomiting, irritation of the eyes, skin, and gastrointestinal tract, headaches, and a general ill feeling. Other clinical symptoms include pain in the limbs persisting for several months, irritability, nervousness, and enlarged and tender liver are reported. For reviews of acute health effects see USEPA (1985), Hay (1982), and Reggiani (1982).

Long-term Health Effects on Humans

Severe and persistent acne is the most common observed chronic or long term effect of exposure to high concentrations of dioxin. The skin disease chloracne is caused by the body trying to rid itself of the poison through the skin. Several years after exposure to dioxin reports of excessive hair growth and pigmentation, and elevated levels of blood constituents are documented associated side-effects.

Nerve disorders including problems or loss of some vision, hearing, taste, and smell. Also a loss of sexual drive and general weakness are among the other reported occurrences after some years. The reported chronic effects of exposure to dioxin are generally inconsistent, with the exception of chloracne, although most cases do indicate such symptoms like enlarged and/or liver damage and neuromuscular problems. For the reviews of chronic health effects associated with industrial exposure or short-term exposure to high concentrations, see USEPA (1985), Hay (1982), Reggiani (1982).

Some epidemiological studies find a correlation between cancer and occupational exposure to dioxin. A study in Sweden finds an abnormally high level of soft-tissue sarcoma in men exposed to chlorophenols and chlorophenol-based herbicides (reviewed in Hardell, 1983). Other researchers report finding such an association with nasal cancer (Hardell, 1983), stomach cancer (USEPA, 1985), and cancer of circulating cells (Hardell, 1983). The USEPA's Health Assessment for Dioxin, however, considers this available evidence on other cancers as inadequate (USEPA, 1985).

The evidence regarding the human carcinogenicity of dioxin at levels found in the environment is mixed and remains controversial. However, a significant amount of research on the assessment of risks from exposure of humans (as well as terrestrial and avian wildlife, and aquatic life) to dioxin is available. USEPA (1990) is an important example of the many studies on this subject. This paper reports the findings of several government agencies' assessment of these risks from dioxin exposure. Risks are evaluated using approximately 120 potential pathways to exposure to pulp and paper products, pulp and paper mill sludge, and pulp and paper mill effluent.

The report references the best estimates for 'no extra risk' thresholds developed by several agencies. Expressed as a rate of ingestion of dioxin per unit of body weight per day, these estimates are the following: 1.6×10^{-4} (pg/kg/day)$^{-1}$ (USEPA), 1.8×10^{-5} (pg/kg/day)$^{-1}$ (FDA), and a value of 1.8×10^{-5} (Consumer Product Safety Commission). Thus, it is assumed that if a person's daily rate of ingestion of dioxin through food, water and air is less than these amounts, then no long term adverse health effects will be observed.

An alternative way to express the risks associated with dioxin exposure is to estimate the rate of ingestion at which the risk of getting cancer is raised. The ingestion rates which are estimated to cause an increase in the lifetime likelihood of getting cancer by a factor of 1×10^{-6} are the following for 2,3,7,8 TCDD: 0.06 pg/kg/day (EPA), 0.06 pg/kg/day (FDA) and 0.015 pg/kg/day (CPSC). Thus, if the daily rate of dioxin ingestion exceeds these limits, the risk of getting cancer is increased by a small amount. If the rate of daily ingestion is less than these limits but exceeds the 'no extra risk' threshold, then the individual runs an elevated risk of contracting some chronic illness other than cancer. It should be noted, however, that while these exposure levels are indicative of the toxicity of dioxin, the belief that there is a 'safe' threshold of exposure is not supported by all scientists. In the recent past, improved understanding and enhanced measurement techniques have frequently lowered the estimated 'no effects' thresholds of many environmental contaminants.

D.2 Regulatory Chronology

The regulation of the class of dioxin compounds and, in particular, 2,3,7,8-TCDD, posses special difficulties for governments. Unlike DDT and lead, dioxin is not an intermediate input and occurs largely as an unwanted by-product of specific industrial processes. This means that governments have had to choose between controlling the use of compounds that are known or suspected of causing dioxin to be formed, prohibiting the processes known or suspected to cause dioxins to be formed or mandating that dioxin emissions must be below a prescribed level independent of the compounds or processes involved. Past regulatory efforts have been a combination of all three approaches.

An example of a ban on dioxin-producing compounds is found in governments' response to a group of pesticides called phenoxy herbicides (includes such commercial products as 2,4,5-T, 2,4,5-TP, 2,4-D, MCPA, mecroprop, and fenoprop). All the phenoxy's are derived from benzene (a known human carcinogen) and synthesized from chlorophenols. In the United States, the manufacture and use of 2,4,5-T and the related herbicide Silvex was restricted in 1970 and prohibited in 1984.

Phenoxy herbicides were also used in Ontario to control broad-leafed weeds, hardwood trees, and brush. They are also used to control undergrowth in coniferous forests, and are sprayed along highways, railways, pipelines, in parks and playgrounds and even on home lawns. The use of 2,4,5-T and 2,4,5-TP (Silvex) has been banned in Ontario as well as in British Columbia, New Brunswick, and Saskatchewan. This decision to ban 2,4,5-T closely followed the release of a report in 1978 by the National Research Council of Canada (1987). The government decision had been also influenced by the USEPA's decision to limit American usage. In January of 1981, Ottawa followed the American lead and banned a range of uses of pentachlorophenol in wood preservatives, weed killer, and disinfectants.

In contrast to the banning of herbicides and wood preservatives, Canadian regulations towards the pulp and paper industry have imposed limits to dioxin emissions without placing restrictions on the inputs or processes used by the industry (Environment Canada, 1991). Until 1992, restrictions existed on pulp and paper mill effluent. Unfortunately, these applied only to the mills built after 1971 (13 of 154 mills in Canada). In 1992, regulations promulgated under the Canadian Environmental Protection Act established a timetable for reducing dioxin levels to "below detectable levels". Unfortunately, since the establishment of the regulation, the timetable has been revised to allow more time for industry compliance.

In Ontario, the basis for current efforts to virtually eliminate the release of toxic contaminants into the water is the government's Municipal and Industrial Strategy for Abatement (MISA). This program sets contaminant limits for industrial effluent and the outfall of municipal sewage treatment plants using technology-based criteria. That is, for a wide variety of contaminants, the government regulations will determine the emission rate associated with the "Best Available Technology" that is Economically Achievable (BATEA) and specify that rate as the maximum that is allowed. Unfortunately, there is considerable uncertainty regarding what criteria are to be used to identify the BATEA (cf. the discussion in KPMG Peat Marwick Stevenson and Kellogg, 1990). Aside from this uncertainty, another criticism of MISA is its failure to incorporate benefit information into the determination of maximum allowable effluent standards.

One of the industries whose effluent are being monitored and regulated under MISA is pulp and paper. More recently, however, the Ontario government has gone further than the federal government by combining a "zero-discharge" policy on pulp mill's emissions of dioxin with a policy aimed at eliminating chlorine as an input to pulp and paper production. The pulp and paper industry will be required to eliminate the discharge of organochlorines in stages and create plans to eliminate chlorine. Zero discharge of organochlorines by 2002 is Ontario's goal (IJC, FOCUS, March/April 1993, 18(1)).

In addition to government efforts to control the release of dioxins into the environment, there are currently a variety of environmental standards aimed at controlling exposure to dioxins. Current Canadian objectives, guidelines, and regulations cover various media like fish, other food, drinking water, ambient water, and ambient air. National Health and Welfare sets a 20 ppt of 2,3,7,8-TCDD (edible portion) regulation in fish, while all other foods must not contain any detectable levels of dioxin.

The Interim Drinking Water objective of the Ontario Ministry of Environment is 15 parts-per quadrillion (ppq) of 2,3,7,8-TCDD toxic equivalents. The ambient water level or water quality objective given by the International Joint Commission is 10 ppq of 2,3,7,8-TCDD (sediment or tissue of aquatic organisms). The provisional air quality guidelines of the MORE are 30 picograms (total dioxin and 1/50 total furans per cubic metre as an annual average).

D.3 Assessment of Regulatory Decision-Making

Much of the effort by governments to cope with the problem of dioxin in the environment has gone into documenting its existence, establishing the pathways through which it enters the environment and estimating its toxicity. This concentration on the physical characteristics of dioxin is not surprising given the multiplicity of sources of dioxin, the complexity of its behaviour in the environment and the very small concentrations at which it can have adverse health effects.

To date, however, there has not been the same amount of attention paid to the economic aspects of the dioxin problem. As in the case of many environmental problems, the costs of reducing dioxin emissions are relatively straightforward to estimate. For example, the Ontario government has recently released a report that provides detailed analyses of the expenditures required by the pulp and paper industry to meet the government's goal of zero discharge of organo-chlorines. The conclusions of the report are reviewed below. On the other hand, estimating the benefits of virtually eliminating dioxin has proven to be much more difficult. The demonstrated toxicity of dioxin means that removing it from the Great Lakes ecosystem will unarguably yield benefits in the form of improved recreational opportunities, avoided health damages, avoided ecosystem damage and avoided future remediation costs. However, lack of data, incomplete understanding of dose-response relationships, shortcomings in economic valuation techniques and a lack of concerted effort by government have combined to inhibit the estimation of the size of these benefits.

In order to reduce or eliminate the discharge of organo-chlorine compounds (including dioxin) by pulp and paper mills, a variety of changes to manufacturing processes must take place. These include input substitution, process change, and investment in new capital.[5] The cost to industry of meeting more stringent dioxin regulations can be estimated because many of these activities are understood by engineers and economists and because there are market prices that can be used to evaluate changes in outputs and inputs.

In a report to the Ontario Ministry of the Environment, McCubbin Consultants (1992) employs detailed engineering models to estimate the costs of Ontario pulp and paper mills moving to production processes that reduce or eliminate AOX (Adsorbable Organic Halogens) emissions. AOX is a commonly used index of the quantity of chlorine-related compounds in effluent. In 1991, Ontario pulp and paper kraft mills recorded AOX emissions varying between 0.9 kg/tonne of output to 3.1 kg/tonne of output. However, this average masks the fact that Ontario mills have been fairly successful recently in reducing their dioxin emissions. For example, during the year-long monitoring of effluent undertaken under the MISA program only 2 of 11 mills reports detectable levels of 2,3,7,8 TCDD. The following discussion of pulp and paper mills' costs is based on the report by McCubbin Consultants.

The capital and operating costs necessary to reduce AOX emissions depend on the specific plant configuration and the degree of abatement desired. According to the engineering estimates, the total annual cost of the regulatory program is predicted to be between $1 billion and $2 billion, depending on the degree of AOX reduction that is mandated. It should be recalled from the discussion in chapter five, however, that these pollution control and abatement expenditures may not be accurate estimates of the true cost to Canadian society of the regulation. For example, if Ontario pulp and paper firms have some market power and are able to pass on some of these costs to foreign customers, then the social costs of the regulations will be lower than the engineering cost estimates.

For any individual plant, costs are reported for five production processes that achieve differing degrees of abatement beyond what is currently achieved. That data can be used to calculate approximate marginal abatement costs for different degrees of AOX reductions. The specific case of the Boise plant in Fort Frances is used as an illustrative example.

The first level of abatement reduces AOX emissions from 1.7 kg /tonne of output to 1.56 kg/tonne of output and corresponds to a "no detectable dioxins" objective. To achieve this reduction, Boise is forecasted to need a capital expenditure of $13.1 million and an annual operating and maintenance cost of $2.1 million. Thus the annualized cost of the first level of abatement is approximately $24.1 million per kg of AOX.[6] Repeating the same calculation for the next four levels of abatement (each one being more stringent than the previous one) yields the following marginal abatement cost (MAC) estimates. Level two has a MAC of $3.97 million/kg of AOX removed and lowers emissions to 0.49 kg /tonne (this eliminates emissions of molecular chlorine). Level three has a MAC of $17.2 million/kg of AOX removed and reduces emissions to 0.32 kg/tonne. Level four has a MAC of $33.5 million/kg of AOX removed and reduces emissions to 0.28 kg/tonne. Finally, level five has a MAC of $150 million/kg of AOX removed and reduces emissions to 0.23 kg/tonne of output (this eliminates emissions of molecular chlorine and includes oxygen delignification and extended cooling technologies).

The Boise example is characterized by increasing marginal costs of abatement. That is, the first level of abatement reduces AOX emissions at an approximate cost of $8.5 million per year per 0.05 kg/tonne. However, the annual cost of achieving the same reduction using the level five abatement technology (that is, once emissions have already been reduced to 0.28 kg/tonne) is $150 million. This is a seventeen-fold difference.

Despite the demonstrated toxicity of dioxin our understanding of the societal benefits arising from its elimination lags significantly behind the information available regarding the costs of elimination. This is an unsatisfactory position for at least two reasons. First, it means that a complete economic analysis of the proposal to eliminate dioxin from the Great Lakes basin cannot be conducted. Second, the absence of empirical estimates regarding the benefits of reducing dioxin levels may lead some people to conclude that they are insignificant or even nonexistent. The remainder of the discussion in this case study describes the categories of benefits that can be expected to accrue from a decision to virtually eliminate dioxins and to provide some estimates of the orders of magnitude for the economic value of some of these categories of benefits.

In order for the health benefit of reduced pollution to be measured, the link between pollution and health must be established. When a pollutant is released into the environment, it increases the risk of exposure to a contaminant, thus increasing the risk of adverse health effects. The reduction in the risk of these health effects is an economic benefit from reduced pollution. Once the environmental link between pollution and the health damage it causes is made, the data can be used for economic analysis. Some of the preliminary efforts to determine the economic value of the health damages associated with dioxin exposure are discussed here.

An example of how this epidemiological data could be used in an economic assessment of a policy aimed at dioxin emissions is suggested by Kask (Bingham, et. al., 1992). The author reports on an ongoing study of a large integrated pulp and paper mill in North Carolina which is responsible for significant dioxin emissions. Champion Paper Ltd. currently discharges approximately 43 million gallons of waste-water containing dioxin into the Pigeon River daily. This research reports on the first stages of an analysis of North Carolina's proposal to raise the maximum allowable dioxin limit currently set at 14 parts per trillion (ppt). In particular, Kask is interested in the viability of applying health benefit estimates derived by other researchers to the problem of contamination of the Pigeon River. While the study has not yet provided concrete estimates of the economic value of avoided health damages, it is instructive to review its methodology in order to see both the strengths and weaknesses of this approach.

In the Kask study, the principal benefit of a decision not to raise the allowable dioxin emission rate stems from the avoided increase in the likelihood of morbidity and mortality effects from long-term dose levels of dioxin exposure. Based on this definition, Kask intends to estimate the value of avoiding an increase in the probability of chronic morbidity or cancer mortality or both. The study is currently collecting the data necessary to determine these probabilities. Once this is done, the epidemiological data will be combined with economic data concerning the value of a statistical life and the value of avoided illnesses. In order to obtain estimates of these economic values, the study intends to transfer the benefit estimates derived in earlier studies (recall the discussion of the benefit transfer technique in chapter 5).

The work by Viscusi, Magat, and Huber (1991) and Smith and Desvousges (1987) is chosen because these authors provide economic values for a reduction in the risk of morbidity or mortality. These studies are attractive sources of estimates because they both value chronic or latent health effects, which are similar to the effects from dioxin exposure. A benefit transfer equation is to be used to convert WTP estimates from these two studies to values based on the data relevant to the Pigeon River area. A transfer equation relates

WTP estimates to the site specific variables such as exposure and conditional risk levels, and socioeconomic variables such as age, income, number of children in a household, and attitudes to hazardous wastes. Preliminary results from the application of this method of transferring valuation estimates implies that, for Pigeon River, the mean values for the WTP to avoid a 1×10^{-6} increase in death risk range from $31.90 to $85.51 using Smith and Desvousges (1987) estimates. The Viscusi, Magat, and Huber (1991) observation values provide an interval of $2.13 to $113.67 per 1×10^{-6} decrease in the probability of chronic bronchitis, with a mean of $12.55. Kask indicates that future work is needed to improve the quality of the risk data, to determine whether the WTP estimates 'transferred' from other studies are accurate representations of local individuals' preferences and derive final estimates of the benefits of not loosening the South Carolina water quality guidelines.

The Kask study is a valuable reference as it indicates the 'state of the art' of estimating the economic value of the human health benefits associated with environmental pollution. It is, however, only indirectly relevant to assessing current proposals to remediate the Great Lakes.

While there are no extant studies which calculate the health benefits from eliminating dioxin from the Great Lakes, some tentative calculations are possible. It is important to remember, however, that these are meant to be only suggestive rather than definitive.

The USEPA has conducted a risk assessment of exposure to a number of toxins found in the Great Lakes[7]. This assessment uses recent observations of toxin concentrations but restricts its attention to the question of quantifying the probable increased incidence of cancers amongst the exposed population living in the Great Lakes basin. The basic method is to determine the risk of cancer from each toxin by making the following calculation: risk of cancer = (cancer potency) x (chronic daily intake). The second component can be further broken down into the following formula: chronic daily intake = (environmental concentration) x (human consumption rate) x (bioconcentration factor). These risk of cancer estimates are obtained by applying data to these formulas which represent average values for people living in the Great Lakes. Once the risk of cancer estimate is derived, then it can be multiplied by the size of the exposed population to determine the approximate number of excess cancers which are predicted to occur annually due to the presence of the toxin. In the case of PCB's, the annual number of excess cancer deaths for the Great Lakes basin is between 300 and 400. In the case of dioxins (which are more toxic and have a higher bioconcentration factor than PCB's but are found in much smaller concentrations), the approximate number of predicted excess cancer deaths is 8-10 annually.

These estimates serve as a starting point for the calculation of the economic value of the health benefits from eliminating dioxin (or other toxins) from the Great Lakes. It will be recalled from the discussion in chapter five that the recent economic literature indicates that direct and indirect estimates of the value of a statistical life are in the range of $2 million to $5 million. This range of values would suggest that the annual benefit of eliminating dioxin from the Great Lakes is approximately $16 to $40 million if only the avoided fatal cancers are considered.

A consideration of the health benefits from eliminating dioxin that accounted only for the avoided fatal cancers would be seriously incomplete. As indicated above, there is some evidence that exposure to dioxin may also be related to disruption of the human immunological system, reproductive impairment as well as developmental abnormalities.

Unfortunately, neither physical scientists nor economists have advanced their state of understanding of the impacts of dioxin exposure to the point where the economic value of avoiding these health effects can be quantified.

Given the difficulty of computing health benefit estimates in this context, it may be valuable to recall the estimates presented in chapter five associated with the proposed changes to Ontario's air pollution guidelines (DPA Group, 1990). The analysis reviewed there indicated that reductions in the levels of air-borne contaminants would lead to avoided health damages which have an economic value between $1.0 billion and $4.0 billion. The lower estimate corresponds to less stringent regulations and also to lower estimates of the value of each avoided premature death or illness. In aggregate, then, the economic value of the health effects associated restoring with environmental quality in Ontario is quite significant.

In addition to the health benefits associated with eliminating dioxin, there are a number of other sources of benefits. These include both use and non-use recreational benefits, avoided <u>in situ</u> remediation costs and avoided water treatment costs for municipal utilities and private industries. While it is difficult to assess the magnitude of these benefits in the absence of sound data, the review of benefit estimates in chapter five suggests that they also might be significant.

As an example of the kind of analysis that would have to be done in order to obtain estimates of the non-health benefits from controlling dioxin emissions, consider the value to be assigned to avoiding species extinction. For example, the presence of dioxin in the Great Lakes ecosystem has been identified as a cause of observed eggshell thinning for certain species of predatory birds (hawks, peregrine falcons, and bald eagles). Suppose further it could be argued that, in the absence of virtually eliminating dioxin, these species face extinction in the foreseeable future. It will be recalled from chapter five that estimates of existence or non-use values for bald eagles had been reported to be on the order of $20-$25 per person per year. If we extrapolate to the adult population of the Great Lakes basin, then we have an estimated aggregate WTP (based on non-use values alone) to avoid extinction of this one species of approximately $50 million annually. For a complete accounting of the value of this species to humans, of course, we would have to add in the benefits derived from the bald eagles' contribution to value from other forms of recreation (bird watching, hiking and camping), as well as the value that humans assign to the bald eagles' role in the ecosystem of the Great Lakes (for example, in maintaining rodent populations).

It should be emphasized that these benefit calculations are very rough and are meant to be illustrative rather than definitive. Nonetheless, they are very conservative indications of the magnitude of the benefits that can be expected to be experienced from controlling exposure to dioxin in the Great Lakes. Researchers will be able to generate more accurate measures of these and other benefits once improved scientific and socioeconomic data are available.

Endnotes

1. Or 1,1,1-trichloro-2,2-bis-(p-chlorophenyl)ethane

2. U.S. Public Health Service, (1989), No.205-88-0608, p8

3. The choice of threshold is very important here. If a threshold of 20μg/l had been chosen, the numbers would have been 476,000 and 500,000 respectively. This would imply a tenfold increase in this category of benefits.

4. The reader should recall that the discussion regarding the evaluation of health benefits in chapter five indicated that losses in lifetime earnings are a very poor indicator of the full social cost of serious illness or premature death. As was also pointed out in chapter five, the lifetime earnings approach will, in most cases, seriously underestimate the social value of an avoided illness

5. A complicating factor regarding the control of dioxin emissions is the fact that firms do not use dioxin as an input. Rather, dioxin emissions usually occur due to the combination of chlorine and oxygen in the presence of heat. This observation, combined with the difficulty of modelling the creation of specific dioxin compounds, has led some groups to argue that dioxin emissions must be seen as the necessary result of using chlorine as an industrial input. The implication, these groups contend, is to regulate (or even ban) chlorine use.

6. This is calculated in three steps. First, we determine an annualized value for the capital expenditures by applying a 10 % interest rate to the total amount of capital expenditure of $13.1 million. This gives an annual capital expenditure of $1.31 million. Second, we add this annual value to the annual operating and maintenance cost of $2.1 million. Thus, the first level of abatement reduces AOX emissions by 0.14 kg, with an annual cost of $3.4 million. Third, we prorate the annualized cost to find the annualized costs for a 1.0 kg reduction in AOX emissions.

7. The analysis of the USEPA and the use of those calculations to calculate excess cancer rates is based on Gillett and Rish (1991) and Peck, Hulsey and Savagian (1993).

7. ASSESSMENT OF APPLYING CBA TO ENVIRONMENTAL ISSUES

A. INTRODUCTION

The purpose of this chapter is to provide an assessment of the strengths and weaknesses of applying economic valuation techniques to measure the costs and benefits associated with environmental regulations. A separate, but related, issue concerns how these estimates are to be used in the political decision-making process. These issues have been studied by many authors, both economists and non-economists, including Sagoff (1993) and Kneese and Schulze (1985).

Conceptually, this chapter presents a brief cost-benefit analysis on a specific application of cost-benefit analysis. In order to do this, two complementary perspectives are adopted. The first examines the environment-economics nexus from the 'outside' and addresses the broader question of the validity of the assumptions and techniques of welfare economics, which serves as the basis for cost-benefit analysis. The second examines the environment-economics nexus from the 'inside'. This approach accepts the assumptions of welfare economics and addresses the more narrow set of issues associated with identifying the most comprehensive theoretical models and most appropriate empirical methods to assess and evaluate changes in environmental quality.

B. THE OUTSIDE VIEW

The conduct of cost-benefit analysis represents an application of the theories of welfare economics. From the outside view, the fact that CBA is an application of these theoretical models is both its strength and its weakness. These two features are reviewed here briefly. A more detailed treatment of these issues is found in Sagoff (1991).

Welfare economics rests on a set of theoretical models of decision making by individuals and firms. These theories form the basis for all neoclassical economics and have been the subject of extensive research. Few other social sciences have developed a set of theoretical models that have the same degree of acceptance within the discipline and the same history of empirical testing. Thus, when an analyst conducts a CBA, he\she is doing applied economics; to the extent that economics is a coherent and comprehensive model of decision making and valuation, so too is CBA.

On the other hand, because CBA is an application of welfare economics, it takes on the assumptions and limitations contained in welfare economics. For example, CBA reflects the rather narrow utilitarian, human centred, ethical position inherent in welfare economics. From this perspective, something has value only if it contributes to the welfare of some individual or set of individuals. Thus, trees, bald eagles and rain forests do not inherently have value (or the right to exist) aside from that value derived from providing humans with satisfaction. In addition, actions such as polluting a river are evaluated only by referring to how they alter the welfare levels of individuals. Whether these actions are inherently morally right or wrong is irrelevant to the analysis. Finally, policies are evaluated solely by their consequences rather than by the process by which they achieve their results.

There are a number of grounds or bases for criticising this set of ethical assumptions or positions. First, the fact that CBA is human centred implies that humans have some superior right to existence that transcends the rights of other species. Many environmentalists argue that all living things share the same right to exist and to place the rights of one species above the rights of others helps to explain many environmental problems today. Roughly speaking, a more inclusive CBA would solicit the willingness to pay to avoid damage or to acquire certain benefits from all species and place them along side whatever values are espoused by humans.

A second basis for criticism stems from the consequentialist nature of CBA. To many people, the process 'counts'. That is to say, the means by which decisions are made and policies are implemented are important and perhaps as important as the outcomes of those decisions and policies.

A third issue concerns the amoral character of CBA. In our society, it is believed that certain actions such as rape and murder are inherently wrong. This position is partially based on the notion that people have certain inalienable rights such as the right to live. Furthermore, this position could imply that CBA is unnecessary in certain situations. For example, if the position is taken that it is morally wrong to pollute a water course, then a goal of zero discharge is the only ethically defensible policy and no CBA of alternatives is required (other than perhaps in order to determine the least cost means of achieving this goal).

Recent attempts to define what is meant by sustainable development may be interpreted as an effort to address a number of these concerns. One way to interpret sustainable development is to see it as economic development that is designed so that it is not in conflict with the preservation of environmental quality. Part of the effort to define sustainable development involves the extension of conventional economic theories and measurement techniques to incorporate a broader ethical perspective and a broader definition of what constitutes economic development.

C. THE INSIDE VIEW

The inside view critically assesses the application of CBA techniques to issues concerning environmental quality while accepting the set of behavioral assumptions contained in welfare economics. This view is the one taken by most academic environmental economists who work within the disciplinary framework of neoclassical economics to improve the accuracy of techniques to measure environmental costs and benefits (cf. Bockstael, McConnell and Strand, 1991; Braden and Kolstad, 1991; Cropper and Oates, 1992; Smith, 1993) .

In the last twenty years there have been a number of notable advances in the techniques used by applied economists to measure the costs and benefits associated with changes in environmental quality. The development of the travel cost and contingent valuation methods has allowed researchers to estimate recreational benefits systematically, and to correlate recreational benefits with environmental quality. The CVM is particularly noteworthy as it allows researchers to investigate the proportion of total recreational benefits accounted for by non-use motivations (Randall, 1991).

Another significant advance concerns the valuation of changes in the risk of premature death. It is clear that government policies to reduce human exposure to a number of environmental contaminants will mean fewer premature deaths. The economic significance of this benefit has long been a source of controversy. The human capital approach was the initial attempt by economists to evaluate this type of benefit and was conceptually flawed from the beginning. Fortunately, the work done on estimating the value of a statistical life altered the definition of the problem away from "valuing a human life" and towards valuing small changes in the probability of the risk of death or serious injury. This technique has provided economists with a technique which is consistent with microeconomic theory and provides credible estimates of individuals' willingness to pay to acquire small reductions in risk.

Another significant advance concerns the measurement of the costs associated with environmental regulations. The use of dynamic computable general equilibrium (DCGE) models has significantly extended the ability of economists to consider the long run consequences of regulations and simultaneously incorporate the complex interactions of an economy's different markets and industries. A particularly attractive feature of these models is that their measures of aggregate welfare change are consistent with microeconomic theory and are sensitive to distributional concerns. The analysis of Jorgenson, Slesnick and Wilcoxen (1992) is representative of the recent developments in this area.

Despite these advances there are a number of areas where economic theory and measurement techniques require further development. These problem areas have been highlighted throughout this report and three of the most pressing needs are mentioned here. The ability of economists to value avoided illnesses and diseases lags behind the ability to value avoided premature deaths. There are a number of conceptual problems including how to define a disease and how to obtain objective measures of disease severity which currently inhibit broader application of existing economic estimation techniques. In addition, there are the supplemental problems arising from a frequent lack of scientific data concerning dose-response relationships for many contaminants in the Great Lakes ecosystem.

A second area concerns the measurement of the costs of environmental regulations. In the research reported in chapter five, it is commonly assumed that capital diverted for pollution abatement purposes has no or no measurable economic productivity once installed. The problem is that the capital is producing a form of output (pollution abatement) but that output is not valued by conventional accounting or economic procedures. This specific problem is part of a larger problem associated with incorporating changes in the value or quality of environmental assets into economic measures of aggregate income and wealth. Fortunately, both the problem of accounting for environmental quality changes and the problem of valuing avoided illnesses are areas of significant research interest and activity.

The final area of concern has both theoretical and empirical dimensions and relates to the definition and measurement of nonuse benefits. From the available evidence from consumer surveys, it is fairly clear that nonuse benefits are a significant component of total benefits. In fact for some households, nonuse benefits are the entire basis for valuing the preservation of wilderness areas and individual species. It turns out, however, that defining nonuse benefits in a way that is both consistent with theoretical models of consumer decision making and amenable to empirical implementation is a significant challenge (cf. the discussion in Kopp and Smith , 1993; Smith, 1993).

D. IMPLICATIONS FOR THE USE OF CBA TO EVALUATE VIRTUAL ELIMINATION

The most attractive feature of CBA is that it provides a conceptual framework in which the economic impacts of a policy can be identified, quantified and denominated in a common unit of measurement. This is done in a way that is consistent with economic theory and that reflects individuals' valuations of costs and benefits. The principle limitations of CBA stem from its failure to include some categories of benefits, its difficulty in measuring other categories of benefits and its relatively narrow ethical basis. It should be asked whether these limitations imply that CBA consistently errs in its measurement of specific components of costs or benefits.

With respect to costs, the answer most likely is 'no' largely because most of these costs can be readily observed in market transactions and needn't be inferred indirectly. Nonetheless, the analysis of Pollution Abatement and Control expenditures in chapter five indicated that these often cited figures can be inaccurate indicators of the societal costs of environmental regulation. In particular, by emphasizing the opportunity cost of diverted investment funds, the economic analysis was able to demonstrate that PAC expenditures underestimate the long run costs of regulation.

With respect to benefits, the answer is a guarded 'yes'. CBA does not evaluate impacts of a policy unless it is demonstrable that humans place significance on these impacts. Thus, if a policy implied the loss of a species towards which humans were utterly indifferent, then the CBA of that policy would assign no value to the loss of that species. A possible exception would arise if the loss of that species implied changes in some ecosystem or in the population of another species which people did value. Unfortunately, the economist's ability to evaluate the loss of the first species is limited by what is frequently quite limited information regarding the relationships existing within complex ecosystems.

A second type of omission arises when the magnitude of a specific impact of a policy is unknown. For example, it was pointed out in the lead case study that in utero exposure to lead is feared by some health scientists to impair fetal neurological development. However, because of a dearth of data on this impact, the assessment of the benefits of reducing lead in gasoline did not include the economic value of this impact in its final estimate of total benefits. Avoiding these effects clearly has some value to society but because health experts were unable to estimate the number and severity of avoided adverse effects, no estimate of their economic value could be made.

As stated earlier in this report, the current inability of physical sciences and economics to combine and estimate the magnitude and value of certain aspects of a policy's impact on society should not be interpreted as implying that these aspects have no social value or significance.

Along these lines, it bears repeating that the purpose of conducting a CBA of an environmental policy is to provide an assessment of the economic aspects of the policy. A CBA seeks to determine (with some margin of error) whether aggregate welfare rises or falls as a result of a policy. A CBA, however, can not and is not intended to determine whether a policy is morally correct, fair, constitutional, environmentally benign or consistent with any other noneconomic criterion.

The final issue to be considered in this chapter is the appropriate role for CBA in assessing environmental policies in general, and the virtual elimination program in particular.

In order to frame an answer to this question it is necessary to distinguish between CBA's role as an information gathering tool and CBA's role as a decision-making rule. With respect to the former, it is clear that CBAs will be necessary so long as the economic aspects of any environmental policy are of interest to policy makers. Thus, if the economic dimension to virtual elimination is relevant to decision makers, then it should be characterized and measured as accurately and fully as possible.

With respect to the use of CBA as a decision making rule, even economists would be surprised if governments used only economic data in coming to decisions. As indicated above, there are a variety of relevant criteria by which a policy can be assessed, including its efficiency properties, its fairness, its consistency with other government objectives, etc. The difficulty arises in trying to determine how these alternative criteria should be balanced. One extreme position is to argue that each separate criterion represents a necessary but not a sufficient condition for acceptance of a policy. A less extreme position would allow some amount of trading off between criteria. For example, if a policy marginally failed a CBA test, then this might be offset by its successful performance according to some other criterion (e.g. participation of local community groups in decision making or consistency with international treaty obligations).

8. CONCLUSIONS

1. REPORT SUMMARY

This report has provided the reader with a critical review of how economic theory and measurement techniques are used to evaluate changes in environmental quality. It is hoped that the reader now has a better understanding of the strengths and weaknesses of economic analysis, and of the contribution that economics can make to assessing environmental policies.

This report has eight chapters and these chapters have addressed three separate, interrelated topics. Chapters three and four were concerned with developing the theoretical economic models which support the measurement of costs and benefits and with applying these theoretical models to the specific problem of evaluating changes to environmental quality. The important starting points for these theoretical models are individuals' preferences and firms' technologies and the behavioral assumption that economic agents act to maximize their private welfare. From these fundamental concepts, the notions of consumer and producer surplus were defined and exploited to measure costs and benefits. It bears repeating that the general conclusion from the theoretical models is that a benefit is represented by an individual's maximum willingness to pay to acquire the good or service and a cost is represented by the minimum amount of compensation required to have an individual be willing to accept some burden. In this way, the measurement of costs and benefits is based firmly on individuals' preferences and values.

Chapters five and six comprised the second part of the report. These chapters presented the results of past efforts to assess the magnitudes of the costs and benefits associated with changes in environmental quality. In chapter five, the estimates of these costs and benefits were reviewed and the empirical techniques used to generate them were critically assessed. It was demonstrated that both the benefits and costs arising from government environmental policies have the potential for being quite large. With respect to benefits, human health benefits and recreational non-use benefits were seen to be both empirically important and technically difficult to measure. Fortunately, benefit estimation is an active field of research for environmental economists. One particularly exciting area of research concerns the problem of 'benefit transfer'. This refers to the technique of generating benefit estimates in a specific context or ecosystem and then transferring these estimates (possibly after some adjustment for differences in circumstances) to another context or ecosystem.

With respect to costs, it was reported that the measurement difficulties are not nearly as daunting. This is due to the fact that the resources used by firms to comply with environmental regulations usually have market prices which accurately reflect their opportunity cost. This stands in contrast to the benefits of environmental regulations because avoided health damages and avoided ecosystem damages are not commodities for which there are well-defined markets or market prices.

The empirical estimates of the costs of compliance with environmental regulations indicated that these regulations have not thus far imposed a significant burden on most North American industries. This conclusion must be balanced against the finding that in some cases individual industries have borne significant abatement costs (the petrochemical industry in Ontario is an example). In addition, it was noted that environmental regulations have the potential to redirect private sector investment away from contributing towards increases in marketed outputs and towards the production of non-marketed outputs such as improvements in environmental quality. Due to the restrictive way in which economic development or progress is currently measured in Canada, this diversion of capital from producing marketed outputs to producing non-marketed outputs will appear to correspond to a decrease in the rate of economic growth.

In chapter six, government efforts to regulate the emission of, and human exposure to, DDT, lead and dioxin were reviewed. In addition, the economic analyses supporting these government actions were critically assessed. The case studies demonstrated that the amount and quality of economic analysis going into policy making has been quite uneven over the last twenty years. In the cases of DDT and dioxin, the costs of a ban were considered in depth but relatively little was (or is) known regarding the magnitude of benefits. In contrast, the decision to ban lead as a fuel additive was supported by a detailed understanding of both benefits and costs.

Another observation that comes from looking at the three case studies is that the degree and sophistication of economic analysis concerning government policy has increased significantly in the last twenty years. In part, this is due to theoretical and empirical advances made by economists. In addition, governments have grown more willing to use economics to assess both the costs and the benefits of environmental policies. In the case of lead, it was this kind of analysis that demonstrated clearly that even conservative estimates of the benefits from stringent regulations exceeded the costs of those regulations.

Finally, chapters seven and eight provided assessments of two different sorts. In chapter seven, the application of economic theories and measurement techniques to environmental questions was considered. It was seen that while the cost-benefit framework provides some significant insights into policies which are not supplied by other types of analysis, there are also limitations to this kind of approach. These limiting factors are both conceptual (e.g. the difficulty of incorporating ecosystem benefits) and empirical (e.g. the difficulty of using hypothetical questions in surveys to elicit willingness to pay estimates). In the portion of chapter eight that follows, the policy of virtual elimination is critically assessed from an economic perspective.

B. IMPLICATIONS FOR VIRTUAL ELIMINATION AND THE SUNSETTING OF TOXIC CHEMICALS

The decision whether to proceed with virtually eliminating toxic chemicals from the Great Lakes will have serious environmental and economic consequences. This report has been concerned with the economic dimension of that decision. It bears repeating that there are, of course, other dimensions to the issue of virtual elimination and also other ways to assess the social desirability of pursuing this policy. Nonetheless, the purpose of economic analysis is to attempt to measure the costs and benefits arising from such a policy and to infer whether it will increase social welfare (as defined from an economic perspective). Thus, economic analysis assesses the efficiency of a proposed policy and is not suited to determine whether it should be supported on ethical, ecological, political or other grounds.

If it is assumed that the efficiency properties of the proposal to virtually eliminate persistent toxic chemicals is relevant to the decision whether to proceed, then the costs and benefits of the policy must be identified and measured. These costs and benefits have the potential to be very large. On the cost side, the report by VHB Research and Consulting (1991) on the abatement and control expenditures required under the MISA program was summarized in chapter five. That report predicted that the annualized capital and operating costs for Ontario industries to meet the water quality regulations under MISA will be in excess of $1 billion and may reach $2 billion.

There is less information regarding the scale of benefits from environmental remediation in the Great Lakes but the available evidence suggests that they too can be expected to be very large. Annual recreational benefits in Ontario stemming from water quality improvements under the Great Lakes RAP plan are estimated to exceed $270 million. In addition, the annual health benefits for Ontario residents stemming from the proposed revisions to Ontario's air quality regulations are estimated to range from $1 to $4 billion. Finally, in the dioxin case study, USEPA risk assessment calculations were reported to show that eliminating PCB's and dioxin from the Great Lakes could result in 300 to 400 avoided premature fatal cancers annually for the population residing in the Great Lakes basin. If it is assumed that a reasonable estimate for the value of a statistical life is $2.5 million, then the annual benefits from avoiding only this one adverse heath effect would amount to $700 million to $1 billion.

Despite the magnitude of these benefit estimates, it is important to remember that several categories of benefits are poorly understood and have not yet been fully measured. As detailed elsewhere in this report, benefit categories that are currently poorly measured include non-use recreation activities and changes in the incidence of chronic illnesses. The imbalance between the relative ease of measuring the costs of abatement and the difficulty of measuring benefits is particularly worrisome because it leaves the possibility that the absence of empirical estimates for certain benefit categories will be mistaken as an indication that they are empirically unimportant.

A separate concern stems from the use of estimated total benefits and costs as a means of assessing the efficiency of a plan to virtually eliminate PTC's from the Great Lakes. It is correct to say that if it can be demonstrated that total estimated benefits exceed total estimated costs, then a policy to virtually eliminate PTC's can be expected to increase efficiency. However, relying on estimates of only total costs and benefits means that an important set of efficiency-related questions could go unasked. Specifically, an analysis which concentrates only on total costs and benefits and asks only whether virtual elimination should proceed will neglect the behaviour of marginal costs and marginal benefits at different levels of abatement and avoid asking what is the efficient degree of abatement.

In the case where the degree of desired abatement is to be considered, information regarding marginal costs and benefits must be used. This is because economic efficiency requires that the degree of abatement be chosen where marginal benefits equal marginal costs. In many cases, this rule implies that the efficient level of abatement is less than one hundred per cent, for reasons that are explained below. Thus, if a policy such as virtual elimination is to be supported on economic grounds, it must be demonstrated that expected marginal benefits exceed expected marginal costs at 100% abatement (or some level very close to 100%).

What is known about the marginal costs and marginal benefits of abatement at differing levels of abatement? Economic theory and the empirical evidence presented in chapters five and six suggest that marginal costs of abatement increase and marginal benefits decrease with greater degrees of abatement.

In order to see the theoretical reasoning behind this contention, contrast what can be expected to be the costs and benefits of the first 1% of pollution reduction (relative to the current level of pollution emissions) and the last 1% of abatement. The cost of the first 1% reduction is usually quite small for most firms as they can achieve this goal through improved materials handling, more frequent maintenance of machines and equipment and similar adjustments. In contrast, the benefit from the first 1% reduction is probably quite large if the people in the exposed population differ in their relative risk from exposure to the contaminant. Individuals who are particularly susceptible to injury from exposure to the pollutant will enjoy relatively large reductions in risk from the initial reduction in emissions. As a result of these factors, the marginal benefits from the first 1% of abatement frequently exceed the marginal costs.

Now, consider the situation for the costs and benefits of moving from 99% abatement to 100% abatement (or to a level which cannot be detected with current measurement technology). This increment to pollution abatement will likely be more expensive than previously reductions because the firm can be expected to employ means to reduce emissions in increasing order of their costliness. Profit maximizing behaviour dictates that the firm will first use the least costly method of reducing emissions and will only employ more expensive methods when compelled to do so. McCubbin (1992), for example, provides examples of these kinds of adjustments and their costliness for the Ontario pulp and paper industry.

In contrast to rising marginal costs of abatement, the benefits from the last 1% reduction are likely to be small relative to those enjoyed for the first 1% reduction. The members of the exposed population who are susceptible to injury have already been protected from injury and the last 1% reduction will only alter the probability of being harmed for those who are the least susceptible to harm. Thus, whether the benefits exceed the costs for the last 1% of abatement is much more difficult to determine.

This report has presented some empirical evidence that supports the above argument. In the case of the Ontario government's regulations directed towards restricting the emission of organochlorines, marginal abatement costs for a representative firm differed by a factor of 17 between the first and last units of pollution abatement (McCubbin, 1992). In the case of efforts reported in chapter six to restrict lead levels in drinking water supplies, it was argued that the marginal benefit from successive reductions in lead levels decreased. For example, Schwartz et. al. (1984) estimated that the marginal benefit of reducing childrens' blood lead concentration to 50 μg/l was approximately $5,100 while the marginal benefit of reducing the concentration to 30 μg/l was $4,600 and to 15 μg/l was $1,900.

In should be pointed ut that the use of marginal cost and benefit estimates to identify the efficient level of abatement is most easily applied to 'flow' pollutants like ground-level ozone, which have the characteristic that damages in any period depend only on that period's emissions. Persistent toxic chemicals, on the other hand, are considered to be 'stock' pollutants because damage in any period is a function of not just that period's emissions but also on the existing stock of contaminants in the environment. For persistent and bioaccumulative pollutants such as mercury or dioxin, the size of the existing stock of contaminants is a function of past depositions net of any reductions due to removal from the environment or through chemical reactions with other elements.

The significance of this distinction between flow and stock pollutants is best illustrated by referring to the health-related benefits accruing from a reduction in the emission rate of a contaminant. In the case of a flow pollutant, the benefit is the reduced risk of health damage in that period. In the case of a stock pollutant there are two sources of benefits. The first is the reduction in risk of adverse effects on health during the current period. The second is the reduction in risk of adverse health effects in future periods due to the reduction in the size of the future stock of contaminants. Thus, if the measurement of benefits for removing a stock pollutant is to be complete, it must include not only the value of the current period's avoided health damages but also the present value of the discounted future stream of avoided health damages. For a long-lived substance such as lead or DDT or a toxin whose health effects remain unobserved for long periods of time, this need to include both sets of calculations can be very important from an empirical perspective.

From a theoretical perspective, however, the fundamental rule that the efficient degree of abatement be chosen where marginal benefit equals marginal cost is unchanged. The insight gained by differentiating between conventional flow pollutants and long-lasting stock pollutants is that in the case of the latter, however, marginal benefits are far more complex and require greater care in their definition and measurement. This situation presents an added degree of difficulty for regulators grappling with the question of the desirable strategy for restoring the health of the Great Lakes' ecosystem.

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