Chapter 3

Distributional Implications of Environmental Protection Programs: Methodological Issues

In this chapter we take a comprehensive look at the methodological issues related to the measurement of the incidence of the costs and benefits of environmental programs. While Section 3.1 discusses the issues associated with the incidence of benefits from environmental programs, Sections 3.2 and 3.3 cover different aspects the pollution abatement efforts made by the government and industries respectively and the incidence of the costs incurred by them. Section 3.4 explores the possibility of a trade-off between environmental protection and employment.

Section 3.1

Incidence of Benefits from Environmental Programs

Environmental regulations usually impose significant direct as well as indirect costs on the industries and households. It is, therefore, important to estimate the benefits associated with them. In this Section we take at the various issues involved in assessing the incidence of benefits from environmental programs.

3.1.1 Measurement of the Benefits from Environmental Programs

We face at least two basic problems when environmental goods have to be evaluated. (1) There are no markets on which these goods are traded. Therefore it is impossible to observe prices. (2) Even if there were prices they would not reflect the consumers’ marginal WTP since demand for environmental goods is typically rationed. These goods have to be consumed in certain, fixed quantities which cannot be influenced by consumers. Economists have developed a variety of techniques to value nonmarket amenities consistent with the valuation of marketed goods. These techniques are based upon either observed behavior (revealed preferences) toward some marketed good with a connection to the nonmarketed good of interest called the indirect methods; or, stated preferences in surveys with respect to the nonmarket good called the direct methods. The stated preference approach is frequently referred to as contingent valuation (CV) especially when it is used in the context of environmental amenities (Carson 2000).
**Indirect methods of estimation:** Several studies (e.g. Heintz *et al.* 1976) have been done on the damages associated with water pollution. The methods used to estimate the benefits from water pollution control necessarily differ as among the effects associated with water quality deterioration (Kneese 1985). An estimate of these damages would give us an idea about the extent of potential benefits from water pollution control on account of avoided damage. In order to estimate damages of this nature the **dose-response valuation procedure** may be used (Markandaya 1992). If there is some damage that is linked to a cause, the relationship between that cause and effect is a dose-response linkage. E.g. we may consider the linkage between water pollution and loss of industrial output. Once this production damage effect is established the value of the loss of production can also be estimated.

Foremost among the damages avoided by public water treatment are the **human health problems** associated with contaminants in drinking water. Contamination of water may cause chronic diseases which in turn may contribute to higher death rate (mortality) or non fatal sickness (morbidity). The dose response method may be used for estimating the health benefits of water pollution control programs (Gerking and Stanley 1986). The concept of a **health production function** has been used to estimate the benefits of air pollution abatement in urban India (Kumar and Rao 2001). A household may also take various actions in order to reduce or avoid the use of water of deteriorated quality. Purchasing bottled, distilled water, using water purifier are the actions most commonly adopted to deal with water pollution (Um *et al.* 2002). The health benefits of an improvement in water quality can be estimated using the **averting behavior method (ABM).** Research has shown that averting expenditure cannot be taken as the total cost of environmental unpleasantness (Courant and Porter 1981). Wu and Huang (1998) in comparing stated WTP with actual averting expenditure found that averting expenditure is a lower bound of the WTP measured from CV experiment. Studies have also shown lack of empirical correlation between averting costs and WTP (Laughland *et al.* 1996). Breshnahan *et al.* (1997) in studying averting behavior found evidence that people adjust daily activities to defend against pollution.

The recreation benefits from water quality improvement like swimming, boating and fishing is usually estimated using **Travel cost recreation demand models.** A recreation demand function is estimated on the premise that even though most public recreation sites have zero (or nominal) entry fees, recreationists nonetheless pay an implicit ‘price’ for a
sites’ services when they visit it. This implicit price includes vehicle related and time costs of the trip (Smith et al 1986, Hanley et al 2003 etc).

The effect of water pollution is also found to be reflected in the value of nearby properties. Application of the property value approach (a variant of hedonic pricing technique) for the measurement of benefits from improvement in environmental quality is based on the assumption that since different locations have varied environmental attributes such variations will result in differences in property value (Markandaya 1992). Hedonic models of housing and hedonic prices are often used as measure of environmental damage (McConnell 1990). Clark and Kahn (1989) have also extended the hedonic price model to a two stage model in which WTP may be estimated for specific amenities by means of a multiple regression analysis in which usually cross section data is taken on a large number of diverse properties at a point of time. Using this method the individuals’ bid curves and the suppliers’ supply (offer) curves for housing with different levels of an environmental attribute (say z) can be developed. The equilibrium is given by the points of tangency between the two curves and the households’ WTP for environmental quality improvements can be estimated.

These are the various indirect methods of estimating the benefits from improvement in water quality.

**Direct methods of estimation- CVM:** Contingent valuation is a survey based method frequently used for placing monetary values on environmental goods and services not bought and sold in the marketplace. A CV survey can create an idealized market whereby the respondents face a choice between two different quantities of a public good. Under CVM individual respondents are asked hypothetical questions about how much they would be willing to pay for access to or for preservation of the resource or conversely, how much compensation they would demand i.e. they would be willing to accept to forego this. These are the WTP and WTA measures of valuing environmental goods and services as discussed in Section 2.1.3. The choices made by survey respondents are analyzed in a manner similar to the choices made by consumers in actual markets.

Three types of elicitation procedures are used in CV surveys: open-ended (OE) where the respondent is simply asked to name the sum, sequential bids where the respondents are asked whether or not they would pay or accept some specified sum and the question is then repeated using successively higher or lower sums depending on the previous
response and close-ended or dichotomous choice (DC) where the respondent is asked only whether or not they would pay or accept a single specified sum. CVM has been in use for over 35 years, and there are now over 2000 papers and studies dealing with the topic. However, there are several technical as well as philosophical issues still surrounding the use of contingent valuation method (Carson 2000):

(a) Inclusion of Passive-Use Considerations: Unlike private goods there are many environmental goods that have non-use or passive use characteristics. First discussed by Krutilla (1967), many people value natural wonders simply for their existence (existence value). Aesthetic and ecological values refer to the positive values which non-users place upon improved environmental quality. These benefits may be derived from altruistic motives i.e., one derives utility from the knowledge that others benefit (Poop 2001). People may also value preservation of the environmental resources from more intrinsic motives (i.e., one derives utility directly from the knowledge that a natural resource is being protected).

Using the CVM we can derive WTP or WTA values that give us the total economic value (TEV) of an environmental resource when \( TEV = \text{Use Value} + \text{Existence Value} + \text{Bequest Value} + \text{Option Value} \). Until recently, the only method to estimate WTP for passive use values was CVM. Now there are other stated preference techniques, like conjoint analysis, contingent ranking and choice experiments. Nonetheless, CVM remains the most popular way to estimate passive use values.

(b) Technical Issues Surrounding the Use of CV: The measure of economic value produced by a CV study should conform to several economic criteria and various tests have been proposed for this. This includes the price sensitivity test, the scope test and others (Carson and Mitchell 1995, Kahneman and Knetsch 1992). There have now been a considerable number of tests of the scope insensitivity hypothesis and recent review of the empirical evidence suggests that the hypothesis is rejected in a large majority of the tests performed (Carson et al 1998). A major focus of the technical debate concerning CV has been on the choice of format used to elicit information about the respondent's preferences. Estimates from binary DC questions could be higher than those from OE questions. The argument made by some is that if agents had well-defined preferences for the good, both formats should result in similar estimates (Boyle et al 1996). The counter argument is that incentives for truthful preference revelation are different for these two formats, and as consequence, one should expect the estimates should be different with the
binary discrete choice question predicted to yield truthful responses if other conditions typically associated with a referendum are met (Hoehn and Randall 1987). There are several other issues surrounding the use of CV like the issues of yea-saying, protest zeros, nay-saying, and calibration. The presence of 'untruthful' responses, for whatever reason, leads to arguments that CV responses should be 'calibrated' to potentially correct for either an upward or a downward bias (Blackburn and Lynch 1994).

(c) Quality of a CV Study: The first consideration in evaluating the quality of a CV study is the survey instrument. The environmental good and the scenario under which it would be provided should be described clearly and accurately, and the tradeoff that the respondent is asked to make should be a plausible one, the institutional setting in which the good will be provided and the manner in which the good will be paid for must be clarified and a set of questions given regarding respondent characteristics including attitudes and demographic information. Studies that do not follow good survey practices often produce results that are difficult to use and to interpret (Mitchell and Carson 1989). Pretests and pilot studies are conducted to assess how well the survey works as a whole. The NOAA Panel that was set up to examine the issue of the use of CVM for valuing environmental resources recommends in-person interviews and that WTP questions be worded in a referendum format. According to the NOAA, CV studies can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resource damages including lost passive-use value.

A number of studies (e.g. Brookshire et al (1982), Smith et al (1986), Wu and Huang (1998) and many more) have made a comparison of the direct and indirect methods of estimating environmental benefits. CV estimates tend to be slightly lower and highly correlated with corresponding estimates based upon revealed preference methods such as travel cost analysis. Seller et al (1985) made a study of the comparative validity of estimates derived from travel cost and contingent valuation methods and derived WTP estimates for the value of recreational boating in East Texas. The results suggest that the two methods should be used as a validity check on the consumer’s surplus measures estimated whenever possible. Smith et al (1986) found that the indirect methods can also have substantial variability in their estimates and the question format affects the relationship between contingent valuation and travel cost results. For private goods, surveys tend to predict higher purchase levels than actually observed (Cummings et al 1995), which is the same as the result from comparing survey indications of willingness
to make voluntary contributions and actual contributions to provide a public good (Carson et al 1996). Given the limitations of both the direct and indirect approaches for measurement of environmental amenities and change, there is an increasing use of combined stated preference and revealed preference models (e.g. Sandstorm 1998, Shaikh and Larson 1998, Hanley et al 2003).

Various issues like different types of biases, appropriate welfare measure and elicitation procedure, protest bids, choice of payment vehicle etc associated with the designing of CV surveys have been discussed further in Chapter 6.

3.1.2 Statistical Methods for Benefits Estimation

In recent years, increasing attention has been given to the statistical aspects of contingent valuation (CV) survey design and data analysis. The main reason for the growing interest in statistical issues is the shift in CV practice from using an OE question asking about WTP to using a DC question format. Valuations using referendum data have greater plausibility because they require only yes / no responses, not an estimate of what the consumers will pay (Seller et al 1985, 1986). Further, referendum data help draw CVM into the mainstream of economics because they require that models of preferences, that is, cost functions, be specified and estimated (McConnell 1990). At the same time DC survey design are relatively complex. Survey design elements include total sample size, bid range, specific bid levels, and the allocation of the total sample among the bid levels (Duffield and Patterson 1991). There are a number of welfare measures that have been suggested for referendum type models (Hanemann 1984 and 1989).

With the OE format the survey responses yield a direct measure of WTP which requires little or no further analysis. With the closed-ended format, by contrast, since the CV responses are binary variables, one needs a statistical model appropriate for a discrete dependent variable (Maddala 1983, Greene 1993). The models must also make sense from the point of view of economic theory and this places significant restrictions on the statistical models that can be used Hanemann and Kanninen (1999) give a detailed discussion of the alternative methods for statistical analysis of discrete-response CV data. Traditionally the DC WTP responses (Y) are regressed against a constant, the bid amount (BID), and a vector of socioeconomic variables using a logistic function (e.g. Cameroon and James 1987). While Bishop and Heberlein (1979) first used the DC format and a logit model to analyze the data, Hanemann (1984) exploited the similarity of the DC data and
qualitative response models to give a utility based interpretation to the respondents' decision and the framework for the derivation of a welfare measure.

When a parametric functional form is assumed for the WTP distribution, summary statistics such as mean and median WTP can be estimated. The common structure of the statistical models used in DC survey data can be summarized as follows:

\[
(3.1) \quad \Pr \{ \text{response is 'yes'} \} = H(B; Z; \gamma) \\
\text{and} \quad \Pr \{ \text{response is 'no'} \} = 1 - H(B; Z; \gamma)
\]

where B is the bid on that occasion; Z represents other covariates describing the subject, the item being valued, or any other pertinent aspect of the survey, and \( \gamma \) is a vector of parameters to be estimated from the data.

It is common to write the \( H(\cdot) \) function as the composition of two sub functions

\[
(3.2) \quad H(B; Z; \gamma) = 1 - F[T(B; Z; \gamma)]
\]

which permits the statistical response model to be cast in the form 'yes' if

\[
(3.3) \quad T(B; Z; \gamma) - \eta \geq 0 \quad \text{Response = 'no' otherwise.}
\]

where \( T(\cdot) \) is some function of B and Z, \( \eta \) is some random variable with cumulative distribution function (cdf) \( F(\cdot) \), \( \gamma \) represents both coefficients associated with \( T(\cdot) \) and parameters of the cdf. The composition ensures that the RHS of (3.1) returns a value within the range \([0,1]\). Different discrete response models involve different formulas on the RHS of (3.1).

Economic perspective requires that the survey responses be economically meaningful in the sense that they constitute a utility-maximizing response to the survey question. To satisfy both statistical and economic perspectives we must formulate a statistical model for the CV responses that is consistent with an economic model of utility maximization. For this, we assume individual consumers with a utility function defined over both market commodities, denoted \( X \), and water quality denoted \( Q \). The preferences for \( X \) and \( Q \) would also be affected by other socio-economic characteristics of the individual which we denote by \( S \). The corresponding indirect utility function depends on the prices of the market goods, \( P \); the individual's income, \( M \), and her characteristics, \( S \); and the non-market item \( Q \). The other key component of the indirect utility function is a stochastic component representing the notion of random utility maximization (RUM). In a RUM model it is assumed that, while the individual knows her preferences with certainty and
does not consider them stochastic, they contain some components which are unobservable to the econometric investigator and are treated by the investigator as random (Hanemann 1984). These unobservables could be characteristics of the individual and/or attributes of the item; they can stand for variation in preferences among members of a population and measurement error. The stochastic component is represented by \( \varepsilon \) without specifying whether it is a scalar or a vector. The indirect utility function is \( V(P, M, Q, S, \varepsilon) \).

The individual is confronted with the possibility of securing a change from \( Q^0 \) to \( Q^1 > Q^0 \). We assume she regards this as an improvement, so that \( V(P, Q^1, M, S, \varepsilon) \geq V(P, Q^0, M, S, \varepsilon) \). She is told this change will cost rupees \( B \), and she is then asked whether she would be in favor of it at that price. By the logic of utility maximization, she answers ‘yes’ only if \( V(P, Q^1, M - B, S, \varepsilon) \geq V(P, Q^0, M, S, \varepsilon) \) and ‘no’ otherwise. Hence,

\[
\text{Pr \{response is 'yes'} = \text{Pr \{ } V(P, Q^1, M - B, S, \varepsilon) \geq V(P, Q^0, M, S, \varepsilon) \}
\]

An equivalent way to express this same outcome uses the compensating variation measure, which is the quantity \( C \) that satisfies

\[
V(P, Q^1, M - C, S, \varepsilon) \geq V(P, Q^0, M, S, \varepsilon)
\]

Thus, \( C = C(P, Q^0, Q^1, M, S, \varepsilon) \) is her maximum WTP for the change from \( Q^0 \) to \( Q^1 \). It follows that she answers ‘yes’ if the stated price is less than this WTP, and ‘no’ otherwise. Hence, an equivalent condition to (3.4) is

\[
\text{Pr \{response is 'yes'} = \text{Pr \{ } C(P, Q^0, Q^1, M, S, \varepsilon) \geq B \}
\]

In a RUM model, \( C(P, Q^0, Q^1, M, S, \varepsilon) \) itself is a random variable—while the respondent’s WTP for the change in \( Q \) is something that she herself knows, it is something that the investigator does not know but treats as a random variable. Let \( G_C(\cdot) \) be what the investigator assumes is the cdf of \( C \), and \( g_C(\cdot) \) the corresponding density function (the investigator will estimate the parameters from the CV data). Then, (3.6a) becomes

\[
\text{Pr \{response is 'yes'} = 1 - G_C(B)
\]

Equations (3.4) and (3.6a & 3.6b) constitute not only an economic model of respondent behavior but also a statistical model, since the RHS defines a particular form of the \( H(\cdot) \) function in (3.1), viz. \( H(B; Z, \gamma)) = 1 - G_C(B) \) (Hanemann, and Kanninen 1998).

The individuals will respond ‘yes’ to a particular bid ‘\( B \)’ if

\[
\text{WTP}^b(P, Q^0, Q^1, M, S) \geq B \text{ & 'no' otherwise.}
\]
WTPh\textsuperscript{h} indicates the willingness to pay of the affected household if households are taken as the appropriate unit of observation instead of an individual.

E.g. the probability of accepting the bid related to the quality of water Q\textsuperscript{1} will be

\begin{equation}
\Pr^h(Y) = \Pr \{B - \text{WTP}^h < \varepsilon\}
\end{equation}

As discussed \( \varepsilon \) is the unobserved random component of the WTP function.

Similarly we also have a variation of the DC CVM i.e. the double-bounded dichotomous choice (DBDC) CV method. In the DBDC elicitation procedure the respondent is engaged in two rounds of bidding (Hanemann et al. 1991). Hanemann et al. showed that the use of this elicitation procedure improves the efficiency of the WTP estimates. They modeled the DBDC data using the method of multinomial logit. However a newer technique called multi-level modeling (MLM) is now being increasingly used for this purpose. Multi-level models are a subset of random coefficient modeling in general and are particularly appropriate for the type of data collected in CV studies. We have used the MLM technique for analyzing survey data in Chapter 7.

### 3.1.3 Incidence of Benefits of Environmental Protection Programs

The concept of a potential Pareto improvement has traditionally been one of the major economic tools for evaluating alternative environmental policies. However, increasingly a need has been felt for a valuation and decision making framework that explicitly considers interpersonal welfare comparisons (Gowdy 2004). Interpersonal comparisons of utility were part of the welfare theory as late as the 1920s. However over time the moral content of welfare theory was abandoned and interpersonal comparisons of utility were avoided as being 'normative' value judgments not 'positive' statements of fact.

When a large scale environmental protection program for the improvement of the water quality of a river or controlling the air pollution of a region is undertaken, several sectors of the economy have to be involved. There are significant costs as well as benefits attached to such programs. The distributional impact of these programs is also likely to be quite significant particularly for developing countries that are usually characterized by extreme inequality of income. In this section we look at the methodologies that have been used for studying the distribution of benefits from environmental programs. The basic question we wish to answer here is: "Is the distribution of benefits derived from any environmental protection program 'pro-poor' or 'pro-rich'?"
As mentioned in Chapter 2 the early studies on the incidence of the benefits of environmental policies (Dorfman 1977, Freeman 1972, Baumol 1974 and also Baumol and Oates 1988) have in general argued that the benefits from environmental programs are distributed progressively, i.e., they favor the rich.

Gianessi et al (1979) analyzed the distributional effects of uniform air pollution policy in the United States. They found striking locational differences in benefits since most of the benefits from efforts to improve air quality are concentrated in the more industrialized urban areas. The study finds that, when the distribution of policy benefits is considered along with the distribution of costs, there may be substantial differences in the net impact of the policy when one compares income groups or geographical areas. For the households sector alone there are only four areas in the nation that enjoy positive net benefits. These four areas account for about 8% of the population. For the nation as a whole, only about 29% of the population gain from the entire air pollution policy.

In general the empirical studies on air pollution control have found that in urban areas the lower income households have derived greater physical benefits. This is logical since the poorer families tend to reside in areas of lower air quality (e.g. Asch and Seneca 1979 and Freeman 1972). However the distribution of benefits measured in economic terms could be substantially less pro-poor than the physical benefits suggest. Some of the benefits that initially accrue to renters will eventually accrue to owners, who tend to have significantly higher incomes. Also lower income households may also place a lower value on environmental benefits than higher income households as found by Gianessi et al (1977 and 1979) about air quality improvements.

Markandya (1998) has also explored the literature on distributional impact of environmental policy. He finds that while most studies have found that the distributive impacts of the costs of environmental policies are regressive, by and large the benefits are more evenly distributed. The early literature on the relationship between environmental quality and income focused on the income elasticity of demand for environmental good.

Mishra (1996) in measuring the benefit from industrial water pollution abatement for a small industrial estate in India found that the WTP per annum for user and non-user values increased with an increase in income in both urban as well as rural areas. However in a cross country analysis Kristöm and Riera (1996) found a persistent result that people in lower income brackets have a relatively higher willingness to pay for preserving
particular environmental goods. One explanation could be that the richer people might use substitutes to a much larger extent than poorer people do. According to Johnstone (1997) also while in the past it has often been argued that preferences for environmental quality are income elastic, overall there is increasing reason to believe that this is not the case across a wide spectrum of indicators. This is even more likely to be true of those aspects of environmental quality which are more closely identified with basic needs like potable water and sanitation facilities.

For an investigation of the distributional effects of an environmental good we have to solve two problems, namely: we have to measure the benefits enjoyed and to describe their incidence. A recent paper by Ebert (2003) presents a thorough theoretical investigation of the benefits from environmental commodities and their effect on the income distribution as well as the determinants of progressivity / regressivity. His model comprises consumers having different incomes and takes into account the consumers’ evaluation of the environment though for simplicity he uses an equal-preference model that is, consumers possess the same preference ordering.

The paper assumes that there are $m$ private market goods $X_1, \ldots, X_m$ and a nonmarket good $G$. The latter should be interpreted as a purely public commodity (like e.g., water quality measured appropriately) for the population under consideration. A typical consumer has a strictly convex preference ordering $R$ over commodity bundles $(X,G) = (X_1, \ldots, X_m, G)$. $R$ is represented by an ordinal, twice differentiable direct utility function $U(X,G)$ which is strictly quasi-concave and strictly increasing in each of its arguments; i.e., preferences are nonsatiated and the environmental good $G$ is desired and not a “bad”. $M$ denotes a consumer’s income.

Ebert examines two different maximization problems. to start with the problem actually faced by a consumer is investigated. Typically, a single consumer is not able to influence the quantity $G$. In her view it is fixed. Given the prices $p = (p_1, \ldots, p_m)$ of market goods and her income $M$ a consumer solves a conditional or constrained maximization problem:

\[
(3.9) \quad \max U(X,G) \\
\text{s.t. } pX = M \text{ and } G \text{ fixed, where } pX = \sum_{i=1}^{m} p_i X_i.
\]

Its solution yields the observable conditional demand function $X_i(p_1, \ldots, p_m, G, M), i = 1, \ldots, m$. They depend on the exogenous variable $G$. Inserting $X_i(p,G,M)$ in the direct utility function $U$ we obtain the conditional indirect utility function $V(p,G,M)$ and the
conditional expenditure function \( E(p, G, u) \). Given \( U \), they are unique, but they cannot be recovered from the conditional demand system. Though it is not realistic it is then assumed that \( G \) is also treated as market good which gives us a pseudo-choice problem whose solution turns out to be helpful. Then the consumer maximizes her utility given prices \( p, p_G \), and income \( M' \):

\[
\text{(3.10)} \quad \max U(X, G) \\
\text{s.t.} \quad pX + p_G G = M'
\]

We get the ordinary (unconditional) Marshallian demand system as solution:

\[
\text{(3.11)} \quad X_i^*(p, p_G, M') \text{ for } i = 1, \ldots, m \text{ and } G^*(p, p_G, M')
\]

It should be stressed that the demand function \( G^* \) is not directly observable, but could be derived e.g., by means of contingent valuation surveys or other methods. The demand system can be employed to define the (unconditional) indirect utility function and the corresponding expenditure function \( E'(p, p_G, u') \). Using \( E' \) we obtain the compensated demand functions

\[
\text{(3.12)} \quad X_{ci}^*(p, p_G, u') = \frac{\partial E'(p, p_G, u')}{\partial p_i}, \quad i = 1, \ldots, m \text{ and } G_{ci}^*(p, p_G, u') = \frac{\partial E'(p, p_G, u')}{\partial p_G}
\]

by Shepard's lemma.

These functions, though hypothetical, will be used since they describe the underlying preference ordering: The systems (3.11) or (3.12) allow us to recover the underlying preference ordering.

If \( G \) is a market good consumers are willing to pay the market price \( p_G \) for a(nother) unit because of the assumption of utility maximization. When \( G \) is exogenously fixed (and the consumer is rationed) the marginal willingness to pay for \( G \) can be derived by an extension of Roy's identity from \( V \):

\[
\text{(3.13)} \quad w_G(p, G, M) = \frac{\partial V(p, G, M)/\partial G}{\partial V(p, G, M)/\partial M}
\]

and the compensated one from \( E \): \( w^c_{G}(p, G, u) = -\frac{\partial E(p, G, u)}{\partial G} \).

\( w_G \) and \( w^c_{G} \), respectively, can be interpreted as virtual prices. Suppose that a consumer faces prices \( p_1, \ldots, p_m \) for market goods, has income \( M \), and that the quantity \( G \) is fixed. Then she chooses the commodity bundle \((X_1(p,G,M), \ldots, X_m(p,G,M))\). Now one can ask the question under which conditions the consumer would buy \((X_1, \ldots, X_m, G)\) voluntarily. The answer is given by the virtual price and the virtual income. Assume that the market price \( p_G \) is equal to \( w_G(p,G,M) \) and let the consumer's income be given by

\[
\text{(3.14)} \quad M' = M + w_G(p, G, M)G.
\]
In this case she would buy the commodity bundle \((X,G)\); i.e., we have
\[
(3.15) \quad X_i (p,G,M) = X^*_i (p,w_G (p,G,M),M + w_G(p,G,M)G) \quad \text{for } i = 1, \ldots, m
\]
and \(G = G^*(p,w_G(p,G,M),M + w_G(p,G,M)G)\).

These identities hold for all \(p,G,M\). In other words, if \(M\) is increased by \(w_GG\), the amount necessary to pay the bundle \(G\) at the virtual price \(p_G = w_G(p,G,M)\), the consumer is able to buy \((X,G)\) and, indeed, chooses this bundle. Thus virtual price and virtual income are helpful in connecting the unconditional demand system with the conditional demand system. Equation (3.16) is well-known in the theory of household behaviour under rationing. It can be uniquely solved for \(w_G = w_G(p,G,M)\).

Various benefit measure like equivalent variation (EV), consumer's surplus (CS) and willingness to pay (WTP) can be defined. Since no market exists for the environmental good the market price is replaced directly by the marginal willingness to pay. Then benefits enjoyed can be measured by
\[
(3.16) \quad \text{WTP} (p,G,M) = w_G(p,G,M)G.
\]
It is an income-equivalent of the benefits received from the consumption of the environmental good. \(w_G G\) equals the amount the consumer would be willing to pay per unit times the total number of units consumed.

Similarly EV and CS may be defined. However the informational requirements of calculating EV, CS and WTP are different. For EV and CS the marginal willingness to pay is used on the entire interval \([0,G]\), whereas for WTP only the marginal willingness to pay for \(G\) in the status quo is required. Moreover whenever the environmental good is essential the integrals estimating EV and CS tend to infinity. Also the expenditure function \(E(p,G, u)\) is not defined for \(G = 0\). In such a case EV and CS cannot be employed. The WTP is, however, well defined. Ebert therefore suggests the choice of the income-equivalent WTP \((p,G,M)\) as the benefit measure.

The problem of benefit incidence can be tackled using the general notation \(B(p,G,M)\) for the benefits enjoyed (equal to EV, CS or WTP). In accordance with the methods employed for taxes and benefits in public finance we may consider the (imputed) benefits from the environmental good, expressed as a fraction of income: \(b(p,G,M) = B(p,G,M)/M\). The rise or fall of the average benefit rate \(b(p,G,M)\) with income determines the incidence of benefits.
If benefits increase faster than income their
distribution is progressive or pro-rich. Otherwise benefits are distributed proportionally,
i.e., they increase at the same pace as income, or average benefits decrease with income,
their distribution is pro-poor. Equation (3.17) tells us that the relevant factor is the income
elasticity of benefits from \( G \). In the light of this result, the elasticity denoted by \( \eta(B,M) > 1 \) (or \( < 1 \)) is a sufficient condition for overall progressivity (or regressivity). In that case
benefits increase faster (slower) than income.

Thus the paper investigates whether the benefits from environmental goods, expressed as
a fraction of income rise or fall with income in accordance with the theory of tax
incidence. For a single environmental good it turns out that this fraction increases
(decreases) if and only if the income elasticity of the benefit measure is greater (less)
than one. Then benefits are distributed progressively, i.e., pro-rich (regressively or pro-
poor). The result is broadly the same as that obtained by Kristöm and Riera (1996).

The question of who benefits from public expenditure and therefore what is the benefit
incidence of public goods has also been explored in public finance literature (e.g.
Musgrave 1953, Maital 1973 etc.). The public sector engages in multiple activities
including provision of public goods. Pure public goods satisfy wants which, on account
of their technical characteristics (e.g. non rivalry in the consumption, externalities,
impossibility of exclusion), cannot be provided by the private market. For the purpose of
benefit incidence studies the definition of public goods can be taken as any publicly
induced or provided collective good.

The aim of benefits incidence is to assess how each individual household values the
goods and services provided by the public sector and for this the analyst must answer
these two questions: (i) How much total benefit do particular expenditures generate (ii)
To whom should these benefits be assigned? For decades economists have estimated the
proportion of the burdens of taxation borne by various income classes or social groups
and the proportion of the benefits of government expenditures received by them.
However as pointed out by Aaron and McGuire (1970) authors while studying the
distributional effects of governmental fiscal operations are often driven into countless
compromises in order to obtain any results at all.
How should the value of public goods be imputed to households? Consider a model in which commodities are either pure private goods (i.e., goods that are exhaustively apportioned among individuals and produce no external effects) or pure public goods all of which enter the utility faction of every person. Assume that some resources are taxed away from households (thereby diminishing each household’s consumption of private goods) and devoted to the production of public goods. Aaron and McGuire (1969) showed that the resulting distribution of public and private good consumption may be analyzed into two conceptually discrete steps. The first is an implicit redistribution of private goods or income by taxes and transfers; the second is a purchase for public goods, paid for by other taxes levied on each household at a rate equal to its marginal rate of substitution between private and public good (at the final private-public good position of the household).

The difficulty of measuring the distributional impact of actual taxes and expenditures in practice surpasses that of the procedure described above for a variety of reasons. Government taxes citizens not only to produce pure public goods, but also to produce and distribute specific goods. The tax bill of each household, however, cannot be separated into identifiable components, one supporting public goods, the other supporting specific goods. Further difficulty arises from the fact that the redistributive element of government budgets can be estimated only as a residual once the income value to each individual of public goods, supplied to all, has been estimated. Aaron and McGuire (1970) very correctly point out that this requires knowledge regarding marginal rate of substitution (and, hence, household utility functions) necessary to assign to household the value of public goods from which one can make assumptions regarding the shape of utility functions and generate estimates, however arbitrary. Past studies have failed to recognize that the estimation of the benefits form government expenditures implies certain implicit utility functions applicable at least on the average.

Aaron and McGuire (1970) first used the voluntary exchange approach to estimate the distributional impact of public goods. The voluntary exchange theory of decision-making in the public expenditure is drawn upon for answering the question: “who benefits from public expenditure”? In this an attempt is made to measure how much a person would be willing to be taxed in return for a given public good.
Aaron and McGuire's voluntary exchange model (AM model) deals only with public goods, defined as those goods that are subject to joint consumption. It assumes that these goods are produced efficiently: i.e., the sum of the marginal rates of substitution of all households between the public goods and income (the private goods bundle selected as numeraire) equals the marginal cost of the public good. This basic assumption is needed to permit the model to operate with the basic accounting identity between costs of supplying public goods and the benefits derived from them.

After adopting several simplifying assumptions, the AM model yields the simple conclusion that to each household should be imputed a fraction of the total value of the public goods, proportional to the reciprocal of its marginal utility of private goods expenditure. The shape of the utility of income function then determines the distributional implications of this analysis. If the marginal utility of money is assumed constant, all household benefits are identical, which provides the theoretical framework to an allocation scheme often used in earlier studies. With a declining marginal utility of income of schedule, the quantitative outcome of the analysis depends on the variable $\alpha$ and in the equation $\text{MU} = \frac{C}{Y^\alpha}$ where MU is marginal utility of income, $C$ is a constant, and $Y$ is income.

The previous studies of expenditure incidence divided up the benefits from government spending in proportion to some criterion such as gross income, disposable income, or net assets. Clearly no convincing a priori considerations exist which favour one criterion over another and results are always sensitive to the particular criterion chosen. The AM model is a considerable improvement over these previous studies (Maital 1973). However Aaron and McGuire's results are sensitive to the value of a parameter related to the marginal rate of substitution between public and private goods. The AM model was vague about the real magnitude of $\alpha$ and used $\alpha = -1$ for illustrative purposes. With $\alpha = -1$ benefits can be distributed in proportion to income.

Maital (1973) has quoted several studies that have estimated $\alpha$ which show that $\alpha$ converges to -1.5 for the United States. De Wulf (1981) points out that while the availability of various estimates for $\alpha$ would be no great problem if their distributional implications were minimal but this is not the case. This reduces the operational value of this approach. De Wulf (1981) also points out that one major problem is identifying the beneficiaries of public goods $a$ that has been ignored in the AM model, by assuming that
all public goods enter each household’s utility function in the same way. This assumption was needed to allow the AM model to analyze public goods as an undifferentiated whole. Yet, public goods are not an undifferentiated lot, certainly not if they are defined as “publicly supplied goods” which include not only traditional public goods (pure externality and joint consumption), but all such goods as transfers, education and health, whose beneficiaries are more readily identifiable. The identity between costs and benefits of public goods is basic to the AM model. However the issue of efficient resources allocation poses so many problems as to make the usefulness of costs as a good approximation of benefits doubtful.

Section 3.2
Incidence of Pollution Control Costs Incurred by Government

All the costs of meeting environmental standards and the costs of various environmental programs are ultimately borne by households. They must pay for those costs which are incurred by the government through the process of taxation and expenditure cuts and those incurred by business by a shifting mechanism which includes higher prices, lower dividends and changed job opportunities. An important issue in this context is whether this ‘tax’ on society is progressive, regressive or neutral (Dorfman 1977). In this section we will examine the various issues involved in the determination of the incidence of pollution control costs incurred by government.

3.2.1 Water Pollution Control Programs Carried Out by the Government

Governments of different countries have sought to deal with environmental problems in several ways and the performance of the policy instruments vary widely, depending on the context of their use and the magnitude of the problem. In our study the focus is primarily on local water pollution problems. These can differ depending on the agents responsible for the pollution, the nature of the sources and sinks of these pollutants and the type of pollutants being analyzed.

Abatement of Water Pollution from Domestic Sources: While for the control of industrial effluents the government has to set the pollution norms for different industries, pass laws regarding penal action against defaulting industries and also set up pollution control bodies for the monitoring of these industries. Sewage treatment, the treatment of
water that is used for domestic, agricultural, industrial and other purposes are usually the responsibility of the municipal bodies of the towns and cities. For this the government has to be directly involved in the construction and operation of water and sewage treatment plants. Increasing public pressure for effluent control, generated from a growing perception of environmental degradation, has led to the installation of many new municipal waste treatment systems throughout many countries of the world. However, often the treatment solutions are adopted without adequate evaluation of alternative treatment methods or the potential redistribution of responsibility for waste treatment between different players in the municipality. While the relationship between effluent treatment cost and level of treatment achieved represents an essential element in evaluating water pollution control polices, relatively few empirical studies are available on this subject.

Fraas and Munley (1984) have developed a model for the estimation of wastewater treatment cost. This paper develops estimates of the marginal cost for conventional pollution control by new municipal wastewater treatment plants. Ordinary least-squares regression equations are estimated relating treatment cost to operating characteristics of Publicly Owned Treatment Works (POTW) using data obtained from EPA. The marginal cost to society for the removal of conventional pollutants by new POTWs can be calculated from these regression equations.

Pollution Control from Industrial Sources: For the regulation of industrial water pollution by the government a number of policy instruments are used. These can be broadly classified into the following. (a) Command and control instruments which consist of technology-based standards which define specific abatement techniques for each source of pollution or to performance standards, which set specific ceilings on emission for each source, leaving the source to determine the most effective way to achieve them. Punishment for excess is administered through the legal system. (b) Economic instruments that consist of price instruments, quantity instruments and hybrid instruments. The first refers to taxes and subsidies, the second to marketable permits, the last to combinations of the two. Price instruments include emission charges i.e. taxes on emissions specifies a rate of tax per unit of effluent discharged by the industrial unit into a water body. Where the monitoring of effluents is not feasible a tax may be levied on a proxy commodity like the output of the firm or an input used in production. For marketable permit system, a given area is divided into a number of permits, which are
then distributed (e.g. by auction) to the polluting firms in the area. Firms which are comparatively more efficient in controlling their wastes may sell extra permits to firms which are less efficient. (c) Suasive instruments consist of voluntary pollution agreements between industry and the government or between industry and the local community, in which the industry 'volunteers' to reduce its waste emissions by a certain amount within a specified time (James and Murthy 1996).

Figure 3.1 shows the marginal abatement costs of pollution control. Figure 3.2 shows how we get aggregate effluents from firms' emissions (Boyd 2003). The equivalence of the two instruments and their ability to generate the efficient outcome however break down in the face of real-world constraints. The most important of these constraints is that regulators have limited information regarding both firm specific control costs and the benefits arising from reduced emissions. When this informational constraint is taken into account, policy goals change. Figure 3.3 shows how pollution taxes give the least costs solution.
Economists have long advocated pollution taxes as a policy to improve water quality (Boyd 2003). One of the reasons is that sources of water pollution are varied and difficult to assess individually in terms of control costs. Taxes overcome this problem. With a tax applied to pollution emissions the firms compare this price to the costs they have to incur for emissions control. The high-control-cost firms therefore abate less and low-control cost firms abate more and a given level of pollution reductions is achieved at the least cost. Taxes also promote innovation and generate revenue which is particularly pertinent for developing countries since most water quality improvement projects involve large investments.

Nemetz (1980) carried out a study on a small municipality in British Columbia in order to demonstrate a methodology for the identification of least cost treatment systems for water borne effluents generated within a municipality. The paper constructs a materials balance model for identification of major pollutant sources. Then an assessment of alternative capital and operating costs of treatment is carried for the identification of the optimal treatment solution. The study finds that only the end-of-pipe waste treatment processes should not be compared. Rather a broader system perspective should be adopted since industrial process changes or other methods of pollution abatement may be cheaper and more efficient than end-of-pipe treatment by municipal waste treatment systems. A comprehensive systems analysis of the urban wastewater treatment problem is likely to yield more useful results.
Since technologies, locations, and other circumstances differ in important respects from one facility to the next, for the first-best outcome to be achieved, different pollution sources must be assigned sources-specific tax levels. In practice a regulator cannot easily determine these differences. For these reasons the second best policy option is often chosen namely, what is the best way to achieve a minimum level of ambient water quality in a water body? Achieving ambient standards is a more limited and realistic goal for which the states must establish surface water quality standards. Taking these standards as given the ways in which these standards can be met at minimum cost must be determined (Kneese and Shultze 1975). One may adopt the uniform treatment approach where all firms are required to reduce emissions by an equal percentage or amount, sufficient to meet an ambient standard.

In spite of its advantages effluent tax has not been widely adopted. Political opposition to taxes as a control instrument comes from the regulated firm themselves as well as environmental interests who are wary of the flexibility inherent in a price-based approach. One reason why, in spite of their apparent efficiency, economic instruments like environmental taxes have been adopted less frequently to mitigate environmental damages than direct regulations could be that some of the distributional issues involved in the incidence of environmental taxation have been neglected. Nevertheless interest in taxes as instruments of environmental protection has grown considerably in recent years. In many industrialized countries, environmentally motivated taxes either have been introduced recently or are being seriously contemplated (Johnstone & Alavalapati 1998). The actual and potential reforms include taxes on fossil fuels (such as carbon taxes and fossil-fuel-based BTU taxes) and taxes on gasoline. The growing importance of environmental measures means that their distributional implications can no longer be ignored, particularly in the face of increasing economic inequality in many countries (Johnstone and Alavalapati 1998).

3.2.2 Incidence of Pollution Control Costs Incurred by Government

As pollution controls become an increasingly important part of the government's regulatory activity in different countries all over the world, the question of their ultimate incidence should also be assuming a larger part of the public's concern. Depending upon the particular circumstances, the burden of these controls may be shifted forward onto the consumers of the polluting products or backwards onto the various factors of production (Yohe 1979). However despite their potential significance these incidences have not
attracted a great deal of attention in literature. In order to determine the pattern of incidence of water pollution control expenditure incurred by the government among different income groups, we must determine not only the absolute level of that expenditure but also the way in which the expenditure is financed (Markandaya 1998). Water pollution control facilities are financed by central, state and local governments. The first task then is to allocate costs among these three levels of government (Roberts 1984).

Increase in taxes and expenditure cuts are basically the two modes of financing any government program. If the total expenditure is to be financed via tax increases then net incidence can be found by comparing the distribution of taxes among income classes along with the distribution of the water pollution abatement benefits. If on the other hand the new expenditure is made at the expense of some old expenditure then we should try to estimate the differential incidence of the pollution control expenditures incurred by the government. We may make a set of assumptions about what public goods would have been provided without the pollution control expenditure. We can then compare this estimated public budget with the actual budget to determine the source of pollution abatement funds. The conclusions would clearly depend heavily upon the choice of assumptions (Roberts 1984).

Accurate analysis of the social cost of environmental regulation requires a sophisticated application of welfare economics (Pizer and Koop 2003). Hazilla and Kopp (1990) point out that often the cost of regulations calculated by agencies is not based on the theoretical concept of social costs. Rather agencies equate social cost with annualized engineering costs of installed capital and related operating and maintenance expenses and general equilibrium impacts and intertemporal effects of regulations are typically not included in the evaluation as discussed in Section 2.1.2.

All the costs, whatever their point of initial impact, are defrayed ultimately by households. The governmental expenditures are transmitted by increases in taxes and by reductions in government expenditures in other sectors. Business expenditures are shifted to households primarily by increases in prices (Dorfman 1977). A proper evaluation of the incidence of the costs of environmental protection is difficult because most measures impact on firms, which result in changes in relative prices that are complex to model in a way that is empirically accurate (Markandya 1998).
Incidence of Taxes and Public Expenditure in General: In allocating the tax burden associated with a specific government program between different income groups, the analyst confronts two problems: (1) identifying which taxes finance that program, and (2) determining their incidence. A mistaken assumption in solving either problem introduces error into the program incidence calculations. The tax incidence assumptions can very across studies since economists are not fully agreed as to the final incidence of some taxes, particularly the corporation income tax (Long and Settle (1982). Hansen and Weisbrod (1971) have examined how the distribution of benefits from a public expenditure program compares with the distribution of taxes that pay for these benefits. They have questioned the equi-proportional approach followed by most tax and expenditure incidence studies since it cannot be assumed that what is true for a total is necessarily true for each of its parts. They point out that the equi-proportional allocation assumption has no logical foundation. If a particular program were to be eliminated from the government budget, it is not at all clear which tax or taxes would be cut as a consequence. They conclude that since "there is simply no non-arbitrary way to say which tax dollars go to finance any particular expenditure, these types of calculations should not even be performed". To see the distributive effects of any public expenditure program the authors suggest that one should concentrate only on the distribution of benefits.

Long and Settle (1982) suggest that the sensitivity of the incidence estimates to any particular set of assumptions can be determined. However by adopting different tax incidence assumptions like the equi-proportional and the incremental assumptions about tax mix they get very similar results and therefore conclude that researchers undertaking studies of burden distribution are probably safe in adopting any compromise set of assumptions.

In the traditional partial equilibrium approach the taxes on products are taken to affect households from the uses side for their accounts only, the burden distributed in line with the distribution of consumer expenditures. Further effects on factor prices, which may affect the position of households from the sources side, are disregarded, as are second-round effects on relative commodity prices. Thus, it is concluded that a sales tax on a luxury item will be progressive whereas one on a necessity will be regressive. Similarly, taxes on factor income such as the income tax are taken to affect household positions from the sources side only. The underlying argument in these studies is that the burden
distribution of a tax which initially impacts from the source side will be dominated by source-side effects. Similarly, it is postulated that the effect of a tax which initially impacts from the uses side will be dominated by uses-side effects (Musgrave and Musgrave 1989). These hypotheses have been tested by Devarajan et al (1980). The found that the general equilibrium approach takes into account the total effects and as a result the net effect of a progressive change in income tax is less progressive than intended and they get similar results for commodity taxes.

In any incidence analysis apart from assumptions about the direction of shifting of the various taxes under consideration researchers must also determine how to rank people using some measure of ‘well-being’ (Metcalf 1999). As pointed out by Poterba (1989) household income measured over long horizons is less variable than annual household income and the failure to distinguish between lifetime and annual incidence overstates the degree of inequality in tax burdens between groups. In fact a repeatedly stated qualification to annual calculations in the empirical tax incidence literature is that it would be more satisfactory to make calculations on a lifetime basis (Davies et al 1984).

In public finance literature the focus has mostly been on the distribution of tax burdens and not on expenditure benefits since there are many additional problems associated with the estimation of the latter. De Wulf (1981) described four different approaches that have been used for the analysis of the distributional implications of budgetary expenditure: (a) the money flow approach or impact incidence, (b) the study of ‘on whose behalf are expenditures made’, (c) the expenditure incidence, and (d) the benefit incidence.

The first two approaches are cast within an ‘accounting’ framework. The money flow approach or impact incidence studies allocate the cost of government outlays among the direct recipients of these outlays; the second approach goes one step further and analyzes on whose behalf are public expenditure made without inquiring how these beneficiaries value these goods. The other two approaches depart from the ‘accounting’ framework. Expenditure incidence studies deal with the income distributional effects of public outlays on input and output prices, but continue to express the results in income terms. The benefits incidence studies attempt to measure how public expenditures affect the welfare of the beneficiaries and express their results in terms of utility to the beneficiaries. De Wulf points out that there has apparently been no systematic empirical research on this issue, because data requirements and other theoretical implications are overwhelming.
There is a large volume of literature on the incidence of specific taxes as well on the estimation of the differential incidence of a number of general taxes (Mieszkowski 1967). However, firm scientific judgment regarding the distributional implications of alternative tax measures, is exceedingly difficult to come by. Approached in rigorous terms, the problem involves full-fledged general equilibrium analysis with all its difficulties. The general equilibrium problems associated with government expenditure are related to output effects due to resulting changes in techniques, changes in labour supply, changes in savings and capital formation, or in the efficiency of resource use (Meerman 1978).

In the studies that have been carried out for analysis of the incidence of government expenditures fairly standard distributional patterns have been reported. The combined specific expenditures (those that can be attributed to readily identifiable beneficiaries) are almost always found to be pro-poor. The distributional impact of public good expenditures may vary from pro-poor to pro-rich, yet in most studies the net result is that total government expenditures are pro-poor (Crane 1983). E.g Gillespie (1965) found that the federal pattern of fiscal incidence generally favors low incomes, burdens high incomes, and is mainly neutral over a wide middle income range. The state and local pattern also favors low incomes, but is essentially neutral over both the middle and high income ranges. Crane (1983) however concludes that given the methodological difficulties encountered in distributing government expenditures, we really know very little about the distributional consequences of government expenditures.

**Incidence of environmental regulations**: There is usually a strong interest in regulatory reform to enhance the efficiency of the nation's pollution control policies. However it is also a political reality that to achieve environmental improvements legislations must be passed and regulations implemented and enforced. The process may be aborted if key interest groups take offence (Shortle and Willet 1986).

The distributive effects of environmental policies have been studied in some depth in the industrialized countries but not in the developing countries. Markandya (1998) has reviewed these studies and found that in most studies the distributive impact is mildly regressive and the degree of regressiveness varies according to the policy implemented. Dorfman and Snow (1975) studied the distribution by income of the burden of the national environmental protection program 1972-80 in the US and found that the national environmental protection program exacted a substantial and continuing price from the
public. The study assumes that tax increases financed the government's share of the pollution control programs since studies have shown that across the board budget cuts at any level of government are likely to be more regressive than across the board tax increases. Each governmental level was assumed to rely on the major revenue sources excluding those such as social security taxes which are not likely to be drawn upon to finance the environmental program. A theory of incidence based on the traditional approach was then used to determine how the burden would fall on individual households depending on their sources and uses of funds and the study found that the distribution of government shares of pollution costs as a percentage of family income for the years 1972, 1976 and 1980 was decidedly regressive except for the very highest bracket.

The control costs for mobile source air pollution through vehicles also tend to be regressively distributed (Harrison (1977). For stationary sources the distribution of costs is more complex to model because the incidence structure is more involved but in essence the US studies reveal the same finding of regressivity of costs (Gianessi et al 1979). A review of the literature on the social incidence of environmental costs and benefits was also done by Pearce (1998) with similar conclusions. Pearce also looks at the studies done on the physical incidence of pollution. Several studies have shown an inverse relationship between air pollution and income classes. Overall there is evidence of pro rich distribution of environmental quality. This could be because while the poor remain 'locked-in' to polluted areas the rich are able to move to cleaner areas. Thus broadly speaking the existing literature finds the cost distribution of pollution control policies tend to be regressive.

The possibility of the use of green taxes or environmental tax reform (ETR) has created great interest both within political discussion and economic research (Pirttila and Toumala 1997). ETR include (a) emission and effluent taxes which are targeted directly at the source of environmental damages like ozone-depleting chemicals tax, water effluent charges etc. (b) disposal and waste taxes which are targeted at the unwanted by-products of production processes like landfill levy and hazardous waste charges; (c) taxes on inputs that are closely associated with particular environmental damages like carbon taxes, fertilizer charges, battery charges etc. (d) charges and fees which are levied on the use of services with significant environmental impact like water charges, municipal waste user charges etc. However such taxes can only be used extensively provided the distributional impact of these taxes is fair or are separately dealt with.
It has been argued that ETR tend to be borne disproportionately by the poor, it is also argued that the benefits are realized disproportionately by the rich. Part of the reasoning behind this assertion however arises from a belief in the income-elasticity of demand for environmental quality discussed in Section 2.1.4.

In analyzing the distributional effect of environmental tax reform, Johnstone & Alavalapati (1998) carried out a study of three environment-intensive (both in terms of resource use and pollution emissions) sectors: residential energy, private road transport and agriculture. They found that in energy and agriculture ETR is likely to have adverse distributional consequences at least in terms of direct tax burdens. This arises from the fact that domestic energy consumption and food are basic needs, consumed in a relatively greater proportion by lower income households. In case of transport ETR is unlikely to have regressive consequences in terms of tax burdens.

According to Metcalf (1999) too environmental taxes tend to be regressive: poor people pay a disproportionate share of their income in these taxes relative to rich people. While some authors have challenged this perception by taking into account lifetime considerations (e.g. Menchik 1991, Davies et al 1984) etc., it is clear that distributional concerns limit political support for the greater use of environmental taxes. Metcalf (1999) however argues that while it might be the case that the imposition of an environmental tax by itself is regressive, it is quite possible that a revenue neutral tax reform, where an environmental tax replaces some other tax, could be progressive and environmental taxes must not be viewed in isolation. The paper analyses the impact of a moderate shift in income tax base in which 10% of federal receipts is replaced with a cluster of environmental taxes like taxes on carbon emissions, gasoline consumption, air pollution, and a tax on unrecovered waste. The incidence results show that the lowest income group faces an increase in taxes equal to 7% of income, while the top income group faces an increase equal to 1.6% of income. However from further analysis it appears that many distributional concerns about the greater use of environmental taxes can be addressed through a careful menu of tax reductions that are targeted to low-income households and distributionally neutral (or even mildly progressive) GTR are quite feasible.

Gasoline tax’s regressivity is often cited as one of the strongest arguments against increasing this tax. However West (2004) found that a tax on miles or gasoline is regressive only across upper income groups. This is because many lower income
households do not own any vehicles and in response to price increase poor households reduce miles by more than do wealthy households. West and Williams (2004) have also recently studied the distributional effects of a gasoline tax increase. They found that the increase in gasoline tax will generally be regressive though it can become somewhat progressive if the additional revenue is used to provide a lump-sum transfer to households if the transfer is sufficiently progressive to outweigh the regressivity of the tax increase. Using the additional revenue to lower taxes on labour yields an efficiency gain and makes the policy more progressive, but not enough to overcome the regressivity of the gas tax.

The tradable permits approach pioneered by US is receiving a great deal of attention throughout the world. Parry (2004) has studied if emission permits are regressive. The study finds that if revenue recycling occurs through broad income tax deductions there is a strong equity as well as efficiency argument for using emissions taxes or auctioned permits. On distributional grounds it is better to use emissions taxes or auctioned emissions permits in stead of grandfathered permits.

To conclude, most empirical studies find that the costs of environmental policies are borne disproportionately by lower income groups. This appears to hold across a range of policy instruments, especially grandfathered emissions permits and energy taxes. The finding is less pronounced for taxes on intermediate products than for taxes on final goods, and when some measure of lifetime income is used rather than annual income, though measuring lifetime income remains problematic especially in cross-section studies (Parry et al 2005). It is evidently very important to address distributional consequences when introducing ETR particularly as they start to play a larger role in the economy.

**Incidence of Water Pollution Control Programs:** The benefits from any regulatory intervention undertaken for the improvement in the quality of any water body (which depend on a variety of chemical, physical, biological and economic factors) are likely to be quite uncertain and difficult to estimate (Hammitt 2000).

Gianessi and Peskin (1980) carried out a study of the distributional consequences or 'equity' of the costs of Federal Water Pollution Control Policy of the US. The analysis uses a partial equilibrium approach and considers only initial distributive impacts on different regions and on different income groups. According to the Federal Environmental Policy in US the municipal treatment of wastewater is financed partly by a federal subsidy and partly through mandatory cost sharing by industrial users of municipal
facilities. For most of the water pollution control policies analyzed, family cost percentages declined with increases in income showing a regressive pattern of incidence. Comparing the results of this study with the studies carried out to estimate incidence of air pollution control costs the authors find that the water pollution control policy is less regressive than the air policy because of the municipal treatment subsidy policy. The share of water pollution control costs actually borne by federal and local governments is about 45% of the total costs. In contrast government outlays are only about 10% of the air pollution control costs.

Generally economists do not support the use of government subsidies to offset the polluters' costs of cleanup on the principle that the polluter should bear the costs of the control activities necessitated by his own polluting actions. This has strong justification on the grounds of efficiency. However the results of the paper by Gianessi and Peskin (1980) suggest that central governmental subsidies may tend to alleviate the regressive burdens that result either because of the polluters' ability to shift costs to consumers or because of relatively regressive state and local tax structures.

Roberts (1984) studied the distribution of costs and benefits of water pollution control in the Boston Metropolitan Area. The study focused on the distribution among income classes of water pollution abatement costs borne by local governments since all operating and maintenance costs of sewage treatment plants were incurred by local municipalities. Data from two large towns in the Merrimack River basin area: Nashua, New Hampshire and Leominster, Massachusetts was used. The study found that: (a) In general, costs which are covered by (property) tax increases are more regressive than those financed by alternative expenditure reduction. (b) The regressivity/progressivity of the expenditure substitution financing depends critically on the pre-pollution expenditure allocation of the government. Thus expenditure substitution is more regressive in towns in which health and welfare expenditure is a prominent item, than in towns in which highways was a major portion of the town budget. Water pollution abatement expenditures will therefore tend to be least regressive in those towns whose budgetary patterns are least pro-poor in the period prior to pollution control expenditures.

Like Gianessi and Peskin (1980), Shortle and Willet (1986) too have studied the incidence of the federal water pollution control policy of the United States. However the latter have restricted their study primarily to industrial wastewater discharges and not
covered municipal sewage treatment facilities. Shortle and Willet (1986) suggest that it is important to use a general equilibrium framework. This paper compares two computations of the distributional effects of a fully implemented federal water pollution control policy: one based on an extreme partial equilibrium method used by Gianessi and Peskin (1980) and the other based on the use of a CGE model. A comparison of the results from CGE model and partial equilibrium model shows some differences.

In fact GE analysis has long been recognized as an ideal structure for assessing the implications of policy changes on resources allocation and distribution of income. However as partial equilibrium models have been the norm in applied work, there has been a significant gap between theory and practice. Kochanowski et al (1976) pointed out numerous partial equilibrium biases in economics of pollution abatement which suggests strongly that the problems of pollution control cannot be approached satisfactorily through partial equilibrium analysis. While Gianessi and Peskin found that the expenditure burdens of federal water pollution control policy in US are regressive, Shortle and Willet (1986) using GE computations to evaluate the same policy found that water pollution control policy is only regressive over the lower income classes and it actually has a progressive impact at the upper end of the income distribution.

Collins (1977) has examined the distributional impact of federal grants for water pollution control in US to determine who benefits from these government subsidies and by how much. The model used for the analysis of these distributive effects is essentially the same as the one suggested by Gillespie (1965). The primary objective of the paper is to estimate the potential effects of the federal subsidy program for construction of POTWs on the distribution of family income. He uses an equiproportional approach in which a cross section of general revenues is used to represent the burden of a public expenditure funded out of general revenues. Collins suggests that this is the only viable method for solving this problem. He studied the EPA region VII (Iowa, Missouri, Kansas and Nebraska) and found that the incidence of the federal construction grant program mainly favored the rich by redistributing income from the middle income groups to primarily the very rich. Ostro (1981) also carried out a study for estimating the distributional impact of federal grants for water pollution control for Boston Metropolitan area using a similar methodology as Collins. He, however, found a substantial redistribution from upper to lower and middle income groups.
Section 3.3
Incidence of Pollution Control Costs Incurred by Industry

In this section we take a look at the water pollution abatement activities of industries and the costs of these activities. The direct compliance cost borne by the industries is the change in production costs entailed by the policy. This cost will depend on the particular technological alternatives available to the firm. Techniques for modeling and estimating production technology are well described in economic literature. Surveys of business and governments can be used to provide direct estimates of pollution control costs. However, for determining the distributional implications of environmental regulations we must also determine the impact of these regulations on the growth of industrial output, decisions about location of industries and the competitiveness of the regulated firms. Doing an exhaustive study of all these aspects was beyond the scope of this research work. Instead we have reviewed some studies on each aspect in order to draw some general conclusions about the impact of environmental regulations on the industrial sector.

3.3.1 Water Pollution Abatement by Industry

As discussed in Section 3.2 environmental policy may be implemented through a combination of instruments including emission charges, direct controls on emission levels, rewards for pollution removal, and deposit-refund systems. Most studies (e.g. Pittman 1981) show that there are serious inefficiencies resulting from the current system of pollution control regulations based on command and control instruments and more market-oriented control regimes, using either effluent charges or transferable emission permits, would potentially result in a more efficient allocation of pollution control resources.

Lanoie et al (1998) examined if Canadian legislation involving inspection of factories, initiation of proceedings against firms, imposition of penalties on firms transgressing pollution limits, and creation of new emission limits, was able to reduce emissions for the pulp-and-paper industry during the period of 1985-1989 in Ontario. The findings are that firms seem to respond by announcing investments, but these investments do not seem to have much effect on emissions which suggests that policy may have placed undue emphasis on the abatement technology itself rather than its results. On the other hand when McClelland and Horowitz (1999) studied the costs of water pollution regulations in the pulp and paper industry they found that aggregate BOD emissions from pulp and
paper plants in 1992 were roughly 50% of the amount allowed under the Clean Water Act in US indicating a very high degree of over compliance.

Paragal et al (1997) have used survey data in India to examine the impact of monitoring and enforcement efforts of provincial pollution control authorities. The study finds that high levels of pollution do elicit a formal regulatory response in the form of inspections in India but there appears to be no impact of inspections on emissions. Similarly Gray and Shadbegian (2004) used a plant level data set consisting of 409 pulp and paper mills to examine the determinants of environmental regulatory activity like inspections and enforcement actions. They found that plants with more kids, more elders, and fewer poor people nearby emit less pollution indicating that where there are higher marginal benefits of pollution abatement the pollution levels are lower.

Most studies (like Dasgupta et al 1996, Goldar et al 2001 etc) assume that the BOD effluent concentrations are exogenously determined and the plants are designed to maintain BOD effluent concentration at the mandated levels. However some studies like McConnel and Shwarz (1992) have analyzed the determination of pollution control from wastewater treatment as an economic decision facing the regulators. The study shows local jurisdictions may remain out of compliance, be slow to comply, or be lax on enforcement efforts if the perceived benefits are less than the costs. On the other hand, some jurisdictions will perceive benefits of reduced pollution to be high and control more than the federal minimum.

In many developing countries like India the informal sector is responsible for significant amounts of air and water pollution as well as solid waste generation. In most developing countries, the informal sector has grown swiftly over the last several decades as a consequence of population growth and rural-urban migration. Today it accounts for over half of non-agricultural employment in virtually all Latin American and African countries (Blackman 2000). Certain types of informal activities like leather tanning, electroplating, metalworking, brick and tile making etc can create severe pollution problems. Evidence suggests that informal sources are more pollution-intensive than larger sources since they use inputs relatively inefficiently, lack pollution control equipment and access to basic sanitation services such as sewers and waste disposal, and are operated by persons with little awareness of the health and environmental impacts of pollution. Controlling pollution created by informal firms is especially difficult due to distributional
considerations as they sustain the poorest of the poor. As a consequence, they may appear to both regulators and the public as less appropriate targets for regulation than larger firms.

**Costs of Industrial Water Pollution Abatement:** Industrial facilities can abate pollution by scaling back polluting activities or by diverting resources to cleanup. In either case, pollution will entail costs. Economists mainly focus on marginal costs of pollution abatement. Marginal costs tell us the cost, primarily in foregone profits or rents, of relaxing or tightening the relevant regulations. In general diminishing returns will apply: more resources will have to be devoted to cleaning up each additional unit of pollutant. Hence, the marginal abatement cost (MAC) function slopes upward from right of left as pollution falls. The position and slope of the MAC function are affected by factors such as the scale and sectoral composition of production and the efficiency of waste treatment technologies. For any given level of pollution, more costly pollution control is associated with rightward movement of the MAC function. Conceptually, abatement cost functions are dual to abatement functions which relate inputs of capital, labor, energy and materials to pollution reduction.

Several studies like Mehta et al (1993), James and Murthy (1996) etc have estimated water pollution abatement costs for Indian industries. The relationship of the abatement cost with the volume of water treated, the characteristics of the influent and effluent streams (i.e. the extent of pollution abatement done) and the prices of inputs used in the pollution abatement activity (labour, capital, energy etc.) has been analyzed in order to estimate the MAC. Goldar et al (2001) have used an alternative framework in which each pollutant is taken as one type of output since the output of the plant is given by the extent of pollution reduction achieved for the given volume of wastewater treated, and thus the plant is thought of as having multiple outputs. This methodology is applied to derive cost estimates for small scale industries in Nandesari Industrial estate. Unlike the study by Dasgupta et al (1996) which uses a joint abatement cost function, this study estimates a cost function based on COD only since the reductions made by firms in concentration levels of different pollutant are correlated and including COD, BOD and other pollutants would affect the cost estimates by multi-collinearity. The study also assumes (unlike McConnel et al 1992) that the extent of pollution abatement is exogenously determined since the extent of abatement done by firms in this area is largely determined by the technical requirements imposed by the industry association for the proper functioning of the CEPT.
In general firms can adjust to the threat of higher pollution-related costs along many dimensions, including new process technology, pollution control equipment, improved efficiency, and allocation of more resources to legal representation or negotiation. At the plant level, all these options will register as changes in the scale and mix of inputs and, consequently, total production costs. Thus, pollution control will have an impact on conventionally-defined total factor productivity (TFP) which may be significantly different than directly-reported abatement costs. However, the focus of this study is not on issues related to measurement of the costs of pollution abatement and in the next section we look at the factors likely to influence the incidence of pollution control costs incurred by industries.

3.3.2 Impact of Water Pollution Regulations on the Industrial Sector

The conventional wisdom is that environmental regulations of any form imposes significant costs, slows down productivity growth, causes unemployment and also hinders the ability of domestic firms to compete in international markets. In this Section we study the impact of environmental regulations on growth of industrial output, location of industries and international competitiveness that have been documented in several studies across countries. This clearly has important implications in terms of the distributional impact of pollution control activities of industries.

Growth of Industrial Output: There was a significant productivity slowdown in the United States that began in the latter part of the 1960s and accelerated in the 1970s. Environmental regulations have been cited as one of the contributing factors in this productivity decline. For the aggregate economy, the estimates of the productivity decline in the 1970s resulting from environmental regulation have ranged from 0.10 to 0.35 percentage points annually (Barbera, and McConnell 1986). While these numbers do not represent a large proportion of the productivity decline for the whole economy during this period, they may imply substantial impacts for particular industries most affected by this type of regulation. Empirical evidence reveals that environmental regulations have reduced both average productivity of labour (APL) and average productivity of capital (APk) growth in a number of industries (Henderson 1996, Gollop and Roberts 1983).

Barbera and McConnell (1986) estimated systems of factor demand equations derived from a CES production function for four separate US manufacturing industries – paper, chemicals, primary metals, stone, clay, and glass – that are heavily affected by the environmental regulations. They found that abatement regulations retarded both APk and
AP_t in three of the four industries, with paper the only exception. Those industries in which AP_K was slower after 1973, chemicals and SCG (stone, clay & glass), a large proportion of the slowdown may be attributed to the abatement regulation.

Gray (1987) has examined the impact on productivity growth of worker health and safety regulation by the Occupational Safety and Health Administration (OSHA) and environmental regulations by environmental Protection Agency (EPA). Looking at data for 450 manufacturing industries between 1958 and 1978, this study has found evidence that OSHA and EPA regulation reduced productivity growth in the average manufacturing industry by 0.44 percent points per year and over 30 per cent of the slowdown in the 1970s. Similarly Gray and Shadbegian (1998) based their study on the pulp and paper industry to analyze the relationship between environmental regulation, investment timing and technology and they found that plants with relatively high pollution abatement capital expenditures invest less in productive capital over that period. The reduction in productive investment is greater than the increase in abatement investment, leading to a lower total investment in high abatement cost plants.

Location of Industries: Levinson (1996) makes a systematic attempt to measure the effect of state environmental regulations on new manufacturing plant locations. He however finds little evidence that stringent state environmental regulations deter new plants from opening. This is in spite of the fact that on an average the industries studied spent about 4% of their investment dollars on pollution abatement equipment. Levinson suggests that the explanation could be that firms manufacturing products in a variety of jurisdictions find it most cost effective to operate according to the most stringent regulations, eliminating the necessity of designing a different production process for each location.

Deily et al (1991) carried out an empirical study to examine the EPA’s enforcement actions towards US steel plants to test the hypothesis that the regulating agencies direct less enforcement activity toward plants that are likely to close and found that firms’ plant-closing decisions were indeed influenced by the enforcement activity of the regulators. Becker and Henderson (2000) in investigating the effects of environmental regulation on firm decisions concerning plant locations, births, sizes, and investment patterns in major polluting industries found that environmental regulations vary across regions and this has led to significant relocation of polluting industries from more to less polluted areas to

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avoid stricter regulation in more polluted areas; there has been relative proliferation of small-scale, less regulated enterprises in some industries, altering industrial structure; and, in regulated areas, the timing of plant investments by new plants has also been dramatically altered.

**International Competitiveness:** It is believed that there is loss of international competitiveness in industries facing environmental regulations which is reflected in declining exports, increasing imports, and a long-term movement of manufacturing capacity to other countries, particularly in pollution-intensive industries. However under a more recent, revisionist view, environmental regulations are seen not only as benign in their impacts on international competitiveness, but actually as a net positive force driving private firms and the economy as a whole to become more competitive in international markets.

Jaffe et al (1995) have examined the impact of environmental regulations on the competitiveness of US manufacturing sector. To quantify the overall, long-run social costs of regulation a general equilibrium perspective is essential in order to incorporate inter industry interactions and cumulative effects of changes in investment levels. The study finds that overall there is relatively little evidence to support the hypothesis that environmental regulations have had a large adverse effect on competitiveness. This could be because (a) the existing data are severely limited in their ability to measure the relative stringency of environmental regulations (b) for all but the most heavily regulated industries the cost of complying with environmental regulations is a relatively small fraction of total costs of production and (c) the difference in environmental regulations between US and other developed countries is not much.

A study by Pittman (1981) by estimating the production function for a sample of 30 pulp and paper mills in Wisconsin and Michigan, finds a positive association of pollution control intensity with economies of scale. Treatment requirements increase the minimum efficient size of plant, thus increasing barriers to entry and exacerbating any lack of competition in the industry.

The overall findings of this section highlight some of the factors that need to be considered while making policy decisions in the domain of environmental regulations of industries particularly with a view to their distributional considerations.
3.3.3 Incidence of Pollution Control Costs Incurred By Industry

Apart from the water pollution control expenditure borne directly by the government, industries bear the bulk of the initial costs of water pollution control. How and to what extent the industries are able to pass these on to the people is of interest to economists, policy makers and general public alike. To study the distributional effects of these water pollution control costs incurred by the industries one approach could be to see control costs as a tax on producers and then to evaluate what proportion of this tax is passed on to the consumers in the form of higher prices. To start with we may use a micro static partial equilibrium framework to study the process of shifting of control costs and their secondary effects. Thus by recourse to traditional monopoly and perfect competition models of the firms we may evaluate the process of shifting and incidence.

Imposition of environmental regulations and adoption of water pollution control measures by industries are likely to increase the cost of production of the output of that industry. A part of this increased cost is passed on to the consumers in the form of higher prices. In a study of the incidence of price changes in the US, McElroy et al (1981) evaluated the income redistribution effects of environmental policies. The percentage impact on the welfare of rich consumers relative to that of a poor consumer ranges from 0.3 to 10.3. The quantification of income distribution consequences permits the debate to shift from the issue of what the effects of an environmental policy might be to the question of whether those effects are desirable.

Drofman (1975) in analyzing the distribution of the burden of the portion of national environmental protection program that was incurred by the private sector found that the extent of forward shifting will depend on the conditions of demand and supply in the relevant industries but it is assumed that in the long run they are fully passed on. The industries found to be most affected by antipollution measures were the intermediate goods industries primary metals, paper and allied products, petroleum refining and clay products. Also affected were leather tanning and food processing industries. An effort to trace the inter-industry effects of cost increases in the relevant industries to specific categories of consumption goods and services, using an input-output matrix, led to the conclusion that increases in prices of final products will be so widely diffused over the spectrum of commodities that their impact on particular income groups will tend to follow the pattern of general consumption expenditures. Consequently the study distributes the total abatement costs to industry in any year in proportion to estimate of
total consumption exclusive of rent. The shape of the distribution of costs of industry is therefore as regressive as that of a general sales tax which.

Yohe (1979) and Forster (1984) have analyzed the backward incidence of pollution control i.e., it tried to determine the impact of more restrictive pollution control on the relative income shares of the factors of production. The study finds that stronger pollution controls do have backward incidence onto the other factors of production. The direction of that incidence depends on the relative factor intensity of the nonpolluting sector, and its magnitude depends upon (among other things) the relative price elasticities.

Robison (1985) presents a more recent study on this issue. Titled "Who pays for industrial pollution abatement?" this study presents direct measurements of the distribution of industrial pollution abatement costs across twenty income classes, as a percentage of consumers' income and expenditure, for the years 1973 and 1977. The central assumption of this study is that all firms pass off abatement costs into prices charged for goods or services produced, and therefore, that consumers ultimately bear the full burden of these costs. The assumption of full pass-through of abatement costs is a reasonable long-term assumption.

The two major findings of this study were: (1) Total abatement costs per dollar of output in 1977 range from nearly zero for some service sectors to about 5.4% for electricity; and the distribution of abatement costs across income classes is fairly regressive. Indirect payment for industrial pollution abatement range from 0.218% to 1.090% of income for the highest and lowest income groups, respectively, in 1977 and (2) contrary to earlier studies assumptions, the cost are also distributed regressively when measured as a percentage of expenditure, ranging from 0.423% for high income groups to 0.510% for low income groups in 1977. Industries with high abatement costs per dollar of final sales are generally the basic manufacturing industries, such as paper, plastics, agricultural fertilizers, chemicals, copper, ferrous metals, petroleum refining, and electrical utilities. Service sectors, such as eating and drinking places, retails trade, and finance and insurance, have relatively low abatement costs. Most final assembly industries such as motor vehicles, household appliances, and computers, fall in the middle range of abatement costs. This study provides a framework for measuring the distribution of industrial abatement costs and therefore moves one step towards a more accurate assessment of the distributional effects of any environmental policy.
Section 3.4
The Employment Impact of Pollution Control Activities

Though pollution control regulations imposed on industries can have numerous effects as seen in the previous section the aspect that has received maximum public attention is the possibility of a job versus environment trade-off. The visibility and emotion associated with potential job loss make it a crucial issue in ongoing policy debates. In the US the total spending to protect the environment in 1992 was 2.8% of the GDP and chief among the perceived costs of regulation is the loss of employment. At the individual firm or plant level, business and labor experts typically argue that environmental regulation increases a company’s production costs and puts upward pressure on prices. Price increases, in turn, result in a loss of sales and at least some reduction in plant-level employment. There is also the possibility that environmental regulation may be an important secondary factor in plant closure decisions. Potential job loss due to regulation is clearly an important phenomenon to understand since those who stand to lose their jobs as a result of the introduction of environmental policies could, if overlooked, prove to be a formidable political stumbling block.

Environmental regulations may also create jobs—sometimes in the same industry, or even in the same firm since pollution abatement activities themselves require labor input. In addition, environmental regulation may cause firms in a particular industry to shift production and jobs to different areas. Though labor unions and trade groups typically focus on gross job changes and the cost of rearranging workers within an industry, net job loss within an industry which recognizes all intra-industry employment changes associated with environmental regulation is also relevant.

For measuring aggregate environment-related employment effects we must look at environment-related activities which include a heterogeneous set of industrial and service activities which are carried out to clean-up existing processes and production (end-of-pipe equipment and technologies), treat water and effluents and to control air pollution. There is also a set of waste management and recycling technologies and services to deal with waste material and past environmental damage, and a growing range of environmental services such as research, design and engineering services. Similarly there is a group of activities which may be associated with environmental protection, although their primary
purpose is not environmental protection, e.g. energy saving, organic farming, sustainable forestry, or eco-tourism (OECD, 2004).

Thus environmental programs can be associated with both positive and negative employment effects. Preserving and improving the environment can create new or preserve existing jobs. On the other hand, as discussed in Section 3.3.2 environmental programs can force plants to close down, they can cause price increases, and thus lower demand and production and they might induce firms to shift new production capacity to foreign countries which have less stringent pollution control regulations. All this would have impact on employment.

The employment effects associated with environmental expenditure and environmental policies could be direct or indirect. Direct employment effects are the first-round changes in demand, output and employment induced by increased expenditures in environmental protection. Environmental expenditures also involve indirect (second and third-round) effects which include employment effects due to: the demand for intermediate goods and services induced by environmental expenditures; multiplier effects through increased wage incomes generating further demand and employment; relative wage and price effects; and displacement effects due to the diversion of regular investments by pollution control investments. While the direct effects emerge relatively rapidly, indirect effects take much longer to work their way through the economy. A part of the employment impact is usually transitory. In fact experience shows that only a proportion of the employment effects can be expected to endure over even the medium-term.

Markandya (2000) analyses the environment–employment trade-offs in a transition economy. He points out that in developing countries and transition economies often the persons made unemployed will not automatically be employed elsewhere in the economy. There is structural unemployment in the system and the assumption of full employment of resources cannot be maintained. He also mathematically derives environment-employment trade-offs.

Berck and Hoffmann (2002) have made a study of five basic approaches to evaluating the effect of any environmental policy action on employment. These are: (1) Supply and demand analysis of the affected sector; (2) Partial equilibrium analysis of multiple markets; (3) Fixed-price, general equilibrium simulations (input-output (I-O) and social accounting matrix (SAM) multiplier models); (4) Non-linear, CGE models; and (5)
Econometric estimation of the adjustment process, particularly time series analysis. These five approaches differ in the way they capture adjustment and economy wide interactions.

By definition, single and multi-market analysis do not access economy-wide impacts and will underestimate effects if used in situations where sectors other than those modeled are affected. Depending on model specification and the nature of the data set, these models estimate either short-run or long-run demand. I-O, SAM and CGE models represent a continuum of closely related models. I-O and SAM models provide an upper-bound on employment impacts because their Leontief production functions do not allow for adjustment through factor substitution. They can be thought of as simulating very short-run adjustment. CGE models allow for factor substitution in response to changes in relative price. They simulate labor movement between industries and therefore impacts on labor income and include migration or labor force participation equations that allow aggregate employment to change in response to changes in wages. Since CGE models allow capital to be substituted for other inputs, they represent long-run equilibria. In theory, the choice among these alternatives should be dictated by the type and scale of the action being evaluated. In practice, the cost of the evaluation, the time available to complete the analysis, and the convenience of using the tools at hand often dictate the method used. The employment impacts of environmental policies can be evaluated using a number of econometric models like HERMES, QUEST, LIFT and PANTA etc (Markandya 2000).

Doeleman (1992), in analyzing employment concerns and environmental policy, suggests that whilst environmental policy will bring both job losses and job gains the overall evidence suggests a net gain in employment. This is because the general finding is that the economic sectors most likely to be affected by environmental policy are relatively capital intensive like energy, chemicals, mining and exports. By contrast, the economic activities likely to benefit from environmental policy are relatively labour intensive, e.g. sewerage schemes, rural rehabilitation, energy conservation, recycling etc. Theoretically however it is difficult to be categorical on this point.

Pearce (1991) suggested that environmental taxation could lead to a double dividend as they would not only produce improvements in the environment, but also generate substantial amounts of revenue. This revenue would allow governments to reduce the rates of other, distorting, taxes and hence improve economic efficiency. In a situation
with involuntary unemployment, employment will only increase if the environmental
taxes that partially replace existing taxes result in an increased demand for labour. Most
applied models indicate that an employment dividend is possible when the revenues are
recycled in the form of lower labour taxation, and particularly in the form of reduced
payroll taxes.

Berman and Bui (2001) point out that estimating the effects of environmental regulation
is difficult for a number of reasons. They have worked out the estimating equations in
order to determine demand for labour in response to environmental regulation.
Regulations are expected to affect labour demand by affecting the output elasticity of
labour demand and the marginal rates of technical substitution between abatement
activity and labour. Costs incurred to comply with environmental regulation i.e. pollution
abatement capital investment and abatement costs are treated as ‘quasi-fixed’. Labour,
materials and regular capital are the variable factors. They assume a cost-minimizing firm
operating in perfectly competitive markets. There are J variable inputs and K ‘quasi­
fixed’ inputs. The variable cost function is:

\[
CV = H(Y, P_1, \ldots, P_J, Z_1, \ldots, Z_K)
\]

Where \( Y \) is output, the \( P_j \) are price of variable inputs, and \( Z_j \) are quantities of quasi-fixed
inputs. Profit maximization implies the first order condition that yields demand for the
variable input labour, \( L \), as a function of output, quantities of the other inputs, and prices,
which we approximate by the linear equation.

\[
L = \alpha + \rho_Y Y + \sum_{k=1}^{K} \beta_k Z_k + \sum_{j=1}^{J} \gamma_j P_j
\]

The reduced form effect of regulation (R) on labor demand can be written:

\[
L = \delta + \mu R
\]

The effects of regulation on employment are through the mechanism.

\[
dL/dR = \rho_Y [dY/dR] + \sum_{k=1}^{K} [\beta_k dZ_k/dR] + \sum_{j=1}^{J} \gamma_j dP_j/dR = \mu
\]

If input markets are large and competitive, regulatory change will have no effect on input
prices so the final term in (3.21) will disappear, leaving the first two. The first term
reflects the effect of regulation on demand for variable factors through its effect on
output. This output effect of environmental regulation is widely believed to be negative
(though theory gives no clear prediction: if compliance is achieved through an investment
that reduced marginal costs, \( dY/dR \) could be positive). The second term reflects the effect
of regulation on demand for variable factors through its effect on demand for quasi-fixed
abatement activities, \( Z \), and the marginal rates of technical substitution between

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abatement and variable factors. The change in demand for abatement activity due to an increase in regulation $dZ/dR$, must be positive. The signs of the $\beta_k$ which reflect whether abatement activity and labor are complements or substitutes are not known a priori. Abatement technologies fall into two general categories, 'end-of-pipe' and 'changes in processes'. End-of-pipe technologies such as scrubbers and precipitators remove pollutants from existing discharge streams before their release into the environment and probably complement labour, particularly production labour. Improvements in production process, such as the installation of more efficient boilers which operate at lower levels of emissions, often reduce demand for production workers due to a general skill-bias of technological change. Hence the sign of the $\beta_k$ are ambiguous, which is the main reason that the sign of $\mu$ the employment effect of regulation, cannot be predicted from theory alone.

Berman and Bui have gathered detailed micro data on local air pollution regulations in a specific region. They focus on the manufacturing and directly estimate the effects of local air pollution regulations using a quantitative approach that includes comparison plants in the same precisely defined industry. The study finds that while regulations do impose large costs they have a limited effect on employment. The authors suggest that the small employment effects are due to the combination of three factors: (a) regulations apply disproportionately to capital-intensive plants with relatively little employment; (b) these plants sell to local markets where competitors are subject to the same regulations, so that regulations do not decrease sales very much; and (c) abatement inputs complement employment.

In an ingenious study on the employment effects of environmental regulations Morgenstern et al (1998) have decomposed the employment consequences of increased spending into distinct components using reported environmental spending as a measure of regulation. By looking across several industries and decomposing employment effects into distinct supply-side and demand-side components, the study by Morgenstern et al is able to look for patterns of employment changes when environmental regulations are tightened. Employment will adjust to both a rearrangement of production activities as well as a potential output contraction. The more traditional concern is that as production costs rise in response to increased environmental regulation, output prices will rise, quantity demanded will fall, and plants will reduce employment levels. The extent of this effect depends on the cost increase passed on to consumers as well as the demand.
elasticity of industry output. These two features may not be independent: industries facing elastic output demand due to stiff competition may prove more adept at lowering the cost of environmental compliance. Less competitive industries with inelastic demand may be less concerned about cost increases associated with regulation. This is referred to as the demand effect.

On the production side, there are two arguments for increased employment. First, environmental regulation usually raises production costs. If production costs rise, more inputs, including labor, are used to produce the same amount of output. This is referred to as the cost effect. Second, environmental activities may be more labor intensive than conventional production. For example, cleaner operations may involve more inspection and maintenance activities, or reduced use of fuel and materials. In both instances, the amount of labor per dollar of output will rise. This argument obviously can go the other way: cleaner operations could involve automation and less employment, for example. This is referred to as a factor shift.

Thus the effect of increased regulation is considered in three distinct steps. Plant-level data is used to estimate a cost function that allows them to assess the first two components. These estimates are combined with estimates of industry wide demand elasticities to calculate the third component as well as the overall change in employment associated with increases in reported environmental spending. Estimates are developed for four heavily polluting industries (pulp and paper, plastics, petroleum, and steel).

This study finds that there are strong positive employment effects in industries where environmental activities are relatively labor intensive and where demand is relatively inelastic, such as plastics and petroleum. In others, where labor already represents a large share of production costs and where demand is more elastic, such as steel and pulp and paper, there is little evidence of a significant employment consequence either way and the authors suggest that a million dollars of additional environmental expenditure is associated with an insignificant change in employment.

Various studies in the past have also underlined the growing importance of jobs in the EGS (Environmental Goods and Services) sector. Projections in Japan for 2010, for instance, see private environment-related employment increasing by about 25%. Assuming a more moderate 10% increase in the public sector, about 1 million jobs would be directly related to environmental activities by 2010. Overall environment-related
activities in the business, public and third sector have become a significant source of employment in a number of OECD member countries. Existing data indicate that the direct employment effects in the EGS sector alone vary between 0.4 and 3.0 per cent of total employment (between 1 and 1.5 per cent in the majority of countries).

Thus theory alone yields an ambiguous prediction of the over-all employment effects of environmental regulation. Existing empirical studies likewise yielded mixed results on these employment effects but on the whole there appears to be an emerging consensus on three main points:

(1) At the economy wide level there is no trade-off between environmental protection and employment
(2) The number of workers laid off due primarily to environmental regulations has been quite small
(3) Few firms relocate to regions that have lax environmental regulations.

Section 3.5
Concluding Remarks
In this chapter we have covered a broad spectrum of issues involved in estimating the incidence of the benefits and costs associated with environmental protection and pollution control programs and carried out a review of the existing literature on the methodological issues associated with these aspects. In Chapter 4 we look at the dimensions of the water pollution problem of Yamuna River in Delhi and in Chapter 5 we have given a detailed description of the water pollution control programs that are being carried out in Delhi to tackle the problem. In Chapters 6-8 we use appropriate methodologies and statistical methods to estimate the incidence of the costs and benefits of the water pollution control programs being carried out for the protection and restoration of the Yamuna River in Delhi.