Chapter 2
Review of Literature

Environmental regulations that lead to the reduction in the emissions of air and water pollutants clearly generate overall welfare gains for the people. However depending upon the distribution of costs and benefits, environmental policies may be regressive or progressive. A regressive distributional impact implies that net benefits are larger for individuals with high than those with lower incomes. There are a number of conceptual as well as practical issues involved in the study of the distributional impact of environmental change. This chapter deals with those issues.

Section 2.1
Conceptual Issues

The conceptual issues covered in this section are: the meaning and measurement of the benefits and costs of environmental change; the choice between WTP and WTA (willingness to accept) measures for valuing environmental resources; the significance of income elasticity of demand for environmental goods; and the possible trade-off between equity and efficiency in environmental policy.

2.1.1 Benefits of environmental policies

Since pollution and other forms of environmental degradation cause material damage, loss of health, reduced recreational benefits, and decrease in aesthetic pleasure as well as decline in the value of the unexperienced environment; the benefits from environmental policies aimed at controlling pollution of natural resources and degradation of the environment also cover a wide spectrum. These can be broadly classified into user benefits and non user benefits. User benefits are derived from the direct or indirect use of environmental resources and amenities. Nonuser benefits, also called passive use values, include existence value, bequest value and option value (discussed further in Section 3.1.1).

Broadly speaking the benefits from water pollution abatement would include (a) decline in the incidence of water borne diseases; (b) enhanced aesthetic qualities like improvement in taste, odor and appearance of the water bodies; (c) avoided costs of averting behavior like expenditure incurred on water purification equipment for homes.
etc.; (d) avoided materials damage e.g. reduction in the corrosiveness of water would lead
to decline in pipe breakages, lesser damage to meters and water facilities and therefore
reduced water loss from leaks etc.; (e) lower production costs for food processing and
other manufacturing establishments who treat water themselves to improve its quality; (f)
nonuse benefits and information benefits. Monetary valuation of these benefits from
improvement in the quality of fresh water are necessary for the proper evaluation of
government projects and the laws and regulations intended to improve a city's rivers and
lakes.

While the measurement of control costs is itself no simple task, environmental
economists have turned most of their attention to the estimation of benefits (Cropper and
Oates 1992) and the theory and performance of the methods used for nonmarket valuation
have been major preoccupations of environmental economists over the past few decades.
These techniques fall into two categories: indirect market methods and direct questioning
approaches. We take a look at these estimation techniques in Section 3.1.

Environmental resources like clean air, clean water, and biological diversity are public
goods in the sense that they exhibit consumption indivisibilities and, additionally, are
accessible to all. The free market mechanism, that ensures efficient allocation of
resources and optimum production and consumption of private goods, fails in the case of
environmental resources. The inefficiency clearly results from the free-rider problem.
The problem of the pollution of environmental resources as well as the source of the basic
economic principles of environmental policy is found in the theory of externalities.

An externality exists whenever the welfare of some agent, either a firm or a household
depends not only on his or her activities, but also on activities under the control of some
other agent. This causes a divergence between the private and social marginal costs and
benefits. If there is market allocation of commodities causing pollution externalities, it
results in: (a) too large an output of the commodity (b) too much pollution produced (c)
too low prices of products responsible for pollution (d) lack of incentives to search for
cleaner technologies for producing the commodity and (e) recycle and reuse of the
polluting substances are discouraged because release into the environment is so
inefficiently cheap (Tietenberg 1994).

A simple mathematical exposition of the theory for modeling pollution and determining
the first-order conditions for a Pareto-efficient outcome is given in Cropper and Oates
The standard approach in the environmental economics literature characterizes pollution as **public bad** that results from the waste discharges associated with the production of private goods. In deriving the first-order conditions characterizing a competitive market equilibrium we find that competitive firms with free access to environmental resources will continue to engage in polluting activities until the marginal return is zero. The policy implication of this result is clear. Polluting agents need to be confronted with a “price” equal to the marginal external cost of their polluting activities to induce them to internalize at the margin the full social costs of their pursuits. Such a price incentive can take the form of the “Pigouvian tax,” i.e., a levy on the polluting agent equal to marginal social damage. The unit tax or effluent fee must be attached directly to the polluting activity i.e., per unit of waste emissions into the environment, and not to some related output or input.

### 2.1.2 Costs of environmental policies

The real and perceived economic costs associated with environmental protection are easily the greatest obstacles to cleaner air and water, improved preservation of ecosystems and biodiversity, and slower depletion of natural resources. The accurate measurement of costs is challenging conceptually in terms of defining what we include as costs and also in practice. Studies use very different methodological approaches to estimate these costs. Popular debate over environmental protection often centers on the out-of-pocket expenditures incurred by firms, governments, and households or other negative consequences directly associated with pollution reduction (Pizer and Koop 2003).

*The real cost of environmental measures is, ultimately, the additional resources needed to run the economy in a less polluting way.* Whether the industry is required to pay a pollution tax or is simply ordered to install pollution control equipment, its costs will increase, and this puts an upward pressure on the prices of its products. From the point of view of effective environmental protection, these increased prices are desirable: they encourage shift in purchases away from the products of polluting industries to goods and services whose production is less harmful to the environment. However this usually sets in motion a complex chain of price increases, and no one knows precisely what the final price pattern will look like or who will bear what share of the burden. This is particularly the case where the product whose price has risen is an intermediate good rather than a final consumer good. As costs and prices rise, wages too are likely to be affected in a complex pattern. Wages in the industry whose operations are directly affected by an
environmental program may not go up due to a contraction in output. Wages in other industries may however go up due to the increase in the general price level and therefore higher cost of living of the laborers. This complicated pattern of wage changes will in turn influence the prices of goods and services (Baumol and Oates 1988).

A distinction may be drawn between transitional and continuing costs attached to pollution control programs. The most striking feature of transition costs is the very uneven pattern of their incidence. Some industries are hit much harder by the environmental regulations than the others. The same is true of the geographical distribution of the effects. Both high and low salaried workers may find their jobs threatened. But when environmental interests halt, or slow down, the construction of a factory or refinery, the protests about lost jobs usually come from blue collared workers or the jobless. Both opinion polls and election results confirm that there is an income related difference in attitudes towards environmental legislation.

The overwhelming focal point of the literature to date has been measurement of the direct compliance cost to firms. Environmental policies applied to firms either directly force changes in production methods (command-and-control policies) or provide incentives to do so by changing prices (market-based policies). Various instruments of environmental regulation by the government and their relative merits are described in Section 3.2. The direct compliance cost is the change in production costs entailed by the environmental policy. This cost will depend on the particular technological alternatives available to the firm. Engineering models of pollution abatement and surveys of business and governments can be used to provide direct estimates of pollution control costs and when the study is retrospective, historical data can be used to estimate relationships between observed or estimated emissions and production costs. In the case of prospective cost analysis often the only way to estimate costs is to pose the question to engineers familiar with abatement technology. The U.S. Environmental Protection Agency (EPA), for example, in its regulatory impact analysis of proposed regulations usually estimates the costs of alternative policies based on linear programming models of firm response using specific technology options enumerated by engineering experts (Pizer and Koop 2003).

However economists have pointed out the existence of additional unaccounted burdens associated with environmental protection. E.g. regulations that required large capital expenditure could crowd out other investments in newer and more productive equipment.
There is also a general concern that environmental regulation reduces operational flexibility and reduces productivity. These aspects of the costs of environmental regulations, i.e., their impact on industrial productivity, location of industries, international competitiveness, overall economic growth etc, are discussed in Section 3.3.

Cropper and Oates (1992) point out that in the short run the effect of ignoring these adjustments is to overstate the costs of environmental regulations. Simple abatement expenditures overstate the loss in firms' profits if firms can pass on part of their cost increase to consumers. Consumers in turn can avoid some of the welfare effects of price increases of ‘dirty’ goods by substituting ‘clean’ goods for ‘dirty’ ones. When environmental regulations affect sectors like electricity production, that are important producers of intermediate goods, it may be important to measure the impacts that the regulations have throughout the economy and CGE models may be needed to measure correctly the social costs of environmental regulations.

E.g. Hazilla and Koop (1990) used an econometrically estimated a CGE model of the U.S. economy to compute the social costs of Clean Water and Clean Air Acts as implemented in 1981. The effects of these regulations on firms are modeled as an upward shift in the firms’ cost functions, to which the firms can adjust by altering their choice of inputs and outputs. The total social cost obtained by this approach was $28.3 billion whereas the EPA estimated the cost of complying with these regulations as $42.5 billion (in year 1981). In the long run, however, the social costs of environmental regulations are expected to exceed simple expenditure estimates because of the effects of decrease in income on savings and investment. In their analysis of the effects of environmental regulations on U.S. economic growth Jorgenson and Wilcoxen (1990) measured this effect. Using a CGE model of the U.S. economy they estimated that mandated pollution controls reduced the rate of GNP growth by 0.191 % p.a. over the period 1973-85.

Some literature can also be found on negative costs of environmental regulations. E.g. Porter and van der Linde (1995) argued that the firms are not always operating efficiently and that environmental regulations can lead firms to recognize and correct these inefficiencies and benefit through increased productivity. Most economists are, however, skeptical of the possibility of profitable environmental improvements. As Palmer et al (1995) said that despite the presence of some cost-saving offsets, environmental regulation generally must increase costs and reduce profits.
Apart from pollution control costs borne by industries, costs are also borne by governments directly for solid waste disposal, water and sewage treatment etc. These cost estimates can usually be compiled from appropriate government reports. In addition to direct expenditures on environmental protection the government also spends resources on enforcement and monitoring. These costs are typically small (about 2%) compared with national expenditures on environmental protection (Vogan 1996).

The benefits and costs of a proposed environmental regulation are necessarily uncertain because it cannot be known in advance how individuals and institutions will respond to altered legal conditions. The common wisdom seems to be that the costs of environmental regulation are systematically overestimated (e.g. Porter and van der Linde 1995). Before a regulation is adopted, information about response options and costs is asymmetrically distributed. In many cases the potentially regulated parties will have better information about alternatives for meeting requirements than will regulatory agencies and advocacy groups. This asymmetry provides an opportunity for firms that expect to bear compliance costs to exaggerate the costs of environmental regulations (Hammit 2000). Even good-faith estimates may be biased upward if firms do not fully anticipate costs saving measures they may discover once they direct resources to compliance. Regulatory agencies may also overestimate costs if in preparing a record an agency bases its cost estimates on the use of currently available technology, even if lower-cost alternatives are foreseeable.

Thus this section gives an overall view of the various aspects involved in assessing the costs of environmental policies. Sections 3.2 and 3.3 discuss different aspects of the costs of environmental policies incurred by government and the industrial sector respectively. The employment impact of environmental regulations is explored in Section 3.4.

2.1.3 WTP vs. WTA

Because utility is not directly observable environmental values have typically been defined in terms of the maximum amount of money that an individual is willing to pay to avoid a proposed loss or the minimum amount of compensation that he or she must be paid to accept it. Both measures refer to changes in the area under a person's demands curve for the good or activity (Starting with Hicks 1943). The usual presumption is that, aside from small differences due to income or wealth effects, estimates of value will be invariant between the two measures. As Freeman (1979) said, "Practically speaking, it
does to appear to make much difference which definition is accepted”. The work of Willig (1976) has been especially influential in confirming the anticipated equivalence of WTP and CD (compensation demanded) estimates of welfare. Willig not only extended the analysis of Hicks and other early welfare theorists, but he was also able to calculate strict bounds for estimating the magnitude of differences in competing measures of consumer’s surplus. Willig’s calculations show that the differences, in most cases, will be less than 10%.

In carrying out CV studies typically, WTP measures are chosen because they seem to correspond more closely to most of the market exchanges people make and therefore involve people in a more familiar transaction. However over the years various investigators have reported substantial differences in people’s responses to payment and compensation questions in the context of potential economic losses; with compensation-based measures generally exceeding payment-based measures by a factor of 3 or more. E.g. Rowe et. al. (1980) found that the mean level of compensation exceeded the indicated average WTP to maintain various levels of air quality by amounts ranging from about five to sixteen times. Because the choice and interpretation of these measures is fundamental to the prediction of anticipated changes in social welfare, the dispute holds important implications for environmental policy.

Gregory (1986) points out that the belief of equivalence between WTP and WTA has persisted even in the face of a growing number of empirical studies suggesting that differences between payment and compensation measures of economic loss are both frequent and substantial. The observed differences between WTP and CD measures of economic value are usually dismissed on the following grounds: (a) Consumer’s surplus is a normative construct and does not require proof of descriptive validity using empirical tests. (b) Measurement errors are endemic to CV and the variations in WTP and CD are due to weakness in the survey instrument. (c) Since the situations posed in a CV survey are hypothetical the responses that they elicit might not be trustworthy. The participants may be unfamiliar with evaluating the goods in question or they may find the survey context artificial.

Knetsch and Sinden (1984) tested the contention that the large variation in the value measurements recorded was due to the hypothetical nature of the alternatives posed to the respondents in a series of experiments where outcomes were dependent on their making
an actual cash payment or receiving real and immediate monetary compensation. Even these suggested a wide disparity between the two bases for measuring economic values. Coursey et al (1987) did claim that after a series of learning trials using Vickery auction they derived WTA and WTP values that were close but Knetsch and Sinden (1987) have given reasons why these experiments are not convincing.

Psychologists have long argued that people are much more averse to a loss than attracted to an equivalent gain. This behavior, termed loss aversion, has been formalized by Kahneman and Tversky (1979) in their reformulation of expected utility theory called prospect theory. In prospect theory the utility function is replaced by a value function that evaluates changes in income from the current level. This implies that a kink occurs in relationship between utility and income at the initial income or reference point and that the slope of the utility function for losses in income is steeper than it is for gains. In indifference curves the rate of commodity substitution at a point on an indifference curve is assumed to be the same for movements in either direction. The recent empirical findings of asymmetric evaluation of gains and losses would however imply that the presumed reversibility may not accurately reflect preferences. Knetsch (1989) conducted tests of the reversibility of indifference curves and found that in every case, people exhibited valuations that varied systematically and substantially with the initial reference entitlement and the direction of exchange offers.

Thaler (1980) labeled the increased value of a good to an individual when the good becomes part of the individual’s endowment the endowment effect. This effect is a manifestation of loss aversion, the generalization that losses are weighted substantially more than commensurate gains in the evaluation of prospects and trades (Kahneman and Tversky 1979). Kahneman et al (1990) also present evidence of what may be called an instant endowment effect: the value that an individual assigns to any objects appears to increase substantially as soon as that individual is given the object.

Hanemann (1991) reconsidered the work of Randal and Stoll (1980) on value measures given changed quantities of the good. Hanemann proposed that the difference between WTP and WTA is due to the existence of a lack of substitutes for the commodity being valued. Hanemann shows that when there is a perfect substitute for the good being valued, WTP and WTA will be equal in the absence of any income effect. Adamowicz et al (1993) have also investigated the role of substitutes in the difference between WTP and...
WTA measures of value as suggested by Hanemann and found that though the availability of substitutes is a significant factor in determining the disparity between WTA and WTP for a good, it was not sufficient to erase the significant differences between these two measures.

Boyce et al (1992) argued that kinked or inflected indifference curves between the commodity and other money expenditures results since intrinsic values are included in WTA measures of value but (at least partially) excluded from WTP measures of value. Attributes that have been associated with intrinsic values include nonsubstitutibility (uniqueness), irreversibility (replication of the specific commodity is impossible if it is destroyed), feelings of sentimentality, and a sense of moral obligation. Amiran and Hagen (2003) have shown that in the case of exogenous changes in the quantities of public goods, large deviations of WTA from WTP, including infinite WTA, can arise. All that is required is the assumption of asymptotically bounded utility which in fact avoids a rather implausible implication of unbounded utility functions, namely that a sufficient quantity of any one good can substitute for the loss of nearly the entire quantity of all other goods. Mansfield (1999) demonstrated that the alternative explanations for the disparity between WTP and WTA can be tested within a single analytical framework. Starting form the assumption that individuals make constrained utility maximizing choices, equations are developed for WTP and WTA that yield simple parametric tests of various hypotheses.

Thus the difference between WTP and WTA has thus been widely studied through both theory and experiments. However as pointed out by Horowitz and McConnell (2002) although large WTA/WTP ratios are well documented, the findings do not seem to have had much of an effect on either economic models or discussion of policy design. In spite of the persistent differences in WTP and WTA values both losses and gains are usually assessed by the payment measure on the pragmatic grounds that “Generally speaking, willingness to pay is easier to estimate than required compensations” (Kneese 1985).

The National Oceanic and Atmospheric Administration Panel on Contingent Valuation (NOAA, 1993) also recommend that researchers ask only WTP questions though this is inappropriate if WTP and WTA are not close in value. Gowdy (2004) has pointed out that even according to the authors of the influential NOAA panel reports (Arrow et al. 1993), “the conceptually correct measure of lost passive-use value for environmental damage that has already occurred is the minimum amount of compensation that each affected
individual would be willing to accept.” In spite of this, the NOAA panel recommends the WTP measure on the grounds that “the WTP format should be used instead of the CD because the former is the conservative choice” and also due to “the cause that respondents would give unrealistically high answers to such (WTA) questions”.

As Knetsch (1990) has pointed out, a large difference between WTA and WTP can have potent effects on environmental policy. These occur when the appropriate welfare measure is WTA (because, in most instances, environmental quality can only deteriorate) but policy analysts use the WTP measure for valuing environmental change. This may well lead to misleading assessments, inappropriate policy decisions, and resources misallocation. If environmental degradation or a contemplated change imposes loss on individuals, this usual practice of using the payment measure will be lead to large understatement of the welfare changes. Environmental control standards are, at least in principle, aimed at levels where marginal gains are matched by the marginal costs of further restraint. Levels based on WTP measures of loss will likely differ from ones chosen on the basis of CD. More damaging activities and fewer preventive measures will be undertaken with the use of WTP measure of loss than would be encouraged with the more appropriate CD measure.

One reason for the NOAA Panel’s recommendation of the use of WTP format is purely practical. Although the WTA figures are higher, they are also more varied with some individuals stating extremely high WTAs, and others stating that no sum would compensate them for the loss. Unless one wishes to give such individuals the right effectively to veto any change, it is hard to see how WTA can be applied for loss of amenity.

2.1.4 Income elasticity of demand for environmental goods
To examine the issue of distribution of benefits of environmental programs theoretically it is necessary to consider how the demand for environmental quality is likely to vary with income. The general belief about the pro rich distribution of environmental benefits mentioned in Section 1.1 is based on the argument that the income elasticity of demand for an environmental good is positive (e.g. Baumol 1974, Baumol and Oates 1988).

Freeman (1972) addressed the question of patterns of distribution of benefits associated with environmental pollution abatement programs. He proposed that “if the benefits are rising faster than income, the elasticity is greater than one and the distribution is strongly
pro-rich; similarly if the benefits are a decreasing function of income, an elastic benefit schedule is strongly pro-poor. On the other hand if benefits are rising at a rate slower than that of income, the elasticity is less than one and the distribution is mildly pro-rich; similarly if the benefits are a decreasing function of income, an inelastic benefit schedule is mildly pro-poor. Thus like Baumol, Freeman too referred to income elasticities greater than zero as being evidence of a pro-rich distribution since benefits rise with income.

Dorfman (1977) got similar results. The paper quotes a public opinion study done by the National Wildlife Federation in 1969 in which a referendum question was asked about WTA a $X increase in a family’s total expenses for the clean up of the natural environment. The paper draws the conclusion that clean environment is a superior good and that any stated price more people at higher income levels are willingness to buy it than at lower income”. Kristrom and Riera (1996) pointed out that a significant problem with this survey was the vagueness of the valuation question.

Kristrom and Riera (1996) clarified the difference between income elasticity of demand for environmental goods and income elasticity of environmental values. They defined the income elasticity of environmental improvements as follows: If we define the share of WTP over income \( y \) as 
\[
s = \frac{\text{WTP}(y)}{y},
\]
then by differentiation we find that 
\[
\frac{ds}{dy} > 0
\]
if 
\[
\frac{\delta \text{WTP}}{\delta y} \cdot \frac{y}{\text{WTP}} > 1.
\]
Thus if income elasticity of WTP for environmental improvement is greater than one, the share of WTP allocated to the environment increases with income. Numerous empirical studies suggest that WTP is an increasing function of income i.e. those with more income tend to express higher WTPs for environmental improvements than those with less income; i.e. environmental improvements are typically normal goods.

Flores and Carson (1997) also emphasized on making the distinction between income elasticity of demand for environmental goods and income elasticity of environmental values when discussing the distributional impacts of environmental policies, it is important to make this The latter are the appropriate concept for understanding the distributional impacts of policies that affect quantity rationed collective goods. Intuitively there is no reason a good with an income elasticity of demand greater than one (a luxury) may not have an income elasticity of WTP that is substantially less than one. The reverse can also hold true, whereby environmental projects exhibit larger income elasticities for WTP than for demand. Understanding this distinction is the first step in resolving the debate over the distributional impacts of environmental policies.
One expects to see a positive relationship between income and WTP if the good being valued is a “normal” good. A frequently made claim, for which there is surprisingly little empirical support, is that most environmental goods are “luxury” goods. If this were the case, one would expect the income elasticity to be greater than one. However, the usual empirical result from CV studies is to find a positive income elasticity of WTP substantially less than one for environmental commodities. This empirical result has even been cited as evidence that CV results are theoretically deficient (McFadden, 1994).

In analyzing the relationship between income inequality and the urban environment in the context of water and sanitation sector, Johnstone (1997) says that preferences for environmental quality are not income elastic across a wide spectrum of indicators particularly for those aspects of environmental quality which are more closely identified with basic needs rather than luxury goods like potable water and sanitation facilities. Bahl and Linn (1992) review a number of studies of water demand which find estimated income elasticities ranging from 0.0 to 0.4 for a number of different countries. Cross-sectional evidence from different countries also indicates that the income-elasticity of water consumption is in the region of 0.3 (Anderson and Cavendish, 1993). Thus, it is quite likely that water demand rises less than proportionately with income. Given the reliance of poorer households on surface water for many basic needs such as cooking and washing, it is quite likely that demand for water quality is also income-inelastic. Johnstone, therefore, says that there is good reason to believe that lower-income groups are willing to pay - or rather are required to pay - a higher proportion of their incomes for water and sanitation services than higher income groups.

Carson et al (2001) have clarified that the problem is that the terms necessary (e.g., normal but not luxury) and luxury are defined in terms of the income elasticities of demand; a measure based on varying quantity, not in terms of the income elasticities of WTP, a measure based upon holding the quantity fixed. The two types of income elasticities are fundamentally different. The income elasticity of demand shows how the quantity demanded increases as income increases while the income elasticity of WTP looks at how WTP for a fixed quantity of the good changes as income increases. The paper by Flores and Carson (1997) shows that the two income elasticities are functionally related using the concept of a shadow or virtual price that responds to changes in the levels of rationed goods. Results show that for any fixed value of the income elasticity of demand, the income elasticity of WTP can differ significantly in magnitude and even
sign. Thus, a good which is a luxury good in a demand sense may have a WTP income elasticity which is less than zero, between zero and one, or greater than one.

As Flores and Carson (1997) put it, “the economic intuition behind the empirical results on income and environmental values found in the existing literature can be put simply: the rich man may buy proportionately more loaves of bread than his poorer brother, but this does not imply that he is willing to pay proportionately more for the same loaf”.

2.1.5 Equity-efficiency trade off?

Most of us who are interested in the equality-efficiency trade-off derive their interest from a view that more equality than currently characterizes the society would be desirable from an ethical viewpoint, but only if it could be engendered without large efficiency costs. Income is the parameter that is usually taken to measure the degree of inequality since it is not possible to make interpersonal comparisons of welfare.

The urban environment is ‘public’ in the sense that residents of an urban area share to some extent the same environmental system. Although there is likely to be some differentiation between levels of pollution in different parts of the same urban area, it is generally true that residents will tend to share exposure to the same environmental bads. Johnstone (1997) points out that the “public” nature of urban environmental goods and bads has important consequences for the relationship between distribution and efficiency. In the case of public goods, the issues of distribution and efficiency can not be separated. This arises from the fact that for public goods an efficient level of provision requires that all residents freely choose to consume the same level. If, however, preferences for public goods differ systematically with income levels the marginal benefits of its provision will not be equal to the marginal costs for each and every resident. Thus, in cases where preferences differ systematically with incomes, distributional issues will play an important role in determining whether or not the level of provision of public goods is efficient. In that case decisions related to the level of provision of a public good become ethical questions. One group will necessarily be favored over another i.e., one group will receive benefits in excess of its costs and vice versa for the other group. Access to clean water and sanitation services is itself an important contributor to a household’s relative economic wealth. Okun (1988) estimates that poor unconnected households can pay up to 30% of their income on water, while connected rich households pay generally less than 2%.
The efficiency conditions derived in environmental economics, as in other fields of applied economics, are all founded on the criterion of Pareto optimality. Theoretically 'compensation' principles are applied to take care of the inequitable impact of public policies whereby a project is deemed to be worth undertaking if aggregate social welfare rises (Maler 1985). However, the compensation assumed in project evaluation is usually hypothetical which is justified either by asserting that over a cross-section of projects distributional issues will tend to be counterbalancing or by asserting that a separate agency i.e., the central government already addresses distributional issues by other means. In practice most projects apply unweighted utility functions (Kanninen & Kristrom, 1992). The combined effects of the use of potential compensation and equal weights in project evaluation means that equity and efficiency objectives are assumed to be separable and, more significantly, efficiency objectives are given priority over distributional concerns. Since demand for urban environmental goods tends to rise with income projects which tend to benefit richer households disproportionately will have a greater chance of being approved than those which benefit poorer households.

As pointed out by Baumol & Oates (1988), even if the pollution control measures are predominantly based on the efficiency criterion, efforts must be made to reduce the iniquitous effects of these measures to the extent possible through provision of adjustment assistance like unemployment compensation and retraining and relocation assistance. This can help in offsetting the heavy transition burdens. Modern theories of asymmetric information and incentives also provide numerous examples of equality-efficiency synergies (Putterman et al 1998). E.g. more equal distribution of wealth has a felicitous effect on educational investment choices. Higher levels of education and awareness change social behaviour which often has a positive impact on the quality of environmental resources. This in turn improves the overall economic efficiency.

**Section 2.2**

**Practical Issues**

When analyzing the distributional impact of large scale environmental protection programs like River action plans we must also address certain important practical issues. These are discussed in this section.
2.2.1 Imperfect Knowledge of Environmental Systems

An issue that is fundamental to all studies on the valuation of environmental costs and benefits is the uncertainties that are associated with environmental systems. Our information about the benefits from environmental protection is really quite limited. Ecosystems are stochastic, that is objectively (as opposed to subjectively) uncertain. Rainfall and weather patterns vary from year to year. Some years the streams will be high, in others they will be low. Some years there will be few atmospheric inversions and others they will be more frequent. Changes in animal populations similarly are determined by uncertain reproductive processes. Even the behavior of business firms (and consumers) may well be more accurately represented by probabilistic as opposed to deterministic models (Georgescu-Rogen 1958). In sum, we are subjectively uncertain about the shape of the objective probability distributions that generate our options.

Natural systems are also imperfectly reversible. The damage done by waste sometimes depends on the slowly decaying stock of pollution in the environment. This stock often cannot be diminished by policy measures taken later. Where we can modify these situations it is often possible to do so only slowly, imperfectly or at great cost. The bottom deposits in an estuary (which use up oxygen in the water) could only be removed by making very large expenditures for dredging. More young redwood trees can be planted, but the stock of mature ones cannot be expanded in our lifetimes. A restored system is never the same and always less valuable than one which has never been contaminated.

Such limitations on 'reversibility' enhance the value of policies designed to preserve options and avoid risks (Loomis et al 1999). If we guess wrong about the steady state population of eagles, the birds could become extinct. Unfortunately, the harm caused by a 'stock' of environmental bads may not become apparent until the stock has grown (irreversibly) to dangerous levels. The biological and chemical processes that link many environmental systems also exhibit relevant nonlinearities and nonconvexities. Such systems often have a substantial capacity to absorb shocks of less-than-critical magnitude but there are 'thresholds' that lead to nonlinear damage functions. The policy problems raised by such nonlinearities are considerable. It is not unusual to find multiple thresholds with increasing marginal damages as we push past some environmental threshold, then decreasing marginal damages until we push through yet another critical area where
marginal damages again increase. In water pollution, for example, this might happen as we successively lose additional species and uses (Baumol 1972).

The available analytical techniques for dealing with such complex systems leave many fundamental issues incompletely resolved. To ‘solve’ the environmental system for ‘optimal’ outcomes is quite impossible that is neither supported by data nor theory (Harsanyi 1955).

2.2.2 The Application of CVM in Developing Countries

Till about a decade back only a very few CV studies had been conducted in the developing countries. At the time the conventional wisdom was that the problems associated with posing hypothetical questions to low-income, perhaps illiterate respondents was so overwhelming that one should not even try. However over the years there has been a surge of activity in the use of CVM to measure the value of environmental and health-related outcomes from projects, policies, and regulations in developing countries (Whittington 1998). Many have, however, expressed concerns about the quality of these CV studies due to the circumstances peculiar to the developing countries.

Whittington (1998) gives a detailed description of the various difficulties a researcher often faces in administering a CV survey in a developing country. These include difficulty of explaining to government officials and interviewers what the study is about and the concepts of economic value and ‘maximum WTP’ or ‘WTA a minimum compensation’; the difficulty in communicating to interviewers the reason behind split sample techniques in referendum elicitation procedure; confusions about ability and willingness to pay of the respondents and also the difficulty of getting well trained enumerators. Whittington (2002) has pointed out that the reasons behind the poor quality of many CV studies conducted in developing countries are (a) the CV surveys themselves are often poorly administered and executed, (b) CV scenarios are often very poorly crafted and lack clarity, and (c) few CV studies are designed to test whether the assumptions that the researcher made were the right ones, and whether the results are robust with respect to simple variations in research design and survey methods.

Whittington et al (1992) found that respondents who were allowed time to evaluate the proposed water system bid significantly less than those who did not have that time. The authors believe that time to think are especially important in the CV studies conducted in
developing countries. This is because there are substantial differences in the education and demographic characteristics of a typical respondent in a developing country and an average consumer in a developed country. Svendsäter (2003) has pointed out that the broader issue is how people interpret, understand and make sense of CV questions. While the hypothetical market may make perfect sense to the researcher, it is not necessarily perceived in the same way by lay people. These concerns are even more relevant in a developing country context. The political factors involved in the provision of many government supplied goods and services in developing countries make crafting a CV scenario and valuation question(s) even more complicated. To engage the respondent and to capture the political reality of a local situation, CV researchers often need to have a carefully nuanced sense of the political forces at play before trying to draft a CV scenario.

Whittington (2004) points out that that CV surveys also have the potential to confuse or mislead respondents caused by inaccurate provision of background information in the CV scenario, the description of the hypothetical market, and the use of the referendum elicitation procedure and split-sample experiments commonly used by CV researchers. The individual respondent who is confused or misled may take actions that could harm him or members of his household. The spread of misinformation or confusion among the study population could also influence the policy process itself in unintended and unfortunate ways. The difficulties of cross-cultural communication add to the complexity of conducting CV surveys in developing countries.

In spite of these reservations CVM is being increasingly used in developing countries. Choe et al (1996) for example, carried out a CV study to estimate the economic benefits of surface water quality improvements in Davao, Philippines. The authors found evidence that respondents in a developing country are sensitive to the scope of the commodity described in the CV scenario and feel that the use of non market valuation methods can be both practical and feasible in developing countries. Similarly Markandaya and Murthy (2004) also successfully used the CVM to survey households in order to estimate the benefits from cleaning the Ganges under the GAP in India.

2.2.3 The Possibility of a Coasean Solution

Conventionally, governmental intervenes in the market process for environmental protection through pollution taxes, marketable pollution permits and command and control instruments. This Pigouvian solution to the problem of externalities has been the
subject of repeated attack along Coasian lines (Cropper and Oates 1992). Coase (1960) argued that if all trades were ‘costless’ and all participants ‘rational’, all mutually beneficial transactions would occur. In a perfect frictionless world the economy must wind up in a Pareto Optimal situation, regardless of externalities. For example, in such a world, if the harm caused by a smoking factory were greater than the cost of cleaning up, those harmed would organize to pay the factory to limit its emissions to the ‘correct’ level. In the real world, there are two main reasons why such bargaining does not occur (Roberts 1984). (1) The structure of the bargaining game which the externality gives rise to might include **perverse incentives** and we may find ourselves faced with a free rider problem. (2) High transaction costs could prevent such trades from being negotiated.

The free rider argument also helps to explain why it is so hard to organize private action to control widespread external effects like urban air pollution when the beneficiaries of cleanup are also jointly the source of the difficulty and each must bear the costs of cleaning up his own wastes. Littering and polluting toilets in public places fall in this category. More often than not voluntary trading does not take place not because of the free rider problem but on account of the information and decision costs of arranging such transactions. Environmental externalities usually affect large numbers of individuals. This greatly increases the costs of reaching an agreement due to the complexity of options and the multiple uncertainties-involved. The benefits of environmental protection to any one individual are often small, difficult for the individual to quantify and evaluate and uncertain.

While the Coase solution may be one end of the spectrum with self enforcing behaviour of the individuals; contractual arrangements with complete state regulation is the other end. In a developing country context it has been suggested that an in-between arrangement of **Collective Action** could work. Murthy (1994) has shown that collective action can result in the efficient management of common property resources. Also, it can efficiently control the externality problem in situations of non-benevolent government (Murthy 1996). Collective action is possible if there can be institutional arrangements in which all the relevant agents viz. polluters, affected people and the government play an important role. The expansion of Coasean theory to include collective action may make it possible to deal with the problem of environmental pollution abatement with positive transaction costs (Becker 1983). For example, in the case of water pollution abatement there can be collective action in the fixation of standards but implementation may be
through economic instruments or Coasean bargaining approach may be used for both fixation and implementation of standards.

The case of water pollution abatement in small scale industries provides an example of this approach (Mishra 1996). Common Effluent Treatment Plants (CETPs) have been set up at places where the necessary legislations are in place to define the property rights of factories and affected people and active political groups are formed of factories owners and the affected parties. The emergence of active political groups depends upon the benefits each group gets from the bargaining.

Collective action for industrial pollution abatement may not be possible due to inadequate and ambiguous environmental laws; lack of public awareness about the magnitude of damages to the community from pollution; lack of resources to local communities to organize themselves as politically active groups for taking recourse to legal acting against polluters etc. The government can play an enabling or catalytic role in this process by removing these constraints. Government or Non Governmental Organizations (NGOs) can extend soft financial loans and technical help for setting up CETPs and also help in working out mutually agreeable methods of sharing the cost of construction and running of CETPs.

Section 2.3
Concluding Remarks
The review of literature carried out in this chapter has highlighted the various conceptual and practical issues involved in a study of the equity aspects of large scale environmental protection programs. These issues are again taken up in Chapter 9 to see how they were dealt with in this study.