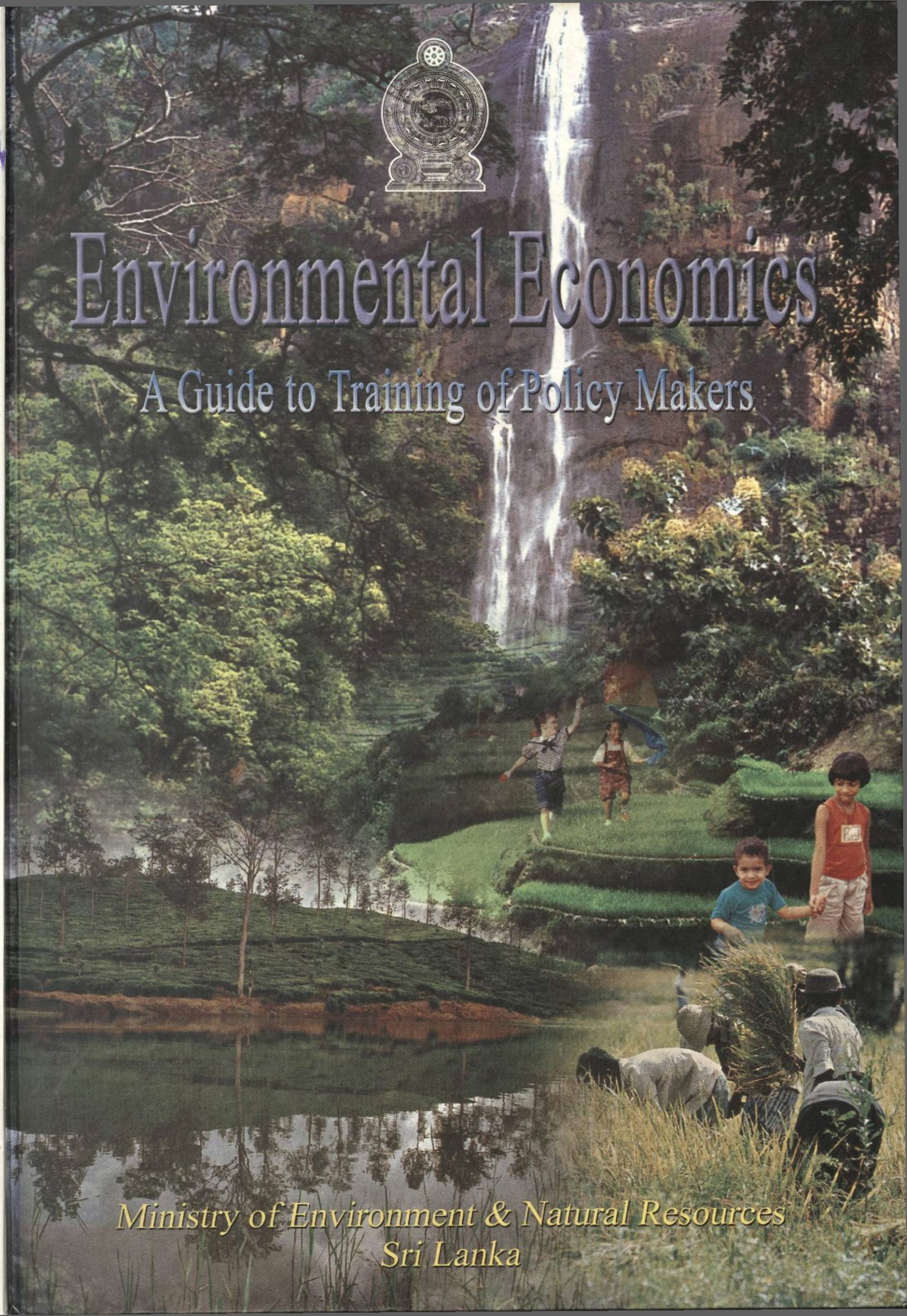

# Environmental Economics

## A Guide to Training of Policy Makers

*Ministry of Environment & Natural Resources*
*Sri Lanka*

ENVIRONMENTAL ECONOMICS A Guide to Training of Policy Makers

# ENVIRONMENTAL ECONOMICS

## A Guide to Training of Policy Makers

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MINISTRY OF ENVIRONMENT

Environmental Economics and Global Affairs Division,
Ministry of Environment and Natural Resources,
Sri Lanka.

in Collaboration with

Department of Economic and Environment
University of Bath
England

# ENVIRONMENTAL ECONOMICS

## A Guide to Training of Policy Makers

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# ENVIRONMENTAL ECONOMICS

## A Guide to Training of Policy Makers

2002 Edition


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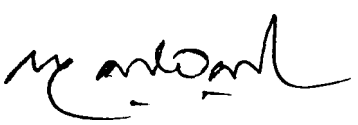
# Foreword

Policies and investment decisions concerning projects and individual resource uses need to be based on the efficient allocation of scare resources including environmental resources. In the market economic system, the goods that are bought and sold in the market have a market price, which determines the allocation of resources to provide goods and services to meet the needs of the people. Most of the environmental resources such as clean environment, urban air etc. are in the category of non-market goods and do not have appropriate price tags. Although most of the environmental resources are provided free of charge they provide important life support functions. In the case of these environmental resources there is a market failure i the present day market due to externalities and free riders since most of the environmental resources do not have prices. This leads to the problem of maintaining the continuous flow of natural capital to ensure the human wellbeing and the persistence of humanity. The only way to rectify this problem is to develop techniques and methods to value the non-market environmental resources and internalize them into the market system. Creating awareness among the policy markers on the importance of recognizing non-market environmental benefits in economic terms is critical in this endeavour. Therefore, we decided to launch a programme to create awareness among the policy makers on environment economics. A training of trainers workshop was organized as an initial step.

This Environmental Economics Workshop was a collaborative effort by the Ministry of Environment and Natural Resources and the University of Bath, UK, in fulfilling the need for training suitable trainers in this field in Sri Lanka. Publishing this guide is among the follow up activities in relation to the above important workshop. Sri Lanka lacks detailed references of Environmental Economics pertaining to the country. This guide is a comprehensive document comprising the lecture material distributed at the workshop by a team of eminent resource economists from the University of Bath UK, and other reading material. Although it is meant for the trainers in Environmental Economics it may also serve as a valuable reference for the readers at various levels of knowledge in this field.

Publishing of this document is an endeavour by the Environmental Economics and Global Affairs Division of the Ministry of Environment and Natural Resources, and it is the outcome of hard work, dedication and teamwork.



Thosapala Hewage

Secretary
Ministry of Environment & Natural Resources

# Preface & Acknowledgement

This guide is an outcome of a joint effort by the Environmental Economics and Global Affair division of the Ministry of Environment and Natural Resources and the Department of Economics and Environment, University of Bath, U.K. A Training Workshop of Environmental Economics was organized under the leadership of Prof. Anil Markandya, University of Bath, UK. for trainers of Environmental Economics and the policy makers in the field of environment in Sri Lanka from 19th to 24th April 2000.

Prof. Markandya and his team consisting of Mr. R. Boyd, Mr. A. Hunt and Ms. P. Mason have carried out an extensive analysis of materials related to Environmental Economics particularly in relation to policy makers and produced substantial training materials for use at the training workshop. Considering the importance and quality of the materials, Ministry of Environment and Natural Resources decided to publish the proceedings of the workshop and use in future training programmes.

The Environmental Economics and Global Affairs Division of the ministry of Environment and Natural Resources is grateful to Prof. Markandya and his team for their excellent work.

Mr. K. A. S. Gunasekara, the then Secretary of the Ministry of Forestry and Environment gave us invaluable advice and encouragement for successful completion of the workshop. We acknowledge his guidance and leadership with great honour. We are also grateful to Mt. Thosapala Hewage, Secretary of the Ministry of Environment and Natural Resources for his technical advice in organizing the workshop. Our deep appreciation also goes to the participants of the workshop who represented a wide range of institutions, both government and non - government.

Most importantly, we warmly acknowledge the financial assistance given by the Environment Action 1 Project, which is funded by the World Bank, for conducting the workshop and publishing this guide.

Recognition is extended to the staff of the Ministry of Environment and Natural Resources for editing, designing and producing of this valuable guide.

B.M.S Batagooda
Director
Environmental Economics & Global Affairs division
Ministry of Environment &Natural Resources

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# Chapter 1

# INTRODUCTION TO ENVIRONMENTAL ECONOMICS

*by*

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## 1 INTRODUCTION

Environmental economics is concerned with analysing environmental issues within an economic framework. As a discipline it is about 40 years old, although the ideas it is dealing with are much older than that. There are three key areas of the study of environmental economics:

- Valuation of the environment
- Regulation of the natural environment and of environmental pollution
- Integration of the environmental dimension into the mainstream of decision-making

In this first Chapter we will say something about each of these. The remaining Chapters will be devoted to discussing these issues in more detail.

## 1.1 Valuation of the Environment

The word value is used loosely and in many senses in our everyday language. Even in economic there are several uses of the word. We distinguish, for example, between **value in use and value in exchange**. We also have the concepts of cost and price.

The cost of something is what you have to pay for it or what you have to pay to acquire the inputs to make it. In economics, however, we have a more specialised use of the word cost as well. The cost of good 'X' is given by the value of the best alternative use to which resources could be put. Such a cost is called **opportunity cost**.

The world 'price', has a number of different interpretations. We distinguish between **producer price**, consumer **price and shadow price**. The first two are self explanatory. The third is the price of the good based on the opportunity cost of producing it. Under 'perfect markets' the shadow price is equal to the producer price which is equal to the consumer price. In practice the three differ because of monopoly power on the part of the buyer or seller, subsidies given by the government, taxes charged on inputs and outputs, and a lack of markets for some of the inputs.

The basis of valuation in liberal economics is

- **The maximum willingness of an individual to pay for receiving something (WTP), or**
- **The minimum willingness of an individual to accept payment for giving up something (WTA)**

The basis lies in taking individual preferences as the foundation of choice and value. In terms of the different concepts of cost and price discussed above we can say:

**Value in use** is the critical concept for environmental valuation and is related to WTP/WTA in an obvious way.

The shadow price of a good is based on the WTP/ WTA for the resources used in making it.

WTP/WTA are central concepts in environmental economics. The controversial aspect of this is the use of money to value environmental resources and natural capital. There are two main objections to this. The first goes along the lines, "how can you value in money terms, something which has no price, for which no money is exchanged and which is not associated in the public mind with money?" The answer to this is that a money value can be placed on something for which an individual **would be willing to pay** something, even if he or she does not actually make the payment. Related to that is the point that such values are relative and anothropocentric[1]. A fine view or a clean environrnent is not being given an absolute value that will hold for ever, irrespective of who cares about the resource and how that concern compares with his or her concern for other goods. For both these reasons it is not unreasonable to want to place values on the environment, although actually doing so may be difficult. That is an issue we discuss in Chapter 6.

[1] As noted the system of valuation is based on individual human beings and is anthropocentric. This is to be contrasted with a system of valuation that gives things an intrinsic value as, for exarnple, is the case with some ecologists. It is assumed here (and indeed in all economic analysis) that the appropriate system of values is the individualist/anthropocentric one, rather than the intrinsic one. This debate is beyond the scope of this review, although it will be argued that such a system is not inconsistent with many of the goals that ecologists would wish to pursue, such as the protection of key natural resources.

The second criticism against the use of money values, especially ones based on WTP is that they are 'income constrained'. Since you cannot pay what you do not have, a poorer person's WTP is less than that of a richer person, other things being equal. This occurs most forcefully in connection with the valuation of a statistical life (VOSL) where the WTP to avoid an increase in the risk of death is measured in terms of a VOSL. In general one would expect the VOSL for a poor person to be less than that of a rich person. But is more or less objectionable than saying that a rich person can and does spend more on health protection than a poor person; or that individuals of higher social status and wealth live longer on average than person of lower status; or that better neighbourhoods will spend more on environmental protection than poorer neighbourhoods. The basic inequalities in society result in different values being put on the environment by different people. One may object to these inequalities, and wish to adjust ones decisions to reflect them, but most economists would argue for that to be done outside the system of valuation of the impacts[2].

## 1.2 Theoretical Foundations of Environmental Economics

The area of economics to which environmental economics is most closely tied is welfare economics. Welfare economics deals with 'normative' issues -- i.e. with issues about which allocations of resources are 'good' and which are not and in ranking and comparing allocations.

A key concept in environmental economics is that of **Pareto efficient allocations.** An allocation is Pareto efficient if there is no other feasible allocation, which makes at least one person better off without making anyone else worse off. Related to that is the idea of a **Pareto Improvement. A Pareto improvement** occurs when a move is made to an allocation at which at least one person is better off and no one is worse off. Figure 1 below shows Pareto efficient allocations and Pareto improvements.

All allocations on the utility possibility frontier are Pareto efficient allocations. A move from $\alpha$ to $\gamma$ is a Pareto improvement, but a move from $\alpha$ to $\beta$ is not; nor is one from $\gamma$ to $\beta$.

2 One could argue, for example, that increased expenditure on high technology medicine in Europe is unethical, even though the citizens of that region have a WTP that justifies such expenditures, because the same expenditure on preventative medicine in a poor developing country would save more lives. Society does not accept fully such an argument, taking the view that most decisions about allocation of resources are predicated on the existing inequality of income and wealth, both between and within societies. It does, however, justify some subsidy to R&D in this area for tropical diseases on grounds of equity.

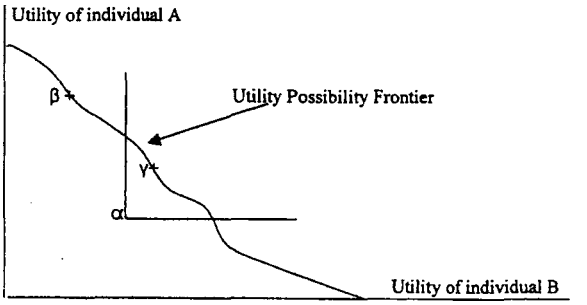


Figure 1: Pareto Efficient Allocations and Pareto Improvements

There is an important link between Pareto efficient allocations and a competitive economy. We define an economy as competitive if

- It has many producers and consumers who take prices as given.
- All producers and consumers are self-interested; producers seek to maximise profits and consumers seek to maximise their own utility.
- There are well-defined prices for all goods and services, including goods and services which will be provided in the future.

Such an economy is said to be in a **General Competitive Equilibrium** if the demand for each good or services is equal to the supply, or less than the supply. If the supply exceeds the demand, however, the price has to be zero.

There are two important links between Pareto efficient allocations and a general competitive equilibrium.

First, all general competitive equilibria are Pareto efficient allocations.

Second, any Pareto efficient allocation can be obtained as a general competitive equilibrium provided certain conditions are satisfied. The most controversial of these are: (a) that the government should be able to allocate resources to individuals in such a way that they can acquire the goods and services consistent with the Pareto efficient allocation and (b) the production of goods should take place under conditions of 'convexity'. The

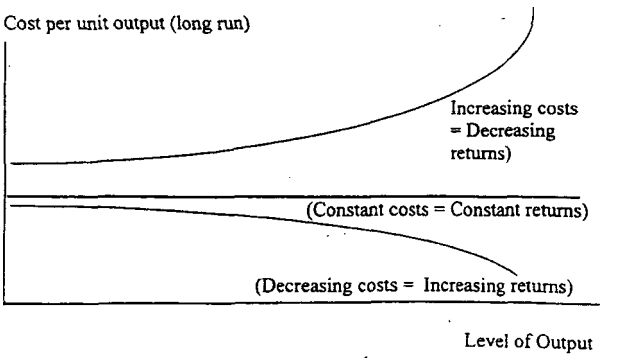


Figure 2: Convexity , Average Costs and Returns to Scale

second condition implies that producers face increasing or constant long run average costs, and that costs per unit of output do not fall with output. Figure 2 below shows the cases of increasing, constant and decreasing costs. The last is the case where production does not meet the 'convexity' criterion.

These propositions are called the **fundamental theorems of welfare economics.** It should be noted that: (a) the system only determines relative prices of goods -- i.e. the price of one good in terms of another and (b) the system has no role for money.

It is instructive to see how the Pareto effficient allocations and competitive general equilibria (CGE) are related. The CGE ensures that all consumers pay the same price for the same good. This in turn ensures that the willingness to pay (WTP) at the margin is the same for all individual who consume the good. Then there is no way we can redistribute the good and make everyone better off. Similarly under CGE producers will set the price of a good such that it is equal to the cost of producing the last unit (i.e. equal to the marginal cost). This ensures that profits are maximised. Hence CGE ensures that the market equilibrium is as shown in the Figure 3. In that figure there is no way we can change the level of production and make both parties (consumers and producers) better off.

In environmental economics we take the fundamental theorems of welfare economics as our point of departure. We know that the conditions for a CGE to exist are not all satisfied, or that the conditions to attain a particular Pareto effficient allocation are not all met. The most important condition that is not satisfied is that of **complete markets.**

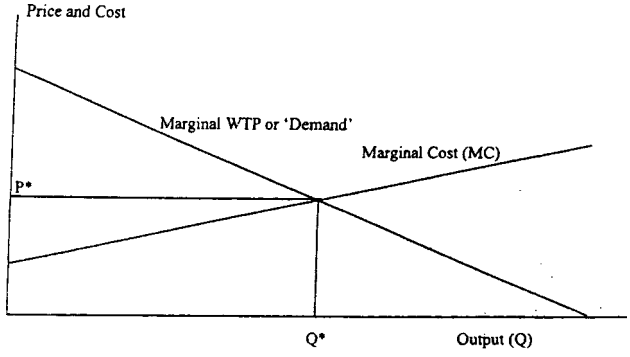


Figure 3: Production Under Competitive Conditions

We say that if there is an activity that impacts on producers or consumers, and if that impact is not directly allowed for in an organised market, then there is an **external effect or an externality.** Many externalities are environmental in nature:

- Air pollution from electricity generation
- Water pollution from industrial activity
- Agricultural runoff affecting water resources
- Use of CFCs affecting the ozone layer.

What should we do about such externalities? This takes us into the second area of environmental economics, namely regulation. It will be dealt with in greater detail later. At this point we should note two things: (a) that externalities may give rise to Pareto inefficiency and (b) that externalities may be addressed through private negotiations. In Figure 4 below curve AB represents the marginal damages from air emissions (damages to other industries, the general public etc.) and the curve CD represents the marginal costs of abating emissions on the part of the polluter. If the rights to emit are with the polluter the level of emissions will be OD. This is not, however, a Pareto efficient allocation. The sufferers can bribe the polluters to reduce emissions and still be better of themselves. The optimal position is OE.

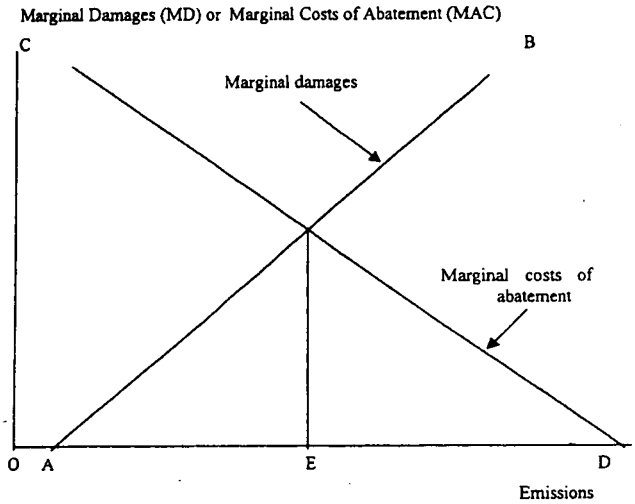


Figure 4: Marginal Abatement Costs and Marginal Damages

Some economists, notably Coase and the 'Chicago school' believe that the point OE can be attained without regulation, since it is in the interest of both parties to negotiate to that point. Moreover, Coase has argued that the final solution does not depend on who has the rights, the polluter or the sufferer.

Problems with the Coase solution are:

- Transactions costs
- Information
- Issues of justice.

Nevertheless Coase type negotiations do take place and policymakers need to be aware of them in designing and enforcing regulations.

In several Chapter in this workshop we will look at ways in which the curves AB and CD can be measured. and we will discuss policy for the regulation of the environment.

## 2 KEY ISSUES IN ENVIRONMENTAL ECONOMICS

The discussion above has focused on how environmental economics 'fits in' with more conventional economics. In this section we will look at a number of areas where the two disciplines do not always sit easily together. These are:

- Individual preference based valuation
- Recycling and the linear versus circular views of the economy
- Discounting in project appraisal and policy analysis
- Sustainability and economic policy

### 2.1 Individual Preference Based Valuation

Many ecologists and 'ecological economists' object to a valuation system based on individual preferences. They argue that environmental values should not be based just on the preferences of the present generation, but should take account of future generations and, in some cases should be based on the 'intrinsic worth' of the environment.

Individual preference based valuation, properly interpreted, is not just about present generations values, but also about future generations values. The problems has always been, how to measure the values of generations that are as yet unborn and, more importantly, how to account for them in the present allocation of resources. Economic models for dealing with this rely on 'overlapping generations' so that each generation is linked to the next (and possibly the one after that) through family and social ties. Such models are complex to analyse but they show that, in general, the allocation of resources that emerges will tend to favour the present. In other words, compared to an intertemporal Pareto efficient allocation the result will be suboptimal under market conditions. This suggests that future generations may need to be guarded in some special way. One way is to use a lower rate of discount in valuing projects that are concerned with allocating resources over time (see below). Another is for the government to act as a guardian of future interests, ensuring that its best estimate of their values is fed into the decision-making process. Typically, one can expect that many environmental goods, which are in fixed supply and of which there is an increasing scarcity, will be in greater demand in the future, for that reason and for the reason that *per capita* incomes will be increasing, especially in developing countries., as a result of which the WTP for these goods will increase.

As far as intrinsic worth is concerned, there is more of an *impasse* between the ecologists (of varying levels of depth) and the more conventional economists. It is difficult-to overlay on a system of individual values a system of intrinsic values. If the later are a reflection of the need to take account of future needs than that can be satisfied as described above. But it is sometimes more than that. Some natural resources, it is argued, have a value irrespective of human beings and their needs. This takes us away from the **anthropocentric** value system of economics to one that is based on non-anthropocentric principles. As yet there is no sound way of accounting for such values in an economic framework.

### 2.2 Recycling and the linear versus circular views of the economy

The economist Boulding noted that conventional economics does not take account of the fact that production and consumption generate waste, and the capacity of the plant to provide natural resources and absorb the waste are limited. The conventional model is as shown in Figure 5a whereas the more accurate description is a shown in Figure 5b.

The circular view of the economy takes account of the fact that the waste is fed into the environment, which has a limited capacity to absorb that waste and that, through recycling the amount of that waste can be reduced.

There are two aspects of the waste flows that can be incorporated into the conventional analysis. First, we can estimate the amount of the waste through the principle of the conservation of mass. All materials produced are transformed through use but not destroyed and statistics on waste in all forms (liquid, solid and air) can be estimated at least on a non-spatial basis. Second, by the principle of increasing entropy, materials can never be fully recycled. The process of production will always generate some waste.

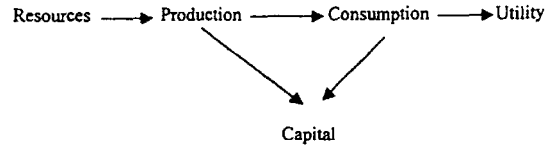


Figure 5a: A 'Linear View of the Economy

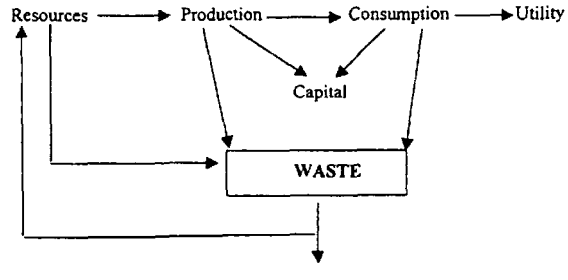


Figure 5b: A Circular View of the Economy

Allowing for both of these is possible in an economic framework. Where waste sink capacities are limited relative to the amounts of waste being generated, the costs of using that sink can be included in the costs of the disposal of the products, so sink capacities are not exceeded. The issue of whether or not the management of sinks can be handled solely by economic instrurnents is left to Chapter 3. The level of recycling should be determined by full economic and environrnental considerations, so that materials are recycled whenever the social cost of disposal exceed the costs of recycling. In practice levels of recycling are not determined by such factors. Either there is not enough recycling (when the facilities for doing so are not there) or possibly too much (when the facilities are heavily subsidised).

## 2.3 Discounting in project appraisal and policy analysis

Discounting is the act of placing a lower value on future goods and services than on present goods and services. The value of one pound of something one year from now is put at

$$\frac{1}{(1+r_1)}$$

where $r_1$ is the discount factor for year one. In two years time the discount factor is

$$\frac{1}{(1+r_1)} \cdot \frac{1}{(1+r_2)}$$

where $r_2$ is the discount factor for year 2 so on. In most analysis the discount factor is treated as independent of the year, so that $r_1$, $r_2$ etc. are independent of the time period and can be replaced by $r$.

The justification for discounting is:

- time preference (impatience)
- productivity of capital and growth in the economy

For both these reasons individuals and societies place a lower value on future costs and benefits than on present costs and benefits. In developing countries the productivity based rationale for discounting results in very high discount rates. Investments in the private sector yield rates of return in excess of 20%. Hence, it is argued that in making investment decisions we should discount future costs and benefits by that amount. The agreements against that view are: (a) that the high returns are in part a compensation for the high risk and (b) there is a scarcity of capital because capital markets are inefficient and distorted. Likewise individuals have a very high private rate of discount because the future is highly uncertain and because the needs of the present are so pressing.

Based on extensive discussion and analysis, most economists accept that there is a divergence between the private and social rates of discount. Whereas the private rates may be in excess of 20%, social rates are more likely to be around 3-5%. These are **real rates**. This means that the nominal rate to be applied is equal to that plus the rate of inflation.

What are the environmental implications of a high rate of discount?

- Whereas low rates can result in higher value for future benefits (and therefore more conservation), low rates can also promote more investment, which may imply a higher rate of exploitation of natural resources.
- Many long term investments can only be justified at low rates of discount
- If the environmental benefits in the future are quantified, the use of higher rates is, to some extent mitigated.
- For some projects with very long term impacts low discount rates will not resolve the problem. Even with a rate of less than one percent the costs and benefits hundreds of years from now will have no impact on present day values. But a rate of zero will make the problem unbounded (i.e. the future costs and benefits can become infinite). In these cases one needs to make use of other constraints based on physical and other restrictions on the exploitation of resources.

Discounting is not necessarily an enemy of the environment but the use of lower is justified for all projects (not just environmental ones). Furthermore, it may be sensible to look at the issue of time varying rates for projects with very long term environmental impacts.

## 2.4 Sustainability and economic policy

The concern with sustainability was raised by the Bruntland Commission. The underlying thinking was that the development paths being followed were resulting in a level of environmental degradation that would cause future generations to suffer from a reduced level of welfare, as resources ran out and/or as life support systems were severely damaged.

In economic terms the notion of sustainability is a difficult one to operationalise. Certainly one does not want to pursue a path that results in lower welfare in the future. But then no economist would ever have advocated such a path. So what does sustainability add to the debate?

The one operational notion is to see what sustainabilty has to say about the use of natural resources and accumulation of capital over time. In this context

there is the notion of sustainable income. It can best be understood in terms of ensuring that present generations pass on to future generations enough assets to ensure that the future generations can enjoy the same living standards as the present. What are the assets? The issue of sustainability is discussed further in Chapter 13 and 14.

## 3 THE ENVIRONMENT IN A POLICY FRAMEWORK

The environmental dimension enters policy at many levels, each of which requires different tools of analysis. This course provides only an outline of the linkages between environmental and other policy areas, and an in- depth look at some of the key issues of interest. The structure is usefully represented by Table 1 below.

**Table 1: Links Between Environment and Socio-economic Policies**

| Level | Environmental Policies | Socio-economic Policies | Linkages |
|---|---|---|---|
| Global | Climate change, Biodiversity, Ozone layer, International waters | International trade Globalisation | Impacts of liberalisation/ globalisation on natural resource use. Impacts of environmental policies on economic growth, welfare. |
| National | Sustainable development strategies National environmental quality targets Methods of regulation of environment. | Interest rates, taxation, government expenditure, external balance. Policies on equity. | Env. targets and regulations impact on competitiveness, employment, growth, poverty. Macroeconomic policies and structural adjustment impacts on use of environment |
| Sectoral | Many sectors: Forestry: pricing and taxation Agriculture: pricing, land use, subsidies, land reform. Transport: MBIs Vs.C&C, infrastructure investment Energy: prcing, subsidies, investment, MBIs Vs. C&C. Manufacturing: C&C Vs. MBIs | Export versus domestic oriented development. Employment creation National security pricing, Poverty C&C, alleviation | Environmental policies can hurt vulnerable sections through lower employment, higher prices etc. Sectoral development policies can effect the environment through increased degradation overuse of natural resources |
| Project | Environmental impacts are as wide as those for the sectors: air and water pollution, soil erosion, loss of biodiversity. | Projects are linked to development goals of provision of infrastructure and key inputs into the economy -- Water, energy etc. | The environmental impacts can reduce the effectiveness of the investment under consideration as well as other investments. At the same time the projects can have a detrimental impact on individuals by reducing their environmental capital. |

The highest level of policy making is the global, where national governments need to negotiate and agree on actions to protect the global environment. Such agreements represent the environmental dimension of international policy-making. The other issues that are discussed at the international level and that have a bearing on the environment are trade and 'globalisation' more generally (as shorthand for increased international movements of capital and other factors of production). Links between the two arise through trade restrictions in environmental policy (e.g. CITES and the Montreal Protocol on Substances that Deplete the Ozone Layer); as well as through the environmental implications of liberalisation of trade and of international capital flows. Some of these issues are covered in Chapter 12 on the Economics of Climate Change.

The next level of policy-making is the national one, where decisions on macroeconomic variables such as interest rates, the national taxation and expenditure, monetary policy and exchange rate policy are taken. It also includes national policies in the areas of poverty and equity. Environmental policy at this level can consist of commitments to national targets, and to approaches to environmental decision-making, including sustainable development. Links between the two are explored through environment-economy models, including links between poverty and the environment, structural adjustment and the use of natural resources and so on. In this course we cover some of these topics in Chapter 6 on environmental valuation at the micro and macro level, although there are more macroeconomic linkages that we will not deal with, especially on structural adjustment.

The third level of decision-making is the sectoral one, where environmental policy is relevant to virtually all sectors of the economy, but especially the agricultural, forestry, energy, 'transport and manufacturing industry sectors. Economic and social policies in the areas of pricing, subsidies and investment regulation all have environmental impacts. Equally environmental policies on eco-labelling, animal welfare etc have economic implications. Some of these issues are covered in Chapters 3, 4 and 5. The key tools are sectoral models of supply and demand, with physical linkages that look at the environmental impacts and the feedback from those impacts on supply. In some cases these linkages are quantified in physical and monetary terms, which would make the analysis similar to the next level, where a more formal assessment if the costs and benefits is carried out.

The lowest level of decision-making where the environmental dimension comes in is at the appraisal of projects such as investments in infrastructure. Here the analysis focuses on the benefits and costs and attempts to estimate the rate of return on the project, or some other summary measure, inclusive of the environmental impacts. The aim is to represent in money terms those impacts that can be so represented, using methods of valuation that will be discussed in some detail in Chapter 6. Where impacts cannot be measured in money terms, the data are presented in physical terms and some method of valuation based on multi-criteria is needed. In any event a partial monetisation can usually be achieved for the environmental impacts and that provides at least a partial picture of the net benefits of the project.

A full appreciation of all the linkages is not possible in a short course such as this. We hope, however, to provide a flavour of the key issues, and a detailed coverage of some case studies of particular problems.

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# Chapter 2

# RESEARCH METHODS IN ENVIRONMENTAL ECONOMICS MACROECONOMIC POLICY AND THE ENVIRONMENT

*by*

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## 1 INTRODUCTION

Macroeconomic policies have a widespread impact on the use of a country's resources and ecological services. The policies change economic signals throughout the economy, influencing decisions on which and how much of a resource or service will be used. Macroeconomic policies, whether or not successful in generating economic growth, also indirectly impact the environment due to changes in income, public revenues, and innovative capacity.

Frequently governments in developing countries have been forced to undertake major macroeconomic reforms in order to obtain support from international financing agencies such as the IMF, the World Bank and the EC. These measures, which are described briefly below, have an effect on the environment. The issues that are of interest here are: (a) how can one assess these impacts and (b) what actions if any need to be taken to correct any negative environmental effects.

The Chapter is organised as follows. The next section (Section 2) describes the major macroeconomic reforms and their likely effects on the environment. Section 3 describes the tools that are available for the analysis of such impacts and their respective strengths and weaknesses. Section 4 discusses the different instruments that could be used to correct the negative environmental effects. The whole discussion of the Environmental impacts of macroeconomic policies and the correction mechanisms suffers, however, from the fact that the basic information on macroeconomic indicators, such as GDP, is flawed. It is flawed because it does not account for the damages done to the environment in the process Of generating economic benefits to society. Hence a major area of research in recent years has been that of integrated economic-environment accounting. This is not dealt with here but is covered in Chapter 14. Section 6 concludes the main chapter.

## 2 MACROECONOMIC POLICIES AND THE ENVIRONMENT

Most macroeconomic reform programs have two distinguishable time horizons. In the short term, the focus is on the stabilisation program aimed at rapid restoration of macroeconomic balance through demand management policies. At the same time, there is the longer term goal of improving efficiency and economic growth through liberalisation of trade and domestic pricing policies. and institutional reform through the structural adjustment and sectoral adjustment loans. The detailed components and policy tools comprising a typical adjustment program are reviewed below. Any potential conflicts between the short and long term goals are highlighted.

### 2.1 Short Term Stabilisation Programs

This is a short-term program (12-18 months) to achieve **rapid** internal and external balance via adoption of policies to reduce both the level and composition of aggregate demand.

Measures to achieve rapid balance of payments relief would primarily address any imbalance on the current account. A deficit on the current account implies that domestic absorption of goods and services exceeds domestic production. In the short term, the analysis underlying the formulation of such programs assumes productive capacity to be fixed. Hence, the stabilisation program focuses on demand reducing and expenditure switching policies.

A number of policy options are available to reduce domestic absorption and typically might include all or a combination of the following:

#### 2.1.1 Fiscal policy

Large budget deficits have been a primary source of external disequilibria and macroeconomic instability. A major objective of the stabilisation program has therefore been to reduce the budget deficit both via reductions in government spending and increases in tax revenue. Government spending may be cut on recurrent expenditures (eg. reductions in public sector wages, food subsidies) and/or public sector investment and lending to the private sector.

#### 2.1.2 Monetary policy

In its simplest form, the rate of domestic credit

expansion has been the primary monetary policy instrument of the short-term stabilisation program. The desired effect is to reduce domestic absorption via reductions in private consumption and investment. The rate of domestic expansion can be controlled through direct regulation of the money supply, as well as thorough indirect measures such as rationing credit, regulating the banking system and controlling interest rates.

### 2.1.3 Exchange rate policy

These fiscal and monetary instruments are **demand reducing** but do not necessarily ensure that simultaneous balance is achieved on both the government budget deficit and the balance of payments. To ensure this, an additional instrument is needed. This acts to change the composition of demand from foreign to domestic goods and to increase incentives for production of exports and/or import substituting goods. Balancing the government budget is sometimes referred to as **internal balance** and balancing the external budget account is referred to as **external balance.**

Managing both external and internal balance involves adjustments to **both** sets of instruments and is difficult to achieve, even in developed countries. Moreover, it entails costs, at least over the short run, in the form of reductions in real income for certain sectors of society. Partly to mitigate these, and partly to protect special interest groups, import/export controls are sometimes employed as additional instruments. However, their use is generally not supported by the international financial institutions as they tend to be discriminatory in their impact and result in inefficient resource use.

### 2.1.4 Summary of short term stabilisation impacts

The stabilisation program would therefore typically target reductions in fiscal deficits, money and credit restrictions and real exchange rate devaluation. The emphasis is on achieving rapid reductions and switching in the composition of aggregate demand to achieve internal and external balance. The success of such a program in the short term is very much dependent on political and social acceptability and the ability of the economy to respond to this new set of price incentives. Any transitional phase, during which the balance of payments may even worsen (depending on the elasticities of imports and exports) will be cushioned by the provision of conditional funds under the adjustment program.

It is also important to stress that these contractionary policies will not ensure long-term balance of payments stability or increase the capacity of the economy to respond to external shocks. The medium term structural and sectoral adjustment packages address these issues.

## 2.2 Medium Term Structural and Sectoral Adjustment Programs (SALs)

The stabilisation program focuses on reducing demand whereas the medium term strategy focuses on relaxing supply side constraints. The program aims at both increasing output from existing capacity and increasing the rate of growth of capacity by prescribing a greater role for prices, markets and the private sector in the development process.

The market-based argument for improving efficiency is to reduce distortions that drive a wedge between prices and marginal costs. To encourage growth of existing capacity the SAL conditionality will target policies that increase the rate of and return to investment.

The array of policy instruments to achieve these ends is varied and country specific. Some of the major instruments highlighted in the literature are briefly reviewed below:

### 2.2.1 Trade liberalisation

Those countries that adopted the import substitution development path popularised in the 1950s and 1960s now have a legacy in the form of a comprehensive package of import and export taxes. Such protectionist policies are now regarded as a major source of trade distortions and as a constraint on efficiency and growth. The gradual dismantling of import and export taxes is thus a central component of the medium term strategy. Trade issues are discussed further in Chapter 11.

### 2.2.2 Domestic pricing policies

Price controls and subsidies have frequently been used in developing countries both to stabilise markets and to achieve income support for the poorer groups in society. They are, however, also regarded as a major domestic distortion and source of economic rigidity.

Agricultural input and food subsidies have been particularly popular and in many countries are now being dismantled as a central part of their structural adjustment program. Indeed subsidies worldwide are estimated to have fallen by almost half since 1990 (UN, 2000).

### 2.2.3 Non-price incentives

'Getting the prices right' via trade and domestic pricing reforms often has to be combined with non-price

measures if there is to be the desired supply response. Therefore, in addition to pricing policies, the medium term strategy may also identify the need for reforms and developments in other areas to relax supply constraints. To varying degrees attention has focused on institutional reforms (eg. land tenure, credit, marketing and distribution; research and extension), infrastructural developments (transport, education); stimulation of technological innovation (high yielding crop varieties, irrigation) and expansion of education and manpower training program.

### 2.2.4 Conflicts between the short term and long term strategy

Conflicts between the short-term objectives of macroeconomic management and the long-term objectives of structural adjustment can arise. Where the pressure for an increase in foreign exchange earnings is very strong, it can take precedence over all other considerations, leading to a non-sustainable use of the natural resource base and the acceptance of environmental standards that are so low as to cause lasting damage to the absorptive capacity of the local environment. Examples of the former might be the mining of forest resources and the export of wild animals and animal products. Examples of the latter are the environmental damage caused in some places by tourism, low cost mining of minerals and the uncontrolled discharges of industrial effluent into water and land bodies from industrial activities.

The pressure for an increase in foreign exchange can arise for a number of reasons. These include the demands of the economy for imported inputs for its continued growth and development, as well as the need to finance debt overhangs built up as a result of past imprudent borrowing and inappropriate domestic economic policies.

In any attempt to foster a sustainable economic policy in a developing country it is precisely this conflict between the short and long-term objectives that has to be resolved. There is also a potential conflict between macroeconomic and sectoral policies. Examples are plentiful in the literature of inappropriate macroeconomic policies undermining the success of sector-based

**Box 1 - Effects of Structural Adjustments on the Environment**

| | | | Envrironmental Impacts |
|---|---|---|---|
| SHORT TERM MACROECONOMIC POLICY | FISCAL | GOVT.EXP | +/- Drought relife/food aid/Agric. extension Public Infrastucture (e.g. Roads)/Env. Management |
| | | TAXES | +/- Reduces demand for resources/Env. Charges |
| | | SUBSIDIES | +/- Input Effect. Machinery, Fertilizers, Pesticides, Water, Energy, Credit, Output Effect : Depends on Crop Characteristics |
| | MONETARY | CREDIT | +/- Reduced credit for inputs e.g Fertilizers, Pesticides: and Investments e.g. Leveling irrigation, machinery |
| | | INTEREST RATE | +/- Reduced investment and resource demand |
| | EXCHANGE RATE | DEVALUATION | +/- Import Effect: Increases prices of imported inputs e.g. Fertilizers, Pesticides Output effect : Increased export crops-depends on crop characteristics and farming practices |
| | TRADE | IMPORT/EXPORT TAX | +/- Removal of protectionist taxes-same effects as devaluation but on selected commodities |
| | | TRADE CONTROLS | +/- Same effects as trade taxes. Technological lock-in |
| | PRICING POLICY | PRICE CONTROLS | +/- Rise in official prices: impact of output responses depends on crop charateristics, farming practices Fall in Official prices : reverse effect |
| | | REDUCE SUBSIDIES | + Reduced use of pesticides, fertilizers, credit irrigation water, energy, machinery |
| | | INCREASE TAXES | + Indirect impact via reduced resource demand |
| MEDIUM/LONG TERM | INSTITUTIONAL REFORMS | LAND | + Encourage on farm investment and long-term sustainable resource management |
| | | FINANCIAL | + Improved credit mobilization may benefit rural farmer |
| | | RESEARCH & EXTENSION | + Improved extension services promoting sustainable resource management |
| | INVESTMENT POLICY | TRAINING | Human investments in agricultural extension, wildlife and resource management |
| | | VALUATION | + Project evaluation to include environmental cost and benefits |
| | | TECHNOLOGY | + Industrial Pollution abatement technologies; adapted agricultural technologies |
| | | PUBLIC INFRASTRUCTURE | +/- May increase access to natural resources and encourage their exploitation, Supporting infrastucture for producers enabling greater price responsiveness |

environmental policies. For example, when the Ghanaian government introduced a system of royalties in forestry aimed at improving the efficiency of logging, the over-valued exchange rate rendered the stumpage prices negligible, almost a free good, and trees continued to be felled at the same rate.

## 2.3 General Environmental Impacts of the Macroeconomic Policies

A summary of the linkages discussed above is provided in Box 3.1. The same box also summarises the major environmental linkages. These are described below. Where adverse impacts are clearly present a (-) sign has been inserted and where they are clearly beneficial a (+) has been put in. Where the impacts are unclear a (+/-) has been used. Further discussion of these issues can be found in Reed (1992, 1996).

### 2.3.1 Short term stabilisation programs

The policy instruments of short term stabilisation programs are aimed at a short sharp reduction and, if necessary, a change in the composition of aggregate demand. In recent years, such policies have been criticised on the grounds that the burden of adjustment has often fallen on those sectors of the population that are least able to bear it. (Cornia, Giovanni et al., 1987). The three main areas of policy that impinge directly on the environment are: public sector expenditures, taxation and exchange rate adjustments.

***Reductions in public sector spending***

The economic rationale behind reductions in public spending is to relieve possible deficits on both the internal and external accounts. Exactly which areas of public spending will suffer is the subject for negotiation, but there is considerable evidence to suggest that special environmental programs, food relief, agriculture extension services, welfare, and similar programs tend to get cut. This has negative effects on the poor and, in the short run, on the environment. Investments in infrastructure, education, health, and reforestation are also likely to suffer. Yet as the price of resources increases, resource use may become more efficient, and projects that impact the environment negatively, such as inappropriate irrigation schemes, could become unprofitable. Thus, the impact of each of these scenarios is country and site specific. The following are some examples of possible impacts.

- Reductions in drought relief, food aid, subsidies and general welfare programs may intensify rural poverty and lead to unsustainable resource management and farming of marginal lands. Analysis of the incentive effects of the removal of various subsidies is reviewed in greater depth under the domestic pricing section. (-).

- Reductions in agricultural extension services and environmental and resource management programs will obviously have a direct and adverse environmental impact (-). In many countries there have- been major cutbacks in extension staff, which undoubtedly will affect the adoption of improved practices and land conservation methods.

***Changes in taxation***

Taxes and subsidies affect relative prices of resources. They also affect the amount the government has available for environmental protection. (+). Sometimes special extra-budgetary funds can be set up so that when tax revenues fall, the environmental sector is not impacted.

***Exchange rate devaluation***

Exchange-rate devaluation directs expenditure away from foreign imports and encourages exports by increasing the domestic price for both exports and imports. This new set of price incentives will affect the economy across the board and can indirectly affect natural resource management. Increased export prices may favour the export of cash crops. If property rights are well established and the export crops are perennials, the effects for soil conservation purposes are neutral to positive. (+). In the case of timber, which is often produced on public lands, environmental effects will depend on the strength of the government institutions and the land tenure arrangement. A crop's price increase will cause higher land values, which may induce conservation practices, but, again, only if property rights are well established. Increasing the cost of fertiliser and pesticides by, for example, reducing subsidies, reduces their use and improves the quality of the environment (+)

***Monetary policies***

Monetary policies aimed at increasing the cost of capital directly impacts the management of natural resources by enforcing rapid use rather than a conservation approach. However, because attention is given to the long-term interest rate, long-term stability is more important than a short-term interest rate change in a stabilisation program. In this respect, inflation caused by bad monetary policies does not help. An increase in inflation raises nominal rates immediately as well as raising long terrn expected interest rates. Targeted credit affects the resource in a similar way to a subsidy. Environmental impacts from monetary policies cannot

be generalised but depend on the sector and the particular circumstances of land tenures, informal availability of credit, and other factors.

*Sectoral Policies*

Various sectoral policies are associated with macroeconomic reform. These include reductions in subsidies to energy, agriculture etc. They are discussed further in the individual Chapters devoted to those topics.

## 2.3.2 Structural adjustment and trade liberalisation

Although trade-related measures have been discussed above, this section looks more closely at the possible effects of trade liberalisation in more detail, as they are the focus of considerable debate. Trade issues are also discussed in greater detail in Chapter 11.

Stabilisation techniques focus on reducing demand, whereas structural adjustment (SA) programs, over the medium term, relax supply-side constraints, which increases efficiency. By prescribing a greater role for prices, markets, and the private sector, SA aims both at increasing existing output capacity and increasing the capacity's rate of growth. SA policies attempt to establish the right price through trade liberalisation and domestic pricing policies. Domestic pricing is discussed in the sector analysis Chapters. However trade liberalisation through a floating exchange rate is a major force behind domestic prices. Removing protectionist quotas and tariffs, and export taxes and incentives are discussed here.

Trade liberalisation's macroeconomic objective is to open the economy to foreign competition and technology, to encourage foreign exchange earning capacity, and so to move towards a more sustainable balance of payments. Trade instruments are often a central component of fiscal and industrial policy. Trade levies account for a substantial portion of government revenue and play a role in the promotion of certain industries. If these levies are removed, alternative sources of taxes have to be found. The relative price changes within sectors in which protection or incentives are removed and between sectors will affect resource use. If trade liberalisation occurs in conjunction with sound environmental and macroeconomic policies, this can contribute to sustainable development.

Import barriers (tariffs and quotas) increase the price of goods and services in an economy but more particularly in the specific sector. These barriers reduce the general level of economic activity, increase producer surplus, but decrease consumers' welfare. Producers are protected, often in terms of the infant industry argument. Given a monopolistic situation, less may be produced and less pressure put on the environment, but producers are nevertheless ineffficient users of resource input, which makes the net impact on the environment uncertain. Tariffs can be levied against the import of old technologies, which can prevent pollution and thus beneficially affect a country's environment. The environmental impact of import barriers is similar to that of devaluation, which was discussed earlier. These impacts cannot be predicted on a general level and depend on the economy's structure and institutional capabilities. In the agricultural and natural resource sectors, the environmental impact of import barriers depends heavily on the property rights system.

Trade liberalisation can help or hurt the environment depending on which sectors expand or contract, on the effectiveness of the county's environmental policies and on the extent to which increases in income are spent on improving environmental quality. Trade liberalization (by letting the economy use its comparative advantage) tends to expand the export sector, while causing the import-competing formerly protected sector to contract (a sector composition effect). The environmental impact of the composition effect depends on the country specific conditions, i.e. whether the industries that expand following liberalization are more environmentally damaging than those that contract. In many developing countries the export industries, e.g. tree crops and labor intensive light manufacturing, are relatively "clean" industries, in particular when compared to import-competing heavy industry. In the Philippines, for example, trade liberalization reduced soil erosion and consequent water pollution by shifting resources away from import-competing annual crops like upland rice to export-oriented tree crops, where soil erosion rates are lower by a factor of between five and ten. In many African countries export crops such as tree crops grown with grass cover in West Africa cause much less erosion than food crops such as cassava.

. Global agricultural liberalization is likely to be very environmentally beneficial since much of the agricultural industry that has developed behind high protective barriers in developed countries has become very intensive in the use of chemicals and pesticides because high producer prices create incentives to increase yields. Agricultural liberalization of the type begun under the Uruguay Round of GATT will reduce worldwide pollution by reducing the artificial stimulus to agricultural production in developed countries. This will come about as world prices are raised when overproduction in developed countries is reduced, and output in the developing countries, where fertiliser use per hectare is frequently one twentieth of the level in the highly protected countries, is increased (Anderson, 1992).

Where environmental policies are inadequate, trade liberalization can have an adverse impact on the environment in the short run. However, a dynamic positive effect of trade liberalization will also occur for two reasons (Grossman and Krueger, 1993). First, trade liberalization leads in the long run to an increase in income that is vitally important both for reducing poverty and for improving environmental quality. Higher economic growth is likely to increase the political demands for a cleaner environment. Second, foreign producers may transfer modern technologies to the local economy when restrictions on foreign investment are relaxed. Industries in developing countries with open trade regimes have been found to be cleaner than in countries with closed trade regimes (Lucas, Wheeler and Hettige, 1992).

Trade liberalisation causes a scale effect by expanding economic activity, and if the nature of that activity remains unchanged, the total amount of pollution generated must increase. To the extent that economic growth gives rise to an increase in the demand for energy, generated by means similar to the prevailing methods, output of harmful pollutants will increase. Furthermore, if expanded trade results in increased demand for transportation services without any change in transportation practices, increased trade will contribute to a deterioration in environmental quality.

The key to ensuring that trade liberalization improves the environment as well as raising real incomes is to have sound environmental policies in place. These policies need to deal with the externalities that are at the root of most environmental problems. Key policy reforms needed in developing countries include the establishment of secure land tenure to encourage sustainable use of land and resources such as forests and fisheries, and the introduction of policies based on the Polluter Pays Principle (PPP), which implies that the polluter should pay for the environmental damage caused.

If they are to be used at all, trade restrictions should only be used with extreme caution in support of environmental goals. In general, trade policies are not the best policy measures for environmental objectives. Even if trade policies are environmentally beneficial, they will generally be inferior to policies targeted directly at an environmental problem. Where consumption of the good causes pollution, an import barrier reduces consumption, but will also create a distortion that favors domestic production of that good. By contrast, a consumption tax will reduce consumption without creating an incentive to expand domestic production of that good.

Export barriers (taxes) are often levied to raise revenue and finance price stabilisation and marketing activities. Exchange rate devaluation is equivalent to an export tax removal. Again, the environmental impact of export barriers is sector specific. **A priori,** it is difficult to say whether the long run response to export taxes will beneficially or adversely affect the environment. The arguments that apply to import tariff liberalization also apply to export taxes.

### 2.3.3 Debt

The severity of the debt problem cannot be measured by the absolute level of the debt burden. Debts are only problematic when repayments become difficult due to poor economic performance and foreign exchange constraints. It is for this reason that they are often regarded as a symptom rather than a cause of economic mismanagement or structural imbalance. For example, the newly industrialized countries in South East Asia incurred significant levels of external debt in the 1970s when funds were readily available and interest rates were low. However, they have not run into the same financial difficulties as many countries in Africa and Latin America. Their success has partly been due to restructuring and diversification of their economies together with higher growth rates and greater access to foreign exchange via expansion in exports.

Once a country moves into a financial crisis situation, it is possible that the financial distress induces an economic response that may be in direct conflict with longer term economic and environmental objectives. In such circumstances, debt itself becomes the source of further economic and environmental stress as it actually changes economic behavior. Both bankers and their debtors may give priority to their survival over other long term objectives, including profitability.

The empirical evidence on this issue is unclear. A cross section of 45 countries over the period 1967-1985, Capistrano and Kier (1990) showed that there was a significant relationship between the rate of industrial logging and the debt service ratio, the real rate of devaluation and the export price of agricultural crops. Although these studies do not establish any causality between these variables, the evidence cannot be ignored as indicative of a possible relationship between debt and the environment.

Debt for environmental projects or sustainable development policies swaps could be a win-win proposition if the relief is used both as a stabilisation and a structural adjustment tool. The underlying principle is that a portion of a country's external debt is bought at a discount on the secondary loan market. The market value of the debt is likely to be considerably below the face value, depending on the risk of default attached to the debt. The local currency equivalent of the debt is

then set aside for investment in environmental projects or to undertake policy reforms beneficial to the economy and the environment. Such projects have been successfully implemented, although their overall impact remains small.

### 2.3.4 Poverty, Population and Environmental Degradation

*The poverty-environment nexus*

The link between poverty and the environment is important because many macro policies have an impact on poverty and thereby, it is claimed, on the environment. This section looks at some of the evidence on this issue. Poverty and environmental degradation can be studied by looking at how the two are correlated over time, as well as how they are correlated across society at a given point in time. For the latter, there are few serious cross section studies. In one of the earlier efforts in this area, Jaganathan (1989) reports on a study between rates of deforestation and the level of poverty in West Java, which finds little evidence that poverty was a driving force in the deforestation of that region. Likewise, the main factors driving land use change in two regions in Nigeria were infrastructure projects such as roads, market incentives, such as increased producer prices for food, and institutional factors that determined access to land. Poverty as such appears to have played little part.

There are studies that point to the role of agricultural expansion by (generally) poor farmers as the cause of deforestation and other environmentally damaging land use changes (Southgate 1988, Mink, 1993). The argument is that small (poor) farmers tend to occupy fragile lands. If there is any increase in the number of such framers, or if peasants are displaced from other occupations, they will occupy more of such lands. But there is no empirical evidence to suggest that it is an increase in the level of poverty per household is the determining factor. Rather, it is because of an increase in the number of poor households, either through population increase, or through policies that increase landlessness, that the problem may arise.

Furthermore, researchers in this area correctly stress the importance of the policy context in determining the nature of the agricultural expansion and its environmental impacts. Land that is cleared for agriculture can be managed in a sustainable way, or it can be 'mined' and then abandoned in a degraded state. What happens depends on the property management regimes that are in place and whether they provide the right incentives for proper land management. These could be through secure private property rights, but could also be through common management systems (Bromley, 1991, Dasgupta and Mäler, 1990). Where the management regime is secure the impacts on the environment are not necessarily negative and may even be positive. One study from Nigeria (Mortimore, 1989) shows how small farmers there adopted sustainable management strategies on new land even when the short term costs of doing so were high.

*The population-environment nexus*

As with poverty, for a fuller appreciation of macro policy impacts it is important to look at the linkages between population and the environment. There is a widespread perception that high population density and high rates of population growth are a direct cause of environmental degradation. Some of the poorest countries in the world are also densely populated. Statistics on rates of fertility and mortality indicate rapid growth in human populations over the coming decades, especially in the developing countries. High rates of growth imply increasing demand for employment, education, health and many other public services. Population growth also implies an increasing demand for natural resources. The question arises, can the environment bear the strain of ever increasing human pressures?

There is little doubt that increases in population exacerbate the pressure on natural resources. The latter are fixed and as more people make demands on them, the quality and possibly the volume of the resources declines. Hence policies that reduce the rate of population growth will, *ceteris paribus,* reduce the pressure on the natural resource base. As far as macroeconomic policy is concerned, the key issue is to ensure that resources are efficiently devoted to reducing fertility, especially where it is having a detrimental environmental effect. In this context, there is considerable evidence to suggest that education (particularly of women), the level of agricultural employment and level of nutrition and the extent of civil liberty all act to reduce the levels of total fertility (Sen, 1994; Dasgupta, 1995). Policies, therefore, that act to improve these factors can be expected to reduce total fertility and, thereby, pressure on the natural resource base. Macroeconomic adjustment measures should therefore be sensitive to these links.

In addition, general economic growth has been negatively associated with population growth and it has been argued that the former will act to reduce population pressures over time. The problem with this argument, however, is that while *average* population growth rates may decline with per *capita* GDP, sections of the community that depend on natural resources may find themselves locked into a cycle of poverty in which high fertility rates are maintained and that, in turn, exacerbate the pressure on the natural environment. Dasgupta (1995)

has argued that this cycle could work in the following way. As common resource management systems break down, so individuals are more able and willing to make family size decisions that do not take full account of the social costs of child rearing, with the use of common resources treated as a free good. Over time, the natural resource base is increasingly depleted and the family unit requires more members to achieve the same level of welfare. Thus a cycle of increasing degradation is established. The theory has some plausibility but clearly needs to be tested with real data.

***The dynamics of institutional change***

It was noted earlier that at the heart of the environment-poverty relationship is the question of what management systems operate for natural resources and how they evolve over time. There is some evidence that, in pre-industrial societies, natural resources have been managed with some degree of sustainability in cohesive communities. With the onset of major social and political changes in the post-war period, these systems have, for many communities, broken down. Different reasons are offered for this. One is the process of privatisation, which reduced the common land available and destroyed existing systems (Dasgupta, 1996). Another is nationalisation of common lands (particularly forests) where the government took responsibility for the lands but was not able to control its exploitation (Bromley, 1991). A third is agricultural expansion for large-scale production, as in the case of Brazil, Kenya and other countries (Panayotou, 1993). A fourth is simply the pressure of population increase, which may result from families following traditional norms for family size even when mortality rates have declined. In all these cases, the argument runs, traditional systems have broken down and common resources over-exploited.

The situation is, however, dynamic and one would expect institutions to reform themselves where it is in the common interest to do so. There is some evidence that this happens, albeit over a medium to long time frame. Taking data from Western India, Chopra and Gulati (1996) have shown that property rights have evolved in such a way so as to reduce out-migration and improve the management of common resources. Although this is an interesting and fascinating paper, more work needs to be done to understand the dynamics of institutional change for agricultural communities in developing countries.

Again, the macroeconomic policy issue is to ensure that resources supporting the necessary institutional changes are made available. What can be done so as to support the move of these institutions in a positive and sustainable direction? There are a few suggestions. Education again is seen as an important influence, as is empowerment of women. These will result in a 'new order' but one that is more sensitive to the environmental constraints and the imperative of co-operation in the management of natural resources. Another is information and public education in general[1]. A third is legal and other government support for new and reformed property management systems. All these responses are of great importance in setting the right policy framework. Unfortunately the state of knowledge about the dynamics of institutional change, on which to base them, remains weak. Again macroeconomic reforms should take account of these factors.

## 3 METHODS OF MACROECONOMIC ANALYSIS

The analysis of environment-economy linkages is carried out using techniques essentially developed for a better understanding of the relationships between different sectors of the economy. Broadly speaking these fall into three groups: intersectoral input-output models, less restrictive general equilibriurn models more limited macroeconomic/qualitative models.

### 3.1 Input-Output Analysis[2]

Input-Output analysis (IO) focuses on the relationships existing between different sectors in a complex economy. The overall objective of IO analysis is to determine the total output required to meet the total demand. The total demand is made up of the intermediate consumption and the final demand. It is possible to incorporate environmental impacts into the IO model, as demonstrated in Table 1 below. The economy is described as a table of transactions. The horizontal rows are the outputs produced by the sectors and the vertical columns are the inputs received. Each cell gives the physical quantity of each input required by a specific sector.

In the last row and last column (sector 3) an environmental resource is considered as a sector which provides inputs (row 3) for agricultural and industrial activities, but does not require agricultural and/or industrial inputs for its production. It is important to notice that the total output of each sector includes both the intermediate and the final demand of each sector in the economy. This model can be used to calculate the quantities of environmental resources necessary for satisfying human activities, including both productive

[1] As Dasgupta (1995) notes, 'Female education is thus a potent force for creating tradition breakers, as are earning opportunities for women' and 'A more direct route would be social co-ordination through education and public exhortation. In this context the role of television and radio in transmitting information about other life styles can be especially important.

[2] This section draws on Costa and Marangoni (1995).

**Table 1: Integrating Pollution in an I-O Model**

| *Output from Producing Sector* | Sector 1 (Agric.) | Sector 2 (Industry) | Sector 3 (Water Pollution) | Final Demand | Total Prod. |
|---|---|---|---|---|---|
| Sector 1 (Agriculture) | 25 | 20 | 0 | 55 | 100 |
| Sector 2 (Industry) | 14 | 6 | 0 | 30 | 50 |
| Sector 3 (Pollution) | 50 | 10 | 0 | 0 | 60 |

activities and final consumption. A further advantage is that the model allows for the inclusion of prices and an analysis of the cost of mitigation measures[3].

Such models can use information on the flow of resources between different sectors of the economy to answer a range of questions in a medium/long term planning context:

- What are the implications of alternative growth paths on the demand for key commodities such as energy and key factors such as labour;
- What are the regional implications of alternative development policies;
- If resources are likely to be limited for investment, which sectors are the 'priority' ones to give the greatest benefit in terms of economic growth. The same applies if key commodities are also in short supply, such as oil. I-O models have been used to identify the key bottlenecks in the economy that need to be cleared for maximum efficiency to be obtained from a given supply.

These models have also been adapted to examine the environmental implications of medium/long term economic strategies in developed countries. In principle, one could identify the key pollution outputs associated with economic activities, the key pollution abatement sectors and the key constraints in terms of the sink capacity of the environment and incorporate them within an input-output framework to answer question similar to the above but with respect to the environment. Where should scarce investment resources be allocated if there are environmental as well as economic objectives to be satisfied, and how should future economic development be influenced by environmental constraints as well as economic ones?

[3] Most commonly the cells give the inputs in value terms and the analysis is done at constant prices. For the environmental sector this poses the problem of valuation, which we address in Part II. For a more in-depth analysis on methods to include environmental concerns in the Input-Output framework the reader is referred to a seminal paper by Leontief (1970).

The difficulty with this approach is essentially the lack of good recent data. I-O tables exist for most developing countries but they are often out of date and there are rarely enough data on the environmental flows, in the comprehensive and systematic manner required, for them to be useable within this modelling framework. The rigidity of the technique does not lend itself to being able to use partial and incomplete information in a useful way

## 3.2 General equilibrium models

Input-output models incorporate a number of restrictive assumptions about technology that are sometimes considered to lead to invalid conclusions to the sorts of questions raised above. The development of non-linear modelling techniques with a more flexible structure with regard to the number of sectors and the relationships between them has been developed over the last 20 years. Such models (referred to as computable general equilibrium models (CGE)) have reached the stage of being easily applicable with relatively limited computing equipment.

General equilibrium models take, as their point of departure, the data on the demands and supplies of all the goods and services in the economy to the different agents. This information is summarised, partly in the input-output tables referred to above, and in the 'social accounting matrices' which give a breakdown of how the inputs and outputs in an economy are distributed across different groups of individuals in society. The modeller then constructs aggregate demand and supply functions for the different outputs (e.g. manufacturing, services, and agriculture) and inputs (labour, capital, and raw materials). Each aggregate demand and supply can be made up of components (e.g. domestic demand and exports and domestic supply and imports), and is a function of relative prices of all inputs and outputs, as well as income and other economic factors). The model solves for the prices that equate supply and demand in all the markets. Some of the prices are determined outside the system (e.g. world prices for traded commodities). The model can by 'dynamic' in the sense that it solves for prices over time, with the prices in one period determining the levels of some inputs such as capital in the next period.

For environmental problems specific advances have been made in the use of CGE models in recent years. Although such models represent an important tool, one should be aware of their limitations. The key problem is that there are many parameters of the models that cannot be determined empirically with any real accuracy. Hence values have to be assigned to them on a somewhat

arbitrary basis and the results of using them are subject to considerable uncertainty. This can be avoided to some extent by looking for the sensitivity of the policy conclusions to the values of key parameters. Nevertheless a broad area of uncertainty can remain. Another difficulty is that given their complex structure, it is hard to see why certain impacts arise when taxes or other parameters are changed. This makes it unconvincing when explaining the results to a policy-maker.

## 3.3 Macro economic models

Undoubtedly the most frequently used models for economic management are the short-term macroeconomic models linking the key aggregates such as income, consumption, investment etc. Such models are frequently used for forecasting short term; movements in the economy, as well as in analysing the implications of alternative fiscal and monetary policies. Formally such models are formally structured, with equations relating the movements of key variables, and the latter divided into two groups: those whose values are determined within the model (endogenous variables) and those whose values are to be fed into the model from outside (exogenous variables).

More recently, however, the policy analysis of alternative policies has been carried out in a less formal modelling framework. With the help of the World Bank and the IMF, finance ministries in developing countries have adopted an 'inter-sectoral' accounting framework to look at the key questions relating to the impacts of alternative policies on different sectors. Such a framework, which can usually be set up in a spreadsheet, would consist of projected national accounts for the economy by production sector, and sectoral accounts for the government budget, the external account, the private account and the monetary sector. Further sectors can be added as appropriate. For example, if one was analysing the impacts of investments in the energy sector, an energy sector as well as a set of accounts for the key entities in that sector could be appended. The 'model' consists of tracing the flow of funds between the sectors and ensuring consistency in the values of the variables. In terms of predicting future values of these variables, the distinction between the endogenous and exogenous variables remains, except that the endogenous variables are now those determined by the accounting framework in the model and the exogenous ones are those determined outside the framework, including any that are forecast from other macroeconomic models.

Could such models be used for the analysis of environmental questions? Given their flexibility the answer is almost certainly yes, partly in terms of the environmental impact of economic policy but more effectively in terms of the economic implications of environmental regulation. The former may be illustrated with the example of the forestry sector. One could develop a physical accounting framework for such a sector relating stocks and flows of different timbers and a link between the level of harvesting and the economic incentives for doing so. Every time there was a change in a policy instrument, such as export taxes or the exchange rate, its implications could then be worked out for this sector.

The more useful applications of such models, however, lie in dealing with the second linkage between the economy and the environment - viz. the economic impacts of environmental regulation. If taxes and charges are imposed on certain sectors of the economy for environmental reasons, the framework allows one to trace through their implications on the government budget account, the external account and the private sector.

## 3.4 Action Impact Matrices

Another general analytical tool used to help identify important problems is the Action Impact Matrix (AIM). The AIM is a tool for policy analysis and coordination and helps identify important problems and the links between sectors and policy decisions and environmental outcomes. It is a more qualitative tool than some that we have discussed in this chapter but, since it is broader, the link between policy change and environmental impact can be explored in a wider context.

Economic and environmental analyses and policies may be used more effectively to achieve sustainable development goals, by linking and articulating these activities explicitly. Implementation of such an approach would be facilitated by constructing an Action Impact Matrix (AIM)—a simple example is shown in Table 2, although an actual AIM would be very much larger and more detailed (see Annex I). Such a matrix helps to promote an integrated view, meshing development decisions with priority economic, environmental and social impacts. The left-most column of Table 2 lists examples of the main development interventions (both policies and projects), while the top row indicates some of the main sustainable development issues. Thus the elements of cells in the matrix help to: (a) explicitly identify the key linkages; (b) focus attention on valuation and other methods of analysing the most important impacts; and (c) suggest action priorities. At the same time, the organization of the overall matrix facilitates the tracing of impacts, as well as the coherent articulation of the links between a range of development actions— that is, policies and projects.

A stepwise procedure, starting with readily available data, has been used effectively to develop the

AIM in several recent country studies including Ghana and Sri Lanka. This process has helped to harmonize views among those involved (economists, environmental specialists and others), in the country concerned and the Bank, thereby improving the prospects for successful implementation. First, data from NEAPs, EAs, and so forth may be organized into an environmental issues table that prioritise these problems, provides quantitative or qualitative indicators of damage, and helps identify underlying economic causes. Second, using information readily available from country economic and sector work, the main economy-wide policies (current and intended) are set out in a second table, together with a brief review of the basic economic issues that they address and potential environmental linkages. The information from these two tables are then combined to develop a preliminary Action Impact Matrix. In this table examples of typical policies and projects as well as key environmental and social issues are shown. Some illustrative but qualitative impact assessment are also indicated: thus + and - signify beneficial and harmful impacts, while H and M indicate high and moderate intensity. The AIM process helps to focus on the highest priority environmental issues and related social concerns

Note that under complementary measures, we are thinking of market-based instruments such as effluent charges, tradable emission permits, emission taxes or subsidies, bubbles and offsets (emission banking), stumpage fees, royalties, user fees, depositrefund schemes, performance bonds, and taxes on products (such as fuel taxes). Non-market based mesures comprise regulations and laws specifying environmental standards (such as ambient standards, emission standards, and technology standards) which permit or limit certain actions ("dos" and "don'ts").

One of the early objectives of the AIM-based process is to help in problem identification- by preparing a preliminary matrix that identifies broad relationships, without necessarily being able to specify with any accuracy, the magnitudes of the impacts or their relative priorities. For example, in Table 2, a currency devaluation may make timber exports more profitable and lead to deforestation of open access forest. The appropriate remedy might be to strengthen property rights or restrict access to the forest areas. A second example might involve increasing energy prices toward marginal costs to improve energy efficiency and decrease pollution. Adding pollution taxes to marginal energy costs will further reduce pollution. Increasing public sector accountability will reinforce favourable responses to these price incentives, by reducing the ability of inefficient firms to pass on cost increases to consumers or to transfer their losses to the government. In the same vein, a major hydroelectric project is shown in Table 2 as having two adverse impacts- inundation of forested areas and villages, as well as one positive impact - the replacement of thermal power generation (thereby reducing air pollution). A reafforestation project coupled with adequate resettlement efforts may help address the negative impacts. The matrix-based approach therefore encourages the systematic articulation and coordination of policies and projects to achieve sustainable development goals. Based on readily available data, it would be possible to develop such an initial matrix for many countries. Furthermore, a range of social impacts could be incorporated into the AIM, using the same approach.

This process may be developed further to assist in analysis and remediation. For example, more detailed analyses may be carried out for the subset of main economy-wide policies and environmental impact links identified in the cells of the preliminary matrix. This, in turn, would lead to a more refined final matrix, which would help to quantify impacts and formulate additional measures to enhance positive linkages and mitigate negative ones. The more detailed analyses which could help to determine the final matrix would depend on planning goals and available data and resources. They may range from the application of conventional sectoral economic analysis methods (appropriately modified in scope to incorporate environmental impacts), to fairly comprehensive system or multisector modelling efforts.

## 3.5 Economic Instruments for Integrating Environmental and Macroeconomic Policies

The discussion so far has shown that macroeconomic policies can have a detrimental effect on the environment, especially in the short run. They can also have many positive effects, as reforms bring about economic stability and prosperity. Of course, such prosperity will bring with it new and different environmental problems, but no one seriously suggests that growth should be curtailed on the grounds that such problems will occur.

Where there are conflicts between macroeconomic policies and environmental goals, how should they be resolved? While it is important to know what the environmental consequences of major macroeconomic policy reforms are, it is almost impossible to think of arresting such reforms on the grounds that they have unfavourable environmental consequences. The imperative for such reforms is simply too great - it is the need to ensure national survival and long-term economic development.

Hence the only ways in which the environmental objectives can be reconciled with the macroeconomic objectives is to introduce more instruments to deal with the environmental dimension. Following the rules of the theory of planning as set out by Tinbergen, one needs as many instruments as there are objectives. With the environment, one has added a further objective - or set of objectives - which have to be met at the same time as the environmental reforms are taking place. What form should such instruments take? They could include economic measures such as environmental charges and taxes, or legal measures, such as pollution licences, property reforms etc. The general choice of instruments is discussed in Chapter 3, but a few general propositions can be advanced here.

First, when further instruments are introduced one cannot generally 'assign' instruments to objectives. The classical case is when policy makers assign monetary policy to control inflation and fiscal policy to control unemployment. Both policies of course interact and affect both inflation and unemployment. In the case of the environment, devaluation could result in increased deforestation and a possible additional instrument would be export controls on forest products. The levels of such controls have to be taken in conjunction with the level of devaluation, as both have an effect on the natural resource

**Table 2: Simple Example of an Action Impact Matris (AIM) for Sri Lanka**

| Activity/ Policy | Main Objective | Impacts On Key Sustainable Development Issues | | | |
|---|---|---|---|---|---|
| | | LandDegradation | AirPollution | Resettlement | Othe |
| 1. Macroeconomic & Sectoral Policies | Macroeconomic and Sectoral Improve-ments | Positive Impacts Due to Removal of Distortions<br>Negative Impacts Mainly Due to Remaining Constraints | | | |
| • Exchange Rate | • Improve Trade Balance and Economic Growth | (-H)<br>(deforest open access areas) | | | |
| • Energy Pricing | • Improve Economic and Energy Use Efficiency | | (+M)<br>(energy efficiency) | | |
| • Other | | | | | |
| 2. Complementary Measures | Specific/Local Social and Environmental Gains | Enhance Positive Impacts and Mitigate Negative Impacts (above) of Broader Macroeconomic and Sectoral Policies | | | |
| • Market Based<br>• Non-Market Based | • Reverse Negative Impacts of Market Failures, Policy Distortions and Institutional Constraints | (+H)<br>(property rights) | (+M)<br>(pollution tax)<br>(+M)<br>(public sector accountability) | | |
| 3. Investment Projects | Improve Effciency of Investments | Investment Decision Made More Consistent with Broader Policy and, Institutional Framework | | | |
| • Project I (Hydro Dam) | • Use of Project Evaluation (Cost Benefit Analysis, Environmental | (-H)<br>(inundate forests) | (+M)<br>(displace fossil fuel use) | (-M)<br>(displace people) | |
| • Project 2 (Re-afforest and relocate) | Assessment, Multi-Criteria Analysis, etc.) | (+H)<br>(replant forests) | | (+M)<br>(relocate people) | |
| Project N | | | | | |

*Source: Munasinghe 1993b.*

sector and the gains from devaluation. A total ban on forest products, for example, could reduce the benefits of the devaluation by restricting the export of products made with domestic wood. That said, the **goal** is to find instruments that are able to be set **more or less independently of the macroeconomic instruments,** simply because it is too difficult to get the macroeconomic policy-makers to adjust their instruments on the grounds of the environmental objectives.

Second, in setting the environmental instruments one should try not to introduce other distortions in the economy. Taking the same example as above, an export tax on logs will make domestic prices different from world prices and thereby create inefficiencies in the economy. A better instrument would be to ensure that forest loggers pay the full social cost of their actions. Then there would be no need to regulate exports. The latter is called a **Second Best** policy, because it introduces further distortions while solving one set of problems. Where second best policies have to be followed one has to analyse the gains and losses to different sectors rather carefully. There are, however, some general rules. One is in relation to trade. Here the theory of the Second Best establishes that there almost always exists a domestic economic instrument, the use of which will be superior to the importation of a tariff or other trade restriction. Thus, for example, if the manufacturing sector is unionized and wages are above marginal costs, taxes on labour or subsidies on capital are generally preferable to export subsidies or import tariffs. Similarly with a production externality a production tax is preferred to trade interventions (Bhagwati, 1984). This powerful result implies that, where alternative instruments are available, environmental impacts should be dealt with through the use of domestic economic instruments. What are the instruments available for environmental regulation that would be preferable to the use of trade restrictions? There is a substantial literature, which shows how environmental costs can be internalised through the use of economic and non-economic instruments (Tietenberg, 1996).

The third general proposition relates to way in which economic growth can eventually resolve the environmental problems of a country, and that some degradation is the inevitable cost of the pursuit of such growth through various macroeconomic policies discussed above.

There is a strand of literature (Grossman and Kruega, 1991, World Bank, 1992,), which suggests that the relationship between GDP and the quality of the environment is 'U-shaped' — i.e. the quality of the environment deteriorates initially as GDP *per capita* increases, and then improves after a certain critical value of *per capita* GDP has been reached. This critical value varies with the pollutant, and indeed for some pollutants such as VOCs there is no 'turning point'.

This model (also referred to as the 'Environmental Kuznets Curve') can be looked at in conjunction with the original Kuznets curve, which postulated a deterioration in income distribution in the early stages of economic growth, followed by an improvement later. Taking the two together one would conclude that a declining environmental quality and increasing income inequality go hand in hand as part of the 'development process'. In the end it will all work out fine, with improvements in both these indicators of human welfare.

Unfortunately, such a sanguine view is inappropriate and misleading from a policy viewpoint. First, some of the environmental degradation being observed, and sometimes being caused by extreme poverty, is irreversible and will never be recovered. Second, what is a long-term time series relationship is being inferred from cross section inter-country data. There is no reason why a particular country should follow the path characterised by a cross section of countries. Indeed the aim should be to follow a policy based on a comparison of domestic costs and benefits of different options, taking account of their impacts on all aspects of welfare, including poverty/inequality, environmental quality, GDP and other indicators such as those used by the UNDP in its Human Development Reports. The existence of the Kuznets curve is of little relevance in determining such a set of policies.

## 4 CONCLUSIONS

The macroeconomic policies that are essential to achieve stability and growth have environmental effects, some of which are negative, especially in the short run. These effects have to be analysed on a case by case basis, as the impacts are complex and depend on the institutional and regulatory framework that is in place. The longer-term structural adjustment policies are more likely to have positive environmental effects, as they encourage the better pricing of resources.

Where the macroeconomic policies do have negative environmental effects it may be desirable to introduce additional instruments to protect the environment. Ideally the instrument chosen should be the one that internalises the full environmental costs but this is not always possible. In the absence of that, one has to select the instrument in a 'second best' framework. Details of such instrument selection are complex, but a general guide is that one should try and avoid trade restrictive measures if domestic policies to achieve the same goals are available. The impacts of trade liberalisation are some of the most controversial in terms of the environment, but most of the recent research shows

that it should be possible to achieve the benefits of liberalisation while protecting the environment. This can be done with one or more of the range of environmental regulation instruments that are available. The instruments are discussed further in Chapter 3.

Other areas where environmental and macroeconomic polices are in conflict are those of debt and poverty/population. There is some evidence that debt is correlated with high rates of deforestation, but it is not clear that the causal effect is high debt results in high deforestation. It could be that countries with a large level of forest activity borrow more internationally. The links between poverty and environment are also more complex. Macroeconomic policies that increase poverty are to be avoided for obvious reasons but it is not always the case that they damage the environment. It depends on what the arrangements are in place for the management of these resources.

Pressures from growing population on the environment are well documented and it is desirable that macroeconomic reforms allocate sufficient resources to measures that result in reduced fertility. These are, primarily women's education and empowerment and better agricultural employment. Support for institutional reforms that reflect changing realities on the ground is also critical.

The methods of analysis for the macroeconomic problems include input-output models, macroeconomic models, Action Impact Matrices and computable general equilibrium models. Of these the last are the most comprehensive in their evaluation of the effects, but they are also the most demanding in terms of information. Sometimes a simpler sectoral model or a macroeconomic model may provide better information for the policy maker.

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# Chapter 3

# PROJECT APPRAISAL TOOLS
# COST BENEFIT ANALYSIS AND COST EFFECTIVENESS ANALYSIS

*by*

*Richard Boyd*

## 1 INTRODUCTION

*"One can view cost-benefit analysis as anything from an infallible means ; of reaching the new Utopia to a waste of resources in attempting to measure the unmeasurable".*

Prest and Turvey (1966) ;

The purpose of the economic analysis of projects is to bring about a better allocation of scarce resources, leading to enhanced incomes for investment or consumption (ADB, 1999). In some cases, decisions are required within the economy to ensure that projects selected for investment meet a minimum standard for resource generation, and to weed ; out those projects that do not. In other cases, decisions are needed. sometimes within projects, between different means of achieving the same objectives. Economic analysis is used here to choose the means employing the least resources for a given output. All resource inputs and outputs have an opportunity cost through which the extent and value of project items are estimated. In short, projects should be chosen where the resources will be used most effectively. To aid project selection in this context, the decision-maker can draw upon two economic project appraisal tools - namely cost-benefit analysis and cost-effectiveness analysis. The purpose of this Module is to provide the reader with a practical introduction to these tools, with an emphasis on applications in an environmental context.

The Module is structured as follows. Section 2 looks at cost-benefit analysis (CBA). After defining CBA the main discussion is divided into three sub-sections, reflecting the key stages or activities in conducting CBA - these are STAGE 1 - Identification and Quantification of Project Effects, STAGE 2 - Valuation of Project Impacts and STAGE 3 - Apply Project Selection Criteria and Make Decision. The main alterative project appraisal tool to CBA, cost-effectiveness analysis (CEA) is considered in Section 3. Within this section the purpose and scope of CEA is defined ; and methods for measuring cost-effectiveness outlined. The presentation of the results of CEA in cost cu~ves is also described.

## 2 COST-BENEFIT ANALYSIS

### 2.1 What is Cost-benefit Analysis?

**Cost-benefit analysis** (CBA) can mean different things to different people. This divergence of understanding is particularly evident between economists and non-economists. A useful starting point is therefore to develop an understanding what is meant by the term CBA. To this end a collection of definitions of CBA from the literature are provided below:

- "...considers all gains (benefits) and losses (costs) regardless of to whom they accrue, although usually confined to inhabitants of one nation. . ." (Pearce, 1992);
- "...cost-benefit analysis defines costs and benefits in a particular way, and it goes beyond the idea of an individual's balancing of costs and benefits to society's balancing of costs and benefits..." (Turner, Pearce and Bateman 1994);
- "...the fundamental role of CBA is to establish principles by which the costs and benefits of any public programrne are measured..." (Hartwick and Olewiler, 1986)
- "...CBA is designed to show whether the total benefits of a policy or project exceed the costs. including environmental benefits and costs. As far as possible ... all effects are measured as the persons affected would measure them..." (Abelson, 1997)
- "...decisions are made by decision makers, and benefit-cost analysis is properly regarded as an aid to decision making, and not the decision itself..." (Zerbe and Dively, 1994);
- "...general premise that benefits and costs of actions should be weighted prior to deciding on a policy choice " (Tietenberg, 1992);
- " process of identifying, quantifying, weighing up, and reporting costs and benefits of the measures which are proposed to implement a policy " (UK DoE, 1991).

Based on the above quotes, we may define CBA as a set of analytical procedures for measuring and

comparing the advantages (benefits) and disadvantages (costs) of a project or policy. Specifically, CBA is designed to show whether the total benefits of a project exceed the total costs (i.e. does the project have a net benefit?).

Two further points can be drawn from our collection of quotes, both of which should be borne in mind.

- Traditionally, the boundaries of CBA are set at a **national level,** i.e. CBA typically considers the (net) impact of a project on national 'welfare' or 'well being'.
- The purpose of CBA is to **inform** the decision-making process, and it may be used in conjunction with other decision support tools, e.g. Environmental Impact Assessment (EIA) or Multi-criteria Analysis (MCA), to reach a decision.

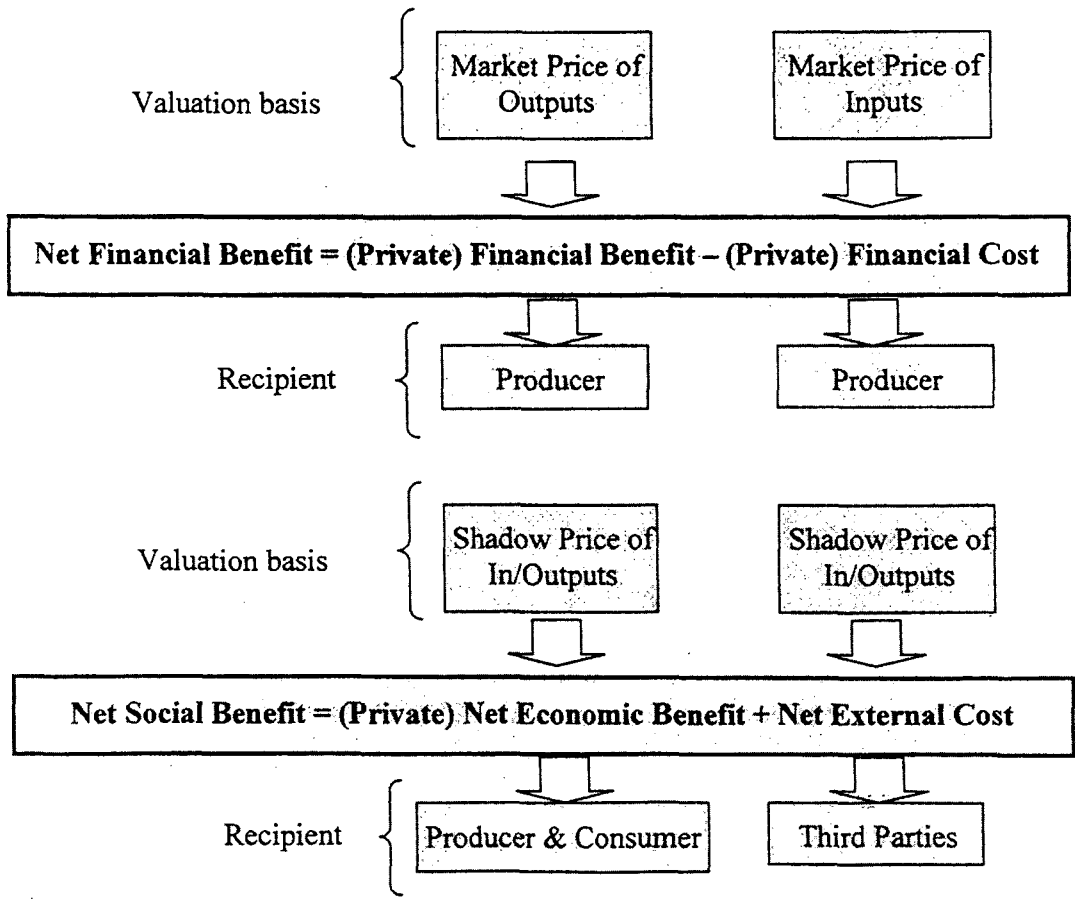
### 2.1.1 Distinguishing Between Financial Project Appraisal and CBA

Cost-benefit analysis is similar in form to the financial analysis of projects: both appraise the net benefits (profit) of a project. The concept of financial net benefit is not the same as economic net benefit. The financial analysis of a project estimates the net benefit accruing to the project operating entity (or to the project participants), whereas CBA measures the effect of the project on society as a whole.

Both types of analysis are conducted in monetary terms, the major difference lying in the definition of costs and benefits (and hence net benefits). The difference between the two concepts of net benefit is illustrated in Figure 1 below.

In short, with financial project appraisal, the inputs (e.g. equipment, labour, energy, etc.) that a project-operating entity must purchase for a project, and the output (goods or services) that these inputs generate for the project-operating entity, are valued using their **market prices**. The term **'private'** refers to the fact that the estimated costs and benefits accrue directly to the project-operating entity and therefore have a direct bearing on any decisions made (i.e. they are **internal** to the decision making process). CBA also values the resource inputs to, and the outputs from a project. However, CBA does

**Figure 1: The Distinction Between 'Social' Net Benefit and 'Financial' Net Benefit**




**Source:** Boyd (forthcoming)

not restrict the analysis solely to those impacts accruing directly to the project-operating entity. CBA is also concerned with the effect project inputs/outputs have on society as a whole. Typically these effects are **external** to the decision-making process - i.e. they are not recognised by the project-operating entity. These so-called **'externalities'** do nonetheless affect society's welfare or 'well- being', and are thus relevant to CBA

Furthermore, the valuation basis of CBA differs from that of financial appraisal. CBA measures a project's dis/advantages in terms of **willingness-to-pay** (WTP) for units of increased consumption, and to **willingness-to-accept** (WTA) compensation for foregone units of consumption[1] WTP and WTA compensation are used rather than market prices actually paid or received because (see Section 2.3.2 below):

- many project effects that are marketed will be bought and sold in markets where prices are 'distorted'; and
- many project effects that are to be included in CBA either will be non-marketed, for example, biodiversity preservation, or incompletely marketed, such as, water supply and sanitation benefits (ADB, 1999).

CBA employs so-called **'shadow prices'** to estimate WTP and WTA compensation values when markets are distorted or absent. With respect to environmental resources - which are typically non-marketed - shadow prices are estimated using one of the specialist valuation methods (e.g. contingent valuation, travel cost, hedonic pricing, etc.).

The different valuation basis also allows CBA to distinguish between private benefits accruing to producers and those accruing to consumers.

### 2.1.2 CBA Methodology

The basic approach to CBA may be divided into three main activities or stages. These are outlined below and shown in Figure 2 - note that each of the stages consists of a sequence of interrelated steps:

- **STAGE 1** - The process of identifying all parties affected by the project and quantifying the 'additional' impact of the project on these parties.
- **STAGE 2** - The process of attaching an appropriate 'price tag' to all economically relevant impacts.
- **STAGE 3** - The process of discounting (at an appropriate discount rate) all costs and benefits that occur in different time periods so that the present value net benefit of the project can be determined and a decision can be made on the relative economic merit of the project.

Each stage is considered in turn below.

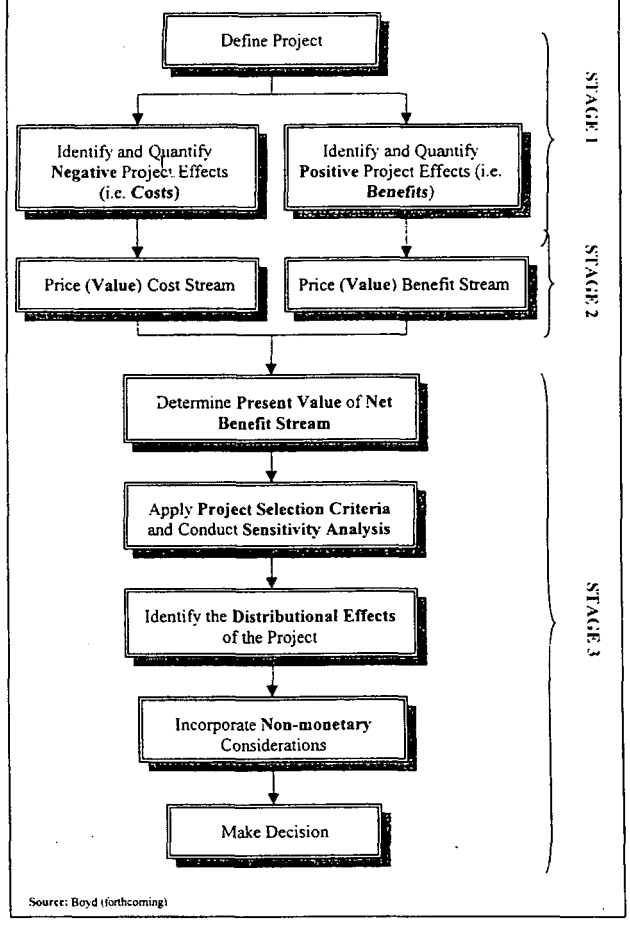
## 2.2 STAGE 1 - Identification and Quantification of Project Effects

### 2.2.1 General

To identify project disadvantages (costs) and advantages (benefits) the situation without the project must be compared with the situation with the project. This is reflected in STAGE 1 above, where we make reference to the **'additional'** advantages and disadvantages of a project. In any form of project appraisal, whether strictly financial or otherwise, costs and benefits are always measured additional, or relative, to some type of baseline. The term **'baseline'** refers to the situation assumed to exist in the absence of the project or policy in question. The baseline is also referred to as the **'business-as-usual'** or **'reference'** scenario.

The baseline scenario (without project situation) is not the same as the before-project situation. The baseline scenario can sometimes be represented by the

**Figure 2: Methodological Framework for Cost-benefit Analysis**




Source: Boyd (forthcoming)

[1] Consumption is defined broadly - consumption may apply equally well to a society's WTP for preservation of plant or animal species, as to society's WTP for the consumption of agricultural produce or clean drinking water.

present levels of productivity of the relevant resources. However, present levels of productivity would frequently change without the project, and this should be taken into account in defining the without project situation.

For example, consider a project to gradually replace diesel-powered buses in an urban area with alternative fuelled buses, which run on compressed natural gas (CNG). (Assume the objective is to improve urban air quality.) The replacement programme defines what we call the 'project' scenario. Your job as a project analyst is to assess the economic merit of the replacement programme. In doing so however, it is wrong to look at the total costs of the replacement programme in isolation (e.g. the cost of new buses, refuelling infrastructure, CNG, etc.), since in the absence of the project, the city would continue to incur the costs of using the diesel powered buses. So, while incurring costs to introduce new CNG-powered buses, the city simultaneously saves the costs associated with running the old diesel buses. Failure to account for the costs that would be incurred under the baseline scenario would lead to overestimation of the true costs of the project scenario, thereby reducing its attractiveness. The same rationale applies equally to the assessment of project benefits.

The relationship between the baseline scenario, project scenario and the measurement of incremental costs and benefits is summarised as follows:

| Project (with) scenario: | | Baseline (without) scenario: | | Project appraisal (CBA): |
|---|---|---|---|---|
| Total benefits | - | Total benefits | = | Additional benefits |
| - | | - | | - |
| Total costs | - | To costs | = | Additional costs |
| = | | = | | = |
| Total net benefits | - | Total net benefits | = | Additional net benefits |

Clearly, the additional costs and benefits of a project greatly depend on the assumptions underlying the baseline definition.

## 2.2.2 Identification and Quantification of Costs

The total social cost of a project or policy intervention includes the 'true' private cost of all resources used by the provider (project-operating entity) of the project over some pre-defined time horizon (usually the operating life of the project[2]), plus any costs imposed on third parties (i.e. the externalities).

The two main categories of (private) costs, which may need to be considered when estimating the cost of a project, are investment expenditures and recurring costs. **Investment expenditures** include initial investments to implement the project, replacement investments during the life of the project, and the residual value of investment assets at the end of the project[3]. Initial investments are generally broken down into subcategories, such as land preparation, buildings and construction, equipment, vehicles, and other costs included in the initial investments, such as environmental mitigation and monitoring. Physical contingencies included in the initial investments, should be allocated to the relevant category, and included in the appraisal. The initial investments may be concentrated in a single project year, or more generally scheduled over more than one year according to the project phasing and implementation schedule. Typically, however, **investment expenditures** do not tend to recur throughout the project's life, hence they are also known as **non recurring costs.**

The operation and maintenance of a project or policy intervention usually incurs expenses. As these expenses tend to be incurred annually throughout the life of the project, they are **termed recurring costs**. Private recurring costs tend to be grouped into three broad categories: annual energy costs, annual labour costs, and annual material costs. These cost items typically vary with capacity utilisation. Other types of recuring costs are fixed - e.g. overhead costs.

The underlying principle of what items to include in CBA is that project costs comprise the difference in all resource inputs between the baseline and project scenario - that is, the extra use of resources necessary to achieve the corresponding benefits. As a result, some types of financial cost items must be excluded, whilst others may be included. Some examples include:

- A project may require the use of facilities already in existence. The costs of such facilities are sunk costs and should not be included, provided their use in the project involves no opportunity cost. In other words, sunk costs are those costs that would exist both without and with the project
- Contingency allowances, which are determined by engineering, also have implications for CBA. Physical contingencies represent the

2. the number of operating years to include can be determined by (ADB, 1999): (1) the market life of the project-i.e. the years over which the project benefits will occur; (2) the technical life of major investment assrt -1/e the number of years of normal operation before the assets are full owrn out; and (3) the economic life-1.e. the number of years after which it is worth reinvesting to obtail the benefits of reduced operating and maintenance costs even if the assets are not fully won out.

3 Different types i of investment asset hae fiffercut replacement periods. For whatever project periof is decided upon, some assets will not be fully worn out at the end of the project period. The remaining value of the assets-i.e. their residual value - is entered as a negative incestment cost at the end of the project. It is callculated as the proportion of the replacement period still remaining for a particular subcategory, times the value of the assets concerned.

monetary value of additional real resources that may be required beyond the base cost to complete the project, and should be treated as part of the cost of a project.

- **Working capital** is commonly defined in financial analysis as net current assets, consisting of inventories. For purposes of CBA, only inventories that constitute real claims on the nation's resources should be included
- Some of the items included in the financial costs of a project are not economic costs, as they do not increase or decrease the availability of real resources to the rest of the economy. They are referred to as **transfer payments**, as they essentially transfer command over resources from one party (e.g. producer) to another (e.g. government) without reducing or increasing the amount of resources available as a whole. Taxes, duties, and subsidies are examples of transfer payments.
- Regarding investment expenditures, the financial accounts of project-operating entities will include provision for **depreciation** on the basis of prevailing accounting practices. In CBA however, the stream of real investment required to realize and maintain project benefits is included in the resource flow in full at the time when the expenditure is incurred. This stream of expenditures generally will not coincide exactly with the time profile of depreciation and amortization in the financial accounts.
- Many projects involve the exploitation of a non-renewable natural resources. The opportunity cost of using these natural resources must be included in CBA. Since non-renewable natural resources cannot be replenished, and when depleted must be replaced by imports or domestic substitutes, the opportunity cost of the resource includes the cost (price) of the substitutes when the resource is exhausted. The **depletion premium** depends on this price and the proportion of the total reserves exploited during each year. It is added to the economic cost of exploitation to arrive at the full opportunity cost of using the non-renewable resource.

### 2.2.3 Idenfffication and Quantificaffon of Benefits

A taxonomy of project benefits is presented in the proceeding section, and is therefore not discussed further here.

## 2.3 STAGE 2 - Valuation of Project Impacts

### 2.3.1 Introduction .

In general, incorporating project impacts, environmental or otherwise, into CBA is a two-step process. (It should be noted that the discussion to follow focuses primarily on environmental impacts.) Before a project's impacts can be valued, they must first be identified and measured. Secondly, the value of the impacts must be estimated in monetary terms to determine their relative economic importance, so that the benefits and costs of various alternatives can be assessed. The entire process is not always easy, since some impacts - particularly environmental impacts - are often dislocated in time and space, making the cause and effect difficult to establish. In addition, the severity of an impact will often depend on an accumulation of problems. Many environmental goods and services also do not enter markets, which causes difficulties for valuation, compounded by the fact that the data, which is available, is often scarce or of poor quality.

The section is divided into three sub-sections. The first sub-section provides an introduction to key concepts, which underp'n the economic approach to valuation. This is followed by an overview of the main techniques for valuing changes in environmental quality or the provision of an environmental resource. The final sub-section looks at some additional issues relevant to economic valuation.

### 2.3.2 Concepts Underlying the Economic Approach to Valuation

Any empirical analysis requires some decisions to be made regarding the scope of the work. Concerning economic valuation problems, this means defining what 'value' to quantify. So let us begin by developing a taxonomy of project benefits.

For some goods and services (e.g. a net or a boat purchased by a capture fishery operator), the market provides prices that reasonably reflect the value society places on that good or service. For other goods and services however, market prices either only partially reflect the value society places on them (e.g. electricity) or do not exist at all (e.g. the use of a Lake or the atmosphere as a discharge sink). To simplify the task of valuation, therefore, economists like to disaggregate project impacts on the environment into individual components of value. The most commonly used approach is based on the concept of *Total Economic Value* (TEV).

*Total Economic Value*

Under the **Total Economic Value** (TEV) approach an impact on an environmental resource, for example, a waterborne pollutant on a river, is broken down into a number of categories of value. The logic behind the approach is that a good or service comprises various attributes, some of which are tangible and readily measured, while others are less tangible and thus more difficult to quantify. The total value of the good or service however, is given by the sum of all categories of value, and not simply those that are easy to measure.

The decomposition and terminology of the components of TEV tends to vary from text to text. Nonetheless, TEV is generally decomposed into three categories of value: (1) direct use value; (2) indirect use value; and (3) non-use value. The former two categories are sometimes collectively referred to as "use value". Further subdivision of these categories is also possible (see Figure 3 below).

**Direct use value** derives from goods, which can be extracted, consumed or directly enjoyed. It is also therefore known as **extractive** or **consumptive** use value. In the context of a river, for example, direct (extractive) use value is derived from the harvesting of fish. In general, direct use values are real, can be measured, and have values. Furthermore, since extractive use typically involves observable quantities of goods, whose prices are also observable, direct use benefits are usually relatively straightforward to value.

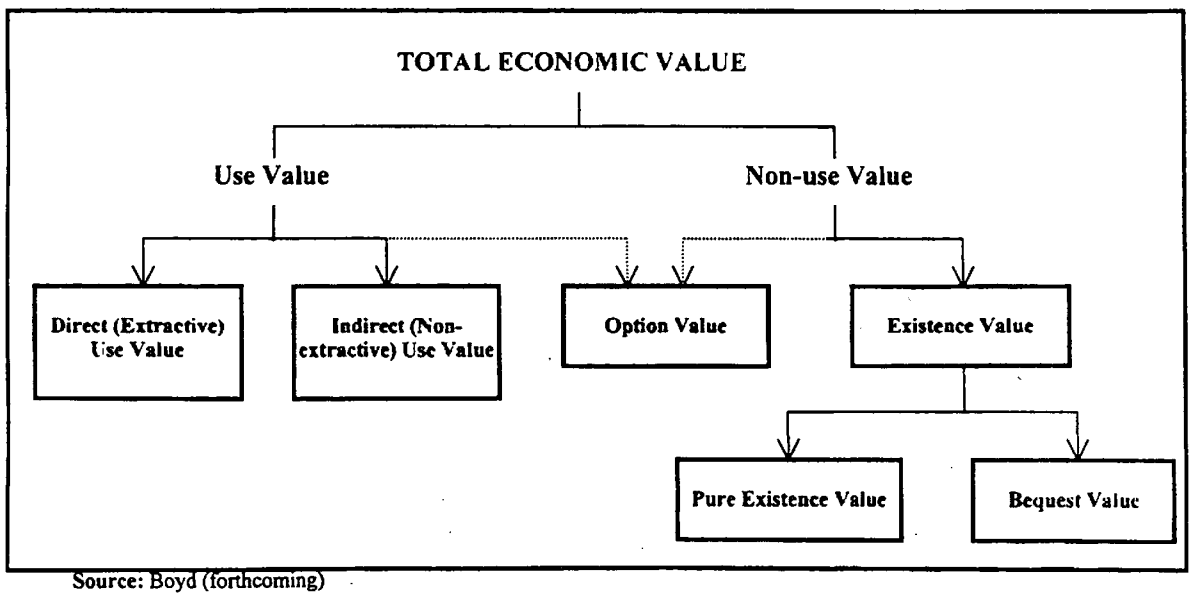
**Indirect use value,** also referred to as non-extractive use value, derives from the services that an environmental resource provides. A wetland, for example, acts a water filter, often improving water quality for downstream users. This service is valued by downstream users, but does not require any good to be extracted/ consumed.

In terms of measurement, indirect use values differ from direct use values in two ways: (1) the 'quantity' of the service provided is often hard to define, and (2) the types of services in question are often not traded in established markets, and therefore have no readily observable 'prices'. For these reasons, measuring indirect use value is relatively more diffficult than measuring direct use value.

**Non-use values** are defined as those benefits or welfare gains/losses to individuals that arise from environmental changes independently of any direct or indirect use of the environment. This category can be further subdivided into (1) **option value** and (2) **existence value.**

The former is a benefit expressed through an option to use the environment - that is, the value of the environment as a potential benefit as opposed to an actual present use benefit. If you are unsure whether you will use an environmental service or not, you might be willing to pay a positive sum to guarantee that the service will still be available in case you desire to use it at a later date. **Option values** consequently arise when you are uncertain about whether you will demand a commodity in some future time period and are faced with uncertainty concerning the future supply or availability of that commodity. It is distinct from a use value in that it arises not from the use of the site itself but from uncertainty over the site's availability to meet future demands. In this way option value is akin to an insurance policy against future uncertainty.

**Figure 3: Components of Total Economic Value**




Source: Boyd (forthcoming)

As option value is the value derived from maintaining the option to use a good or service at some point in the future, it is sometimes treated as a special case of **use value** (hence the dashed line in Figure 3).

**Existence value** can be defined in various ways. Most definitions however contain two main components: (1) pure existence values; and (2) bequest values.

**A pure existence value** relates to the worth you associate with an environmental good or service, which is completely unrelated to current or future use of that commodity, by yourself, your descendants, or others. These values are intrinsic in nature, i.e. they represent a value that resides in something. Some possible motivations or rationales for the presence of such values include the preservation of, concern for, sympathy with, respect for the rights of, any other altruistic motives with respect to nonhuman beings. A number of pure existence values are related to ecological attributes. Support for the protection of endangered species and the protection of critical habitats for those species represents an intrinsic valuation process.

**Bequest value** derive from our desire to preserve the environment for relatives and friends, and also for all other people living today and future generations, so that they may benefit from conservation of the environment.

Since in most cases non-use value is not, by definition, reflected in individual's behaviour and is thus not observable, it is the most difficult component of TEV to measure.

| Total Ecomomic Value = Direct and Indirect Use Values + Option Values + Existence Values |
|---|

It is of crucial importance to assess the change in the TEV arising from a project-induced change in environmental quality, or a change in the provision of an environmental resource. It will often be the case that the 'true' benefits (change in TEV) of a proposed project or policy will be much greater than its direct use value, but the direct use value may be less than the cost of the resource inputs. In order to build an accurate argument for or against the development it is important to measure as many components of TEV as possible, whilst avoiding double counting (see Section O below).

***Measures of Economic Benefit***

The concept of TEV is grounded in neo-classical welfare economics. This ideology of economic thought has two basic premises: (1) all economic activity should be aimed at increasing the welfare of individuals in society; and (2) individuals are the best judges of their own welfare. Individuals themselves derive welfare from the consumption of privately and publicly provided goods and services, and the consumption of what we refer to as 'non-marketed' goods and services, e.g. environmental assets. It therefore follows that the basis for deriving measures of the economic value of changes in environmental resource service flows, is ultimately given by their impact on human welfare. In other words, only those service flows, from which individuals derive value, are included in the valuation framework.

Water pollutants, for example, can affect individuals in many ways pollutants can adversely affect health directly, or they can limit outdoor recreation (and hence tourism) by degrading canals, lakes, rivers and estuaries. The costs of degraded quality are defined by the difference between an individual's well-being before water quality changes, and that individual's level of well-being after the change in water quality[4]. Assuming that individuals are aware of the impacts of the pollutants, a change in water quality can be expressed in money terms by identifying the amount of income an individual would be **willing-to-pay** (WTP), or **willing-to-accept** (WTA) as compensation, that would leave them indifferent between the state-of-the-world before the change (degraded water) and that prevailing after the change ('cleaner' water)[5].

An alternative way to measure the benefits of improved water quality is to identify the decrease in the costs of producing a good or service that uses water as a factor input. Both these effects will be reflected in the demand and supply of goods and services.

**In its simplest form economic valuation is the process of identifying the relevant changes in consumer demand and producer supply arising from a (project-induced) change in environmental quality, or the change in the provision of an environmental resources**

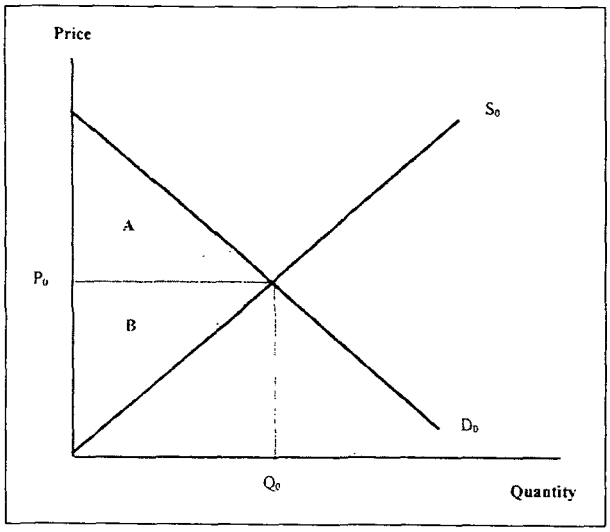
The concept of economic benefits can be illustrated with the aid of a simple graph. Consider Figure 4, which shows a supply and demand curve for a hypothetical good X. The net economic benefits derived from the consumption and production of this good comprise two parts - these are known as consumer surplus and producer surplus. The area, *A*, above the price line and bounded by the demand curve measures the **consumer surplus**. The consumer surplus is essentially

4. Or, conversely, the nenefits of improved water quality are given by the difference between and individual's well-being before water quality changes and theat invidual's leve; of happiness after the change

5 WTP is a measure of the economic value, in terms of income or other goods, an individual is willing and able to forgo to gain or maintain a resource, good, or service. It is accepted as the standard when conducting cost-benefit studies of non-marketed resources.

the net benefit accruing to the consumer, given the existing price[6]. **Producer surplus**, formally defined as the excess of price over production costs, is given by the area, B, below the prevailing price and above the supply curve. The net benefits to society from the production and consumption of this hypothetical good is given by the sum of areas *A* and *B*.

**Figure 4 Market for Hypothetical Good X**




An example of how the framework depicted in Figure 4 above is used to value a change in environmental quality (or the provision of an environmental good or service) is given in Box 1.

## 2.3.3 Economic Valuation Techniques

*Taxonomy of Benefit Valuation Techniques*

Unfortunately, economists working in this area classify the main benefit valuation techniques in different ways. These different classifications can often be a source of confusion; where different terms are used to describe the same techniques. Hence, to avoid any confusion, only one classification is presented below. The taxonomy used herein emphasizes the degree to which the valuation techniques recognise the behavioural adjustments that individuals (consumers or producers) make in responses to changes in environmental quality, or the provision of an environmental resource.

Two distinct sets of benefit valuation techniques can be distinguished. The first set is based on **technical (physical) linkages** that formally describe cause and effect relationships. Included in this set of techniques are the change in the output (or input) of marketable goods, the cost of illness and the replacement cost methods.

6 Formally, consumer surplus is defined as the excess consumers are willing-to-pay over the actual expenditures at the prevailing price.

The second set of techniques is based on **behavioural linkages** between a change in the state of the environment and the actions of individuals, whereby values are either stated or revealed in actual or hypothetical market behaviour. Using revealed behaviour, we examine the trade-offs individuals make between the state of the environment and goods or service traded in actual markets. These so-called **revealed preference** approaches include the hedonic property, wage-risk, travel cost and averting expenditures approaches.

When environmental goods or services cannot be valued, even indirectly using revealed preference approaches, we can ask individuals directly to express how much they are WTP for a certain level of that environmental good or service. Contingent valuation and constructed markets are the two main types of so-called **stated preference** techniques.

Another approach to benefit valuation in CBA is **benefit transfer,** although strictly speaking not a valuation method.

## 2.3.4 Valuation Techniques Based on Technical (Physical) Linkages

*Introduction*

Valuation techniques categorised under this heading exhibit a **technical (physical) linkage** between a change in the state of the environment and ultimately a change in a good or service that is traded in a market, and therefore has an observable price. The physical linkage is typically described by a **cause - effect** (or **dose-response**) function, which links, for example, the level of water pollution (cause) to a change in harvest rates (effect). The expected change in the quantity of fish harvested is then converted into monetary units using the prevailing market price. Examples of benefit valuation techniques that rely on the presence of a technical (physical) linkage, include: (1) change in the output (or input) of marketable goods, (2) cost of illness; and (3) replacement cost.

As these techniques essentially provide an objective measure of 'damage' derived with the aid of cause-effect (or dose-response) functions, they are also called collectively known as damage function methods.

*Change in Output Input Approaches*

In many cases the environment has a direct effect on:

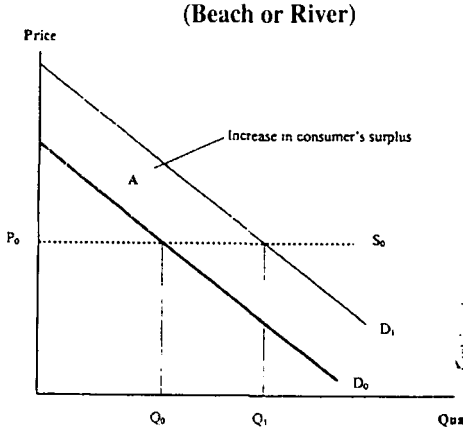
- the capability of a firm to produce (provide) a good (service), and/or
- the costs that the firm incurs in producing (providing) that good (service).

**Box 1: Valuing a Change in Environmental Quality**

Project-induced changes in environmental quality can shift the supply and demand curves from some initial position to a new position. Improved water quality on a beach, for example, may increase tourist use of that beach. Graphically, this is represented by an outward shift in the demand curve for tourist trips to the beach, from $D_0$ to $D_1$ (see below Figure 5)[7]. The gross benefits associated with the improvement in water quality are given by the change in consumer surplus, i.e. area A in Figure 5. Net benefits are found by deducting the costs of improving the water quality.

As mentioned above, changes in water quality can also shift the supply curve. In the case of a river, for example, drinking water treatment plants may benefit from improved water quality through reduced water treatment costs. As treatment costs reduce, the supply curve shifts outward and producer surplus increases (this situation is not shown graphically in Figure 5). In this case, the benefits are measured by the additional producer surplus created by the shift in the treatment plants supply curve.

**Figure 5 Benefits of Improvement in Water Quality (Beach or River)**




The actual distribution of benefits between producers and consumers depends on the relative slopes of the supply and demand curves. In the case illustrated in Figure 5 for example, the supply curve is horizontal, in which case all benefits from water quality improvements accrue to tourists. If a change in environmental quality shifted either the supply or demand curve depicted in Figure 5 however, the valuation of the resulting benefits is not so straightforward[8]. Consequently, in cases in which environmental improvements affect consumers, it is common practice to assume that the supply curve is horizontal in order to simplify the measurement of benefits. For similar reasons, it is often assumed that the demand curve is vertical when measuring the benefits of environmental improvements which affect producers. In the majority of cases, these simplifying assumptions will not significantly affect the benefit estimates.

7 In this case, the horizontal supply curve reflects the assumption that additional visitors can be accommodated at a constant cost per visitor.

8 An assessment of the resulting benefits would require detailed data on the shape and location of the relevant supply and demand curves.

In economics, we say that some attribute of the environment is an argument in the production and/or cost function of some good/service. For example, the amount of whitefish harvested from a lake depends, among other things, on whitefish stocks. In turn, the whitefish stock depends on the quality of the lake's environment (water quality). We can summarise this relationship in the following production/cost function:

$$Q_w = f\{R, S_w(E)\}$$

where

$Q_w$ = the quantity of whitefish harvested,

$f$ = the function that relates environment quality to the quantity of whitefish harvested,

$R$ = the amount of resources devoted to catching whitefish, and

$S_w(E)$ = the stock of whitefish in the lake, which itself depends on the quality of the lake's environment (E).

Now consider the implications of a change in the quality of the lake's environment, which we denote by E. Suppose that, as a result of a pollution incident, water quality in the lake were to deteriorate. Assume further that the deterioration in water quality is sufficient to have an adverse impact on whitefish stocks.

There are two possible consequences of $E$, and the subsequent decrease in whitefish stocks ($\downarrow S_w$):

- If the operator wants to maintain current harvest rates (i.e. $Q_w$ remains unchanged), then it will have to allocate more resources to catching whitefish ($\uparrow R$). Basically, since whitefish are now less abundant in the lake, the operator will have to try harder to maintain harvest rates, e.g. buy an extra fishing vessel, whitefish longer hours, etc.
- If the operator does not increase the amount of resources it allocates to catching whitefish (i.e. $R$ remains unchanged), then we can expect the quantity of whitefish harvested to decrease ($\downarrow Q_w$).

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Either way, the operator suffer an economic loss. Under the latter scenario, they loose the value of the lost output ($\downarrow Q_w$). In the former case, the operator's costs increase as a result of having to increase their level of fishing effort ($\uparrow R$). This provides us with two measures of the cost of the deterioration in environmental quality: the value of lost output or (2) the cost of the additional resource inputs.

Of course, environmental (water) quality can also improve, in which case we might expect whitefish stocks in the lake to increase ($\uparrow S_w$). The benefits of an improvement in water quality are given by either:

- the value of additional output (assuming $R$ remains unchanged) or
- the cost savings associated with using less resource inputs (assuming $Q_w$ remains unchanged).

The impact of changes in environmental quality on the inputs to and outputs from various sectors is illustrated Table 1 below. (Note that the first four rows summarise the water quality example discussed above.) An industry that uses water in the production process, for example, would benefit from an improvement in water quality in that its water treatment costs may decline (-). Conversely, deterioration in water quality may result in increased treatment costs (+). In both cases, it is unlikely that the industry's output would be affected.

**Table 1: Examples of the Productivity Impacts of Selected Environmental Changes**

| Environmental Change | Effect on Outputs | Effect on Inputs |
|---|---|---|
| Increased pollution of fisheries | - | Constant |
| Increased pollution of fisheries | Constant | + |
| Reduced pollution of fisheries | + | Constant |
| Reduced pollution of fisheries | Constant | - |
| Improved water quality for industry | Constant | - |
| Degraded water quality for industry | Constant | + |
| Improvement in soil quality (agriculture) | + | - |
| Conservation of forests | + | + |
| Soil erosion (agriculture) | - | - |
| Loss of forests | - | - |

*Source: Adapted from Abelson (1996) in Boyd (forthcoming)*

In general, when we estimate the benefit (cost) of an improvement (deterioration) in environmental quality by valuing increases (decreases) in output, we are employing what is referred to as the **change-in-productivity approach.** The closely related approach, where we estimate the benefit (cost) of an improvement (deterioration) in environmental quality by valuing decreases (increases) in resource costs, is the **production cost** (or **cost saving) technique.**

In terms of valuation, two situations must be distinguished from each other. These are

- changes in quantity do not result in changes in price; and
- changes in quantity induce changes in price.

They both involve assumptions concerning the size of the expected change in output/inputs.

Let us assume that in the above example the change in the quantity of whitefish harvested - denoted by $\Delta Q_w$ - is small relative to the current total quantity of output available in the marketplace. For instance, the operator might be only one of many small producers of whitefish, in which case it is unlikely that a change in their output will significantly affect the total supply of whitefish in the market. In this case, we can safely assume that the current price of whitefish ($P_w^0$ - we use the superscript '0' to denote time period before the change in output) will not be affected. Given that price of whitefish will remain static, the economic value ($V$) of the change in output is simply obtnined by:

$$V = \Delta Q_w * P_w^0 = (Q^1_w - Q^0_w) * P^0_w$$

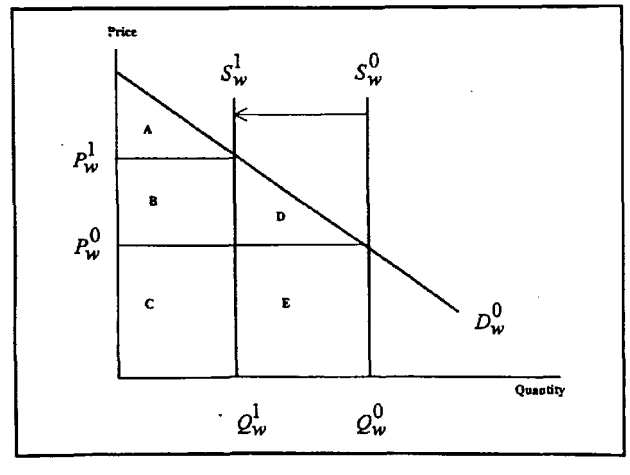
where

$Q_w^0$ = the quantity of whitefish harvested before $\Delta E$, and

$Q_w^1$ = the quantity of whitefish harvested after $\Delta E$ .

In some situations, the change in output may be considerable relative to the current total quantity of output available in the marketplace. This in turn may induce the price of the affected good/service to change. For example, if the deterioration in water quality decimated whitefish populations and subsequently had a large impact on harvest rates, whitefish might become scarce in the local markets. In such circumstances, other things being equal, it is likely that consumers would be willing to pay increasingly higher prices for the whitefish - as 'old' demand exceeds 'new' supply. This would have the effect of bidding up the price of whitefish.

**Figure 6: Valuing a Change in Output When Prices Change**



In general, if the change in output is large enough to affect market prices, then we must resort to the relevant supply and demand curve in order to value the change in output. To help illustrate this consider Figure 6 below, which depicts the market for whitefish. Initially, $Q_w^0$ units of whitefish are supplied at a price of $P_w^0$. Now, suppose that as a result of a pollution incident, and the subsequent change in water quality, output falls to $Q_w^1$. In the market depicted in Figure 6, the corresponding 'new' price of whitefish is $P_w^1$. (Basically, as a result of an initial shortage in supply, consumers bid up the market price, and those consumers not willing to pay the higher price drop out of the market.)

Before the change in water quality consumer (CS) and producer (PS) surplus is given by

$CS^0$=A+B+D,and

$PS^0$=C+E.

After the change in water quality, and the subsequent change in output and price, we have

$CS^1$ = A, and

$PS^1$ =B+C.

The cost of the deterioration in water quality in this case is therefore given by

$[CS^1 - CS^0] + [PS^1 - PS^0]$

[A-(A+B+D)]+[B+C-(C+E)]

-B-D+B-E=-D-E.

In short, the net loss of consumer surplus is equivalent to the area D, while the net loss of producer surplus accruing to FFC is equivalent to the area E.

Of course, in order to conduct the above analysis we must be able to predict the 'new' price of whitefish. This requires us to identify what is referred to as the **price elasticity of demand** for whitefish (which we denote by the Greek letter ' $\xi$ '). If we know $\xi$, and can assume that the demand curve is linear, then it is possible to calculate the induced change in price, and in turn the 'new' price level (see Box 2 below).

While the above illustrations were phrased in terms of changes in output prices, you should note that the same line of discussion could just as easily been constructed in terms of resource input prices. This is relevant in situations in which the change in environmental quality, and the subsequent change in resource (input) use, is significant enough to affect the price of those inputs.

Box 3 provides an example of the use of the change-in-productivity approach to value the damage to agriculture resulting from flooding and damage to irrigation caused by watershed degradation in the Republic of Haiti.

**Box 2: Measuring Consumer Surplus When Prices Change**

In economics, the price elasticity of demand for, say, good X measures the percentage change in the quantity demanded associated with a percentage change in price. Mathematically,

$$\xi_x = \frac{(\Delta Q_x / Q_x^0)}{(\Delta P_x / P_x^0)} = \left(\frac{\Delta Q_x}{\Delta P_x}\right) \times \left(\frac{P_x^0}{Q_x^0}\right)$$

Given $\xi_x$ , $Q_x^0$, $Q^1{}_x$ and $P_x^0$, the above equation can be solved for $\Delta P_x$, from which $P_x^1$ can be readily derived since $\Delta P_x = P_x^1 - P^0{}_x$ .

The net change in consumer and producer surplus in the above example is therefore given by

$$\frac{1}{2}(\Delta Q_x \times \Delta P_x) + (\Delta Q_x \times P_x^0)$$

*Source: Boyd (forthcoming)*

By far the biggest diffficulty in applying this approach relates to measuring the quantity of goods being produced, and in predicting how these amounts will change in response to changes in environmental quality. As mentioned above, this usually involves the use of a cause-effect function, which relates the level of pollution (cause) to a change in output (effect). The cause-effect functions themselves are developed from data obtained from either controlled experimental trials or controlled field studies, known as epidemiological studies in the case of health impacts.

**Box 3 Change-in-productivity Example**

Protecting the Pic Macaya watershed will help reduce damage to irrigation systems downstream, both from siltation and from reduced dry-season flow, which is forcing a reversion to rain-fed cultivation and a switch to lower-value crops. Returns from rain-fed production are US$ 200800/hal/year lower than for irrigated production, depending on the crops being planted. Combined with estimates of the effects of damage to irrigation infrastructure on the irrigated area, the present value of the benefits avoided reduction in productivity, were estimated to be about US$ 2-7 million (the wide range reflecting the weakness of the underlying date).

*Source: World Bank (1998)*

Once the expected changes in output are known, converting them into monetary units is usually relatively straightforward. Ideally, the monetary values used should be based on observed market prices of the affected good/ service. If the affected good/service is non-marketed, then unit values derived from one of the valuation techniques described below can be used. For example, mortality impacts estimated by a cause-effect function could be valued using the Value of Statistical Life derived from contingent valuation studies, or the present value of foregone earnings derived by the Human Capital approach.

The change-in-productivity and the production cost (saving) approaches to valuation have the advantage that they are based on observable data from producers using an environmental parameter as an input. Moreover, they have a sound basis in microeconomic theory and are relatively inexpensive to implement, at least compared to those techniques used to value non-marketed goods/ services. On the negative side, these approaches tend to understate the true change in WTP, since they may not measure all the welfare effects of changes in environmental quality. Also, the analysis becomes more complex when input/output prices changes. Finally, the lack of cause-effect functions relating exposure to a particular environmental parameter to impacts on service flows severely limits the use of these techniques.

*Cost-of-illness*

Many environmental impacts, in particular those arising from air and water pollution, have repercussions for human health. Exposure to unsafe levels of pollutants in water for example, through ingestion in drinking water or other routes (e.g. skin absorption, consumption of contaminated fish), can lead to increased morbidity (sickness) or mortality (premature death). Valuing the cost of pollution-induced morbidity or mortality requires data on the underlying cause-effect (i.e. **damage**) function, which relates the level of pollution to the incidence of the health effect. The cost of pollution-induced health effects can then be estimated by multiplying the estimated change in mortality/morbidity by the corresponding economic (unit) value.

The two main approaches economists have used to value morbidity are based on either individual preferences (WTP or WTA compensation) or the resource or opportunity cost approach (Freeman, 1993). With the latter, known as the **cost-of-illness** (COI) approach, the costs of morbidity are measured by estimating the direct out-of-pocket expenses resulting from illness (e.g. medicine, doctor and hospital bills) and opportunity costs (e.g. any loss of earnings resulting from the sickness). The COI approach does have several important limitations. Firstly, the COI approach disregards the affected individual's preference for health versus illness, for which they may be willing-to-pay. That is, it does not recognise the full disutility of those affected by sickness. Secondly, the method does not account of the fact that individuals may undertake averting expenditures and incur costs to reduce health risks. For these reasons, estimates derived with the COI approach are often interpreted as lower bound estimates[9]. On the plus side, the approach is relatively inexpensive to implement.

When the COI approach is extended to measure the cost of premature death, it is commonly known as the **human capital** (HC) **approach.** With this approach the "value of life" is given by the present value of an individual's (forgone) future income stream. It therefore considers humans as units of capital that produce goods and services for society. For this reason, the human capital approach to mortality valuation is ethically troubling, and is falling out of favour with practitioners and policy makers. However, it is relatively easy to apply, particularly in developing countries. The preferred approach to mortality valuation is to use WTP- based estimates - e.g. the Value of a Statistical Life (VOSL) - derived from wage-risk or contingent valuation studies (discussed below).

The COI and HC approaches have been applied in the context of urban air pollution control programmes in Mexico City, Santiago, San Paulo, Jakarta, Delhi and several cities in Russia. In Mexico and Delhi, for example, the HC approach valued each life lost from TSP pollution at US$ 75,000 and US$ 10,000, respectively.

The valuation of impacts on human health is discussed further in the Chapter on Air Pollution and Energy.

*Replacement or Restoration Costs*

When the benefits of a given environmental impact cannot be measured directly, cost data can be used to yield valuable information. Estimates of the potential costs (or savings) to households and producers, for example, can be obtained by using: (1) the cost of replacing the services provided by the affected environmental resource after the impact has occurred; or (2) the cost of reducing or avoiding the impact before it occurs. The former are known as **replacement costs or corrective expenditures;** the latter are referred to as **averting or preventative expenditures.** We consider the former here.

9 Consequently, a "multiplir" is usually applied to morbidity values based on COI - ranging from 1.5 to 2 - in order to convert the COI-based figure into a WTP-based estimate.

The **replacement cost** (RC) approach is often used to estimate the cost of pollution (or the benefits of avoiding the pollution). RC focuses on potential damage costs, typically measured by *ex ante* engineering estimates of the costs of replacement or restoration, if damage from pollution were to arise. Consequently, the required economic data for sitespecific assessments is typically readily accessible. For example, the cost of restoring a river to a certain water quality standard can be used to estimate the costs of the pollution that lowered water quality below the standard in the first place. Also, the costs of air pollution related soiling (or acid deposition) of buildings could be approximated by the costs of cleaning (or restoring) the damaged buildings. The use of RC to value the benefits obtained from reduced flooding in the Haiti example is shown in Box4.

Several key assumptions underpin the RC approach, these include (World Bank, 1998):

- The nature and extent of the physical damage expected is predictable (i.e. an accurate damage function exists).
- The cost to replace or restore the damaged assets can be estimated with a reasonable degree of accuracy.
- These costs can be used as a valid proxy for the cost of the environmental damage (i.e. the replacement or restoration costs do not exceed the economic value of the asset).

Clearly, these are significant assumptions, and may not always be valid.

In general, the RC technique is best suited to assessing the costs arising from damage to tangible assets, since the repair or replacement costs are easily measurable. Conversely, the technique is less well suited to valuing damage to unique assets (e.g. historical or cultural sites, special natural sites), which cannot be replaced or restored, and for which restoration costs are uncertain.

**Box 4 Replacement Cost Example**

The cost of damage to irrigation and other infrastructure were estimated using their replacement costs (IJS$ 2,500-5,000 per km for irrigated canals and US$ 3,500-7,000 per km for roads). This resulted in an estimated benefit from avoided damage, relative to the watershed protection case, of about US$ 2.5-5 million.

*Source: World Bank (1998)*

## 2.3.5 Valuation Techniques Based on Behavioural Linkages: Revealed Preference

***Introduction***

Valuation techniques categorised under this heading work by observing individual expenditures of money or time to obtain goods, or avoid their loss, and inferring WTP values for environmental goods or services from these expenditures. It is assumed that individuals take the state of the environment into consideration when making consumption decisions with respect to marketed goods and services. In essence, individuals **reveal** their preferences for environmental goods/services through their spending behaviour on marketed goods/services. Hence, valuation techniques that fall under this heading are often categorised as revealed preference techniques. Such **revealed preference** approaches towards valuation, include: (1) hedonic analysis (property and wage-risk); (2) travel cost, and (3) averting expenditures.

***Hedonic Techniques***

It is fair to say that environmental quality affects the price individuals are willing-to-pay from certain goods/services. For example, hotels in the travel brochures often charge a supplement for rooms with a "sea view". Econometric models can be used to examine the contribution of specific "attributes" to the price paid for a property or wage rates. These "attributes" include indicators of environmental quality.

If the hedonic analysis is conducted on housing data, it is referred to as the **property value approach.** When applied to wage data - to measure the value of changes in morbidity/mortality risks - it is often referred to as the **wage differential or wage-risk approach.**

The hedonic property value approach measures the welfare effects of changes in environmental goods or services by estimating the influence of environmental attributes on the value (or price) of properties. In order to obtain a measure of how a specific environmental attribute of interest affects the welfare of individuals, the technique attempts to: (1) **identify** how much of a property price differential is due to a particular environmental difference between properties, and (2) **infer** how much people are willing-to-pay for an improvement in the environmental quality and what the social value of the improvement is.

In attempting to isolate the effects of specific environmental attributes on ; the price of houses we have to "explain" the price of a house as a function ; of its key characteristics. If we take house price to be a function of

all the physical features of the house (e.g. number of rooms, central heating, garage space etc.), neighbourhood characteristics, and environmental attributes, then the following relationship can be identified:

$$P^h = f(S, N, E)$$

where ;

$P^h$ = The market price of the property.

$f$ = The function that relates the house characteristics to price.

$S$ = The different structural characteristics of the property.

$N$ = The different neighbourhood characteristics of the property.

$E$ = The different environmental attributes of the property.

This function is the called a **hedonic price function.** Fixing the level of all the structural characteristics of a property and the neighbourhood characteristics, we are able to focus on the relationship between the property price and the environmental attribute under investigation[10].

By partially differentiating the hedonic price function with respect to $E$ we obtain the **implicit price** (or **implicit price curve**) of the environmental attribute

$$\frac{dP^h}{dE}$$

This partial derivative is interpreted as the price paid by the individuals for the last unit of the environmental attribute, purchased by choosing a given property instead of another one with a unit less of the environmental attribute, **other things equal.** Estimated implicit prices for different properties refer to different individuals. Every estimated implicit price is only one observation of the true individual demand curve and corresponds to the individual WTP for a marginal unit of environmental good only for that specific level of environmental good purchased. Therefore, the implicit price (curve) cannot be viewed as an **inverse demand curve.** Hence, it does not represent the maximum marginal WTP of the individual for one more unit of the environmental attribute, unless we assume that all the individuals have the same structure of preferences and the same income. If this assumption does not hold, the various individuals will have different inverse demand curves. Nevertheless the implicit price can be regressed on the observed quantities of the Environmental attribute and some socio-economic characteristics of individuals. This "second stage" regression could allow the identification of the inverse individual demand function. The area under the inverse demand curve between two levels of the environmental attribute represents the change in the consumer surplus caused by the change in this attribute. By aggregating all individuals' consumer surpluses we obtain the overall value of the environmental change.

In practice, especially in developing countries, only the first stage of the process is usually carried out, and the results used to obtain only rough values for the impact of the attribute in question

A summary of the main steps followed in undertaking a hedonic property value study are outlined in Figure 7 below.

**Figure 7 Step-by step Procedure for the Calculation of the Consumer Surplus with the Hedonic Properly, Value Approach**

| Steps | Assumptions-Notes |
|---|---|
| 1 Collection of data on prices and houses features | Various methods exist to collect these data. For complex studies this data must be complemented with information on the soci-economic characteristics of households investigated. |
| 2 Estimation of the hedonic price function | This relates the price of houses to the characteristics explaining the house price. |
| 3 Calculation of the implicit price of the environmental attribute in question | This is the first derivative of the house price function with respect to the environmental attribute |
| 4 Estimation of the inverse demand curve of the environmental attribute | The price paid is explained by the quantity/quality of the environmental attribute but also by the socio-economic charaeteristics of households |
| 5. Calculation of the consumer surplus | Integration of the implicit demand curve between the former level of environmental quality/quantity and the new one. |

*Souree: Markandya et al (fortheoming)*

The hedonic wage-risk method is very similar, and is only briefly discussed here. Basically, to estimate the relationship between wages and risks we must control for other variables that influence earnings - as in the hedonic property value approach above - except this time we estimate a hedonic wage function:

[10] The estimation of a hedonic price function is usually done using a multivariate regression technique. Data are taken either on a small number of similar residential properties over a period of years (time series), or on a larger number of diverse properties at a point in time (cross section), or on both (pooled data). In practice almost all property value studies have used cross section data, as controlling for other influences over time is much more difficult.

$$W = f(Q,X,R)$$

where

$W$ = Wage rate in each occupation.

$Q$ = Qualifications of workers.

$X$ = Job attributes such as unionisation, desirability, etc.

$R$ = Workplace risk, e.g. risk of death.

The partial derivative of this function with respect to R is the **wage premium** for accepting, say, an additional risk of death of 1 in 10,000. To estimate a 'value of a statistical life' (VOSL) from this, the wage premium is factored by the additional risk (in this case 10,000). For example, if the 'average' wage premiurn is $45 in this case, then the VOSL is given by

$$\uparrow \frac{1}{100,000} \times 100,000 = 1 \Rightarrow \$45 \times 100,000 = \$4,500,000$$

The hedonic technique has several advantages. Firstly, hedonic analysis uses market, i.e. observed, data on property sales or wage rates. The method is versatile and can be adapted to consider several possible interactions between market goods and environmental quality. Moreover, estimated values obtained from one study can be used in other policy areas if the environments have similar demand and supply characteristics. On the negative side, the results of hedonic studies are sensitive to the econometric assumptions adopted. Furthermore, the assumptions necessary to interpret the results as measures of WTP are restrictive and, in many real world settings, unrealistic. From a practical perspective, full hedonic pricing studies require a considerable amount of data, which may be difficult and expensive to collect, such studies tend not to be done quickly.

### *Travel Cost*

The **travel cost** (TC) method is also an example of a technique, which attempts to deduce value from observed (i.e. revealed) behaviour. The TC model, and its many variants[11], is the most commonly used indirect approach to valuing site-specific levels of environmental resource provision. Basically, information on visitors' total expenditure to visit a site is used to derive their demand curve for the services provided by the site. Among other things, the TC model assumes that changes in total travel expenditures are equivalent to changes in an admission fee[12]. Given this, the model is used to predict changes in demand in response to changes in 'admission fees', thereby tracing out a demand curve for the site. This demand curve may then be used to measure the total benefits visitors accrue from the site[13].

There are two main variants of the TC model: (1) the **Zonal** TC model (ZTCM) and (2) the **Individual** TC model (ITCM). The ZTCM divides the entire area from which visitors originate into a set of visitor zones and then defines the dependent variable as the visitor or visitation rate (i.e. the number of visits made from a particular zone in a period divided by the population of that zone). The ITCM defines the dependent variable as the number of site visits made by each visitor over a specified period.

The basic **(zonal)** travel cost model defines a trip demand curve for a given recreational site from zone $j$ as (Markandya *et al*, forthcoming):

$$\frac{V_j}{P_j} = f(TC_j, X_j)$$

where

$V_j$ = the total number of trips by individuals from zone j to the recreational site per unit of time,

$f$ = the function that relates travel cost and socio-economic characteristics to visitation rates,

$P_j$ = the population of zone $j$,

$TC_j$ = the travel cost from zone $j$ to the recreational site[14], and

$X_j$ = the socio-economic characteristics of the population of zone $j$, which include, amongst others, factors such as income levels, spending on other goods, the existence of substitute sites, entrance fees and quality indices of substitute sites.[15]

The visitor or visitation rate $V_j / P_j$ is generally calculated as visits per unit of population, usually expressed in thousand persons, in zone $j$ .

[11] TC methods encompass a wide variety of models, ranging from the simple single-site TC model to regional and generalised models that incorporate quality indices and account for substitution across sites.

[12] The fundamental insight that drives this model is that if a consumer wants to use the (recreational) services of a site ther have to visit it. The travel cost to reach the site is considered as the implicit or the surrogate price of the visit, and changes in the travel cost will cause a variation in the quantity of visits.

[13] The value of the site is not given by the total travel costs; this data is only used to derive the demand curve, from which the consumer's surplus of visitors is calculated.

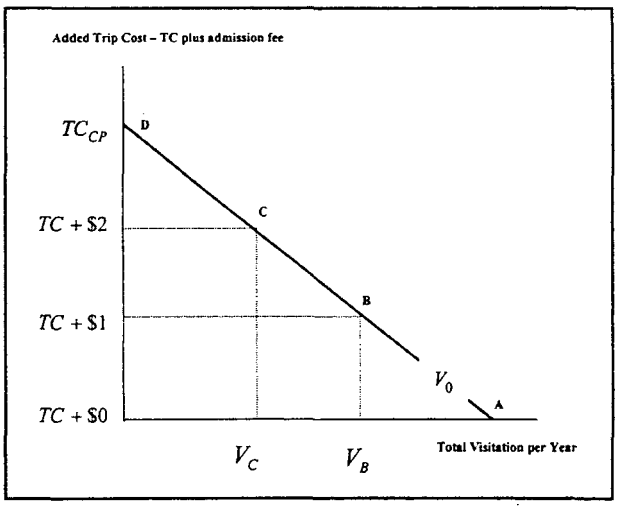
[14] Travel cost is the sum of expenditures incurred for gasoline, opportunity cost of time for travelling and for visit onsite.

[15] All these variables allow the estimation of different demand functions according to the socio-economic characteristics of the visitors, thus enabling better estimates of the consumer surplus. In practice they have rarely been used in zonal travel cost studies, being easier to be exploit in individual travel cost models.

Based on data obtained from a survey of site users, the above equation is estimated using regression analysis. This leads to the creation of a so called **'whole experience'** demand curve based on visitation rates and not the number of actual visits made. To estimate the consumer surplus accruing from the site, the **'whole experience'** demand curve is used to estimate the actual number of visitors and how the numbers would change subject to increases in admissions 'prices' - in essence constructing a classic inverse demand curve.

The base data set, from which the ***'whole experience'*** demand curve is created, defines one point on the inverse demand curve for the study site that is, the intersection of the present zero price line and the inverse demand curve ($V_0$). In Figure 8 this is given by point A, where admission fees or added trip cost is zero.

**Figure 8: Zonal Travel Cost**




As mentioned above, the remainder of $V_0$ is derived by assuming that visitors will respond to increases in admission fees in the same way they would to equal increases in travel cost. For each incremental increase in admission fees, the expected visitation rate from each travel origin zone is calculated using the above equation. The 'new' zone-specific visitation rates are then converted to expected numbers of visitors using data on $P_j$. These values are summed across all travel origin zones to find the predicted total number of visitors to the site at the added trip cost (i.e original travel cost plus, say, $1). For example, a $1 increase in trip costs may lead to point B in Figure 8, a $2 increase in trip costs may lead to point C, etc. This process is repeated until the added trip cost is sufficient to result in zero visitors to the site (the so-called **choke price** given by point D in Figure 8) - until the entire inverse demand curve ($V_0$) is traced.

The area under $V_0$ provides an estimate of the total consumer surplus enjoyed by present users of the study site.

The basic **(individual)** travel cost model relates individual's annual visits to the costs of those visits - that is (Markandya *et al*, forthcoming),

$$V_i = f(TC_i, X_i)$$

where

$V_i$= The number of visits made in a time period, say a year, by individual i to the site.

$TC_j$ = Travel cost faced by individual $i$ to visit the site.

$X_i$ = All other factors determining individual $i$'s visits (income, time, and other socio-economic characteristics).

This demand function can be extended to allow for the specification of a number of explanatory variables. These include the individual's estimate of the proportion of the enjoyment of the overall trip imputed to the specific site under investigation; the individual's view of the availability of substitute sites; size of individual's household; and whether the individual is a member of an environmental organisation, as well as other socio-economic data.

Integrating the demand curve between the actual travel cost $TCi$ and the **choke price** gives an estimate of the individual annual consumer surplus (ICS) for individual $i$

$$ICS_i = \int_{TC_i}^{CP} f(TC,X)dTC.$$

The total annual consumer surplus for the site is obtained by multiplying the ICS by the number of individuals visiting the site annually. The modelling of individual socio-economic features enables the estimation of consumer surplus for different socio-economic groups of visitors. Alternatively, the average ICS per visit can be calculated and then multiplied by the total annual number of visits to the site to get the total annual consumer surplus of the site.

Like hedonic techniques, the TC method has the advantage that it is based on observed behaviour. Also, TC is a well-tried technique which, is it generally accepted, yields plausible results. The individual TC model, the zonal TC model, or similar specifications, have been used to assess changes in site quality, which include the degradation of water quality, changes in fish catches, etc. However, they are more commonly used to value the total benefit of a resource, rather than changes in that resource. The TC method is not without its disadvantages however. To start, in complex situations, especially when changes in environmental quality are being assessed, the data requirements are considerable.

Moreover, *"a whole host of issues arises in the specification and estimation of the model and subsequent calculation of consumer surplus, all of which have enormous bearing on the final benefit estimates"*. These issues include the development of multi-site models, the valuation of travel time and the treatment of non-visitors. As a result, TC studies tend to be conducted as self-standing research studies, with sufficient resources to adequately address these complex issues.

The main application of TC in developing countries is to value tourist's WTP for national parks. For example, in Zimbabwe, a TC study of tourists found that they derived a benefit (consumer surplus) of about US$ 275 per person per trip to national parks. In Costa Rica, a TC valued trips to parks and reserves at US$ 1,150 per person.

*Averting or Preventative Expenditures*

The cost of actions taken to avoid or reduce damage from exposure to pollution provides a measure of the economic losses associated with that pollution. Theoretical explanations of **averting expenditures** (AE) are based on the household production function theory of consumer behaviour. In the context of averting behaviour models, the household produces consumption goods using various inputs, some of which are subject to degradation by pollution. The household may respond to increased degradation of these inputs in various ways that are generally referred to as averting behaviours. For example, the adverse impacts of groundwater pollution can be avoided in at least three ways:

- buying durable goods (e.g. point-of-use treatment systems);
- buying non-durable goods (e.g. bottled water); and
- changing daily routines to avoid exposure (e.g. boiling water for cooking and drinking, or reducing the frequency or length of showers/ baths).

Averting expenditures and the true benefits of a pollution reduction differ, since these expenditures do not account for all the pollution related costs that affect household utility; basically, the direct utility losses are not captured by the change in household production costs. Hence, it is generally accepted that under certain conditions averting expenditures provide a **lower bound** estimate of the true cost of increased pollution[16].

[16] The ability of AE techniques to provide a lower bound to WTP depends on the following assumptions: (1) averting inputs should not serve in the production for only one output that is valued by the household; (2) households should derive direct utility from the averting expenditure; and (3) the purchases of durable goods do not lower costs.

In general, AE (and RC) approaches are relatively easy to apply, since they use primarily engineering or market data. For the same reason, these techniques are also relatively inexpensive. This approach, however, may not measure all the benefits of an environmental improvement; or conversely, it may not account for all the economic losses of environmental degradation.

## 2.3.6 Valuation Techniques Based on Behavioural Linkages: Stated Preference

*Introduction*

While **revealed preference** techniques quantify consumer preferences indirectly by observing individual behaviour in **actual** situations, the other main group of valuation techniques based on behavioural linkages, seeks to measure consumer preferences for environmental amenities in **hypothetical** situations, drawing on responses to surveys. In short, individuals are asked to directly state their WTP (preference) for a certain level of an environmental good or service (see Box 5 below). Accordingly, techniques based on this approach are collectively referred to as **stated** preference methods.

The main type of stated preference approach is the contingent valuation method.

*Contingent Valuation*

In contrast to the techniques described above, which use observed data the contingent valuation (CV) method relies on structured conversations to directly elicit the values respondents place on some, usually non-marketed, goods or services. The basic notion underpinning CV is that a realistic, yet hypothetical market for buying or selling use and/or preservation of an environmental good/service can be described in detail to an individual. Individuals are then asked to participate in this hypothetical market, by responding to a series of questions.

The main features of the hypothetical market include:

- A detailed description of the good/service being valued. The situation before and after any proposed change in environmental quality and subsequent provision of the good/service should be clearly stated. In addition, it is vital that the respondents are perceiving the correct good/service.
- A detailed description of the "payment vehicle", i.e. the means by which the respondent would pay for the change in

provision of the good/service. The payment vehicle should be appropriate to the good/service and the hypothetical market. Moreover, it should be realistic and emotionally neutral.

- The procedure to elicit the respondent's valuation. The actual valuation can be obtained in a number of ways, for example, asking the respondent to name an amount, having them choose from a number of options. The respondent could also be asked whether they would pay a specific amount[17]. In the case of the latter, follow-up questions with higher and lower amounts are often used. Statistical analysis of the responses is then undertaken to estimate the average WTP in this hypothetical market.

**Box 5 Expressing Preferences in CV Studies**

An individual can be asked to express his or her subjective valuation of possible environmental changes in different ways:

**Environmental improvement.** The value of the environmental improvement in such a situation can be measured either by:

- The individual's maximum WTP to obtain the environmental improvement, or by
- The individual's minimum WTA compensation to forgo the environmental improvement.

**Environmental damage.** The value of the environmental damage in such a situation can be measured either by:

- The individual's maximum WTP to avoid the environmental damage; or by
- The individual's minimum WTA compensation to consent the environmental damage.

*Source: Adapted from Markandya et al (forthcoming)*

A general approach to follow when running a CV study is outlined in Figure 9.

The nature of CV means that, in principle, it can be used to value any change in environmental quality. Furthermore, CV can be used to accurately elicit values about very specific changes in the provision of goods/services, since it does not rely on observed data. Of course, this requires that the hypothetical market and elicitation questions are appropriately worded. An additional plus for CV is that, in contrast to the other valuation techniques described above, which only provide a partial estimate of the value of a good/service, CV can provide a measure of the TEV of a change in environmental quality.

CV methods have nonetheless been the subject of much criticism, mainly relating to their reliance on hypothetical markets (see Box 6). In short, some economists argue that asking individuals hypothetical questions only provides you with hypothetical answers, which cannot be meaningfully used to value environmental quality changes. Following controversy of the use of CV to value damages from the 1989 Exxon Valdez oil spill, the US Department of Interior and the National Oceanic and Atmospheric Administration organised a "Blue Ribbon" panel to assess the validity of using CV to value environmental damage. The panel concluded that that CV could provide useful and reliable information for this type of assessment, as long as certain guidelines are followed. In general, the profession as a whole has also given CV qualified acceptance.

In addition to the above conceptual concerns over the validity of CV based benefits estimates, survey-based research is expensive and timeconsuming; valid benefit estimates require properly designed sampling and enumeration procedures.

## 23.7 BenefitsTransfer

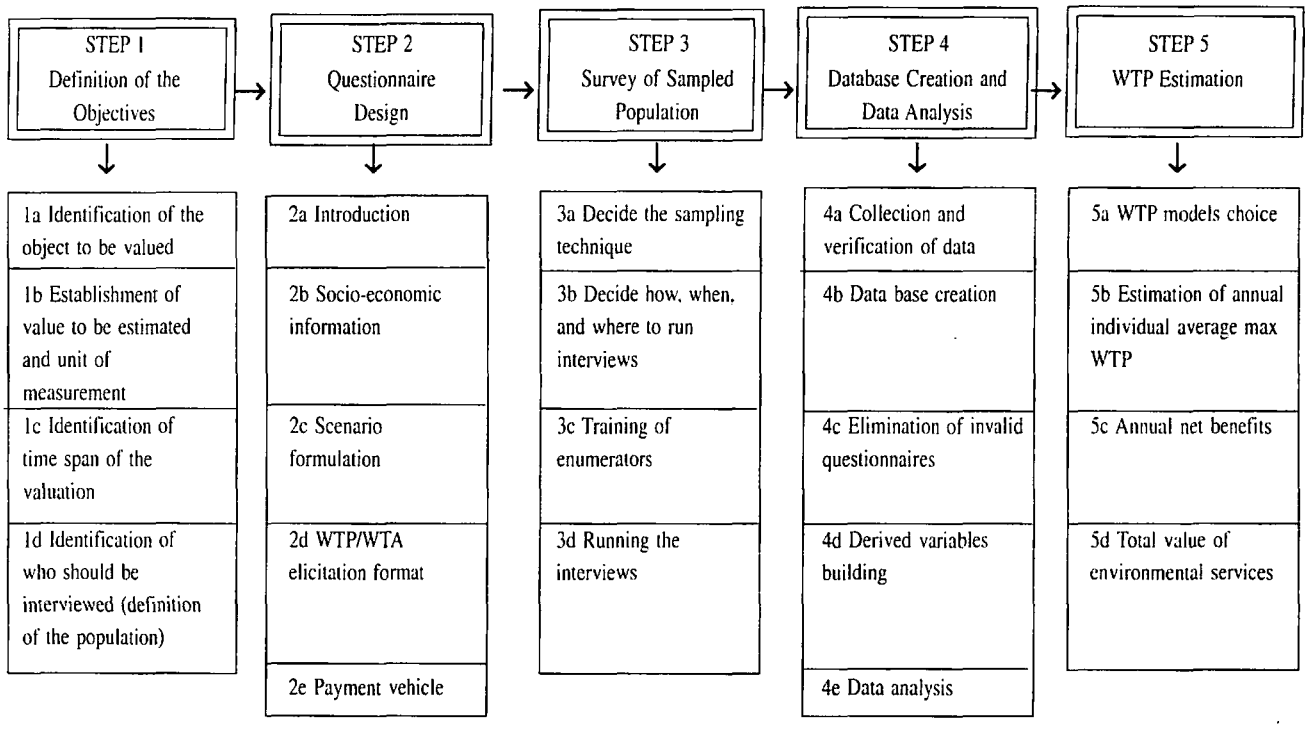
Benefits transfer is not a valuation methodology *perse*, but rather refers to the use of values obtained in one context to estimate values in a different context. A special issue of the *Water Resources Research* was devoted to benefits transfer, *and "provides the best available overview of the conceptual and empirical issues involved'* (World Bank, 1998)[18]

The concept of benefit transfer is best illustrated by way of example. If an analyst were asked to evaluate the cost of deterioration in water quality in a particular setting (known as the **policy site,** the analyst could opt to commission a new (primary) valuation study (e.g. a contingent valuation or averting expenditure study). Such an exercise however, would take considerable time and money. As an alternative, if a valuation study had been undertaken in one region (known as the **study site** that related willingness-to-pay (WTP) or willingness-to-accept (WTA) compensation for a change in water quality to variables such as demographics, income, scale, extent of environmental problems, and so forth, then one could potentially use the measures of WTP/WTA from the study site to approximate WTP/WTA for the deterioration in

[17] This technique is called the "referendum" or "dichotomous choice" approach, since it calls for a yes or no response.

[18] See Brookshire and Nell ( 1992) and all enclosed papers.

**Figure 9 Key Steps in Conducting a Contingent Valuation Study**




*Source: Markandya et al (forthacoming)*

**Box 6 Concerns Over Elicitation Formats in CV Studies**

Several issues concerning the accuracy and reliability of valuations based on CVM have been discussed in the literature. The major concerns regard the biases inherent in the technique, mainly the distortions in eliciting the consumer's preferences. Indeed, in order to obtain answers that reflect the true maximum WTP/minimum WTA of the respondent, different formats have been applied. The main formats are: (1) open-ended questions; (2) bidding games; and (3) dichotomous choice (referendum) questions.

Simple CVM exercises can be based on the so called **"open-ended"** elicitation format, where the individual is simply asked to state his/her maximum WTP or minimum WTA for a described environmental change. In this case, simple descriptive statistics (such as means and medians) can be used to obtain rough estimates of values attributed to an environmental asset. However, the main drawback of this approach is the ease with which the respondent can introduce "strategic bias", i.e. to state a WTP/WTA that is lower or higher than the true one in order to influence the decision making process for sake of his/her own profit. A second drawback of the "open ended" elicitation format is that the individual may not be prepared to express a value judgement starting without a reference point with which to bound his value judgement.

To avoid a high rate of misleading or missing answers caused by the lack of bounds typical of the open ended format, an iterative technique or **"bidding game"** can be used. In this case the respondent is asked whether he is willing to pay or to accept a given amount-of money for a change in the provision of an environmental attribute. If he refuses, the proposed amount is reduced (increased) by a given percentage (say, 10%). The procedure is repeated until the respondent provides a positive answer. The penultimate amount proposed is taken as his maximum WTP (minimum WTA) for obtaining (to give up) the environmental improvement. If instead the individual accepts the proposed amount, it is increased (reduced) by, say, 10%. The procedure continues until the individual answers negatively. Again the penultimate amount proposed is taken as his maximum WTP (minimum WTA) for obtaining (to give up) the environmental improvement. This technique however has the potential to suffer from so-called "starting point bias". It has been observed that the final value judgement is affected by the initial proposed amount. This means that the initial amount somewhat constrains the free expression of the true individual preferences.

To counter starting point bias and strategic bias, the **"dichotomous choice"** (referendum) format is often recommended. Here, a possible range of values for the Maximum WTP (Min WTA) of individuals is pre-set by the analyst. The sample of interviewed individuals is divided into sub-samples. A value within the pre-set range is assigned to each sub-sample. Each individual within a sub-sample is then asked whether he is willing to pay (to accept) the assigned value to obtain (or to compensate for) the environmental improvement (damage). He is not allowed therefore to select a figure as in the case of the open-ended format or to play with subsequent acceptance/refusal answers as in the bidding game format. Besides, he does not know the range of values within which the proposed amount is bounded. In this case however, the outcome of the individual answer is not the maximum WTP (minimum WTA) but only the consent or refusal to pay (to accept as compensation) a given amount of money, i.e. a WTP (WTA) which is not necessarily the maximum (minimum) one. Specific statistical techniques are therefore required to calculate the individual's valuation ofthe environmental change.

*Source: Markandya et al (forthcoming)*

water quality at the policy site. Assuming that the underlying preferences for water quality are the same in both situations, then the analyst could transfer the benefit estimates from the study site to obtain benefit measures at the policy site, hence, the term **benefits transfer** (BT).

Most benefit transfer methods utilised to date involve either the benefit value or benefit function approach. These are explained briefly in Box 7.

The primary appeal of benefits transfer is that it *'provides a low-cost way of estimating values when time and resources do not allow fuller valuation studies"* (World Bank, 1998). Moreover, a consensus seems to be emerging that benefits transfer can, under certain conditions, provide valid and reliable benefit estimates[19].

Also, if a damage estimate is available from one country, say country X and we want to transfer it to (convert it into a damage estimate for) another country, say country Y, then we need to make an adjustment to reflect differences in real income, and hence WTP to reduce damages, between the two countries. Markandya recommends the following equation be used to make such adjustments:

$$\text{Damage}_{Y} = \text{Damage}_{X} (\text{PPP GNP}_{Y} / \text{PPP GNP}_{X})^{E}$$

where PPP GNP denotes the purchasing power parity GNP per capita and E is the elasticity of damages with respect to real income. Two values of the elasticity E are suggested in the literature (1.00 and 0.35). for example, $E$ = 0.35 implies a 0.35% change in WTP relative to a 1% change in real income. An example is given in below.

**Box 7 Main Approaches to Benefit Transfer**

With the **benefit value approach**, a single point estimate (typically, the mean or median WTP per affected unit) is used to represent the results of a study, or selection of studies, that have been developed in a specific context. The average consumer surplus per trip to a water resource, for example, may be taken from a TC study. This unit value is then used to value one aspect of a change in, say, water quality at different locations. With the **benefit function approach**, a relationship between WTP and characteristics of the affected population and the water resource being assessed is modelled. The entire function (model) is then transferred to the site of interest and adjusted to conform as closely as possible to the population and resource characteristics at that site. A TC demand model derived at one site, for example, may be used in conjunction with the average travel costs, income, water quality conditions, etc. at another site, in order to estimate the benefits of changes in water quality at that site.

**Box 8: Example of Transfering Acute Mortality Cost from EU15 to Thailand**

With $E$ = 1.00

Original WTP-based mortality cost (value of a life year lost) from EU15 = $^{95}$ 112.950

$^{95}$ 1 12.950 * [$^{95}$ 6.970 / $^{95}$ 17.907]$^{1.00}$ = $^{95}$ 43.964

$^{95}$ 43.964 * $^{95}$ 1.31 per $^{95}$ = $^{95}$ 57,593 per year of life lost

## 2.3.8 Other Issues Relevant to Valuation

### *Applicability of Valuation Techniques*

Table 2 below summarises the main environmental impacts of projects or policy interventions, along with the main valuation techniques discussed above. A "Y" indicates that the valuation technique can generally be applied to the relevant impact. A question mark means that the valuation technique may apply, an "X" means that the valuation technique generally does not apply.

### *Issues of Time and Space*

The impacts of many environmental changes, whether positive or negative, are often only felt in the future, long after the activity that caused the change has ceased. Similarly, effects are often felt far beyond the boundaries of the project itself. Special attention must be given therefore, to the **temporal** and **spatial** boundaries ofthe analysis. ,

Since environmental impacts extend long beyond the normal life of the project, it ; is important to extend the time horizon of the analysis to include all the benefits and costs associated with environmental impacts even if they go further into the future than the normal life of a project. The time horizon of an analysis is determined by: (1) the number of actual years included in the analysis and (2) the ; discount rate used. Using too short a time horizon effectively ignores many environmental impacts, both positive and negative. For example, an activity that; results in the permanent loss of a capture fishery should include in the analysis the present value of the entire future loss of that resource, even if the activity itself ; only lasts for a few years.

The choice of an appropriate discount rate is also important, since a high discount rate effectively reduces to zero the present value of benefits and costs that occur

[19] See, for example ADB (1996).

many years in the future (see Box 9 below). This does not mean a different discount rate should be used when environmental impacts are important, in fact it is bad practice to mix discount rates within one analysis. But given the significance of the discount rate, it is vital to do sensitivity analysis using different discount rates. Such analysis can yield useful information to the decision maker when comparing altematives that have very different time profiles of benefits and costs (including environmental ones).

When environmental effects are present. careful thought must also be given to the appropriate spatial boundary of the analysis. The analyst often has to look far beyond the geographical boundaries of the project, especially when water or air pollution is involved.

With both spatial and temporal externalities, the important rule is to be transparent in the assumptions being made, and state explicitly the adjustments, which have been used in defining the analytical boundaries for the project or policy.

***Underestimation and Double Counting***

Whatever the actual techniques used to estimate the value of environmental benefits or damages, an important point to be borne in mind is the likelihood of **underestimation**. Inevitably, some components of TEV will prove impossible to estimate using any of the available techniques, either because of lack of data, or to the difficulty of extracting the desired information. For this reason, any estimates of value will likely underestimate the TEV, or that estimates of project benefits will be conservative, while estimates of costs will be optimistic. That some environmental benefits cannot be monetised does not mean that they should be ignored. Rather, any non-monetised impacts should be described quantitatively or qualitatively as far as possible. It may then be possible to use decision-analysis or multi-criteria techniques to compare monetised and non-monetised benefits/costs in a decision-making framework.

Another potential problem, which must always be considered, is the risk of **double counting**. The likelihood that total benefits will be underestimated because some benefits cannot be measured is well recognised. Less well recognised is the opposite danger - that benefits (even if accurately measured) might be overestimated because some benefits are counted twice. An example would be a project, which aims to reduce air pollution at a site by relocating or shutting down polluting activities. The benefits of this reduction could be estimated by predicting the reduction in the prevalence of respiratory illnesses, and valued using the COI approach. At the same time, suppose that a hedonic technique is used to estimate the value of overall environmental quality in the area. Since air pollution is part of area's environmental quality, treating these two estimates as though they described separate problems and adding the corresponding benefits together, would be inaccurate.

## 2.4 STAGE 3 - Apply Project Selection Criteria and Make Decision

### 2.4.1 Introduction

The primary purpose here is to discount (at an appropriate discount rate) all costs and benefits that occur in different time periods so that the present value net benefit of the project can be determined and a decision can be made on the relative economic merit of the project. The contribution of a project to the welfare of society - its economic merit - is typically assessed through the application formal project selection criteria. However, decision-makers are also interested in the certainty of anticipated project outcomes being realised, and in the distribution of net benefits among project participants. These issues are considered below.

In reaching a final decision, it is also important that non-monetised project impacts are considered in conjunction with indicators of project economic performance, uncertainties and distributional effects.

### 2.4.2 Formal Project or Policy Selection Criteria

Once the various impacts of a project or policy intervention have been identified, and the benefits and costs of the various impacts assessed, this information can be incorporated into the broader economic analysis of the project or policy. This is usually done in a cost-benefit framework through the application of formal section criteria, whereby the streams of benefits and costs of the proposed project/policy (including both direct project inputs and outputs, as well as environmental impacts to the extent that they can be identified and monetised) are compared over some period of time. There are three "discounted" project selection criteria widely used in investment (CBA) decision-making, the **net present value** criterion, the **benefit-cost ratio** and the **internal rate of return.** (They are referred to as 'discounted' criteria because they compare discounted streams of costs and benefits that occur at different points in time. This 'time dimension' of project evaluation was briefly highlighted in Box 9 above.)

The principal selection criterion is the **net present value** (NPV) method. The NPV of a project is given by

**Table 2: Applicability of Valuation Techniques in Developing Countries**

| Environmental Impact | Technical Linkage | Behavioural Linkage | | | | |
|---|---|---|---|---|---|---|
| | | Averting Expenditure | Hedonic Property | Wage-nsk | Travel Cost | Stated Preference |
| **Productivity:** | | | | | | |
| Soil loss | Y | Y | ? | X | X | X |
| Crop damage | Y | y | ? | X | X | X |
| Forest loss | Y | Y | X | X | X | ? |
| Habitat loss | Y | Y | X | X | ? | ? |
| Fisheries depletion | Y | Y | X | X | X | X |
| Water quality loss | Y | Y | X | X | X | X |
| Property damage | Y | Y | ? | X | X | X |
| Resource depletion | Y | Y | ? | X | X | X |
| **Human Health:** | | | | | | |
| Mortality | Y | Y | X | Y | X | Y |
| Work days lost | Y | X | X | X | X | X |
| Restricted activity days | X | X | X | X | X | Y |
| Pain and suffering | X | X | X | X | X | Y |
| Medical costs | Y | X | X | X | X | X |
| **Amenity:** | | | | | | |
| Recreation loss | X | ? | ? | X | Y | Y |
| Habitat loss | Y | Y | X | X | ? | Y |
| Aesthetic damage | X | Y | Y | X | X | Y |
| Noise | X | Y | Y | X | X | Y |
| **Other:** | | | | | | |
| Existence values | X | ? | X | X | X | Y |
| Occupational environ | X | X | X | Y | X | Y |
| Access to water | ? | X | ? | X | Y | Y |
| Sanitation services | X | X | Y | X | X | Y |
| Travel time savings | Y | X | Y | X | X | Y |

*Source: Adapted from Abelson (1996)*

the present value of the estimated benefits net of costs. For an **independent project,** i.e. a project that is not in any way a substitute for another project, the decision rule is to "accept the project if its NPV is greater than zero". A positive NPV indicates that a project's total benefits exceed total costs. By undertaking a project with a positive NPV, the welfare of society will increase by the magnitude of the estimated NPV. If, in contrast, the NPV of a project is negative, the funds that would otherwise have been used for this project should be allocated elsewhere. Undertaking a project with a negative NPV will make society "worse off,' than they were to begin with.

The **benefit-cost ratio** (B/C) is simply the ratio of the sum of the project's discounted net benefits (i.e. benefits minus costs) to the discounted investment costs. In terms of the B/C ratio, a project should be accepted "if its B/C ratio is greater than 1", this implies that its NPV is positive.

The **internal rate of return** (IRR) is used by most international financing agencies (e.g. the World Bank and the Asian Development Bank). The IRR is that

**Box 9 The Treatment of Time in Economics**

The service flows from natural environmental resources, occur over multiple time periods. The rate of consumption from one period to the next is referred to as the rate of time preference, which is defined at the individual level. This rate will be positive if people prefer one unit of consumption today as opposed to consuming that unit at some point in the future (typically, they do). In economics we account for this by discounting the value of services created in the future, so that they can be compared with services today, i.e. future values are converted to present ones. A future value (FV) generated years from now is related to the present value (PV) by the following relationship:

$$PV = \frac{FV}{(1+i)^t}$$

where $i$ is the rate of time preference (more commonly referred to as the **discount** rate).

**Box 10: Project selection criteria in CBA**

. The net present value (NPV) of a project is given by

$$NPV = \sum_{n=0}^{N} \frac{(b_n - C_n)}{(1+r)^n}$$

where $b$ and $c$ are benefits and costs in each period $n$ equal to O,l,...,$N$, and $r$ is the chosen rate of discount. For example, consider the road-building project shown in the table below. The initial expenditure on infrastructure is $60 million, including materials, equipment, labour, etc. Recurring maintenance costs are $6 million per year. The benefits fall in three areas: time saving, employment gains and health effects (resulting from, say, lower PM emissions)

**Road Construction Scheme**

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 |
|---|---|---|---|---|---|---|
| **Cost** | | | | | | |
| Investment cost | 60 | 0 | 0 | 0 | 0 | 0 |
| Maintenance cost | 5 | 5 | 5 | 5 | 5 | 5 |
| Total cost | 65 | 5 | 5 | 5 | 5 | 5 |
| **Benefits** | | | | | | |
| Time savinas | 6 | 6 | 6 | 6 | 6 | 6 |
| Health, Benefits | 12 | 12 | 12 | 12 | 12 | 12 |
| Employment Benefits | 15 | 3 | 3 | 3 | 3 | 3 |
| Total benefits | 33 | 21 | 21 | 21 | 21 | 21 |
| **Net Benefits** | -32 | 16 | 16 | 16 | 16 | 16 |
| NPV (10% discount rate) | 26.05 | | | | | |
| NPV (0% discount rate) | 48.00 | | | | | |

The net present value of the road scheme is given by the sum of the discounted net benefit stream. In this simple example, the NPV is positive, which would indicate that the project is economically desirable. As the table shows however, the discount rate applied can have an important impact on the net present value and hence the viability of the project.

The benefit-cost ratio (B/C) is generally defined as ;

$$B/C = \frac{\sum_{n=0}^{N} \frac{(b_n - C_n)}{(1+r)^n}}{\sum_{n=0}^{N} \frac{I_n}{(1+r)^n}}$$

where $I$ is the investment cost in year $n$. Using the same figures in the above example, the $B/C$ ratio is 0.43 for the case where $r$ = 10%.

The internal rate of return (IRR) is the discount rate that satisfies the following relationship

$$\sum_{n=0}^{N} \frac{I_n}{(1+r)^n} = \sum_{n=0}^{N} \frac{(b_n - C_n)}{(1+r)^n}$$

where all the symbols are as before. The IRR for the above example is 41 per cent.

rate of discount, which equates the discounted net benefits to discounted investment costs. The IRR may alternatively be seen as the rate of discount that sets the NPV equal to zero. A project is acceptable if its IRR is greater than the selected rate of discount (the NPV of the project will be positive). The IRR is widely used in business, where decision-makers are use to rate of return concepts. However, one needs to be careful when using the IRR criterion to rank projects relative to one another. In some cases, the IRR criterion will produce different rankings to the NPV criterion.

A numerical example of these three project selection criteria in relation to a road building scheme is illustrated in Box 10 below.

### 2.4.3 Sensitivity Analysis[20]

## Introduction

The NPV or IRR of a project is calculated using the most likely forecast values of benefits and costs. However, the stream of benefits and costs is influenced by a wide variety of factors that may vary from the base case. **Sensitivity analysis** shows the extent to which the project NPV or IRR changes for different values of the major variables.

Sensitivity analysis focus on alternative assumptions that have an unfavourable effect on the project result. It should be applied to all projects with quantified benefits and costs. The purpose is to identify actions that can mitigate the effects of uncertainty, or to redesign the institutional structure of the project to ensure economic sustainability.

***Methodology***

Sensitivity analysis involves recalculating the project results for different values of major variables where they are varied one at a time. Combinations of changes in values can also be investigated - although strictly speaking, this is really **scenario analysis**. Sensitivity analysis involves four steps:

- selecting those variables to which the project decision may be sensitive;
- determining the extent to which the value of such variables may differ from the base case;
- calculating the effect of different values on the project results by recalculating the project NPV/IRR; and
- interpreting the results and designing mitigating actions.

[20] Adapted from ADB (1999).

Some of the variables entering into the project cost and benefit streams will be predictable and small in value compared with total costs and benefits. It is not necessary to investigate the sensitivity of the project to such variables. Other variables may be larger and less predictable. Post-audit studies and previous project experience may indicate both the type of variable that is uncertain and the likely extent of divergence from the base case value. There are also some types of variable in every project that are likely to affect the project result and may be key variables for the project. Some likely candidates for sensitivity analysis include:

- Quantities of outputs and inputs can be affected by changes in technical or market conditions. Quantities should be broken down into their underlying components - for example, vehicle cost savings by type of vehicle, and the sensitivity of the project to each of the components considered.
- Changes in the values of the main project outputs, inputs, and investment costs may occur because of changes in prices for any of these items. Prices for major outputs and inputs can fluctuate considerably from year to year. The influence of the average annual forecast prices on the project worth should be tested by varying the forecasts.
- The timing and coordination of project activities may differ from the base case. The timing of investment costs that occur early in the project life can affect the measure of project worth considerably.
- Project results can be seriously affected by the extent to which the investment assets are utilized. Lower utilization rates (e.g. load factors for power plants) than in the base case will be reflected in lower output levels and lower operational costs, but without any decline in investment costs. The effects of a reduction in the rate of utilization should be investigated through adjustments to both benefit and cost streams.

The following procedure should be followed:

- Variables to which the project is likely to be sensitive should be listed. Alternative values should be assumed, based on previous project data where available. The change in the value of the variable should be calculated and expressed as a percentage of the original value.
- The project NPV/IRR should be recalculated for stated changes in variables one at a time.
- A **sensitivity indicator** (SI) - summarizing the effect of change in a variable on the project NPV - should be calculated where possible.

The SI is calculated as the ratio of the percentage change in the NPV to the percentage change in a variable. A high value for the SI indicates project sensitivity to the variable.

- A **switching value** (SV) may also be calculated. Where the base case shows a positive MV, the SV shows the percentage increase in a cost item (decline in a benefit item) required for the NPV to become zero. The SV is itself a percentage, the percentage change in a variable for the project decision to change. For many variables, the SV will be high, implying a very substantial change in the variable before the project decision is affected. For a few variables, the SV will be relatively low, showing there may be a significant risk for the project outcome.
- The change in the NPV may also be calculated for combinations of variables, for example, an increase in cost together with a lower output price.
- The results of the sensitivity analysis should be presented in a table showing the base case results, the change in each variable considered, the sensitivity indicator, the switching value, and the changes in project worth for cases where these indicators cannot be calculated.

The results of the foregoing sensitivity analysis should be reviewed considering the following questions:

- Which are the variables with high SIs?
- Have the calculations used the likely changes in these variables?
- Do the likely changes come close to, or exceed, the switching values that will change the project decision?
- How likely is it that the combinations of the variables investigated will occur?

These questions will help identify the truly key variables for the project. For the key variables identified in this way, a statement can be made of the likelihood of the variation tested actually occurring and the measures that could be taken to mitigate or reduce the likelihood of such variations from the base case.

### *An Example*

Consider an irrigation rehabilitation project. The project involves a predicted increase in cropped area for irrigated rice, in cropping intensity, and in yield, as a result of rehabilitation, with a compensating decline in vegetable cropped area. The base case result are as follows (see Table 3 below): IRR of 19.0 percent and NPV of Rs 1,440 million at 12 percent discount rate.

On the basis of previous rehabilitation projects, there is uncertainty over the farmer response to improved irrigation. Post-audit studies indicate the possibility of lower values for cropped rice area, cropping intensity and yield by 9, 10 and 6 percent, respectively. There is also uncertainty over the levels of cropping intensity and yield of both vegetables and rice, without the project. Increases in these variables of 10 percent have been included in the sensitivity tests.

The forecast price of rice and fertilizer should be key variables in the project analysis, as the project will increase both the quantity of rice output and the quantity of fertilizer input. In the sensitivity analysis, the forecast price of rice, which declines over the first ten years of the project anyway, is predicted to follow the same pattern but to be at the level of the lower range of the 70 percent distribution given together with the basic World Bank price forecasts. This is equivalent to a price 39 percent lower than in the base case. On a similar basis, the fertilizer price is tested at a price 42 percent higher than in the base case, at the higher range of the 70 percent distribution.

The results of these sensitivity tests on underlying and specific benefit and cost factors are given in Table 3. There are three variables to which the project is most sensitive and to which most attention should be paid. These include the price of rice, the cropping intensity, and the yield for rice with the project. The forecast values for these variables need only be less favourable by 20 and 14 percent for the project decision to change.

**Table 3: Example- Results of Sensitivity Analysis**

| Item | Change (%) | NPV (Rs Mn) | IRR (%) | Sensitivity Indicator | Switching Value (%) |
|---|---|---|---|---|---|
| **Base Case** | | +1,440 | 19.0 | | |
| **Costs** | | | | | |
| Investment costs | +10.0 | +1,291 | 17.9 | 1.03 | +97 |
| Fertilizer price | +42.1 | +7S3 | 15.8 _ | 1.13 | +88 |
| **Benefits** | | | | | |
| Rice price | -38.9 | -1,427 | 1.7 | 5.12 | -20 |
| *With:* | | | | | |
| Rice area | -9 | +1,298 | 18.3 | 1.10 | -91 |
| Rice cropping intensity | -10 | +446 | 14.3 | 6.90 | -14 |
| Rice yield | -6 | +844 | 16.2 | 6.90 | -14 |
| *Without:* | | | | | |
| Rice cropping intensity | +10 | +873 | 16.3 | 3.94 | +25 |
| Rice yield | +10 | +873 | 16.3 | 3.94 | +25 |
| Vegetables yield | +10 | +1,162 | 17.7 | 1.93 | +52 |

***Source:*** *Adapted from ADB (1999)*

### 2.4.4 Distribution of Project Impacts

The costs and benefits of a project are shared among different groups in society. Knowledge of how these impacts are allocated among affected parties is often of interest to policy makers. There are several ways in which the distribution of project impacts can be analysed (ADB, 1999):

- The project impacts can be allocated among different project participants e.g. suppliers, consumers, owners, lenders, producers, and the government.
- For projects that involve foreign investors, lenders, labour, etc., project impacts can be allocated between nationals and foreigners.
- Project impacts can be allocated between the public and private sectors.
- In addition to allocating project impacts among different project participants, they may also be split between participants with different income levels.

Considerable effort has been expended to include the distribution of net impacts by income group in to CBA. The primary purpose is to identify, and give priority, to projects that enhance the incomes of lower income groups. This is accomplished by applying different weights to the net impacts accruing to different income groups - higher weights are given to those impacts accruing to the relatively 'less well-off', lower weights are given to those impacts accruing to the relatively 'more well-off'.

Examples of the derivation of 'distributional weights' for India and Mauritius may be found in Markandya (1998) and Markandya and Boyd (1999).

However, several objections have been raised regarding the use of such weighting systems, including that this form of analysis depends on specifying assumptions that are essentially subjective and open to disagreement. In addition, the distribution of income can be affected more directly by policy changes at the national level rather than through the net effects of new investment projects. Consequently, instead of applying any premium to incomes accruing to particular income groups, it is generally recommended that simply a statement of the distribution of project effects be given.

# 3 COST-EFFECTIVENESS ANALYSIS

## 3.1 Introduction

Once all the resource inputs to a project or policy intervention have been appropriately valued, and the outputs at least quantified, if not also valued, the next step is to assess the (economic) desirability of the project or policy intervention. In the previous section we considered one of the main analytical method used to assess the economic desirability of projects or policies - cost benefit analysis (CBA). The main alternative to this method is **cost-effectiveness analysis** (CEA). In fact, when dealing with the vehicle emission-fuel quality-air quality management nexus, the World Bank advocates the *"use of coste effectiveness analysis as the primary criterion for selecting optimal strategies across various sources and sectors"* (World Bank, 1999, informal issues note "Urban Air Quality Management: Transport-Energy-Environment Nexus").

## 3.2 What is Cost-effectiveness Analysis?

### 3.2.1 Purpose and Scope

**Cost effectiveness analysis** (CEA) is basically concerned with finding:

- the least-cost (most efficient - minimises resource use) way of achieving a predetermined goal, e.g. a given reduction in emissions of a targeted pollutant; or
- the project or policy package that yields the greatest benefit (e.g. a reduction in exposure to certain toxic VOCs) subject to a compliance cost constraint (or other resource constraint).

The mechanics of pursuing both agendas are the same - the difference is primarily a matter of emphasis - as are the cost definitions and the way in which they are quantified.

In contrast to CBA, CEA does not require the desired output (benefit) of the project or policy intervention to be expressed in monetary terms. Only the inputs (costs) of the project are monetised, it is sufficient to express the benefits in physical units, e.g. in tonnes of $CO^2$ equivalent abated per year. This is particularly advantageous when valuation of the outputs is controversial, uncertain, or both. The underling principle of CEA, as illustrated by the first bullet point above, is to determine the effectiveness (in terms of the total required expenditure) of a project in producing a specific level of benefit. At a policy level, CEA is primarily used to identify the 'least-cost' strategy for meeting a set objective[21]. In the context of GHG limitation, for example, CEA may be used to achieve a given reduction target at 'least-cost'. In the health and water resource sector, CEA may be used, for example, to identify the programme that saves the most lives for a fixed investement or 'least-

[21] CEA may equally be used to maximise physical benefit subject to a cost constraint (second bullet point above).

cost' option for balancing supply and demand for potable water, respectively.

CEA is not restricted to private costs, as defined above. It is often desirable in many contexts - air pollution and water supply - to work with a broader definition of costs, e.g. **social costs.** Hence, in addition to resource inputs costs, the cost component may also include avoided costs (e.g. savings in fuel costs), secondary benefits (e.g. the social costs of unemployment avoided, collateral air emission savings) and implementation ('hidden') costs. In this respect, only the monetary benefits associated directly with the physical output (or policy objective) are excluded.

For example, the (purely financial) cost-effectiveness of replacing diesel buses with Original Equipment Manufacture (OEM) Liquefied Petroleum Gas (LPG) equivalents in reducing greenhouse gas (GHG) emissions has been estimated at $US 1,800 per tonne of $CO_2$ equivalent abated (Markandya and Boyd, 1999). However, when the cost component is expanded to include the social costs of unemployment avoided and secondary (collateral) emission savings - i.e. as the cost analysis moves towards the concept of full social cost - the cost-effectiveness of the bus replacement programme becomes $US 600 per tonne of $CO_2$ eq. abated. In both cases, the primary output of the policy (i.e. to reduce GHG s emissions) remains non-monetised; only what is included in the cost component changes.

### 3.2.2 Measuring Cost-effectiveness

The cost-effectiveness of a project or policy intervention in delivering a given output, for example, reductions in GHG emissions can be assessed in one of two ways. In general terms these are given by:

- Cost-effectiveness ($/unit reduction) = the present value of the 'net' incremental cost stream of the project÷ the present value of the stream of annual emission savings associated with the project or policy intervention.
- Cost-effectiveness ($/unit reduction) = the **total annual cost** of the project of policy intervention÷ the average annual emission savings associated with the project.

In most situations, both approaches produce the same measure of cost-effectiveness. However, the (former) net present value approach offers greater flexibility in terms of manipulating key input parameters, e.g. accounting for variations in prices or emission factors over time.

There are also two main approaches given in the literature for calculating the total annual cost of a project. Estimates of the **total annual cost** of a project are a necessary input to the cost-effectiveness calculations. In general terms, total annual costs are given by:

- Total annual cost = annual capital cost (yearly depreciation charge plus average interest cost per year) + net annual operating and maintenance costs.
- Total annual cost = the present value of the total cost stream (investment expenditure plus net operating and maintenance costs) x capital recovery factor.

Again, in most situations, both approaches produce the same estimate of total annual cost, but the second approach offers greater flexibility.

The attached Annex provides an example of how each of the above approaches to estimating total annual costs can be applied to a generic project. The use of these estimates to calculate the cost-effectiveness of the project in abating GHG emissions is also illustrated.

### 3.2.3 (Abatement) Cost Curves[22]

A useful way of presenting the analytical results from a cost-effectiveness analysis to use (abatement) cost curves. While the presentation to follow is in terms of air emission limitation scenarios, it could equal apply to scenarios to balance water supply and demand, meet health targets, etc. In the context of GHG limitation, abatement cost curves express *"the relationship between the minimum cost to society of reducing an additional tonne of GHG and the corresponding level of emission reduction"* (Halsneas *et al*, 1998). Costs and emission reductions are, of course, defened relative to the baseline case. Typically, potential emission reductions (e.g. tonnes of $CO^2$ equivalent) are presented on the horizontal axis, and the cost of abating one tonne of emission (e.g. US $ per tonne CO equivalent) on the vertical axis. Constructing abatement costs curves begins by determining the cost-effectiveness of those limitation measures under consideration (as illustrated above). Limitation measures are then sequentially ranked in order of increasing unit costs. Abatement cost curves therefore rise to the right, reflecting the fact that increased levels of abatement can be achieved at higher and higher unit costs.

Cost curves may take various forms (e.g. 'single' cost curves or 'compound' cost curves), be assessed relative to different baselines (e.g. base year emissions or future baseline scenario emissions), and be constructed in one of several ways (e.g. the 'partial approach, the

[22] Other terms frequently used include "avoidance cost curves", "emission reduction marginal cost curves", "incremental cost of, abatement curves", "abatement supply curves", "elimination cost curves" or "prevention cost curves".

'retrospective systems' approach, or the 'integrated systems' approach).

*Single cost curves are curves that consider only one pollutant. This type of cost curve, for example,* may show only $SO_2$ emission reduction potential and the associated costs. **Compound cost curves,** as the term implies, consider more than one pollutant and hence represent more comprehensive environmental issues. The latter is more likely to be used, for example, in transport scenarios owing to the fact that vehicles produce multiple pollutants, including PM, HC and CO reductions in PM, HC and CO are shown on the same curve, expressed in terms of PM equivalents.

Cost curves can be constructed in several ways, including:

- ***Partial solutions:*** This is where each option for mitigation is taken **separately.** Measures of the cost-effective of each option are computed relative to a baseline. The mitigation options are then ordered by unit costs of GHG reduction, and in this way, the curves are constructed using the (unit) costs of individual options. With this approach, interactions between measures are not taken into account. Consequently, the environmental performance of the listed measures may not be strictly additive. Such interactions need to be addressed before an accurate abatement cost curve can be constructed, from which an effective control programme can be designed. In short, curves based on this construction approach, represent a **starting point** for developing a mitigation programme; they show broad relationships between emission reduction potentials and the cost of those reductions.
- ***Retrospective systems approach:*** This approach extends the partial solutions approach, by explicitly considering the interdependence of mitigation measures and, if appropriate, previous abatement projects. For example, implementing one measure may alter the cost-effectiveness of another measure - by altering the emission baseline when measures are used in sequence or in parallel. In contrast to the partial solutions approach, this approach, through an iterative technique, re-ranks mitigation options making adjusting for interactions between measures that are not mutually exclusive. Abatement curves based on this approach provide sufficient detail to estimate the specific costs of mitigation scenarios.
- ***Integrated systems approach:*** This approach uses a full (e.g. transport) system model to derive the cost curves. With the integrated systems approach, a detailed modelling exercise is undertaken - takcing into account all relevant interactions in the sector- resulting in a smooth marginal abatement cost curve, as opposed to the standard 'step-wise' curves Consequently, any point on the cost curve represents a least-cost solution for the system. Clearly, relatively more accurate mitigation strategies are developed under this approach, in contrast to the two previous approaches.

For a more detailed discussion of these variations see Halsneas *et al* (1998), Riso National Laboratory (1994) or FSO (1996).

An example of a 'partial' solutions cost curve for selected GHG limitation measures in Mauritius is shown in Figure 10 below. The data set from which this curve was derived is given in Table 4.

### 3.2.4 Limitations of CEA

As mentioned above, a relative advantage of CEA is that the direct policy outputs do not need to be valued. This is fine where a prior decision has been made, for example, to reduce $CO_2$ eq. emissions to a desired level, and the issue is solely to do so at 'least-cost'. Cost-effectiveness analysis, however, cannot determine the level of emission reductions to be achieved, nor can it be used to compare different air quality standards. Moreover, CEA says little about the total budget to be allocated to the policy objective. Most projects involve one or other of these issues.

Therefore, when a project has more than one output, requires a decision about the standards to be achieved or the total budget to be allocated, and were acceptable values can be placed on the primary outputs, then CBA has some merit as a decision-support tool over CEA.

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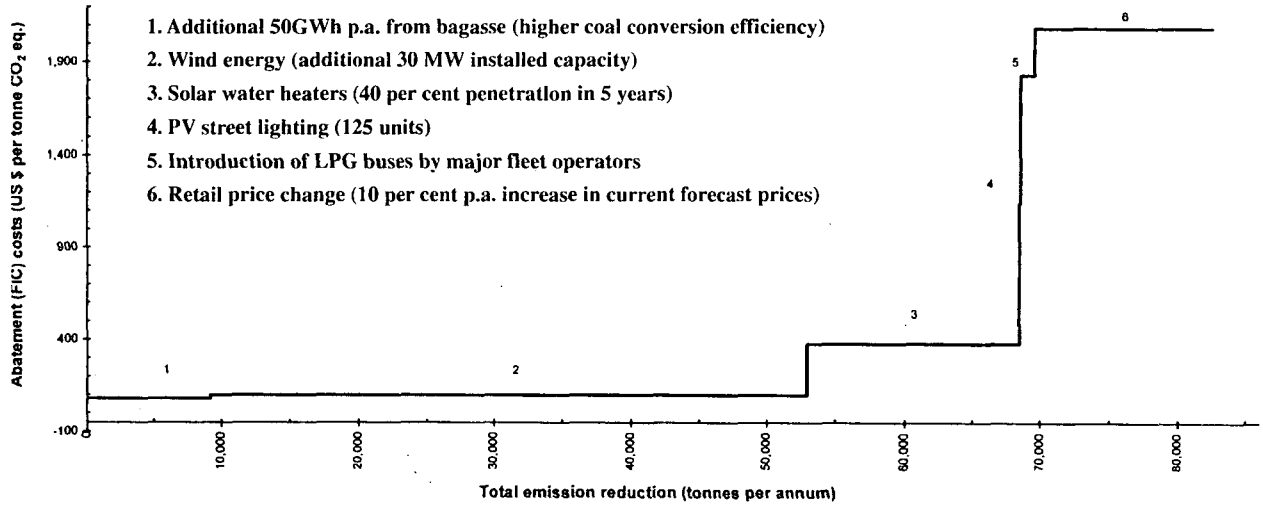
**Table 4: Summary of Financial Cost-effectiveness for the Selected GHG Limitation Measures**

| GHG Project | Unit Abatement Cost (US$ per tonne $CO_2$ eq.) | Total Annual Abatement ($CO_2$ eq. per annum) | Total Abatement Cost (US$ '000 per annum) | Accumulated Abatement Cost (US$ '000 per annum) | Accumulated Annual Abatement (tonnes CO2 eq. per annum) |
|---|---|---|---|---|---|
| 1. Additional 50 GWh from bagasse (higher coal conversion efficiency) | 78 | 9.228 | 723.7 | 723.7 | 9.228 |
| 2. Wind energy(additional30MW installed capacity) | 97 | 43.762 | 4.238.5 | 4.962.2 | 52.990 |
| 3. Solar water heaters(40% penetration over 5 years) | 378 | 15.512 | 5.858.0 | 10.820.2 | 68.502 |
| 4. PV streetlights (125 units to displace 500 kWh of electricity per day) | 1.140 | 122 | 139.1 | 10.959.3 | 68.624 |
| 5. Replacement of diesel buses with OEMLPG buses | 1.832 | 1.064 | 1,949.3 | 12,908.6 | 69,688 |
| 6. Ten per cent increase in the current forecast annual electricity tariff | 2,090 | 12.958 | 27,077.1 | 39.985.7 | 82,646 |

***Source:*** *Markandya and Boyd (1999)*

**Figure 10: Financial Cost Curve for GHG Limitation Measures in Mauritius**




***Source:*** *Markandya and Boyd (1999)*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# 4 ANNEX ILLUSTRATION OF COST EFFECTIVENESS CALCULATIONS

## 4.1 Calculating Annual Costs

The two main approaches for computing total annual costs presented below are based on the following data set:

| | |
|---|---|
| **Investment expenditures:** | $250,000 |
| Infrastructure | $187,500 |
| Equipment | $62,500 |
| **Net recurring costs:** | |
| Energy | $20,000 |
| Labour | $50,000 |
| Materials | $5,000 |
| Offsetting benefits | $0 |
| Equipment lifetime | 5 years |
| Appropriate interest rate | 8% |

For simplicity, it is assumed that the capital goods have no resale or salvage value.

### 4.1.1 Approach 1 - Depreciation Plus Interest Calculation

With this approach the annual cost of the project is obtained by summing the yearly capital and net recurring costs. The capital cost in each year is made up of a **depreciation charge** and the **interest cost** on the outstanding capital balance.

***Depreciation Charge***

The simplest method for depreciating the capital goods is the straight-line method. This method assumes that these goods contribute their services equally to each year's operation so that the total investment expenditure is evenly allocated over the lifetime of the equipment. Thus the yearly depreciation expense is a constant given by:

$$R_t = \frac{W}{n}$$

where $R_t$ is the depreciation charge in year $t$, $W$ is the depreciation base of the equipment, i.e. the difference between the original cost of the capital goods ($C_0$) and the salvage value ($S_n$). and $n$ is the estimated useful lifetime of the equipment in years (or write-off period).

The accumulated depreciation $D_t$ at the end of year $t$ is then given by

$$D_t = t\, R_t$$

The book value $B_t$ of the equipment, i.e. the unamortised portion, at the end of year $t$ is

$$B_t = C_0 - t\,R_t.$$

Using the straight-line method, the depreciation schedule for the capital goods is given below.

| End of year | Yearly depreciation charge | Accumulated depreciation | Book value |
|---|---|---|---|
| *(t)* | $R_t$ | $D_t$ | $B_t$ |
| 0 | - | - | $250,000 |
| 1 | $50,000 | $50,000 | $200,000 |
| 2 | $50,000 | $100,000 | $150,000 |
| 3 | $50,000 | $150,000 | $100,000 |
| 4 | $50,000 | $200,000 | $50,000 |
| 5 | $50,000 | $250,000 | |

*Interest Cost*

In purchasing the capital goods, $250,000 is essentially being tied up. If these funds were not invested in the new equipment, they could be invested either in something else which will earn a return, or, if there are loans which are repayable, this indebtedness can be reduced and the interest cost saved. An annual interest cost should, therefore, be included in the annual cost calculation.

It is incorrect however, to compute the annual interest cost as 8 per cent of $250,000 (i.e. $20,000), as the investment is being reduced each year by the depreciation-recovery charge of $50,000. The $20,000 is the valid interest charge in the first year only. In general terms, the average interest cost per year is given by

$$\text{average interest cost per year} = \frac{1}{2}\left(C_0\, r + \frac{C_0\, r}{n}\right)$$

where $r$ is the interest rate per period. In this case, the average yearly interest cost is $12,000. This is the appropriate amount to use in the annual cost calculations.

The annual capital cost of the capital goods is therefore equal to $62,000, that is, the sum of the average yearly depreciation charge ($50,000) and the average yearly interest cost ($12,000). To this the net annual recurring costs must be added to determine the total annual cost of the project. Hence, the total annual cost of the project should read:

| | | |
|---|---|---|
| **Annual capital cost:** | | |
| Yearly depreciation charge | +$50,000 | |
| Interest cost per year | +$12,000 | |
| Sub-total | = | +$62,000 |
| **Net recurring costs:** | | |
| Energy | +$20,000 | |
| Labour | +$50,000 | |
| Materials | +$5,000 | |
| Offsetting benefits | -$0 | |
| **Sub-total** | = | +$75,000 |
| **Total annual cost** | | = $137,000 |

Expressed in general terms, the total annual cost of a project or policy intervention, using the 'depreciation plus interest' approach, is given by

$$\textbf{toal annual cost} = \frac{(C_0 - S_n)}{n} + \frac{1}{2}\left(C_0\, r + \frac{C_0\, r}{n}\right) + RC$$

where RC is the 'average' net annual recurring costs.

## 4.1.2 Approach 2. Discounted Cash Flow Approach

An alternative to the above approach, and one which offers greater flexibility, involves first determining the present value total cost of the project, and then applying a capital recovery factor. The present value total cost ( PVC ) of an investment is computed as follows:

$$PVC = \sum_{t=0}^{n} \frac{(C_t + RC_t)}{(1+r)^t}.$$

The present value of the total cost stream of the project is $549,500; the calculations are summarised in the table below. This represents the total cost to be recovered in equal annual amounts (denoted by $A_t$) over the lifetime of the equipment. Therefore, the total annual cost of the project is given by:

$$A_t = PVC\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right] = \$549{,}500\left[\frac{0.08(1.08)^5}{(1.08)^5 - 1}\right] = \$549{,}500\,(0.2505) = \$137{,}600$$

The second term in the brackets is the **capital recovery factor.** This approach offers greater flexibility in that it provides a framework for explicitly considering, for example, the effects of price escalation on the various recurring cost components.

In this example, the $137,600 total annual cost is not significantly different from the $137,000 cost calculated through the depreciation plus average interest approach.

| Year | 0 | 1 | 2 | 3 | 4 | 5 |
|---|---|---|---|---|---|---|
| 1 Discount factor | 1.000 | 0.9259 | 0.8573 | 0.7938 | 0.7350 | 0.6806 |
| 2 Investment expend. (a+b): | $250,000 | - | - | - | - | - |
| a Infrastructure | $187,500 | - | - | - | - | - |
| b Equipment | $62,500 | | | | | |
| 3 Recurring costs (a+b+c-d): | - | $75,000 | $75,000 | $75,000 | $75,000 | $75,000 |
| a Energy | - | $20,000 | $20,000 | $20,000 | $20,000 | $20,000 |
| b Labour | - | $50,000 | $50,000 | $50,000 | $50,000 | $50,000 |
| c Materials | - | $5,000 | $5,000 | $5,000 | $5,000 | $5,000 |
| d Offsetting benefits | | | | | | |
| 4 Total cost (1+3) | $250,000 | $75,000 | $75,000 | $75,000 | $75,000 | $75,000 |
| 5 Discounted total cost (1*4) | $250,000 | $69,443 | $64,298 | $59,535 | $55,125 | $51,045 |
| 6 PVC (sum line 51 | $549,446 | | | | | |

Expressed in general terms, the total annual cost of a project or policy intervention may be calculated with the use of a capital recovery factor in one of two ways:

$$\text{total annual cost} = \left[\sum_{t=0}^{n}\frac{(C_t + RC_t)}{(1+r)^t}\right]\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right] \text{ or}$$

$$\text{total annual cost} = C_o\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right] + RC .$$

The latter way is appropriate if the annual recurring costs are constant over the life of the project.

## 4.2 Measures of Cost-effectiveness

The cost-effectiveness of a project or policy intervention in delivering a given output may be assessed in one of two ways: one approach is based on the concept of **present value;** the second approach is based on **annualised cost data**. Using the output from the above cost calculations, we will illustrate how these two approaches are used to estimate the cost- effectiveness of the project.

### 4.2.1 Approach 1. The Present Value Approach

Under this approach, the cost-effectiveness a project in delivering a given output (which we denote by AC ) is formally given by:

$$AC = \frac{\sum_{t=0}^{n}(C_t + RC_t)\cdot(1+r)^{-t}}{\sum_{t=0}^{n}(E_t)\cdot(1+r)^{-t}}$$

where $E_t$ is the unit output (e.g. emission reduction) in year $t$, and all other notation is the same as that used above. The denominator is commonly referred to as the **present tonnes equivalent** (PTE). If the project abates, say, 50 tonnes of $CO_2$ eq. per year, the PTE is therefore 200 tonnes of $CO_2$ eq. We know from above that the present value of the total cost stream of the project is $549,500. Hence, the cost-effectiveness of the project in reducing $CO_2$ eq. emissions is given by:

$$AC = \frac{\sum_{t=0}^{5}(\$250,000 + \$75,000)\cdot(1.08)^{-5}}{\sum_{t=0}^{5}(50\text{ t CO}_2\text{ eq})\cdot(1.08)^{-5}} = \frac{\$549,500}{200\text{ t CO}_2\text{ eq}} = \$2,750/\text{t CO}_2\text{ eq}$$

### 4.2.2 Approach 2. The Annualised Cost Approach

Under this approach, the cost-effectiveness a project in delivering a given output (which we denote by AC ) is formally given by:

$$AC = \frac{\left[\sum_{t=0}^{n}\frac{(C_t + RC_t)}{(1+r)^t}\right]\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right]}{E_t} \text{ or } \frac{C_0\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right] + RC_t}{E_t}$$

where all notation is the same as that used above. Recall that the numerator is simply the total annual cost.

Previously we calculated the total annual cost of the project as $137,600. Given that the project abates 50 tonnes of $CO_2$ eq. per year, the costeffectiveness of the project in reducing $CO_2$ eq. emissions is given by:

$$AC = \frac{\$250,000\left[\frac{0.08(1.08)^5}{(1.08)^5 - 1}\right] + \$75,000}{50\text{ t CO}_2\text{ eq}} = \frac{\$137,600}{50\text{ t CO}_2\text{ eq}} = \$2,750/\text{t CO}_2\text{ eq} .$$

In this example, the two approaches produce the same estimate of AC .

# Chapter 4

# MARKET BASED INSTRUMENTS FOR ENVIRONMENTAL POLICY IN DEVELOPING COUNTRIES

*by*

*Anil Markandya*

## 1. INTRODUCTION

The purpose of this module is to provide the tools, exercises, and case studies relevant to the design and implementation of the economic instruments that are employed in environmental regulation. The module begins by presenting a number of analytical tools commonly used in environmental economics (section 2). These tools provide a simple but powerful analytical framework to which one can refer when discussing the advantages and disadvantages of the ! various instruments and policies typically used by regulators to control the consumption of natural resources. The instruments themselves are reviewed in Section 3. Section 4 provides a brief review of experiences with these instruments, in particular with the implementation of economic instruments for pollution control. These experiences are reviewed with the intent of providing lessons for future use of economic instruments.

## 2. ANALYTICAL TOOLS

### 2.1 Social costs

A key concept behind the case for economic instruments for environmental regulation is that of social costs which are defined as the sum of private and external costs. Private costs are costs that are directly supported by the consumer in consumption or by the producer in production. External costs are costs that are borne by a consumer or producer other than the consumer or producer conducting the consumption or production activity.

Examples of private costs are common. In his/her consumption activities, the consumer supports a large number of private costs: housing, transport, energy, time, etc. Market prices for goods and services are obviously examples of private costs since these must be incurred in order to enjoy the consumption of the said good or service. Similarly, the enterprise must incur input costs (such as labor, capital, energy, etc.) in order to produce output. A large number of these consumption and production activities are also the source of costs that are supported by other agents in the economy. For example, a consumer's decision to drive a car from point A to B (for which private costs are supported, e.g. gasoline), is the source of costs for other individuals: noise, air pollution, contribution to traffic congestion, etc. These are costs external to the car user in the sense that they are not taken into account in the decision to drive or not to drive (or how many miles to drive). Similarly, a large number of enterprises discharge pollution while conducting their production activities. The damages created by this pollution are a cost that is external to the enterprise discharging the pollution - in that it does not support and therefore does not take into account this cost in its production (or location) decisions The social costs include all private and external costs associated with any consumption and production activities.

### 2.2 Social benefits

In order to define social benefits, a distinction has to be made between *private* and *external* benefits. These concepts are defined here. Private benefits are those that are obtained directly by the consumer from consumption activities, or by the producer in production activities. External benefits are benefits obtained by a consumer or producer other than the consumer or producer conducting the consumption or production activities. The social benefit of an activity is the sum of its private and external benefits.

Similarly to the case of private costs, examples of private benefits abound. In fact, it may be said that all consumption activities are undertaken because of the private benefits they generate: the consumer obtains satisfaction (or utility), the producer obtains profits. Some of these consumption or production activities also generate benefits to other individuals or enterprises. For example, a power plant that switches from mazut to natural gas as a source of fuel may do so solely to reduce its production costs. However, in doing so, it reduces the amount of air pollution it generates, which is a source of benefits (or reduced external costs) for those living in the vicinity of the plant. For example, these benefits may be a reduction in the health costs associated with air pollution. The social benefits include all private and external benefits associated with any consumption and production activities.

## 2.2 Externalities

Externalities arise because of market, policy or institutional failures, as shown in module 2. Externalities exist when economic transactions between two or more parties result in an impact on a third party, who is not involved in the transaction. In other words, externalities exist when not a costs or benefits are taken into consideration by consumers and producers when conducting their consumption and production activities. A distinction is made between *positive* externalities and *negative* externalities. Positive externalities exist when the social benefits associated with a consumption or production activity are larger than the private benefits. This happens when an activity generates external benefits. Negative externalities exist when the social costs of a consumption or production activity are larger than the private costs. This happens when an activity is the source of external costs. Pollution is an obvious example of a negative externality.

## 2.3 Public goods

A public good is a good that has the following important characteristic: If the good is made available to one person in the economy, then it becomes available to everyone else. In other words, if the good is made available to one person, it is not possible to exclude other persons from consuming the same quantity of the same good. *Exclusion* is not possible. It is important to note that a public good is *not* defined by the nature of the supplier of the good (public authorities or private firm) but by the *technical* nature of the good. In other words, a public good can be produced by a private firm.

Typical examples of public goods are lighthouses and radio signals. If a lighthouse is operational and through the light generated provides direction to a ship, it is not possible to exclude any other boat from using the same light for direction. Similarly, listening to a radio station does not reduce in anyway the possibility for any other individual to listen to the same radio station simultaneously. In many instances, environmental quality also shares this characteristic. For example, when the air becomes cleaner in a specific region, it becomes cleaner for everyone in that region.

The difficulty with a public good is to get consumers to pay for the provision of such a good. Indeed, given that if the good is provided and it becomes available for everyone, there will a tendency for a consumer not to reveal his/her true willingness to pay for this good. This phenomenon is usually referred to as a *free-riding* problem. As a result, there is a tendency for such a good to be under-supplied by private producers.

## 2.4 Open access resources

One source of market failure arises from a resource with undefined or unenforced property rights. An open access resource is one that is available to everyone to use with uncontrolled access. Typical examples of open access include ocean fisheries, and some types of forest. In many ways, the notion of open access resources is closely linked to the notion of external costs discussed above. For example, when a fisherman has to decide whether or not to take his boat to sea and fish, he takes into account his private costs and benefits to evaluate the profitability of his decision. However, the fisherman does not take into account the fact that if he goes fishing, there will be less fish available for other fishermen (current and future). There is therefore an external cost associated with the decision to fish. The same holds true for forestry. With open-access resources, the problem is typically that *property* rights to the resource are either not defined, or if defined, not well enforced.

Using various examples below, a study of the impact of externalities, public goods, and open access resources on the decisions of private consumers and producers is made. It will be shown that in the presence of externalities, private agents choose to consume or produce quantities of goods and services that are not *socially* optimal. It is on this basis that one is able to justify some form of intervention by public authorities.

## 2.5 Optimality of Private Decisions

This section provides, and works through a number of examples and exercises with the purpose of studying optimality of consumption and production decisions in the presence of externalities. In each example, we show that private decisions lead to a level of consumption or production that is not socially optimal. These examples will ultimately lead to the notion of a socially optimal level of pollution.

### 2.5.1 Problem: Excessive level of production

For the benefit of the example, consider a pulp and paper plant that releases polluting emissions while producing its product. The firm has to decide how many tons of paper to produce. In a market economy, the firm trades-off the costs and benefits (revenues) associated with the production of each ton of paper. Standard microeconomics theory teaches us that the firm seeking to maximise profits will select a level of production such that marginal revenue equals marginal cost. These concepts are defined here. In a perfectly

competitive industry, (which is going to be assumed here), marginal revenue is simply the market price at which the firm sells each unit of its product. More formally, the marginal revenue is the contribution to total revenue associated with the production and selling of an additional unit of the good produced by the firm. In a discrete case marginal revenue is computed as follows:

$$\text{Marginal revenue} = \frac{\Delta \text{ Total revenue}}{\Delta \text{ Quantity produced}}$$

Marginal cost, as is customary, is assumed to be increasing with the additional level of output produced or:

$$\text{Marginal cost} = \frac{\Delta \text{ Total cost}}{\Delta \text{ Quantity produced}}$$

In Figure 1a, at point P the optimal level of production is undertaken by the enterprise that seeks to maximize its profits. While the production of the paper is a source of costs supported by the firm, it is also the source of costs that it does not support: its activity generates a pollution source of environmental damages. Pulp and paper plants are indeed notorious for producing large quantities of water pollutants known as biological oxygen demand (BOD) and total suspended solids (TSS). There are therefore *external* costs associated with the production of pulp and paper. In order to obtain the social cost of pulp and paper production, these external costs must be added to the marginal private costs. This is illustrated in Figure 1b. The distance between the marginal private cost and the marginal social cost may be here referred as the marginal external cost, that is the increase in external cost associated with the production of an additional ton of paper.

While the producer must trade-off the marginal revenue to its own (private) marginal cost of production to determine the privately optimal quantity of paper to produce, the maximization of social welfare requires that the marginal revenue be equated to the marginal *social* cost of production. From society's point of view, S represents the optimal quantity of paper to be produced. One notes that S is smaller than P thus indicating that the firm produces too much paper to be socially optimal.

## 2.6.2 Problem: Excessive exploitation of natural resources

Suppose there is a forest to which access is not controlled through community practices or clear property rights. Villagers, households, enterprises, all have free access to the forest. The wood is used for various purposes: construction, firewood, heating, etc. For the purpose of this example, the unit of measurement is chosen to be the hectare. In Figure 2a both the marginal private benefit and marginal private cost associated with the production (or cutting) of wood are represented. Note

**Figure 1a**
**Example: Optimal Private Choice in Production of Paper**

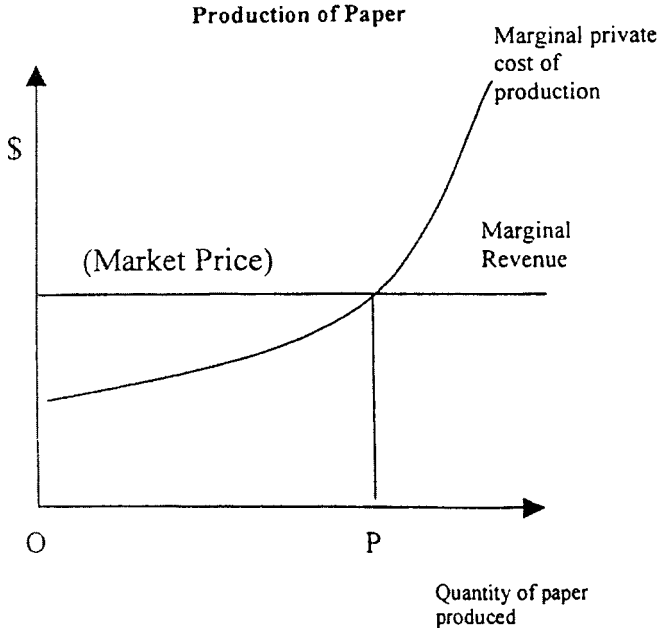
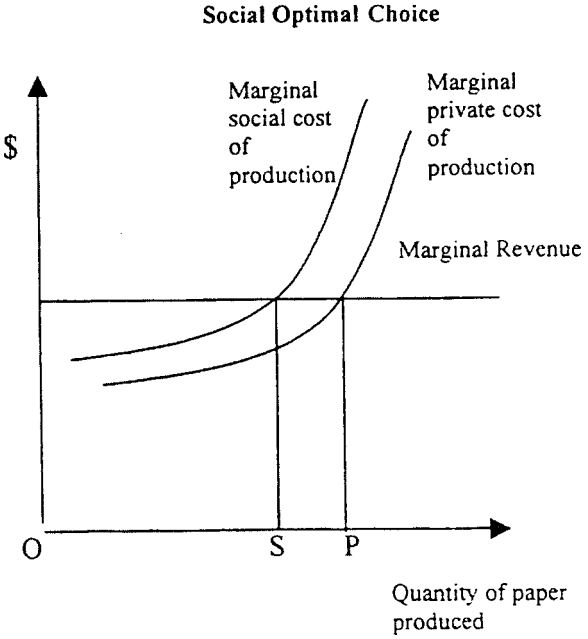



**Figure 1b**
**Example: Optimal Private Choics vs Social Optimal Choice**



that a significant cost associated with wood production is typically the transport cost. For simplicity, we have once again assumed the marginal private benefit to be constant. The quantity A represents the number of hectares of forest that is privately optimal to cut in these circumstances.

In the decision of how many acres of forest to cut, the decision-maker (be it a single individual or a firm) ignores the fact that its decision imposes costs on other users of the forest. For example, for each acre of forest that the individual cuts, it means that other "cutters" must go further away in the forest to obtain the wood thus increasing transport cost. The decision-maker may also ignore that the forest has values other than simply the value of the logs obtained from the forest. For example, the forest may serve as habitat for a large number of species, it may prevent soil erosion, etc. In other words, there are external costs associated with cutting acres of forest: the marginal *social* cost is greater than the marginal private cost. In Figure 2b, S represents the socially optimal amount of acres of forest to cut. As shown, decision-makers that seek to maximize their own private net benefit (defined as benefit minus cost) will cut too many acres of forest. The forest is over-exploited.

**Figure 2a**
**Forest Exploitation: Optimal Private Decision**

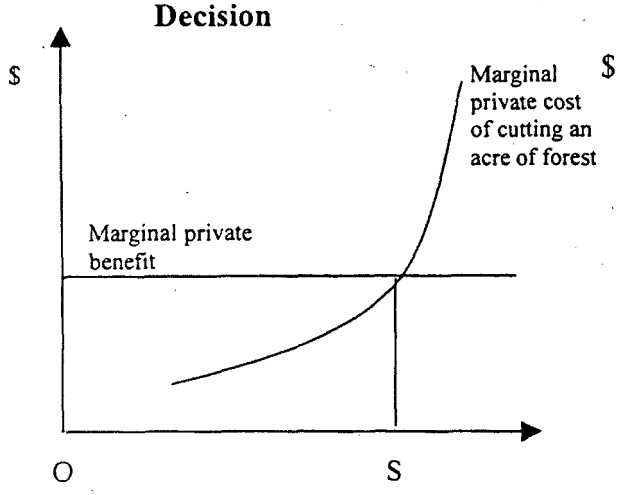



**Figure 2b**
**Forest Exploitation: Optimal Private Choice vs Social Optimal Choice**

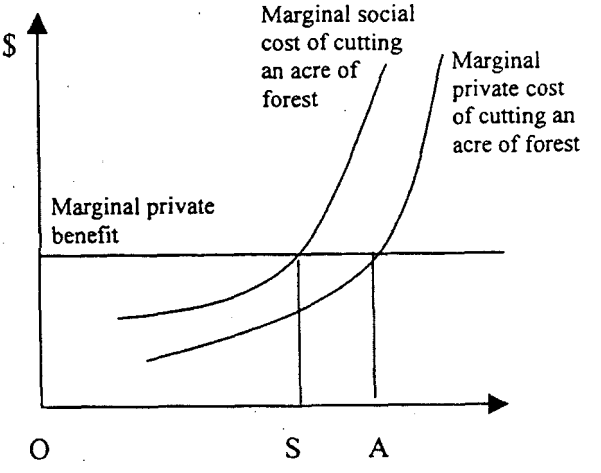



**Figure 3a:**
**Improving environmental quality: Optimal Private Choice**

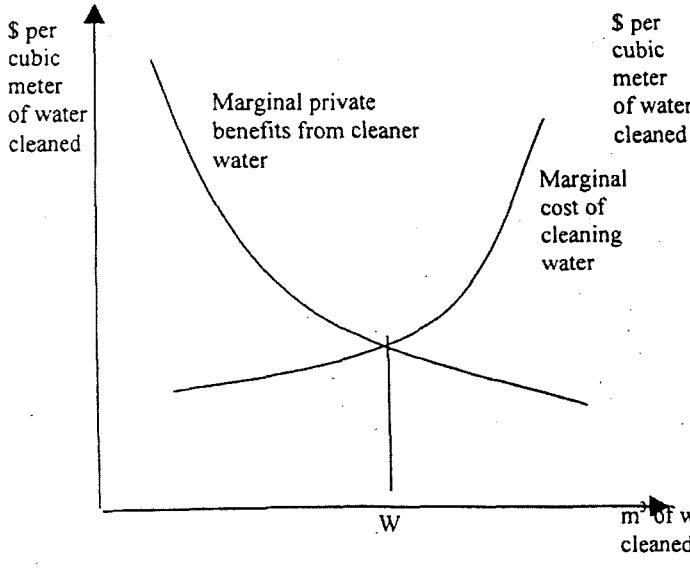
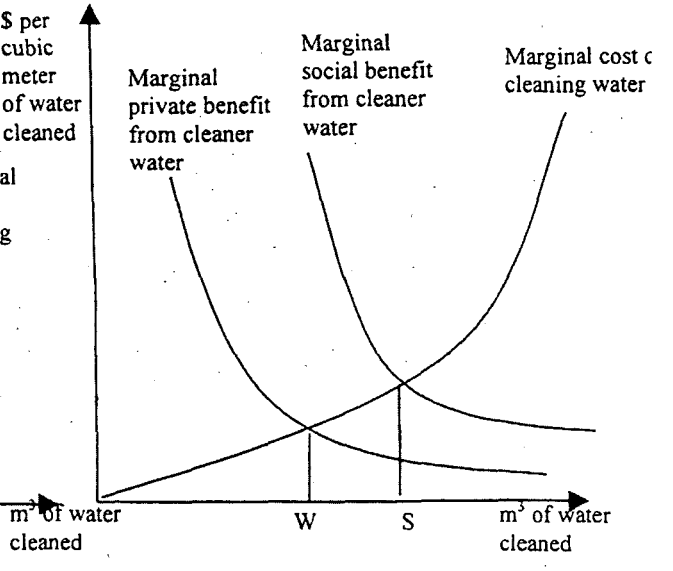



**Figure 3b: Improving environmental quality: Optimal private choice vs social optimal choice**



### 2.6.3 Problem: Insufficient provision of public goods

Consider a number of individuals living around a polluted lake. Because of the degraded quality of the water in the lake, recreational activities (such as swimming, boating, fishing, etc.) will be adversely affected. Each individual 'could invest in cleaning up the lake so as to restore some of these recreational activities. In Figure 3a, the marginal cost of cleaning up the lake is depicted (for simplicity, the unit of measurement of a cubic meter of water has been *i* chosen). As before, it is assumed that the larger the quantity of water cleaned up, the higher the marginal cost of cleaning up. The marginal private benefit associated with cleaner water is also represented. The assumption is that the cleaner the lake becomes, the smaller the marginal private benefit associated with cleaning up an additional cubic meter of water. Any given individual will clean up the lake until the marginal cost of clearing up equals the marginal private benefit. The quantity W of water would thus be cleaned up.

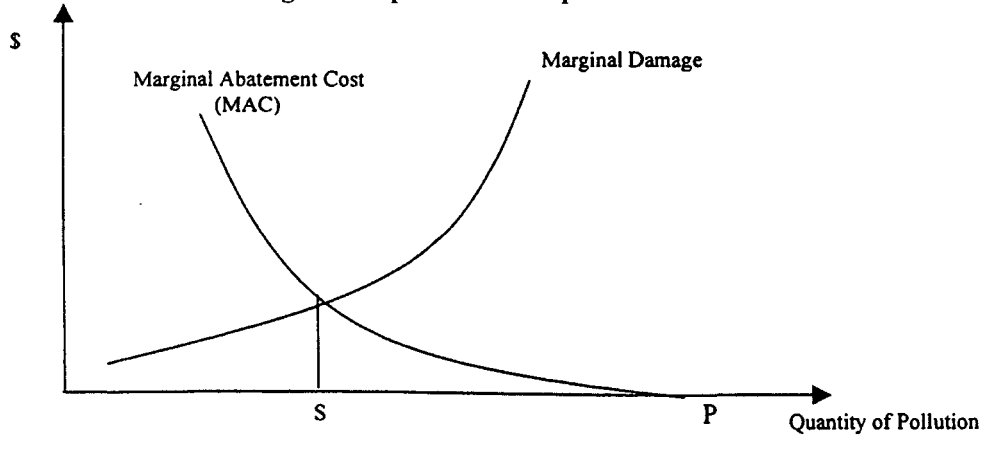
However, if the lake becomes cleaner for one individual, it also becomes cleaner for everyone enjoying recreational activities from the lake: a cleaner lake is a public good. In other words, the marginal *social* benefit associated with cleaning up the lake is larger than the marginal *private* benefit. As shown in Figure 3b, S represents the *socially* optimal amount of water to clean up. Observe that S is larger than W. Private decisions lead to an insufficient amount of a public good being provided.

### 2.6.4 Efficient level of pollution

As for the production of any other goods and services, environmental protection or pollution control involves costs and benefits. Costs are measured in terms of the various inputs that must be devoted to pollution control. Benefits are measured in terms of the reduction in damages (health damages, ecological damages, etc.) that is allowed by reducing the emissions of pollution. Costs and benefits must be traded-off. To do so, the notions or marginal abatement cost and marginal damage need to be defined. The marginal abatement cost (MAC) is the cost of reducing pollution emissions by an additional unit. It is generally assumed that the MAC increases as abatement increases. In other words, the more one abates pollution, the more costly it becomes to reduce pollution by an additional unit. The marginal damage (MD) is the damage caused by an additional unit of pollution. It is generally assumed that the MD increases as pollution increases. It is also generally assumed that the environment has the capacity to absorb a limited amount of pollution without adversely affecting its quality.

Facing no incentives to reduce its level of pollution, a polluter would produce the amount P of pollution, shown in Figure 4, where the marginal abatement cost is zero. However, the efficient, or socially optimal level of pollution is determined at the intersection of the marginal abatement cost and marginal damage functions: S is the optimal level of pollution. Left to themselves, polluters thus produce too much pollution. It is on this basis that interventions by the regulator may be justified. In the next section, the question of whether or not this intervention is indeed justified, and the various forms it may take is examined more precisely.

**Figure 4: Optimal level of pollution**




## 3. ENVIRONMENTAL INSTRUMENTS

The previous section provided a discussion of the basic concepts and ways in which they may be used for the optimal management of environmental pollution and natural resources. This section discusses the different economic instruments in greater detail

Various instruments may be used by a regulator to induce consumers and, producers to undertake a level of activity (be it pollution control, fishing, reforestation, etc.) that coincides with the level that maximizes social welfare. These include the imposition of standards, the use of economic and financial instruments, etc. A large portion of the literature in environmental economics is a devoted to comparing the relative merits of these various instruments. In order to choose among these instruments, a number of criteria may be selected. Considered below some of the criteria that are implicitly or explicitly used to guide the choice of instruments.

## 3.1 Property rights

We have observed that consumers and producers make decisions on the basis of the private costs and benefits they are facing. However, social costs and benefits may differ from the private costs and benefits faced by consumers and producers. Private agents base their decision on private costs as opposed to social costs because they do not have to support in any way the external costs associated with their consumption and production activities. Similarly they base their decision on private benefits as opposed to social benefits because they cannot extract a payment from the recipients of the external benefits.

The next question to ask is the following: why do consumers and producers not have to pay for the external costs their activities generate? Similarly, why can they not extract payment from those enjoying the external benefits? The answer to these questions lies in the absence of property rights. Since many aspects of the environment e.g. a typical fishery, or forest do not belong to any private parties, there is no one to compensate for using these goods. The price of using the environment, a public good, or an open access resource is effectively nil. If these goods were to belong to private parties, a price would have to be paid for using these resources. The simple solution to externalities would therefore appear to be to *privatize property rights.*

A further crucial question however is raised: to whom should private property rights be allocated? To those generating the external costs (such as the polluters), or to those whose welfare is adversely affected by the external costs (such as those whose health is damaged by the pollution)? Is it possible to allocate the property rights to ensure that the social optimum will be reached? As we noted in moduel, Ronald Coase (1960) showed that in the absence of transaction costs, the social optimum can be reached (e.g. the optimal level of pollution, the optimal amount of trees cut, of land protected, of reforestation, of environmental protection, etc.) whether property rights are initially allocated to polluters or to those suffering from the pollution. This result has become known as the *Coase theorem*. It is demonstrated below with the help of a specific example.

Suppose a plant is discharging pollution into a lake, and that the water of the lake is used for consumption and for irrigation by a community living by the lake. Suppose that the benefit for the plant to discharge its pollution in the lake is $1000 ; (for example, this may mean that it would cost $1000 for the plant to stop discharging in the lake by reducing its pollution or by discharging in some other location). Suppose that the benefit for the community of using the lake is $1200 (for example, this may mean that if the community had to stop using the water of the lake, it would have to pay $1200 to obtain water from some other sources). Given these values, the social optimum in this case requires that the lake be used by the community for consumption and irrigation purposes since it is in this use that the lake creates the largest value. In what follows, we will show that if we allow negotiation between the plant owner and the community, the lake will be used for consumption and irrigation by the community irrespective of who owns the property right over the lake.

First suppose that the property right is allocated to the polluter. Since it would cost $1000 for the plant to stop using the lake to discharge its pollution, the plant would accept any compensation above $1000 to stop discharging in the lake. On the other hand, it costs $1200 for the community not to use the water of the lake. The community would be willing to pay up to $1200 to be able to use the water from the lake and still be better off. Since the community is willing to pay more than what the plant would require to stop its discharges, there is room for negotiation. Negotiation will result in the community compensating the polluter to induce him to stop polluting the lake. The lake will be therefore be used for consumption and irrigation by the community. The optimum is therefore reached even if the property right was allocated to the polluter.

Suppose on the other hand that the lake belongs to the community. Since it would cost $1200 for the community to stop using the lake, the community would accept a compensation greater than $1200 to stop using the water of the lake. However, the plant would be willing to pay at most $1000 to obtain the right to discharge in the lake. There is in this case no possibility for negotiation and the lake will be used for consumption and irrigation by the community.

As we have seen, the optimum is reached whoever is allocated the property right. For this result to be obtained however, important conditions must be satisfied. First, it must be possible to define precisely the property rights. Second, this property right must be enforceable, and transferable. Indeed, the property right is of no meaning if abuse of the property right by a third party cannot be prevented. Third, parties to the transaction must be well defined. This may be particularly ! difficult when today's actions affect future generations, by definition, these cannot be part of current negotiation. Fourth, those owning the property rights must be able to capture *all* values associated with the environmental asset they own. In the forestry sector for example, this is generally a problem since the property right is typically defined solely over the wood value of the forest, and not over the entire value of the forest, which goes beyond simply its wood

value. Finally, transaction costs must be small. When the number of polluters and pollutees is large, this condition will likely not be satisfied.

For all the above reasons, the conditions under which the allocation of private property rights may restore social efficiency restrict the applicability of property rights in practice. Hence it is necessary to look at other ways of achieving the social optimal solution.

## 3.2 Command and control regulation

The traditional approach to environmental protection is often referred to as command and control. By means of regulation, the regulator commands a desired behavior, typically by imposing a limit on the amount of emissions that a polluter can produce. These limits are generally called emissions standards. The regulator then controls and enforces compliance with the desired behavior. Under this regime, the incentives for pollution control take the form of penalties or sanctions that the polluter is faced with if it does not comply with the command.

In this section the setting of environmental standards needed to implement the type of instruments are discussed. A classification of regulatory instruments is also suggested for different types of environmental issues. The limitations of command and control mechanisms in comparison with economic instruments are also considered.

Standards are the predominant means for direct regulation of environmental quality throughout most of the developed world. They define environmental targets and establish the permissible amount or concentration of particular substances or discharges into air, water, land, or consumer products. Types of standards include: ambient environmental quality standards, effluent or emission standards, technology-based standards, performance standards, product standards, and process standards (See Box 1). Standards also may include technological specifications for the performance or design of equipment or facilities and the standardization of sampling or analytical methods. In some cases, a regulator takes into account the transfer of pollutants from one medium to another as well as total environmental exposure to specific pollutants in determining ambient standards or discharge limits. Each of the various types of standards is used to provide a reference for evaluation or target for legislative action and control. Generally, standards are established by central governments, in some instances, however, central governments set out framework regulations to be carried out by local, state, or regional authorities. Sub-national standards can be more stringent than those of the central government.

The granting or withholding of permits, licenses, or other authorizations is another important tool for controlling pollution. The permits or licenses are generally tied to an air or water quality standard and may be subject to the fulfillment of specific conditions such as compliance with a code of practice, selection of the location that minimizes environmental and economic impacts, installation of a treatment plant or pollution control equipment within a certain time period, or adoption of other environmentally protective measures. One major advantage of permits and licenses is that they facilitate the enforcement of environmental programs by including in one document all of a facility's pollution control obligations. Other advantages are that they may be withdrawn or suspended according to the needs of the national economy or cover social interests and often require a fee that can be used to cover the costs of the pollution control program. Nonetheless, the use of permits and licenses normally involves regular monitoring and facility reporting.

The command-and-control approach potentially gives the regulator maximum authority to control where and how resources will be spent to achieve environmental objectives. Under ideal conditions, the regulator would be able to identify precisely the marginal abatement cost and marginal damage functions characterizing each and every polluter, and would impose a ceiling on emissions of each polluter corresponding to the optimal level of pollution. In Figure 3.5, firm 1 would be facing a standard equals to $S_1$ and firm 2 equal to $S_2$. The major advantage of this approach is that it provides the regulator a reasonable degree of predictability about how much pollution levels will be reduced.

In practice however, the regulator does not know the firm-specific marginal abatement cost and marginal damage functions. As a result, emissions standards in practice share the following principal characteristics: (1) Standards are defined on the basis of the best available or economically achievable technology; (2) Standards are often defined and tailored to specific industries (for the same pollutant, standards are typically different across industries); (3) Standards are often different for old sources of pollution and new sources of pollution, generally being more stringent for new sources, (4) Standards are usually uniform.

Although command and control strategies have made substantial progress in reducing pollution, this approach has been criticized for not achieving various legislative mandates and deadlines and for being economically inefficient and difficult to enforce. These strategies are inefficient for the regulatory agency, which must have detailed information concerning production processes and the suitability of various pollution control

### Box 1 - Definitions of Standards

**Criterion** - Scientific information (for example, concentration-effect or dose-effect data) used as a basis for setting environmental quality objectives. It assesses the risk to a given victim and the amount of damage caused by a known amount of pollution or dose of exposure.

**Objective** - A designated concentration of a pollutant in an environmental medium or a narrative statement (for example, adequate to support aquatic life, zero pollutant discharge). The objective is based on scientific criteria, local natural conditions, and socioeconomic and environmental factors.

**Standard** - A legally defined regulatory instrument for limiting pollution. Several types of standards are commonly employed:

a. **Ambient Environmental Quality Standard** - Establishes the highest allowable concentration of specified pollutants in the ambient air or water. For example, an ambient standard for a specific river may require that dissolved oxygen, averaged over a 24-hour period at a selected river mile point, must not fall below 4 parts per million on more than one day per year.

b. **Effluent or Emission Standard** - Establishes the legal ceiling on the total quantity or concentration of a pollutant discharged from a pollution source (for example, mg/liter, grams/24 hours, kg/ton)). Effluent standards may include maximum effluent limitations for specified time periods (for example, maximum for any one day, maximum averages of daily values for 30 consecutive days, or for one year) and monitoring requirements.

c. **Technology-Based Standard** - A type of effluent standard that specifies a specific technology a firm must use to comply with environmental laws and regulations. For example, a utility may be required to use a scrubber to control sulfur oxide emissions.

d. **Performance Standard** - A type of effluent standard that defines a performance measure (for example, volume or concentration of a pollutant in a discharge, percent pollutant removal to be achieved) and allows dischargers the flexibility to select the best means to meet this standard. For example, automobile companies may be required to develop a technology for new automobiles that limit tailpipe emissions to no more than .41 grams of hydrocarbons, 12.4 grams of carbon monoxide, and 1.0 grams of nitrogen oxides per mile.

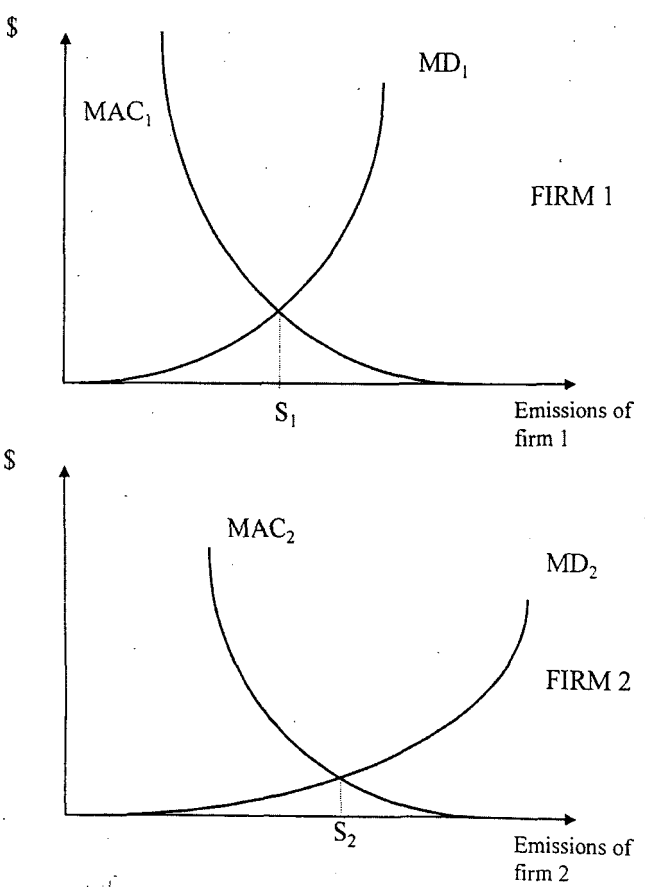
e. **Product Standard** - Establishes a legal ceiling on the total quantity or concentration of pollutants that can be discharged into the environment per unit of product output (kg per 1000 kg of product). Product standards also prohibit the addition of certain substances to products, for example, to eliminate lead discharges from the burning of gasoline, authorities prohibit the addition of lead to gasoline.

f. **Process Standard** - Limits the emission of pollutants associated with specific manufacturing processes (for example, the mandatory replacement of mercury cells by diaphragm cells to prevent mercury emissions from chlor-alkali manufacture).

devices. With diverse industries, it is extremely expensive and time-consuming to obtain the necessary information and expertise on each industry.

Other problems with this approach are the high costs for pollution control that leave little opportunity to take advantage of various forms of flexibility in meeting given ambient quality standards. Although standards may be applied differently depending on the age or type of facility, most polluters using the same production process are required to meet the same standards. Polluters that could reduce pollution at a lower cost are not given the opportunity. Further, there is little flexibility for polluters who already have invested in some type of pollution control system. Consequently, the command and control approach provides little incentive for innovation in pollution control technology once the standards are achieved. Moreover, this approach is insuffficient or ineffective in addressing many of the more recent pollution control and waste management problems confronting environmental managers, such as non-point source pollution (that is, urban and agricultural runoff), solid waste disposal, and global environmental problems (for example, stratospheric ozone depletion and climate changes). The advantages and disadvantages of the above regulatory instruments for different media are summarized in Box 2.

**Figure 5: Optimal standards**



**Box 2 Regulatory Instruments**

| *Instruments* | *Existing Applications* | | | | | *Advantages* | *Disadvantages* |
|---|---|---|---|---|---|---|---|
| Regulatory | Surface Water Pollution Control | Ground Water Protection | air Pollution Control | Solid Waste Management | Hazardous Waste Management | | |
| Ambient Environmental Quality Standards | X | X | X | | | Provide basis for evaluating effectiveness of existing controls | Require highly Technical pollutants;' effects |
| Effluent and emission standards | X | | X | | | Provide maximum government control | Involve high monitoring and enforcement costs |
| Technology based effluent/emission standard | X | X | X | | | provide maximum government control | Allow no flexibility in control technology |
| Performance based effluent/emission standards | X | X | X | | | Promote cost savings Allow flexibility in control Technology | Involve high monitoring and cenforcement costs |
| Products/process standards | X | X | X | | | Eliminate or limit emission of pollutants prior to production | Require close subsitutes for banned products |
| Permits and licenses | | X | X | X | X | Require compliance with standards prior to facility operation Facilitate enforcement of effluent and emission standards Allow government to withdraw or suspend according to national needs | Involve high monitoring and enforcement costs |
| Land and water use control | X | X | X | X | X | Prevent iappropriate siting of polluting activities | Allow govenment to withdraw or suspend to national needs Vulnerable to local political and economic pressure |

## 3.3 Economic, Fiscal and Financial Instruments

There are a large number of regulatory measures, which qualify as economic instruments. Box 3 provides a listing of the main ones. Broadly speaking economic instruments are ones that act **directly** on the pollution, whereas fiscal and financial instruments provide and economic incentive through their effect on the prices paid for the inputs or received for the outputs to a production process. A discussion of the main types of instruments follows.

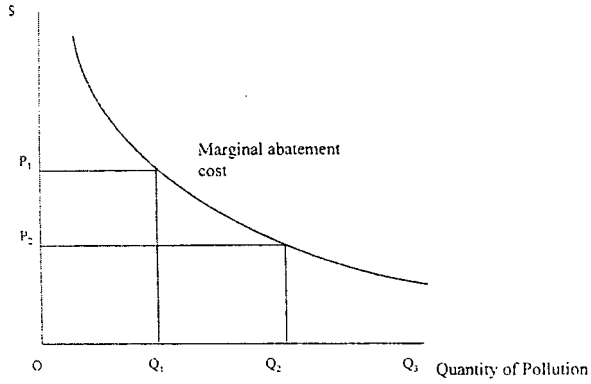
### 3.3.1 Pollution Charges

The principle supporting the use of pollution charges, that is a charge (or price, or fee) per unit of pollution, is relatively simple. It was pointed out above that consumers and producers base their decisions on private costs and benefits as opposed to social costs and benefits because they do not have to support the external costs associated with their activities (or in the case of external benefits, they cannot extract payments from those enjoying these external benefits). In other words, consumers and producers do not have to pay for the external costs they generate. The principle behind pollution charges and tradable emissions permits is simply to create such a price.

Consider first a pollution charge. In Figure 6, upon facing a price $P_1$ for each unit of pollution that it discharges, the firm has a choice between reducing its pollution or paying the price $P_4$. A profit maximizing enterprise will reduce its pollution up to the point where the price to be paid per unit of pollution is just equated to its marginal abatement cost. Facing a price $P_1$, the firm will reduce its emissions from $Q_3$ to $Q_1$. Facing a lower price ($P_2$), the firm will invest less in pollution abatement, and reduce its pollution only to $Q_2$

The level of the charge which equates the marginal abatement cost and the marginal damage functions is

**Figure 6. Impact of pollution charge on firm's pollution**




referred to as the Pigovian charge (after Alfred Pigou the English economist who first worked on this issue). Under ideal conditions, the regulator would be able to identify precisely the marginal abatement cost and marginal damage functions characterizing each and every polluter, and would impose a Pigovian charge corresponding to the optimal level of pollution for each and every polluter.

**Box 3 Economic, Fiscal and Financial Instruments**

**Economic instruments**

| | |
|---|---|
| *CHARGE SYSTEMS* | Pollution charges |
| | User charges |
| | Access fees |
| | Road tolls |
| | Administrative charges |
| PROPERTY | Ownership rights |
| | - land titles |
| | - water rights |
| | - mining rights |
| | Use rights |
| | - licensing |
| | - concession |
| MARKET CREATION | Tradable emission permits |
| | Tradable catch quotas |
| | Tradable land permits |
| LIABILITY SYSTEM | Legal liability |
| | Natural resource damage liability |
| | Liability insurance |
| BONDS AND DEPOSIT SYSTEM | Environmental performance bond |
| | Land reclamation bond |
| | Deposit refund system |

**Fiscal instruments**

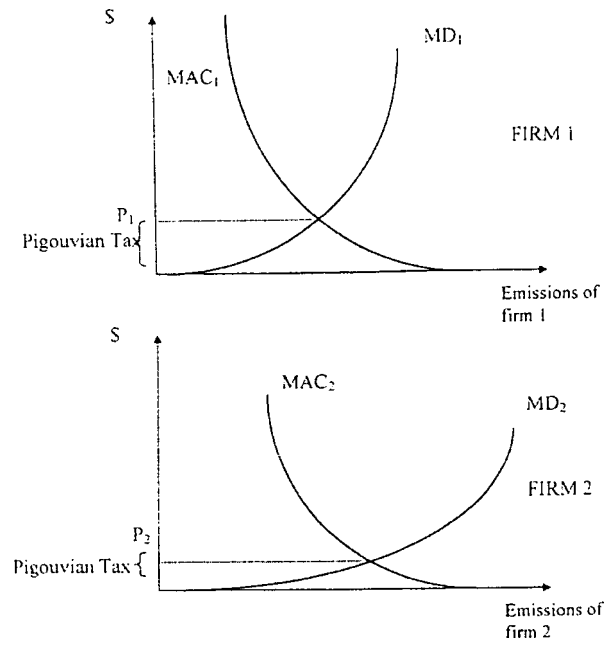
Input taxes, Product taxes, Export taxes, Import tariffs, Tax differentiation, Royalties and resources taxes, Land use taxes, Investment tax credits, Accelerated depreciation, Subsidies

**Financial instruments**

Financial subsidies, Soft loans, Grants, Location/relocation incentives, Subsidized interest, Revolving funds, sectoral funds

In Figure 7, firm 1 would be facing a price P for every unit of pollution it produces, and firm 2 would face a price $P_2$.

**Figure 7 Differentiated Pigovian charge**




In practice however, firm specific marginal abatement cost and marginal damage functions are not known to the regulator. As a result, pollution charges tend to be uniform: a given pollutant will be charged the same price within given geographical areas across industrial sectors.

The regulator does not have sufficient information about marginal abatement cost and marginal damage functions at the firm level to set emissions standards or pollution charges at their optimal level. As a result pollution charges tend to be uniform: a given pollutant will be charged the same rate within given geographical areas and/or across industrial sectors.

## 3.3.2 Tradable Permits

Under this approach, markets can be created in which actors can buy "rights" for producing pollution or where they can sell these "rights" to other actors.

Under a marketable or tradable permit system, the responsible authority determines a target level of environmental quality defined as an allowable level of emissions or an arnbient environmental quality standard. This level of environmental quality is then translated into a total number of allowable emission that can be discharged and then allots discharge rights to firms in the form of permits. Permits are then distributed to firms with each permit allowing the owner to discharge a specified amount of pollution. This permit to discharge may be transferred from one source to another. The demand for the permits is derived from the discharger's marginal costs of treatment: The discharger will treat waste as long as the marginal cost of treatment is less

**Box 4**
**Cost effectiveness**

Suppose there are three plants characterized by the following MAC.

| | $MAC_1$ | $MAC_2$ | $MAC_3$ |
|---|---|---|---|
| 6 | 0 | 0 | 0 |
| 5 | 15 | 40 | 20 |
| 4 | 30 | 75 | 40 |
| 3 | 50 | 110 | 75 |
| 2 | 75 | 150 | 120 |
| 1 | 100 | 200 | 170 |
| 0 | 150 | 260 | 230 |

Total units of pollution: 18

Objective: Reduce aggregate emissions to 9. Denote total abatement cost as TAC.

Solution 1: Require that each plant reduces pollution to 3

Then, TAC for each plant is the following: $TAC_1 = 95$, $TAC_2 = 225$, $TAC_3 = 135$ $\longrightarrow$ TAC = 455

Solution 2: Minimize total abatement cost *(needs clarification)*

| | | | |
|---|---|---|---|
| Unit 1: | 1: 15 | Unit 6: | 1: 50 |
| Unit 2: | 3: 20 | Unit 7: | 1: 75 |
| Unit 3: | 1: 30 | Unit 8: | 2: 75 |
| Unit 4: | 2: 40 | Unit 9 | 3: 75 |
| Unit 5: | 3: 40 | | |

Then, TAC for each plant is the following: $TAC_1 = 170$, $TAC_2 = 115$, TAC3= 135 $\longrightarrow$ TAC = 420

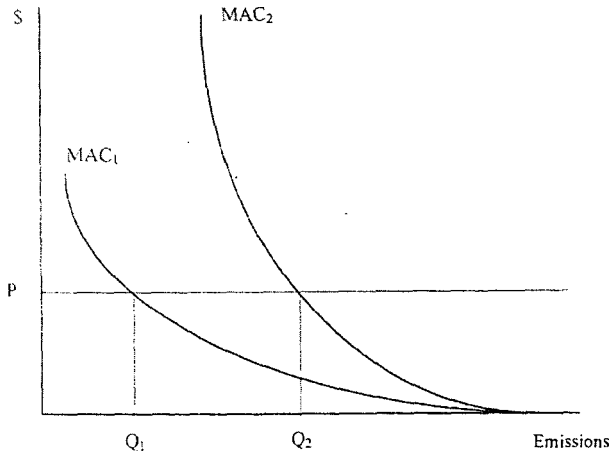
than or equal to the cost of buying a permit. Hence, given a similar aggregate emissions target, the level of the (uniform) pollution charge necessary to achieve the target would be the same as the equilibrium price of a permit, assuming that the market for permits is perfectly competitive. An example is provided in Box 4. The cost of uniform reduction is 455 while the least cost way of getting the same reduction results is a cost of 420.

Assume that the regulator wishes to reach a given level of aggregate emissions by means of pollution charges. Lacking information on the marginal abatement cost, it is not immediately clear what should be the level of the charge that will allow the objective to be reached. In Figure 8, for a given charge P, emissions are going to be $Q_1$, if the marginal abatement cost is $MAC_1$, and $Q_2$ if it is $MAC_2$. None of these may be the desired objective. It has been suggested that the regulator may use an iterative approach by which it keeps adjusting the level of the charge until the target is reached. A difficulty with this approach is that knowing the nature of the process into which the regulator is engaged, firms are going to adjust their emissions in a strategic manner.

Of course, this difficulty does not exist with tradable permits since by definition the number of tradable permits determines exactly the level of aggregate emissions.

With permits, the problem is different. The

**Figure 8**




regulator knows the reduction in emissions with certainty but does not know the costs of the reductions with certainty. Hence a target reduction could be easy to achieve, or may impost a heavy burden on industry. Another issue with permits is that of trading between pollutants whose emissions have very different environmental impacts. In a situation where environmental damages are not a function of the location of the polluter, this does not create problems. However, in many instances damages are a function of the location of the polluter. In such a situation, tradable permits may give rise to what is known as "hot spots".

There are two basic approaches to implementing a marketable discharge permit system: government auction of permits or free distribution of permits to dischargers followed by trading, among, dischargers to establish a market price. Under the first approach, permits are sold for a single market-clearing price, which might be the lowest accepted bid, the highest rejected bid, or some value in between. Alternatively, the permits are allocated to the highest bidders. Under the second approach, permits might be distributed initially to duplicate the effect of uniform removal regulations or to allocate permits to communities on a basis such as population or to firms based on value added. Such an allocation is called "grandfathering". After the initial distribution, the method for exchange of the permits could be a centralized market or bilateral exchanges.

### 3.3.3 Other charge systems

***User charges*** are direct payments for the costs of collective or public treatment of pollution. They are used most often in the collection and treatment of municipal solid waste and for the discharge of wastewater into sewers. With respect to water pollution control, for example, user charges are fees paid to water authorities to allow discharges of industrial wastes into public sewers. Through these charges, which are related to the quality and characteristics of the effluent, the public authority receiving the waste discharge is compensated for the effort involved in its disposal. At the same time, the scale of the charges ensures that the plant has an economic incentive to improve the quality of the effluent. This approach, however, is not appropriate when disposing of certain toxic pollutants (for example, mercury) that never should be permitted to enter watercourses. Another application of user charges is in the area of vehicle-related taxes and charges.

***Product charges*** are fees added to the price of products or product inputs that cause pollution in either the manufacturing or consumption phase or for which a special disposal system has been established. They function like effluent and emission charges in that they allow users to determine their own cost-effective means for reducing pollution. For example, all members of the European Community (EC), except Denmark, have levied a product charge on lubricating oils as a result of a 1975 EC directive to adopt measures to recycle waste oil. The system involves a tax on lubricating oil and subsidies provided for recycling oil. In Germany, this program was highly successful until the country encountered problems with illegal dumping. In France and Italy, where the systems have less of an incentive effect, oil collections have at least doubled since the beginning of the programs. In Norway and Sweden, other product charges connected to environmental goals include charges on nonreturnable containers, batteries, lubricating oil, fertilizer, and pesticides.

The effectiveness of a charge on polluting products or product inputs will depend on the availability of substitutes. For example, where input costs are a small fraction of total costs, doubling or tripling the price through an input tax is unlikely to have a significant effect on consumption unless there are suitably priced substitutes. If less polluting substitutes are available, small increases in input prices may induce substitution and innovation over the longer term. Revenues from product charges can be used to treat pollution from the product directly, to provide for recycling of the used product, or for other budgetary purposes. Product charges also fail to provide an incentive for reducing emissions at the "end-of-the-pipe".

***Administrative charges*** are fees paid to authorities for such services as chemical registration or the implementation and enforcement of environmental regulations. They usually are a component of direct regulation and are intended primarily to finance the licensing and control activities of pollution authorities. In Norway, for example, these charges are levied to

finance the registration and control activity for fish farming and agricultural pollution, control of emissions from industrial sources, and for the licensing, of chemical products. In some cases, administrative charges can have an incentive purpose as well. For example, a registration charge based on the relative hazards of pesticides or other chemicals may be imposed to encourage the use of less harmful products. In practice, administrative charges are similar to product charges in that the levels of such fees are usually low and do not provide significant incentives for changes in purchasing patterns. Revenues from administrative charges usually are added to the general budget rather than the budget of the pollution authority involved.

***Tax differentiation*** is used to promote consumption of products that are environmentally safe. This instrument involves a combination of two surcharges added to other product chases: a positive charge levied on a polluting product and a negative charge on a cleaner alternative. It is used primarily in the context of transport to discourage consumer purchases of polluting vehicles or fuels. Differential taxation of leaded and unleaded gasoline is a common practice throughout Europe. In the Netherlands, for example, unleaded gasoline is taxed at 0.1 ECU per 100 liters (about US$.004 per gallon) and leaded gasoline at 1.74ECU per 100 liters (about US$.08 per gallon 1989). Other taxes are levied on diesel and other fuel oils. The fuel taxes are designed to have some incentive effects. Generally, the levels of the charge are calculated so as to keep the total financial effects budget neutral. Differential taxation of new automobiles based on pollution is practiced in the Netherlands, Norway, Sweden, Finland, and Germany. The practice is now being extended to developing countries.

### 3.3.4 Choosing among instruments

Should we prefer uniform emissions standards, uniform prices, or tradable emissions permits to achieve the desired target? We have listed above a number of criteria that can be used to choose between instruments. We review these three instruments in view of these criteria.

- ***Cost-effectiveness.*** It is easy to demonstrate that uniform pollution charges and tradable permits allow the desired objective to be reached at a lower cost than uniform emissions standards (see Box 3.2).
- ***Dynamic incentives*** certainly exist under economic instruments. Indeed, since the firm has to pay a price for every unit of pollution it produces (or alternatively must buy a permit for every unit of pollution it produces), the firm always has the incentive to reduce its emissions so as to reduce its total tax payments (or sell permits it does not need anymore). Such incentives are much reduced under command and control since under this policy, the firm can pollute up to the standard free of charge. Once it complies with the standard, it has little incentive to do any better.
- ***Implementation issues.*** None of the instruments possess a distinctive advantage in terms of implementability. In all cases, firms must report their level of emissions (self-reporting), the regulator must verify the accuracy of these reports, it must monitor the firms, measure the firm's emissions, and enforce either the standards, mis-reporting by the firm, or compliance with the number of permits held by the firm. These are not easy tasks.
- ***Flexibility.*** While there is a tendency to group together economic instruments, tradable permits may benefit from an important advantage over pollution charges in terms of flexibility. Indeed, with pollution charges, the regulator must intervene to change the level of the charge in order to maintain aggregate emissions at their desired level when economic conditions are changing. For example, in times of increased economic activity, the pollution charge may have to be increased so as to keep aggregate emissions constant. With tradable permits, by definition aggregate emissions will remain constant, however, the price of a permit will reflect the intensity of the demand and adjust itself through market forces. There is in this case no need for the regulator to intervene. On the other hand, for tradable permits to work efficiently, the market for permits must be relatively competitive (with a large number of firms trading the permits). This condition may not always be satisfied.

While pollution charges and tradable permits appear considerably superior, the command-and-control has overwhelmingly been the preferred method of intervention. A large number of reasons may explain this phenomenon. Economic instruments have often been perceived as a "license to pollute" as opposed to a genuine instrument by which social optimality may be restored. The impact of economic instruments on the competitiveness of enterprises has also been a much-debated topic.

### 3.3.5 Disclosure Instruments

The above approaches have relied heavily on economic incentives for environmental regulation. There

is, however, a growing interest in other approaches. An increasing number of authors argue that the traditional view of regulation is misguided because its focus is too narrow. Indeed, the conventional policy discussion, as illustrated by our discussion so far, has focused exclusively on interactions between the regulator and the firm. However, recent research has suggested powerful roles for two additional 'players': the community and the market.

***Community.*** Recent evidence from Asia, Latin America and North America suggests that neighboring communities can have a powerful influence on factories' environmental performance. Communities that are richer, better educated, and more organized find many ways of enforcing Environmental norms. Where formal regulators are present, communities use the political process to influence the tightness of enforcement. Where formal regulators are absent or ineffective, 'informal regulation' is implemented through community groups or NGOs. The agents of informal regulation vary from country to country (they may be local religious institutions, social organizations, community leaders, citizens' movements or politicians) but the pattern is similar: Factories negotiate directly with local communities, responding to social norms and explicit or implicit threats of social, political or physical sanctions if they fail to reduce the damages caused by their emissions. In countries as diverse as China, Brazil, Indonesia and the US, much of the variation in factories' environmental performance is explained by intercommunity variation in income, education and bargaining power.

***Market.*** Factories operate in local, national and international markets, where many agents can affect revenues and costs. Environmental considerations now affect the decisions of many of these agents. In both industrial and developing countries, environmentalism in the middle and upper classes is a significant factor in consumer decisions. With the worldwide advent of environmental legislation, investors are also scrutinizing environmental performance. Among other factors, they have to weigh the potential for financial losses from regulatory penalties and liability settlements. In recent years, the importance of investor interest has been increased by the growth of new stock markets and the internationalization of investment. For similar reasons, international and local suppliers of financing, industrial equipment, and engineering services are increasingly reluctant to do business with flagrant polluters.

Recent evidence from both the OECD and developing countries suggests that environmental reputation matters for firms whose expected costs or revenues are affected by judgments of environmental performance by customers, suppliers, and stockholders. Many factors can affect firms' evaluation of their environmental reputation, including company size, export orientation' and multinational ownership. For reputation-sensitive companies, public certification of good or bad performance may translate to large expected gains or losses over time.

Once the Community and the Market are introduced, we have a much richer and more robust model for explaining the observable variations in factories' environmental performance. Clean factories are perfectly plausible in poor countries, and the survival of dirty factories in rich countries is not hard to understand. In place of the paired State/Factory model of regulation, we therefore propose the 'Regulatory Triangle' which is depicted in Box 5. Once we introduce a world of multiple agents and multiple incentives, we must also rethink the regulator's appropriate role in pollution management. No longer is this role confined to producing, monitoring and enforcing rules and standards. Instead, the regulator can gain leverage through non-traditional programs, which harness the power of communities and markets. Within the 'triangular' regulatory framework, for example, there is ample room for information oriented approaches such as voluntary participation/compliance programs and public disclosure of factories' environmental performance. In fact, we see an increasing number of environmental authorities introducing environmental information programs by which the environmental performance of polluters is rated and revealed to the public. A broader implication is that one size no longer 'fits all' for regulatory policy design: Optimal combinations of regulatory tools will depend on country-specific social, economic and institutional conditions.

When these two sets of factors are taken into account, a different model of regulation emerges. Appropriate regulation for developing countries may incorporate these five key features:

- ***Information Intensity:*** Effective pollution management by the State is impossible unless regulators have reliable data, integrated information systems and the capacity to set priorities which reflect comparative benefits and costs. Markets and Communities need timely, accurate, public information to make appropriate assessments of factories' environmental performance. An effective regulatory agency will therefore allocate fewer resources at the margin to conventional enforcement and more to the generation and distribution of appropriate information products.

  ***Orchestration, not Dictation:*** A pollution control agency is only one player in the

environmental performance game. Agency activities which influence polluters indirectly, through other agents, may be as important as direct enforcement. Potentially high-leverage programs include community environmental education; public disclosure of factory performance ratings; voluntary, public agreements for pollution reduction by industry groups in environmentally-degraded regions; and technical training prograrns for environmental personnel in polluting factories.

- ***Community Control:*** This should be accepted as a current reality, not as the goal of future programs. And in fact, a substantial role for local communities is appropriate from the perspective of environmental economics. Regardless of the state of formal regulation, local 'informal regulation' is stronger in areas with higher pollution loads, larger affected populations and higher incomes. We also find local variations in education and bargaining strength. Taken together, recent findings have three implications:

  1. Strengthening central regulatory agencies should not empower them to impose uniform standards on heterogeneous communities under the guise of 'administrative efficiency.' Much local variation in regulation is legitimate, and should be recognized as such.
  2. Regulatory agencies can play a key role in facilitating negotiations between local communities and neighboring factories. This role includes provision of reliable information on emissions and local ambient quality; technical advice on abatement alternatives, and the transfer of experience from other locations.
  3. Central regulators can use their authority to 'level the playing field' for communities which are excessively polluted because their lack of education, organization and bargaining power prevents them from negotiating effectively with local factories.

- ***Structured Learning:*** Environmental policy reform is a complex business which will inevitably be subject to many uncertainties. Because it is difficult to know exactly what will work in advance, reforms should emphasize structured learning. Rather than pre-committing to broad-based programs, agencies should initiate a variety of pilot projects and build larger programs as the lessons are absorbed.

- ***Adaptive Instruments:*** Newly-industrializing economies can experience rapid changes in ambient quality across air and watersheds. Since regulation should primarily serve environmental quality objectives, it should be focused on adaptation to these rapid changes. Regulators should be empowered to counter environmental degradation by tightening existing regulations. On the other hand, the system should minimize disruption for investors. Meeting both objectives implies:

  1. Transparent adjustment rules, linked to publicly-available data on ambient quality and emissions;
  2. Adjustment which is, as far as politically possible, automatically triggered by deterioration of ambient quality below mandated levels;
  3. Adjustment lags sufficient for smooth adaptation by local economic agents.

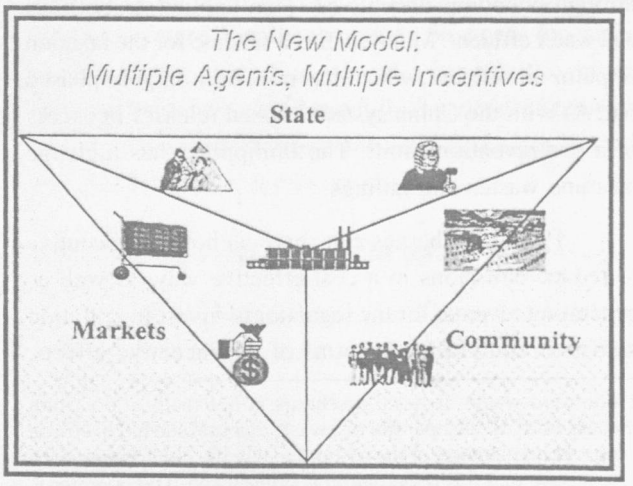
## 4. INSTRUMENTS IN PRACTICE

So far, this module has discussed different economic instruments for environmental regulation, somewhat in the abstract. This section looks at the actual experience with these instruments.

### 4.1 Pollution Charges and Taxes

In practice such charges have been introduced mainly on air emissions, and solid waste with some taxes on water. There are charges for $CO_2$[1] in the Scandinavian countries (Sweden, Denmark, Finland and Norway) and in the Netherlands. Sweden has charges for NOx (for large power stations only), HC and sulphur, Norway has

**Box 5**




[1] Some commentators refer to the carbon tax as a product tax because it is levied at the fuel stage. Given the lack of realistic technologies for collecting carbon dioxide after combustion, this is equivalent to an emissions tax. Where the tax is partly an energy tax, however, the situation is different.

charges on sulphur and lead, and France has very low charges on HCL, $NO_x$, $SO_2$ and $H_2S$ (about one percent of those in Sweden). Switzerland is considering the introduction of a charge on VOCs and $CO_2$ (Jeanrenaud and Stritt (1994)). Rates even within this small group of countries vary widely (OECD, 1995). Water effluent taxes, where the charge is dependent on the composition of the waste water are implemented in France, Germany and the Netherlands. General waste disposal charges exist in all OECD countries but volume based ones are to be found in Australia, Belgium, Canada, Denmark, France, Germany, Italy, Netherlands, Switzerland, and the United States (OECD, 1993; OECD, 1995). The rates vary a lot and also depend on whether the waste goes to landfill or whether it is incinerated (Denmark, Germany, Switzerland). Hazardous waste disposal is charged for on a volume basis in Austria, Finland, Germany and the US[2].

Several countries in Eastern Europe have pollution emissions charges, including the Baltic States, Czech Republic, Slovak Republic, Poland and Russia. The levels range from the extremely low (Baltic States) to higher, but of the same order of magnitude as that of France (Poland, Czech and Slovak Republics). The countries of the former Soviet Union are in the process of reforming their taxes, but the present structure is extremely complex and the number of pollutants taxed is very large. (Markandya and Lehoczki, 1994; Klarer, 1994).

Among developing countries the present use of pollution charges is more limited. China has introduced effluent charges for discharges from industrial plants in a nationwide scheme. Under this scheme, enterprises are required to pay a fee for discharges above a certain norm, with the fee varying according to the pollutant. The system's original purpose was to induce individual firms to reduce the amount of pollutants discharged at source through economic incentives. This applied to both air and water effluent. Malaysia had a scheme for the taxation of palm oil effluent, which has now been largely phased out. As with the China system it taxed releases in excess of a contravention limit[3]. The Philippines has a charge on mine wastes and tailings.

Pollution charges can serve as both an incentive to reduce emissions in a cost effective way, as well as generating revenue for the regulator to invest in pollution reduction equipment. In terms of the incentive effects, these instruments are best used when the environmental goals can be defined in terms of relatively simple objectives with little spatial variation, such as a reduction in ambient concentrations of a pollutant nationally. If the goal is, for example, to limit ambient concentrations over a small area, or to have very different levels of concentrations in different areas, then the pollution charge is not the best instrument, or at best has to be supplemented by direct controls. Pollution charges are also inappropriate where there are many small sources of the emissions, or where the sources are not stationary. Hence they are not appropriate for the regulation of vehicles, or run-off from the application of fertilizers and pesticides. Pollution charges can always be used as a revenue raising device, but in this respect they are less effective than charges on inputs or outputs that generate the emissions in the first place (see below). The latter are easier to tax (there are fewer points of collection) and mechanisms for their taxation are already in place. In practice it has to be said that pollution charges have been used more as a means of generating revenue than as an incentive to reduce emissions.

## 4.2 Environmental Subsidies

Subsidies to reduce emissions and to improve environmental performance are given in many countries, and indeed the rationale for the taxation of effluent is often that the revenues can be used to finance programs of environmental remediation. Examples of such subsidies include: rebate for air and water effluent charges when investment is undertaken (France, Germany, all of Eastern Europe), accelerated tax depreciation of pollution abatement equipment, and rebates on import taxes for such equipment (several developing countries (CSD, 1996a, 1996b).

Environmental subsidies can be effective in achieving reductions in pollution, but the reductions may be bought at an excessive cost[4]. If taxes are earmarked for environmental protection and the revenues recycled, the resulting level of protection may be too low (as in the case of water protection investment in France) or too high (as in the case of the Netherlands). Hence in industrialised countries, where tax regimes are sophisticated and budgetary provision for Environmental protection is possible, earmarking for Environmental protection is generally not desirable. In economies in transition and developing countries, on the other hand, such budgetary provision is very small or non-existent. Consequently the benefits of having access to some funds from this source for the environment probably outweigh

2 For solid waste disposal thecharge is not really a pollution charge if a private operator of a waste disposal site has to ensure proper mainenance of the site and to pay the government a fee to ensure that full remediation is undertaken. The operator is then responsible for collecting the fee from the easte generators. Hence it is better covered under resource pricing.

3 Such a tax system, where the charge is applied only above an arbitray limit is less efficient than one where the tax is applied for all emissions (Markandya, 2000)

4 Raising general taxes to pay the subsidies had a significant welfare cost. In developing countries this cost is even higher than in developed countries.

the costs of an inefficient earmarking system, at least in the short run, and especially in the economies in transition in Eastern Europe where revenues from taxation of emissions are a critical source of funds for mitigation. The use of special tax treatment as a form of subsidy is more questionable. In general they do not provide the enterprises with incentives to choose the least cost options and they increase the polluters profits, which may lead to further pollution problems (Murty, 1996).

In the light of the above discussion, the use of environmental subsidies outside of the recycling of tax revenues to support environmental investments is not an effective economic instrument. Even in the case of the former, strict guidelines are necessary to ensure that the investments that are supported are indeed cost effective in terms of achieving the desired environmental goals.

## 4.3 Charges on Inputs or Outputs that Damage the Environment.

Although ideally the pollution charge should be imposed on the pollution, in practice this is often impossible. Hence, as a second best, the charge can be levied on the input, frequently the energy source or chemical that is generating the pollution. Examples are taxes on petrol, including differentiated taxes where less damaging fuel is taxed at a lower rate[5], taxes on diesel oil based on the sulphur content of the fuel, taxes on fertilizers and pesticides to take account of their environmental impacts, and taxes on ozone depleting substances.

By taxing the source of the pollution rather than the pollution itself, one does not allow for (a) the possibility that mitigation measures can be undertaken at the 'end of the pipe' and (b) the fact that the impacts in terms of damages vary spatially, so that a general tax overtaxes use in a place where there is no environmental problem, and undertaxes it in places where the problem is very serious. The first issue can be addressed by giving rebates for payments when the polluter makes such an investment in end of pipe clean-up. Examples would be rebating a sulphur content tax for polluters who have flue gas desulphurisation equipment. This is possible in a number of cases, but not always. The second problem of spatial variation is more diffcult to address. To get round it one would have to have different rates of taxation depending on where the inputs were used and practically this is difficult to do, except at a rather crude level. The compromise in using input charges is acceptable in a number of situations, although the detailed analysis of the net benefits of moving to such a tax from a command and control situation are not available, certainly not in an *ex post* calculation.

The other big advantage of using taxes on polluting inputs and outputs is that they can be a major source of revenue, which can generate resources for environmental protection (see above), and even help shift the structure of taxation away from taxing 'goods' such as consumption and employment to taxing 'bads' such as pollution. The argument for using environmental taxes in this way has been elaborated and analysed in recent years and there is certainly a case for considering a shift to 'Green Taxation' as a number of countries are doing (Denmark, The Netherlands, Norway). There are, however, concerns that such a shift may not yield the 'double dividend' of improved economic performance as well as reduced environmental damage and these have to be taken into account[6]. A careful analysis of the impacts of the tax modifications has to be made before recommending its implementation. (For such an analysis see Heady and Markandya (2000)).

In conclusion, the case for using input/output charges as an economic instrument for environmental protection, and as part of the program of action for a move to sustainable development, as has been advocated in previous CSD documents is strong but not without qualification. For many applications, such as non-point sources taxes on inputs are the only possibility for dealing with the issue. This will not solve the problem of 'hot spots' or local concentrations of the pollutant, for which direct controls of some kind will be required. **Hence it is clear that the optimal mix of policy instruments is going to be one that combines economic instruments with direct controls.** A second argument in favour of input/output charges is that they are easier to implement and to collect. Since emissions charges can be much more difficult to collect, and spatial variation may be impossible for them, the case for moving to input charges is even stronger. Finally, there may be a case for raising environmental charges as part of a broader tax reform. This has to be considered carefully in a detailed economy environmental model before any conclusions can be made of what kinds of reforms are desirable.

## 4.4 Permit Trading Schemes.

The use of permits that could be traded developed over a period of time in the United States. It started with

5 Arguably this is an example of an environmental subsidy, where the 'clean' fuel pays a lower rate of tax.

6 The idea is that an pollution charge is beneficial, not only because it addresses the environmental problem, but also because it provides revenue to the government from a (paradoxically) 'clean' source; clean in the sense that there is no loss of welfare associated with its collection. One should be careful, however, in using the double dividend argument, as careful analysis shows that the presence of a double dividend is dependent on many complex factors (See Goulder, 1994, for a further discussion of this topic.)

the air pollution control laws in that country in the late 1970s. At first 'trades' were very limited, and a new set of 'instruments' was developed: netting, offsets, bubbles and banking. The point of departure was the notion that a firm had to meet a given reduction from each of its sources of pollution. This was clearly inflexible, and netting allowed the firm that wanted to generate pollution from a new source to count greater than required reductions somewhere else. Initially netting was only an internal trade. Offsets were introduced to allow new forms to set up in areas where the quality standards were not being met and where the normal regulations would not permit another enterprise to establish itself. If, however, the new firm could acquire a reduction in emissions from another firm by more than the amount the latter would have emitted, it could set up operations (as the degree of non-attainment would in fact be reduced). Bubbles were a similar instrument to netting and offsets, except that trading was only permitted within a firm and for existing sources of pollution. Finally there was banking, which allowed a firm to save emissions reductions, over and above those required, for future trading. All these developments took place between 1974 and 1979.

From the present perspective, and with the hindsight of even 15 years, these seem unnecessarily complex notions. Why not simply allow trade in emissions? The reasons of course lie in the fact that policy implementors are naturally cautious, and will not-try something new unless it is forced by circumstances. The need for flexibility was forced upon them and the instruments evolved gradually, as a wide range of administrative issues were addressed (Rico, (1995)). The lessons from this experience are therefore of great importance for this paper in understanding which policies are likely to work in shifting to a greater use of economic instruments.

More recent programs of emissions trading such as that applied to the reductions in $SO_2$ and NOx under the US Clean Air Act take a more liberalised approach to emissions trading. Each trading unit is allocated an allowance based on historical emissions, which are "generally issued in perpetuity" (Rico, 1995). The program is not concerned to address local pollution problems — i.e. the issue of 'hot spots' — places where emission become concentrated, affecting the health of the local population — is not of direct relevance. All utilities are therefore subject to meeting local pollution standards as well, so any spatial problem arising from excessive local emissions is addressed in that way. In Western Europe there is no real experience with tradable permits, although they are being evaluated for the control of nitrogen oxides and VOCs in the canton of Basle in Switzerland (Stritt, 1994). The scheme was set up in 1992 but no trade has occurred as yet. In Eastern Europe, Poland has experimented with this instrument in the city of Chorzow, with some success (Dudek et al., 1992). However, its wider application there, and elsewhere in Eastern Europe, will need some amendments in the law (Anderson and Zylich, 1993). Among developing countries, permit trading is being introduced in some regions of China.

For water pollution, tradable permits have had a more limited success. Hahn (1990) reports on the Wisconsin Fox River scheme where tradable permits were introduced to reduce the costs of limiting biochemical oxygen demand on sections of the river. The scheme did not work, mainly because trades were very restricted, both by location (you could only purchase from firms in your location group, making the market very thin) and by a number of administrative requirements for each trade to be sanctioned. Finally the participants did not conform to competitive market agent profile. Either they were oligopolistic firms or they were municipal waste treatment plants that did not necessarily behave as cost minimisers. As a result, the simulated model, which had predicted benefits of around $7 million a year proved to be incorrect, and the actual benefits were negligible.

Marketable permits have also been used in other areas. For the phaseout of lead in gasoline in the US, it was recognized that the costs of meeting the short deadlines could be very high for some petroleum refiners. Hence the Environmental Protection Agency (EPA) instituted a trading system whereby refiners who achieved more than the required reduction could sell the surplus to those who did not meet the reductions by the appropriate dates. The scheme was very effective and, by end December 1987, complete phase out had been achieved. A similar application was the phase out of CFCs in the US. The international Protocol for the phase out of these ozone-depleting substances demanded reductions by specified percentages by specified dates. Each producer of CFCs was allocated an allowance equal to that percentage of his original production. The allocation was tradable, so that those who could not make the cuts were able to buy from those who overachieved their targets. The trades were, however, restricted; the amounts any producer could increase production were very limited, even if he acquired the permits. The process of acquiring the permits on the basis of past pollution resulted in windfall profits for the producers who were then taxed for the sale of CFCs. A similar scheme has been introduced in Singapore.

More generally, the idea of allowing some trading in an attempt to achieve environmental goals has caught on. One form that has had some applications is that of transferable development rights (TDRs) (Panayotou,

1994). The idea is that if a conservation area is declared, the owner of the land or buildings does not completely lose the right to develop the property. Rather s/he can exercise that right elsewhere, in a place where further development is restricted and the right has some value. Usually the terms allow an increase in development beyond existing legal limits by about 10-15 percent. Such rights have been used in urban areas in the US and for coastal development in Puerto Rico. TDRs are somewhat different from marketable permits, in that they consist of giving groups who would lose out from the development a compensation that is acceptable and that does not involve raising a large amount of resources through taxation. Marketable permits exploit trade between different agents all of whom have to meet a given environmental objective.

From this brief review, permit-trading schemes can be seen as having considerable potential, in industrialised as well as developing countries. They can play a role in the control of air emissions, during the transition to a phaseout of a damaging chemical, and protection for biodiversity. They almost always have to be implemented in conjunction with direct controls, and, like emissions taxes, they are not suitable for dealing with local pollution problems, where the number of traders may be very few. Related to this, if the restrictions on trades is too great, there will be less chance of success. The choice between tradable permits and emissions taxes will depend partly on cultural factors (how acceptable taxes are) as well as on technical considerations about the costs of making errors in fixing the tax rates or the number of permits.

## 4.5 Recycling Schemes

Under recycling schemes are included deposit-refund programs, as well as subsidies to recycling. They could be considered under taxes and subsidies respectively but they are separated because they raise special issues. Many countries have such schemes. In the OECD, deposit refund schemes operate in Austria, Belgium, Canada, Denmark, Finland, France, Germany, Netherlands, Norway, Sweden, Turkey and the United States. (OECD, 1993). Within developing countries deposit refund schemes have been introduced in Bangladesh, Indonesia, Korea and Singapore, among others. Recycling schemes involving manufacturers have been set up in France and Germany. In the United Kingdom disposal authorities have to compensate collection authorities when the latter retain materials for recycling, the payment depending on the net saving in disposal costs.

Both the deposit refund schemes and the recycling there is evidence that they have reduced waste (OECD (1993)). The bottle bills in the US have reduced litter by 10 to 39 percent and solid waste by 1 to 6 percent. In the same country, kerbside collection programmes obtain recycling percentages of around 35 percent for glass containers and 25 to 56 percent for aluminium cans (Repetto et al., 1992). As far as the impact of the producer- based recycling schemes is concerned, it is considered too early to make an evaluation (that was the case in 1993). Nevertheless it is encouraging to note that the German Green Dot scheme has obtained the participation of over 50 percent of households by 1993, and over 80 percent is expected by this year.

Deposit refund schemes are by largely justified in economic and environmental terms but some recycling subsidies, which are quite popular with governments and the public, and which can be very effective, may be oversold, with too much being devoted in resources to collect waste that can be safely and cheaply disposed off. This is one area where the EIs could be too successful. If recycling results in a reduction of one ton of waste per annum at a cost of $100, and the marginal social cost of one ton disposed is $50, then there is too much recycling; it would be better to dispose of the last ton of waste and save $50.

The marginal social costs of the different options that are available: deposit refund schemes, waste landfill, waste incineration and manufacturer recycling schemes should therefore be equalised. In order to know whether this is the case, the correct marginal costs of each of the options have to be ascertained. Such a calculation has not been carried out, although some estimate must have been made of the relative costs of landfill versus incineration. Certainly a full social cost comparison has not been carried out before deciding on the selection and relative use of the different instruments related to recycling. At the same time the rationale for subsidizing recycling has not been fully analysed. As a second best policy it may be justified to subsidise one form of waste reduction when other forms are underpriced. The analogy is with subsiding public transport when private transport is priced below its social cost. But it is generally better to price private transport at its social cost, as it is to price waste disposal properly. The economics of what the subsidy should be and how it should be phased out as waste disposal prices increase has not been studied in a policy context

## 4.6 Conclusions on Different Economic Instruments

In this section the actual experience with the main economic instruments used for environmental regulation have been reviewed. The purpose of the review was to

see where they can be most effectively employed, and where their use is less desirable. In promoting the different instruments it is important to bear in mind what the strengths and weaknesses of the Economic instruments are and how cases where they have been successfully applied evolved.

From this review it appears that **emissions charges and taxes** have a role to play when the goal is a national reduction in emissions, and when the sources of the emission can be easily monitored. They are not the best instruments when the spatial dimension is important, when the sources are 'non-point' and when the goal is to raise revenue. In this context it is important to note that, for historic reasons, the use of such charges in economies in transition in Eastern Europe is justified. Reforms would be desirable but will take time to implement and temporarily the present structure serves a useful purpose. **Environmental subsidies** can be effective in reducing pollution but the costs in economic terms are high. If the subsidies come from earmarking of environmental taxes, this is not a desirable system to introduce, and not as effective as non-earmarked programnmes, for OECD countries. In Eastern Europe earmarking is still desirable because it is virtually the only source of local funds for environmental protection. Programmes of accelerated depreciation and tax write-offs are not as efficient as other economic instruments that could achieve similar goals.

The use of **charges on inputs and outputs** that are environmentally damaging can work as a very effective environmental instrument. Costs of compliance are generally lower than for emissions charges and rebates can be offered to deal with end of pipe clean-up. They would probably be the easiest to integrate into the general system of taxation, although care should be taken in proposing tax increases on the grounds that they will substitute for other economic taxes. As with all economic instruments, they will need to be imposed in conjunction with some direct controls, to deal with local, and specific problems.

**Permit trading schemes** are an important part of the arsenal of tools at the disposal of a regulator. They have somewhat wider application than pollution charges, and can be introduced in stages, making them more acceptable to the affected parties. Applications where successes have been noted are national and international air emissions reductions, phaseout of lead and CFCs, and trading in development rights over conservation and urban land. Gradual phase-in and limited regulation to the process are important ingredients to their success. Recycling schemes can and have been successful in reducing waste. There is definitely a role for such schemes but the danger is that they are too successful and too much emphasis is placed on them, compared to other instruments for the regulation of waste.

## 5. LESSONS AND CONCLUSIONS

A number of useful lessons can be drawn from the implementation of economic instruments in these various countries. We list below what we think to be the most important lessons.

- A growing number of developed and developing countries are adopting a MIX of instruments that combine command-and-control and economic instruments.
- Cost effectiveness is a very important argument in favor of economic instruments. But, it is not a very effective argument in the political / public arena. Indeed:
  - o For the uninitiated, cost effectiveness is a very vague concept. It needs to be explained and made precise in the context in which it is to be applied.
  - o The gains obtained from using economic instruments (reduction in aggregate abatement costs) are likely to be large only at high levels of aggregate abatement.
  - o Except for a very limited number of cases (e.g. $SO_2$ trading emissions in the U.S.), cost effectiveness has not been the most important reason why economic instruments have been introduced, even in developed countries.
- Pollution charges mainly have a financial (revenue-raising) role rather than an incentive effect.
- From a purely theoretical point of view, revenues should NOT be earmarked for specific environmental investments. A dollar of "environmental revenues" should be used wherever it can yield the greatest social benefit (e.g. education, hospital, reducing other taxes, etc.). However, revenues have typically been earmarked for environmental investments: financing of abatement activities by those paying the tax, financing of public abatement activities (e.g. waste water treatment plant), increasing the capacity of the environmental agency to protect the environment, etc.
- The fee structure is sometimes extremely complex, with hundreds of rates, which vary across many pollutants, many polluters, and many regions. This is specially the case in the Eastern European transition economies. A complex structure often opens the door for polluters to challenge the pollution charge

scheme. Administrative costs should be small, and the pollution charge scheme should be transparent and simple. It should not be open to manipulation. Some points to note are:

  - o Including a large number of pollutants is comprehensive but costly to monitor.
  - o Including a large number of (regional) coefficients may approximate damages, but it is costly to implement. Moreover, there is a high likelihood that the value of the coefficients will be set arbitrarily. It may be best in the first phase to use estimated emissions factors for small sources and actual monitoring of emissions for large sources.

- High inflation has reduced the impact of pollution charges. Reviewing the charge to account for inflation is difficult, and often politically difficult. Automatic indexing of the charge (using for example the consumer price index) should be built into the pollution charge program. Removing revisions away from the political arena is generally a good idea.
- Reliable, comprehensive, easy-to-use environmental management systems (EMS) are often lacking in enterprises impacted by pollution charges etc. This is important since it reduces the potential impact of the pollution charge scheme, and prevents the achievement of the overall purpose of the scheme, namely reducing industrial discharges and improving environmental quality. The environmental management system should be able to answer in a very simple way implementation questions as well as management / policy decision questions such as:
  - o How do changes in industrial discharges affect ambient quality?
  - o Which plants have not self-reported their emissions?
  - o What kind of plants are complying with the regulations?
  - o What kind of plants should we inspect more often than others?
  - o Is ambient quality improving as a result of the introduction of the pollution charge scheme?
  - o What is the impact of enforcement actions on emissions?
- Putting in place a reliable and credible environmental management system is absolutely crucial. It gives information to the environmental agency about the behavior of polluters; it gives polluters a greater incentive to control their emissions since the environmental agency knows "what's going on", and it gives the general public some faith in the capacity of the agency to protect environmental quality. The costs of such a system should not be attributed to the introduction of economic instruments. A similar system is required for the proper implementation of a command-and control approach.
- Despite opposition by industries, there is no indication that the introduction of economic instruments has had a detrimental effect on competitiveness and trade.
- How to increase acceptability of the pollution charge scheme by the industrial sector. These include:
  - o Consultation with the industrial sector increasing the charge according to a clear and transparent procedure instead of starting with a very high charge. Clear use of the funding.

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# Chapter 5

# THE ECONOMICS BIODIVERSITY

*by*

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## 1 INTRODUCTION.

Biodiversity is a complicated concept which has, in recent years, become one of the catchwords of environmental policy analysis. In this section we present an overview of the characteristics of biodiversity, and the role that economic analysis and policy can play in helping to ensure that the loss of biodiversity is not excessive.

There are three main types of biodiversity, namely, genetic diversity, species diversity and ecosystem diversity. The first section explains these concepts. We then consider the fundamental life support services that biodiversity and the ecosystems that contains it provide, these being the services that make human life possible. There is much uncertainty and lack of knowledge regarding ecosystems, which has led to calls for a "precautionary approach" in taking decisions regarding biodiversity conservation. Biodiversity and ecosystems provide a wide variety of more observable values and services. These can be described variously as use values, non-use values, option values and bequest values. We provide examples of biodiversity services that fall into each of these categories.

The problems of biodiversity loss is to a large extent a problem of external costs and benefits in that users of biological resources often have no incentive to account for the loss to others caused by the depletion of these resources. The tools of environmental economics can be used therefore to internalise these externalities.

Biodiversity provides services and value at both local and global levels, making the economic analysis of biodiversity conservation more complicated. We consider first biodiversity conservation at the local level, illustrating that one can think of an implicit market for biodiversity, consisting of the benefits and costs of biodiversity preservation. This illustrates the efficient level of conservation. We then consider examples of policies that could be implemented by developing country governments to ensure that the efficient level of conservation, from the point of view of local communities, is achieved.

Many of the services provided by biodiversity are global services. However, most of the biologically rich countries are economically poor. One cannot expect these countries to conserve biodiversity to the detriment of their own wellbeing. This problem constitutes a global environmental externality, and we extend the market analysis to describe this. We then suggest international conservation policies that might be used to give low-income countries the incentive to conserve.

Finally, we draw some conclusions as to the extent to which economic analysis can contribute to solving the problem of biodiversity loss.

## 2 WHAT IS BIODIVERSITY?

Biodiversity, or Biological Diversity, refers to total variety of life that exists on earth. The Convention on Biological Diversity defines it as " the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part, this includes diversity with in species, between species and of ecosystems."

The concept of biodiversity is relatively new, having been introduced in 1980 (Lovejoy (1980). However, its origins go back many centuries, with the first formal attempts to classify species being at the time of Aristotle in the 24th Century BC. (Heywood et al (1995)). Knowledge of species remained very limited until the last 200 years, and even now there is great uncertainty as to how many species there are. The Global Biodiversity Assessment (GBA) takes the figure of 13 million as a working estimate, but estimates have ranged from 2 million to 50 million. Only 1.4 million species have actually been identified. From these figures alone it is clear that one of the fundamental problems when analysing the economics of biodiversity is the scientific uncertainty. More specific aspects of this uncertainty, and its effect on policies relating to biodiversity, will be discussed below.

The unit of analysis of biodiversity is usually, as suggested above, thought of as being the number of species in existence. However, it can be thought of on three different levels, or scales, as follows.

**Genetic diversity** is the total of genetic information contained by all the animal, plant and micro-organism species on earth. This concept is even more recent than the concept of biodiversity itself (Pluknett et al (1987)), and its development has stemmed in part from the fear that the increasing uniformity within certain species, particularly crop plants, reduces their viability and the services that they provide. The clearest example of this is the loss in variability of crop plants since the 1960's due to the increased use of high-yielding but genetically uniform species.

Table 1 below illustrates the process of increasing genetic uniformity in crop plants:

**Table 1: increasing genetic uniformity in crop plants.**

| Crop | Country | Number of Varieties |
|---|---|---|
| Rice | Sri Lanka | from 2,000 varieties in 1959 to 5 major varieties today |
| Rice | India | from 30,000 varieties to 75% of production from fewer than 10 varieties |
| Wheat | USA | 50% of crop in 9 varieties |
| Potato | USA | 75% of crop in 4 varieties |

*Source: World Conservation Monitoring Centre "Valuing Biodiversity" in World Conservation Monitoring Centre, Global Biological Diversity, Chapman and Hall, London.*

One of the negative effects of lower genetic diversity in a crop species is that a larger proportion of a planted crop is susceptible to the same diseases. This means that although uniform varieties may produce higher yields on average, they are more likely, than a genetically diverse crop, to fail altogether. A recent study by Prakash and Pearce (1998) measures the effect of genetic uniformity and biodiversity loss on the stability of agricultural systems. This consists of comparing the variability of harvest yields before and after the Green revolution in India. They show that while average yields have increased, their variability has also increased since the Green revolution, implying that the reduction in biodiversity has decreased the stability of the system.

**Species diversity** in its simplest form refers to the number of species within a system or given area. Species can be defined as populations within which breeding can take place. New species develop most commonly when populations of a species become isolated and evolve separately due to different ecological conditions. A more general definition of species diversity would account also for the extent to which the different species in an ecosystem are related (Reid 1992). This is also known as taxonomic diversity. For example, an ecosystem containing 100 species of birds and 50 species of lizard would, other things being equal, be more diverse than one containing 150 species of birds. Likewise, there are more terrestrial species than there are marine species, but terrestrial species are more closely related to each other. Therefore diversity in ocean systems is higher, relative to terrestrial systems, than the number of species would suggest. However, there is no agreement yet as to the best general measure of species diversity.

**Ecosystem diversity** is the number of habitats, biotic communities and ecological processes in the biosphere, as well as the extent to which ecosystems vary. Ecosystem diversity can itself be thought of on different levels. Functional diversity is the abundance of functionally different organisms. Community diversity, or patchiness, is the number, size and spatial distribution of communities in an ecosystem.

## 3 THE SERVICES PROVIDED BY ECOSYSTEMS AND BIODIVERSITY.

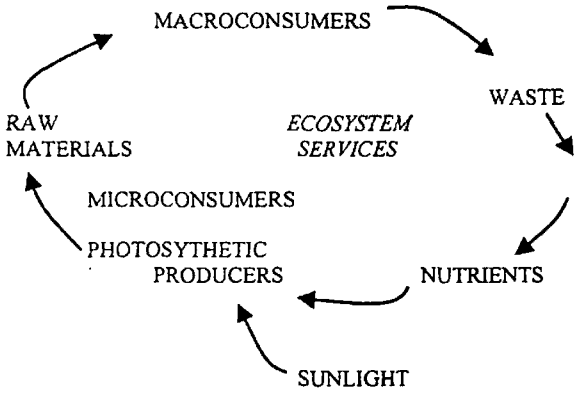
Biodiversity and the ecosystems in which it is contained represent the foundation of all life, including human life. That is, in addition to the direct and measurable services that biodiversity affords, which we will discuss below, it also provides a life support system comprising services necessary for the existence of human life.

Ecosystems, whether natural or managed, are required to capture the sun's energy and the food and raw materials on which we depend. Biodiversity, in turn, is required to maintain the productivity and stability of these ecosystems. Part of this process is the hydrological cycle whereby water evaporates, falls as rain, and is absorbed and used in photosynthesis as well as being diverted for direct human use. Functioning ecosystems regulate this cycle, helping to avoid the extremes of flood and drought

Another part of the natural productive process is the production and maintenance of the fertile soils. Micro-organisms within ecosystems are responsible for breaking down waste, both natural and manmade, into nutrients, which maintain soil productivity. Likewise, the loss of the protection provided by ecosystems usually results in the loss of fertile soils. Ecosystems themselves can be responsible for recycling, trees can capture airborne dirt that washes back down to the ground as productive soil. Some of these ecosystem services are summarised in the figure 1 below.

Another life support service provided by ecosystems and its constituent biodiversity is climate regulation, both on a local and a global scale. As noted, ecosystems are necessary for regulation of the hydrological cycle. On a global scale, they are necessary

**Figure 1: The ecological services of biodiversity.**




to maintain the composition of the world's atmosphere and therefore its system of climate regulation. The mitigation of global warming is the best known example of this service.

## 3.1 Scientific uncertainty regarding the functions and services of Biodiversity.

Scientific knowledge of ecosystem structure is far from complete. This results to a certain extent from the immense complexity of the interdependence of species and systems. Therefore, it is very difficult to predict the effect that removing, or changing the level of a population of one species will have on the rest of the species in a community, or on the viability of the ecosystem.

Some species are known as "keystone species". These are species whose removal would have a profound effect on the structure of an ecosystem. An animal that is the only large predator in an ecosystem is likely to be a keystone species. If such a species is lost or depleted then this will have a more significant effect than the depletion of another species in the ecosystem. However, different species in an ecosystem can be keystone species under different states of nature. Thus, there are latent relationships that cannot be observed under the current state of nature. This constitutes an inevitable source of uncertainty.

A problem that is more difficult still to analyse is the existence of discontinuities or "threshold effects" in the behaviour of ecosystems. Thresholds are defined as critical values for populations or natural cycles at which the ecosystem loses resilience and hence the capacity to maintain its order. This means that a marginal change in one of the parameters of an ecosystem may lead to vast social and economic costs, if the result is a collapse in the ecosystem. The crux of the problem lies in the fact that the critical values for an ecosystem cannot be observed until the moment at which the collapse takes place. Therefore, considerable uncertainty remains as to the threshold values of some of the most important ecosystems.

## 3.2 The precautionary principle.

The uncertainty regarding the extent to which ecosystems, including the global ecosystem, can be depleted of their constituent parts before their integrity and viability is put at risk, has led for calls for a "precautionary principle" to be employed when taking decisions involving ecosystem destruction. This principle implies, among other things, postponing F actions whose effects are uncertain but which could be very damaging, until uncertainty is reduced.

The possibility of sudden, dramatic changes in the structure of ecosystems has led Ehrlich and Ehrlich (1981) to liken the process of species; extinction to the progressive removal of rivets from an aircraft in flight. While no particular rivet is essential, they say, as each one is removed the likelihood of disaster increases.

# 4 THE ECONOMIC VALUE OF BIODIVERSITY AND ECOSYSTEMS

The total economic value of biodiversity consists of the components that are familiar from other areas of economics valuation. These values include use values (direct and indirect), non-use values, option values, bequest values and existence values.

It is worth pointing out at this stage that it is often very difficult to separate the value of biodiversity from the value of the habitat that supports it and, vice versa, that it helps to support. For instance many ecosystem services, such as soil protection and the regulation of water run-off, depend on the existence of certain types of ecosystem, such as forests, and not on any particular species. However, a loss of species diversity may put the ecosystem, and therefore the service, at risk. Likewise, it is often possible to put an economic value on a particular species but again it is difficult to consider this value in isolation from that of the habitat which supports it, and therefore from the rest of the species and populations necessary for the maintenance of the habitat. Therefore to simplify in what follows, we consider various types of value as representing the value of biodiversity.

**Direct use values** derive from direct use of or contact with a biological resource or ecosystem. The most obvious example of the economic value of biological resources in all societies is the use of wild and domesticated plants and animals in production and consumption. The introduction of imported crop varieties has had a tremendous impact on agricultural productivity in the past, with 98% of food production in the US being

derived from crops that originated from outside its borders. (Plotkin in Wilson, 1988).

The world's food production is now based on a remarkably small number of species - 54% of the developing world's calorific consumption is accounted for by wheat, maize and rice. However, in part because of this genetic diversity in agricultural crops has significant economic value. In traditional agricultural systems, a high genetic variability among crop plants creates greater stability in the face of climatic, disease and pest risk. Although yields are lower on average, the variability of yields tends to be lower. For societies that rely directly on the land for food, this "insurance" service provided by genetic diversity is extremely important. As the use of more genetically uniform improved varieties increases, the value of genetic resources will increase, since these varieties have a limited life and need periodic influxes of genetic material to remain viable. This genetic material may be preserved "ex-situ" but, as emphasised by Wessinger (1990), this cannot preserve the whole range of an organism's diversity, but merely a sample of it.

Use values of habitats that support biodiversity are an important aspect of the benefits of preserving biodiversity. These include, in the case of forests, the value of the timber that can be sustainably harvested, and the non-timber products that may be harvested, such as fruit, nuts and mushrooms. They also include the provision of shelter, and the possibility of using the forest floor to graze domesticated animals.

Another significant direct use value of biodiversity is the value of medicines directly or indirectly derived from wild species. Up to 80% of people in developing countries use traditional remedies for their primary health care, most of which is derived from plants (Farnsworth in Wilson 1998) Moreover, many synthetic drugs sold in developed countries are derived from plants.

**Non-consumptive direct use values** of diverse habitats and species can be an important component of the value of biodiversity preservation. These include the use of the reserve of diverse species for scientific research and, significantly, the value generated by tourism. Tourist revenues as affected by the diversity of species that can be seen on a visit, and the variety of habitats and landscapes that can be visited. Certain types of ecosystems, for example rain forests, are particularly likely to attract tourists. Likewise, charismatic species, such as tigers, gorillas and dolphins, attract visitors and generate revenues. Therefore, ecosystems and biodiversity generate tourist revenues for the country in which they are found, and they also generate value for the tourists who visit them. The former is a market value, which can be measured directly. The latter may exceed the money spent on the tourist visit, and therefore could contain a non-market value component, which can be measured using the tools of environmental valuation.

**Indirect use values** of biological resources and ecosystems derive from the role of these resources in supporting economic activity, or in supporting other biological resources which yield direct use value. This relates to the ecological functions performed by ecosystems, such as the regulation of water run-off by forests, and the storage of carbon dioxide. It can also relate to the services provided by species and biodiversity in sustaining the ecosystems that provide direct values.

Thus while, as pointed out by Flint (1997), it is unlikely that any direct use values are generated by, for example, the 1 million species of invertebrates, it is possible that some of these species provide key services in maintaining the ecosystems that provide direct value themselves, and support species that provide direct value.

**Option value** is the value placed on preserving an asset which, while not used at present, may be used in the future. In the case of biodiversity this could refer to the value that potential tourists might place on preserving ecosystems and landscapes that they might wish to visit in the future. Another important option value is the value that might be derived in the future from hitherto undiscovered species. The uncertainty regarding the identity and role of species and the structure of ecosystems is an important aspect of option value. If ecosystems are destroyed, causing the loss of species whose identity and potential use is unknown, then potential value is lost. Therefore the option to capture this value is secured by preserving habitats for future analysis.

**Bequest values** refer to the value that people place on preserving natural assets, including ecosystems and biodiversity, to pass on to future generations.

**Existence values** are the values that people place on knowing that a particular species or ecosystems exists. For instance, people may place value on the knowledge that a certain species of whale has been saved, from extinction, even if they do not envisage every using the species themselves, directly or indirectly.

## 5 VALUING THE SERVICES OF BIODIVERSITY

### *Why value biodiversity?*

We have discussed the ecological significance of biodiversity, and the ' importance of biodiversity to the functioning of ecosystems. We have considered some of the services provided by ecosystems directly and

therefore by the diversity of species indirectly in maintaining functioning ecosystems. Clearly, these services, and therefore natural habitats, have value. However, conserving natural habitats often involves an "opportunity cost", which is defined as the benefit foregone by putting resources to a given use. Assuming that resources are put to their most efficient use, the opportunity cost is the value of putting the resources to their next best use. This cost might be the value that could be obtained by harvesting the ecosystem itself, the most obvious examples being forest ecosystems and fish populations. Alternatively, in the case of terrestrial ecosystems, the opportunity cost could include the alternative uses to which the land could be put.

Economic development requires that land be converted from natural habitats to alternative uses. Sustainable economic development requires that sufficient essential life-support functions remain intact. Even if we accept that this condition must be met, this still leaves the question of which natural habitat ought to be preserved for life support services, and how much and which natural habitat ought to be left intact over and above the bare minimum required for environmental sustainability. We can use the tools of environmental economics to answer these questions by attempting to measure the economic value of in-situ natural habitats, and comparing these to the opportunity costs of conserving them. This section is concerned with measuring this economic value.

## 5.1 How to value biodiversity: some examples

The tools of environmental valuation have been covered elsewhere in this document and therefore we will not repeat the exposition here. The valuation of biodiversity is similar in theory to the valuation of other environmental assets. The major difference perhaps is the uncertainty with respect to the indirect services that biodiversity provides. Here we provide a few examples of the application of environmental valuation to biodiversity.

Contingent valuation involves asking respondents hypothetical questions about their valuation of an asset. Whitehead (1993) carried out a survey including the question "Suppose that a certain contribution... would be needed to support and fund the loggerhead sea turtle program. Would you be willing to contribute this amount each year...": Thus, this method can be used to elicit the value that people place on conserving biodiversity. This is an appropriate method to elicit both use and non-use values of biodiversity. Gutienez and Pearce (1992) used this method to set the existence value of the Brazilian Amazon at $30 billion.

Non-market values such as aesthetic and recreational values can be calculated using the Travel Cost Method. This method uses the theory that the amount of money that people spend visiting a site, such as a particular ecosystem, must represent a lower bound on the value that they place on the ecosystem. Using this method Tobias and Mendelsohn (1990) calculated the valuation of tourist trips to Costa Rica's Monteverde Cloud Forest at $35 per visit. This extrapolates to a flow of value of $12.5 million annually.

Direct use values of habitats, for instance the use of fisheries and forests, can be calculated directly from the value of the products and services derived. Peters, Gentry and Mendelsohn (1989) calculated the value of sustainable harvesting of the Peruvian Amazon at $1987. If the flow of this value over time is converted into a capital value, then this calculates to a value of $6820. This compares to a value of $1000 per hectare for clear felling.

We noted above that the destruction of ecosystems has uncertain effects, and that the actual and potential services provided by species and ecosystems are not fully understood. This means that it is unlikely that the values derived using the tools of environmental valuation are the values that would be derived in a situation of perfect information. In particular, it is likely that certain species and ecosystems have a significant indirect value, but that this value will not be identified in environmental valuation because people are not aware of the asset, or of the services that it provides. Therefore the values derived from environmental valuation of biodiversity should be treated as a partial estimate of the total value.

# 6 THE DEVELOPMENT AND LOSS OF BIODIVERSITY

The existing level of biodiversity in the world has developed over the estimated 4.5 billion years since life on earth began. It developed initially through the process of mutation and expansion of species into previously unoccupied niches. By the end of the first 4 billion years only very simple multicellular organisms had evolved. The development and speciation of complex organisms occurred in the relatively short space of 100 million years. Since then, speciation and extinction have occurred at similar rates, so that evolution has been characterised in the Darwinian tradition as the "survival of the fittest". To this extent, extinction can be seen in a positive light, since inferior species make way, given limited resources, for their more advanced successors. However, the role of humans in the extinction of species has meant that the natural process is distorted, and that this view of extinction is no longer necessarily valid. Estimates put the rate of species loss at between 20% and 50% over the next century. (Swanson 1993)

## 6.1 Causes of biodiversity loss.

The prime reason for the loss of biodiversity is the destruction and degradation of habitats. Related to these are the over-harvesting of species and ecosystems and the introduction of exotic species into ecosystems. Almost 20% of the world's vertebrates in danger of extinction are in danger at least in part because of invasive species. Perhaps the best known example of this phenomenon is the introduction of the Nile Perch into Lake Victoria, resulting in the loss of at least 200 fish species in the lake.

Pollution is another factor related to species loss, in that contamination can cause stress to ecosystems and leads to the reduction or loss of populations. For instance, acid rain in combination with other forms of pollution has virtually eliminated the life in a number of North American and Scandinavian lakes. The role of fertilisers in this process is also important. While their beneficial role in the supply of food is indisputable, they leak into rivers and lakes causing vast increases in algae, a reduction in oxygen (eutrophication) and consequently the loss of other species.

### 6.1.1 The social and economic reasons behind the loss of biodiversity

The immediate causes of biodiversity loss are loss of habitat, overharvesting and pollution. However, the underlying causes of this are the incentives of individual users of natural resources to ignore the effects of their activities on others. That is, the social benefits of biodiversity are not reflected in private decision making.

These factors which stem from this can be summarised under the following three headings.

***Incomplete markets***

The absence from the markets of most of the environmental effects of biological resource use means that these effects are not taken into account by decision-makers. Moreover, Given current institutional structure and property rights, public good nature of many ecological services, and fundamental uncertainty precludes development of markets in the environmental effects of biodiversity loss

***Government failure***

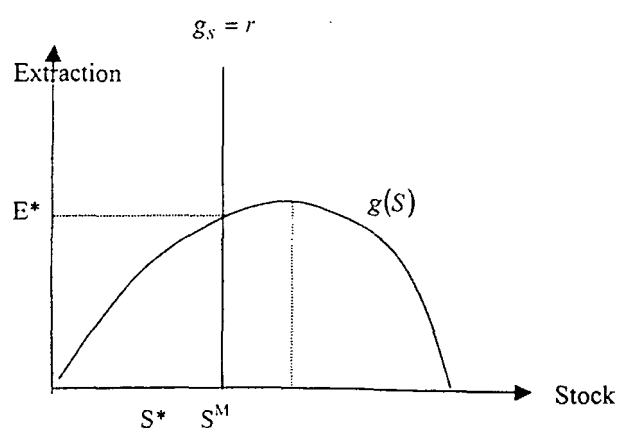
There are forces that influence the prices of commodities for which there do exist markets to the detriment of conservation. One of the most important of these is the effect of government policy. This paper concerns the use of economic tools to propose government and international policies that will help ensure that the appropriate amount of natural habitat and biodiversity is conserved. One of the problems that must be overcome is that government policies have often had "perverse" effects with regard to the efficient preservation of biodiversity. In particular, incentives have been created which accelerate the rate of deforestation and the conversion of forest land to agricultural and other uses. For example the granting of short-term logging concessions provides an incentive to "mine" forests to extract the capital value as quickly as possible, since there is no benefit to the concessionaire from sustainable forestry.

The effects of these policies are often compounded by a failure to enforce logging regulations, to account for the non-timber benefits of forests (Repetto 1993). Governments have used the timber industry to stimulate local economies and maintain employment, despite the fact that these industries are often inefficient, environmentally destructive and unsustainable.

Non-forestry policies have also caused severe forest depletion. This is related to the fact that destruction of natural habitat is necessary for economic development. Agricultural programmes have created incentives to clear forest land for crops. Such programmes have failed in several countries having been implemented at enormous financial cost to the government, and at the cost of deforestation of vast areas. (Repetto and Gillis 1988). Government-subsidised road construction has increased the profitability of timber extraction and from converting land to agriculture or grazing.

We have seen that, due to the public-good nature of many of the services provided by ecosystems and biodiversity, the free market is unlikely to provide a socially optimal level of biodiversity conservation. However, distortionary government policies have meant that the actual level of conservation has in some cases been lower still than it would have been under the free market.

**Figure 2: Efficient resource depletion.**



*Poverty and resource use.*

The fact that many communities who depend directly on using natural resource are very poor means that they may have very high implicit rates of discount. That is, the immediate need for survival may mean that natural resources are depleted. This might happen even if it is to the longer-term detriment of the communities' own well being, as well as causing a loss of biodiversity and therefore wider value.

### 6.1.2 Economic efficiency and the loss of habitat and species.

We saw above that the present level of biodiversity has taken billions of years to develop. Swanson (1993) points therefore that biodiversity occupies a particular place on the interface between renewable and nonrenewable resources. Individual species are renewable resources and may be exploited as such. However, biodiversity itself is a non-renewable resource.

One of the primary reasons that biodiversity loss is a problem is the fact that its conservation has an opportunity cost. In the case of a marine ecosystem this is the value that could be derived from mining a fish stock to extinction. Likewise in the case of the forest ecosystem, the opportunity cost of conserving the forest for sustainable use is the value of clearcutting the timber and putting the land to an alternative use. The economics of this process can be described by the following figure.

In the figures, the curved line g(S) represents the growth of a natural resource stock. This also represents the amount, at each level of the stock, that can be harvested without depleting the stock. Therefore, SM is the level of the stock at which the "maximum sustainable yield" can be harvested.

It can be shown mathematically that the exploitation policy that maximises the net present value of the stock will deplete the stock until the growth rate, or the slope of the curve g(S), is equal to the rate of discount. In the example shown this stock level is S*. However, for a slow-growing resource like a tropical rain forest, there may be no level of the stock at which this condition is met. In this case, depletion is the optimal policy.

This analysis may not account for, in the case of forestry, the non-timber benefits from sustainable resource use, and almost certainly will not consider the indirect benefits provided by a forest to nearby agricultural, land, such as hydrological regulation and the protection of the land from excessive water run-off. Likewise, the global values of the biodiversity contained in the forest and its role in local and global climate regulation are not accounted for in this analysis. More efficient resource use can be achieved if these benefits are internalised. This internalisation is the role of environmental valuation and regulation.

## 7 ENVIRONMENTAL ECONOMICS AND THE CONSERVATION OF BIODIVERSITY.

We have seen that the problem of biodiversity loss can be described as an example of the problem of environmental externality. When the problem is cast in these terms, it is clear that the standard tools of environmental economics can, at least up to a point, be used to identify the efficient level of biodiversity preservation and to suggest policy tools to encourage this level of preservation.

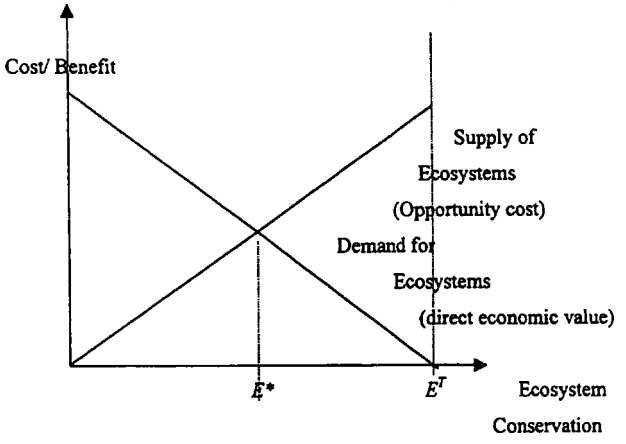
We have also seen that the benefits of biodiversity and ecosystem services are enjoyed at various different levels. The services provided by a mountain forest ecosystem, for example, include regulation of rainwater run-off, which benefits those who farm the land below in the valleys. The fact that the likelihood of floods is mitigated may benefit all those who live near a river that originates in the mountains. The revenues from tourists who visit the forest benefits the country as a whole. The role that the ecosystem plays in global climate regulation, including the absorption of carbon dioxide, benefits the global population.

The maintenance of efficient levels of these various benefits requires different economic tools. We consider first the problem of internalising the external benefits of biodiversity at the local level.

### 7.1 Biodiversity and the appropriation of value: the local problem.

Biodiversity conservation is an issue of fundamental importance to development policy. Because biodiversity increases with reduced latitude, the most species rich countries are also some of the world's poorest countries. Biodiversity, as we have seen, is a land-use issue, in that preservation means that the land cannot be put to alternative uses. Therefore, in formulating biodiversity conservation policies one must account for the fact that the benefits to the local communities who are being asked to conserve must outweigh the costs to them. Otherwise it is unlikely that the policy will be accepted. In the case of very poor communities with high effective discount rates, it is possible that this stipulation might have to be extended to being that the current benefits of the policies must outweigh the current costs.

**Figure 3: The local market for local biodiversity conservation.**




At the same time, economic development policy must bear in mind the values that ecosystems and biodiversity have for local communities. Thus while, as noted, economic development must involve a certain amount of habitat loss, it is important that this should not exceed the amount that is economically efficient from the point of view of local communities. Any conservation over and above this level required of communities by the national or international community must be the subject of alternative policies, as analysed below. We can think of there being a market for conservation at the local level, as depicted in figure 3.

Figure 3 illustrates how the standard market framework can be used to analyse the efficient level of biodiversity preservation in a local framework. $E^T$ represents the entire amount of land in the economy.

The supply of biodiversity represents the opportunity cost of preserving ecosystems. This cost consists of the benefits that can accrue from converting land to agricultural, industrial or residential uses, and from capitalising the value of the ecosystems. This value applies particularly to forest ecosystems. Economic development requires land and therefore the opportunity costs of preservation when very little land has been developed are high. We can assume that as more land is converted, this opportunity cost decreases.

The (local) demand for biodiversity represents the use values that accrue to local communities. These benefits consist of the direct and indirect use values of ecosystems, for example sustainable harvesting of timber and non-timber products, aesthetic and spiritual values, and indirect ecosystem services to agriculture, as discussed above. These marginal benefits can be assumed to be low when there is an abundance of ecosystems, i.e. when land use is close to ET. However, they will increase as development takes the economy further from ET. This analysis could be extended to uncertainty and possible threshold effect. Here, for simplicity, we extrapolate from these.

## 7.2 Economic instruments to reduce biodiversity loss at the local level

*Introduction of Property Rights Institutions*

We have seen that the problem of excessive biodiversity loss can be described as a problem of environmental externalities. This is a problem familiar in environmental economics, and the solution to such problems involves the assignment of property rights. In the case of pollution externalities, the assignment of property rights over the atmosphere may means that a firm must pay local people compensation for polluting the air. Likewise, in the case of biodiversity, assigning property rights to communities who use natural resources would mean that any development of the resource would mean that the value lost to local communities would have to be compensated. Thus the (previously) external costs of development of natural resources would be internalised.

Domestic policies to preserve biodiversity ought to attempt to ensure that a locally efficient level of preservation is achieved. This means establishing a "level playing field" between preservation of natural ecosystems and alternative land uses. In particular this means removing the subsidies and other incentives referred to above which encourage the short-term depletion of natural resource stocks. It also means accounting for the efforts of poorly defined property rights, especially where this means that the benefits that local corurnunities derive from direct use of natural resources is not accounted for in decisions as to whether to conserve or develop land.

This policy might involve providing local communities with land tenure and access to credit, markets and the technology necessary for them to manage forest lands sustainably and profitably. The problem of population pressure on natural resources might also be relieved by investing in nonland based employment.

*Internalising the benefits from eco-tourism*

The benefits from eco-tourism could be internalised to the local community, rather than accruing solely to urban and foreign landowners. This would give local communities a stake in the preservation of natural habitats, and reduce the problem of encroachment. This can be done partly by providing employment to local people as tour guides and forest rangers. and also by allocating part of the revenues from eco-tourism to local employment projects.

*Environmental Bonds*

The use of natural resources can be made more conservation friendly by the use of environmental bonds. These are payments made by developers, in advance of

development, such as logging and mining, to cover potential damage. This means that they have an incentive to minimise damage, and to carry out restoration of the landscape. Moreover, funds are then available to carry out the restoration if developers fail to comply. Environmental bonds have been used as a condition for granting logging concessions in Asia.

***Water charges***

We have seen that indirect benefits of forest ecosystems, such as; watershed protection and hydrological regulation, accrue mainly to farmers and residents downstream from the forest ecosystem. The fact that these benefits are external to the process of forest preservation means that it is underfunded. Local populations have no incentive to account for these benefits in their use of the forest. Therefore, a system of charges for water use could be used to internalise these benefits to the communities who control the forest.

## 7.3 Biodiversity and the appropriation of value: the global problem

Many of the values of biodiversity that we have discussed in this chapter are global environmental values. These benefits include global climate regulation, pharmaceutical uses, option values and existence value. Because the opportunity costs of ecosystem preservation fall mainly on low-income countries, it seems clear that some sort of international transfer mechanism is required to provide developing countries with the incentive to preserve biodiversity, over and above the amount that is efficient from their own point of view.

We can extend the analysis of the market for biodiversity by including in he marginal global benefits of conservation the global benefits of ecosystems conservation. Figure 4 shows the situation in which the marginal benefits of ecosystem preservation are added to the marginal domestic benefits.

Figure 4 shows how the situation changes when the external global values of biodiversity are internalised into the analysis. While the efficient level of conservation from the point of view of the local economy is $E_L$, this illustrates that the globally efficient level of conservation is $E_G$. In the next section we consider international preservation policies, which might ensure that this level of conservation is achieved.

## 7.4 International policies to reduce biodiversity loss.

***International payments***

The ideal system for regulating the conservation decisions of individual states is one that causes them to account for the global benefits of their natural resources. The underinvestment caused by a failure to account for these values could be remedied by means of a system of "strategic international payments." These payments would be used for the purchase and sustainable management of local land. This would achieve the required internalisation without an infringement of the sovereignty of the states in question.

***International parks agreements***

Another, similar example of international transfers is the "international parks agreement." Under this, the international community would pay rent on the land in order that it remain unconverted and managed to maintain diversity. This would require compensation for most of the alternative uses to which the land could be put. This constitutes a stream of payments to compensate for the development opportunities foregone.

***Resource franchise agreements***

Another possible example of international preservation arrangements is a "resource franchise agreement." Under this agreement, which is between the state that owns the resource, the international community and the franchisee, the state would issue a franchise allowing the franchisee to undertake some limited exploitative activity, for example sustainable timber extraction or rubber tapping. The international community then compensates the state for the amount of rent, over and above the amount paid by the franchisee, that the government could have received for a more destructive land-use.

***Transferable development rights***

A final suggested policy for conserving ecosystems and biodiversity is Transferable Development Rights (Panayotou (1992). TDRs have been used for the preservation of historic sites, and have been proposed for the conservation of biodiversity. Developing countries would set aside a section of land to be conserved, and divide it into a number of TDRs, say, one per hectare. Each would state the precise characteristics of the land and any accompanying benefits to the holder, such as the right to carry out scientific exploration for new and commercially useful species.

These rights could then be offered for sale, both locally and internationally, at a price covering the opportunity costs of development. Potential buyers of TDRs include local and international environmental organisations, governments of developed countries,

chemical and pharmaceutical companies, scientific societies, universities and research institutions.

Motivations for holding the TDRs would vary. Some could hold them for the conservation value. Others would hold them for the right to prospects for new chemicals and plants, and the opportunity to develop new medicines. Others might have speculative motives for holding them in that they might expect the value of the TDRs to rise if, for instance, a new plant or species is discovered which increases the value of prospecting rights.

***How to allocate funds: which biodiversity should be preserved?***

A problem that pertains to biodiversity conservation on both the local and the global levels is which biodiversity ought to be preserved. At present, most resources are devoted to preserving those species that are rare and endangered. However, recent work has shown that, even if genetic diversity is valued for its own sake, rather than for the services it provides, this approach is flawed (Perrings and Pearce (1992)). This is because if diversity is the objective which is to be maximised, then effort should be concentrated on those species which are genetically most distinct from other species and which have the greatest probability of survival.

## 8 CONCLUSIONS

We have seen that biodiversity provides fundamental life support services, as well as more observable services. All of these services have value, and the problem of excessive biodiversity loss stems largely from the fact that many of the values that biodiversity provides do not accrue to those who control the resources, but are external. This means that the full value of ecosystems and biodiversity is not accounted for by decision-makers, and biodiversity loss is socially excessive.

Economic analysis can be used to illustrate that accounting for the external costs of biodiversity loss reduces this problem, and creating an implicit market in biodiversity at both local and global levels. This analysis shows that if the external costs and benefits can be valued and internalised into decision making, then the socially efficient level of biodiversity conservation can be identified. We have considered some policies, national and international, which could be used to bring the outcome closer to the efficient one.

There are limits to the extent that economics, in particular environmental valuation, can contribute to biodiversity conservation. These limits stem from two factors that we have discussed above. The first is that many of the services provided by biodiversity are so indirect and little understood that it is not possible for subjects in a valuation study to value them. Therefore, only a subset of the values of biodiversity can be measured in this way. The second problem is the uncertainty regarding the structure of ecosystems and the effects of ecosystem and biodiversity depletion. The fact that the costs of biodiversity loss are not measurable means that they cannot be internalised into a market-type analysis, limiting the contribution of economic analysis.

In conclusion therefore, economics can make a significant contribution to analysing the problem of biodiversity loss, and the extent to which it is caused by external costs and benefits. Although the capability of environmental valuation methods to measure these costs and benefits is somewhat limited, we can suggest policies that will bring individual countries and the global economy, closer to efficient preservation of biodiversity.

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# Chapter 6

# AIR QUALITY MANAGEMENT
# THE ROLE OF ECONOMIC ASSESSMENT TECHNIQUES

*by*

*Richard Boyd*

## 1 INTRODUCTION

Air pollution is a serious, and growing problem in both industrialised and developing countries, in both rural and urban areas. While natural events such as forest fires, release huge amounts of certain pollutants into the atmosphere, the primary cause of air pollution is human (economic) activity, such as the generation of electricity, vehicle use, and industrial operations, mainly in urban areas.

Air pollution is not simply the function of the amount of pollutants released into the atmosphere. Factors such as topography, weather conditions, time of day, and the type of pollutants and any interactions between them all help determine pollution levels.

The effect of air pollution on humans, ecosystems, buildings and monuments has long been recognised, but it is only recently that the relationships between air pollution emissions and exposure have become well understood. The health effects of certain air pollutants have been documented in numerous studies[1], and high concentrations of these substances in many developing countries are known to lead to increased incidence of illness (morbidity), especially among individuals suffering from respiratory problems, and to cause premature death (mortality). Those pollutants which give the greatest cause for concern are carbon monoxide (CO), hydrocarbons (HC), sulphur oxides ($SO_x$), nitrogen oxides ($NO_x$), particulate matter (PM), lead (Pb), and secondary pollutants, particularly ozone ($O_3$). Inhalable suspended PM, for example, particularly particles less than 10 microns in size ($PM_{10}$), can pass through the natural protective mechanism of the human respiratory system. The smallest particulates (2 microns or less - $PM_{2.5}$) are created primarily by the combustion of petroleum fuels. The World Bank (1992) estimates that reducing SPM to safe levels could reduce premature deaths by 300,000 to 700,000 a year in developing countries.

Concentrations of pollutants in ambient air increases with the rate of urbanisation and energy use. Air pollution in some newly industrialised developing countries is also worse than in industrialised countries due to the frequent use, by industries in these countries, of outdated technology. In addition, many vehicles lack emissions controls. Strategies for air quality protection should therefore be given due priority as these countries develop.

Economic growth does not need to be followed by a decline in air quality. Air Quality Management (AQM) programmes or systems can be implemented, which are intended to keep air pollutants below the levels that would damage the environment or harm the health of humans. Such an all encompassing air quality management programme should include: an air quality assessment; an environmental damage assessment; and an economic analysis followed by the selection of an 'optimal' control strategy.

For this to happen, the cost of investment in state-of-the-art measures to control air pollution must be reasonable, and the likely impact of this spending on other sectors must be taken into account. For instance, medical facilities and services, food, education, transport and other social services all need to be provided, as well as clean air. When the costs of investing in air quality programmes are high, and such investments could divert resources from other useful and politically popular programmes, investments which can be justified in economic terms - understood by decision-makers - are more likely to win approval. Problems such as child malnutrition, polluted drinking water, and high levels of unemployment will ordinarily take precedence over programes to combat air pollution. Nevertheless, the benefits of air quality enhancement are very real, and they can be measured, and so they should, therefore, receive just priority.

In short, both the damaging effects of air pollution, and its abatement, have costs associated with them. A knowledge of the damage and abatement costs is the beginning of the process of developing an 'optimal' control strategy for air pollution management. The purpose of this Chapter is to demonstrate the use of economic techniques in measuring the benefits of pollution-induced impacts avoided, and the cost of achieving those reduced impacts, in the context of an Air Quality Management programme.

[1] See, for example, Working Group on Public Health and Fossil Fuel Consumption ( 1997); Cropper *et al* (1997); Ostro (1994); Ostro *et al* (1994 and 1996); Schwarz and Dockery (1992); Dockery and Popetal (1993); and Pope *et al* (1995).

## 2 AIR QUALITY MANAGEMENT (AQM) PROGRAMME

### 2.1 Introduction

An air quality management (AQM) programme or system comprises the following elements (Shah, Nagpal and Brandon, 1997):

- air quality assessment;
- environmental and human health damage assessment;
- abatement options assessment;
- cost-benefit analysis or cost-effectiveness analysis, and
- selection of 'optimal' control strategy (abatement measures).

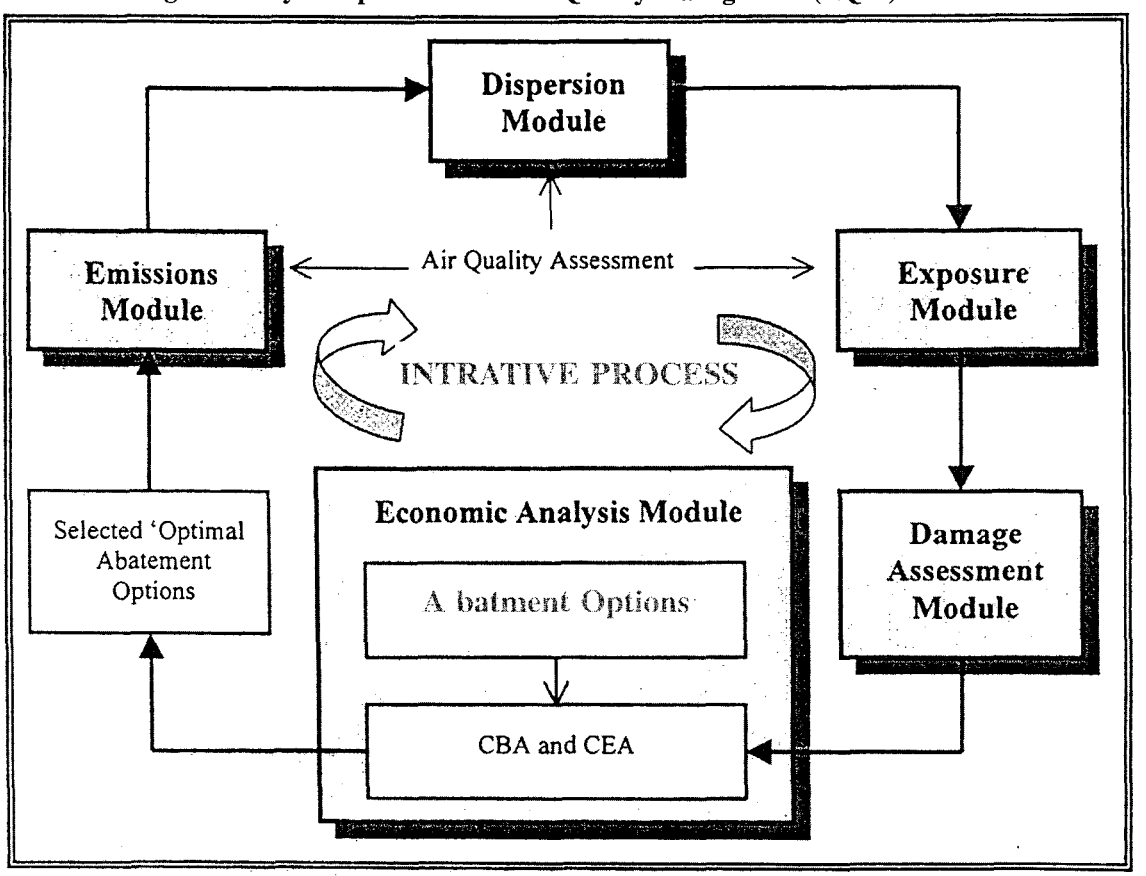
Figure 1 below presents a simple schematic of these elements, indicating the flow of data between them. In short, air quality assessments, environmental and human health damage assessments and abatement options assessment all represent inputs into a cost-benefit analysis (CBA) or cost-effectiveness analysis (CEA). The end result of these types of economic analysis is an 'optimal' control strategy in the form of an action plan, with prioritised abatement measures.

#### 2.1.1 Overview of AQM Programme

The purpose of the **emissions module** is to calculate a spatially distributed (or gridded) **emissions inventory** - for example, emissions can be distributed within a square-kilometre ($km^2$) grid system[2]. The gridded emissions inventory provides the input to a **dispersion model**, which allows air pollution concentrations in an area to be calculated as a function of time and location. It also provides a basis for calculating the effects of specific abatement measures on air quality. The inputs required for emission dispersion calculations include data on meteorology, topography, emission characteristics of the source, and data from the emissions module. The results from the dispersion calculations will be as detailed as the relevant input data

The **exposure** and **damage assessment modules** provide information on the impact of air pollution on human health and damage to ecosystems, buildings and

**Figure 1: Key Components of an Air Quality Management (AQM)**




*Source: Adapted from Shah, Nagpal and Brandon (1997)*

2 An emissions inventory consists of a list of all the relevant air pollutants in an area, broken down by activity sector - ; for example, traffic, industry, construction, etc. Such an inventory is usually based on factors such as the locations of industries, major roads and population distributions.

materials. The exposure, or impact, is defined as the product of the local air pollution concentration, within a grid square for example, and the number of susceptible objects within that location, whether that be people or buildings. The concentrations may be calculated as averages over differing periods, depending on the specific compound. If it is a substance which has an acute effect, such as $SO_2$, ozone, or CO, hourly averages are important; however, for pollutants such as total suspended particulates (TSP), lead, and persistent organic compounds, long term averages (e.g. yearly) are of greater interest.

The assessment of damage in physical terms requires knowledge of dose effect relationships and exposure distributions, for people, objects, buildings, monuments and vegetation.

The purpose of the **economic anabsis module** is assess the damaging effects of air pollution, and the costs of its abatement, in order to begin the process of developing an 'optimal' control strategy for air pollution management. CBA involves a comparison between the costs of abatement, and reduced damage. The monetary assessment of damage can be achieved using methods such as the valuation of productivity loss, defensive or averting expenditures, and willingness-to-pay (WTP) and willingness-to-accept (WTA) compensation for reductions in health risks. An abatement strategy may also be based on a CEA, for example, defined as "the least cost means to achieve a stated air quality target". In all cases, an abatement strategy should be based on analysis of the cost of each abatement measure versus the reduced damage costs (in which case we are employing CBA), or on an analysis of cost for each measure of air quality/ exposure improvement (in which case we are employing CEA).

We will now look briefly look at the main elements involved in the air quality assessment, and then focus our attention on the use of economic techniques within the economic analysis module.

## 2.2 Main Elements of Air Quality Assessment

### 2.2.1 Introduction

Air quality assessment requires an initial identification of all pollution sources, usually by means of monitoring and observation. This is followed by the quantification of the sources, and monitoring of the most important pollutants. Such assessments are intended to provide data, in terms of spatial and temporal resolution. They detail the key polluting compounds, the amount of pollution contributed by various sources in the form of an inventory of sources, and the level of population exposure using air quality modelling. Modelling comprises two parts; pollution concentrations (air quality), and exposure of the population.

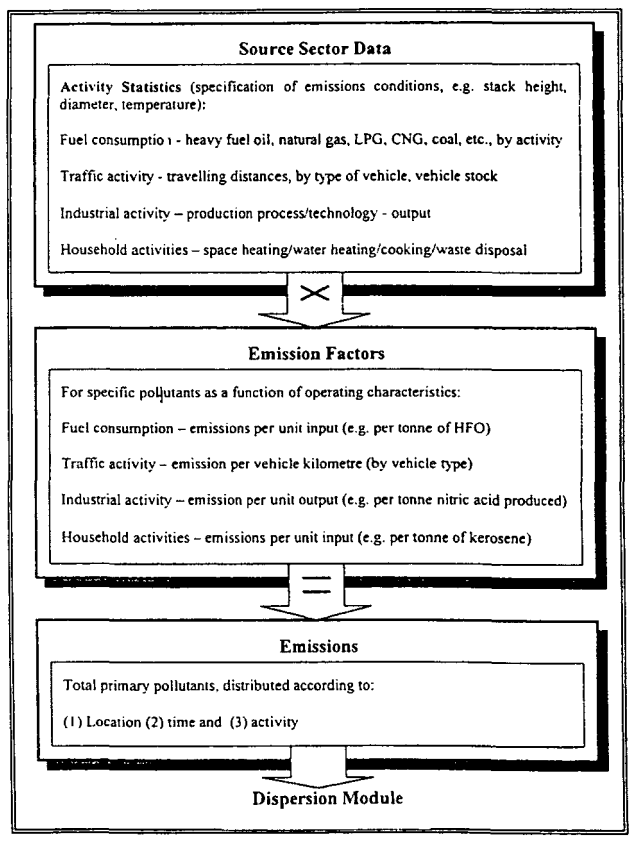
### 2.2.2 Emission Module

As mentioned above, the purpose of the **emissions module** is to calculate a spatially distributed emissions inventory. **An emissions inventory** is basically a compilation of all air polluting emissions in an area. The inventory will include the pollutants, their source (e.g. industry, traffic, domestic, etc.), and/or the energy carrier (e.g. oil type, gasoline, wood, etc.). It will also specify the location of each source, and the time variation in the emissions. The main elements of the emissions module are illustrated in Figure 2 below.

The functions of the inventory are to provide spatially-resolved source ; strength data on each pollutant for dispersion modelling, to predict environmental impacts, to contribute a basis for evaluating emission trends, and to assist in the formulation of air quality management policies.

An emissions inventory for an urban area will need to include the position of the sources of emissions (that is, the spatial distribution of the emissions within

**Figure 2: Elements of the Emissions Module (Emissions Inventory)**




*Source: Adapted from Shah, Nagpal and Brandon (1997)*

the urban area, with a recommended resolution of not more than one kilometre). Such an inventory also requires information on the source strength of all the activities resulting in emissions within a specified geographic area

***What Pollutants to Include?***

There are typically several hundred air pollution compounds in a source ; mix, which might include, for example, traffic, industry, fossil and bio fuel combustion, refuse burning, etc. Most of the compounds are either volatile or non-volatile organic compounds. The World Health Organisation (WHO) has developed air quality guidelines (AQG) for 23 of these compounds, 16 inorganic, and 7 organic, and the compounds on ; the WHO list should take first priority when selecting pollutants for monitoring. In addition, the following pollutants are also typically recommended for inclusion in an AQM programme (Shah, Nagpal and Brandon, 1997):

- Suspended particles, measured as $PM_{10}$ and $PM_{2.5}$; lead; CO; $SO_2$; nitrogen oxides (NO and $NO_2$) and volatile organic compounds (VOC) measured as the integral sum of many compounds.
- Compounds dependent on the specific source mix in the city, which may include specific compounds/elements within the SPM fractions (such as PAH) and VOCs such as benzene, toluene, xylene, and aldehydes.

***Compilation of an Inventory***

In general, the calculation of an emission inventory involves five steps:

1 ) establishing a list of point, area, and mobile sources;

2) contacting each point source operator to obtain quantitative process and emissions data,

3) compiling data on activity levels for area and mobile sources such as number of vehicle kilometres and speed, number of household boilers, monthly heating oil use for a district, etc.;

4) reviewing data to evaluate its suitability and developing alternative strategies or activity level indicators as necessary; and

5) processing the individual source and activity level data to provide a spatially-desegregated source inventory.

There is no 'one' correct way to develop an emissions inventory, and each individual situation will require decisions about the extent to which information is refined into categories. Stationary sources for example, may be treated either as a significant point sources, say an electricity generating station, or a component of an area source, such as a domestic boiler for heating one house, which might not produce significant emissions on its own, but when added to all the other boilers in a city ; suburb, might become important. The emission from the latter would be estimated from fuel consumed in the area for the purpose of domestic heating, a statistic which is usually easily obtained from fuel suppliers. Questionnaires are often used for data collection.

Recently, a number of institutions have developed procedures for the compilation of emissions inventories, including nomenclature, source categorisation and emissions factor compilation. The institutions include: USEPA (USEPA, 1985), IIASA (Amann, 1992), WHO (Economopoulos, 1993) and the European Commission (Bouscaren, 1992). IPCC (1996) also offers detailed guidance on the compilation of emission inventories, albeit for selected pollutants. The European Commission, through the European Topic Centre of Air Emissions of the European Environment Agency, offers a software tool for compiling an emissions inventory, but for a rapid compilation of a first estimate, IIASA and WHO citations may be used (Shah, Nagpal and Brandon, 1997).

**Emission factors** are an important element of inventory development. They relate pollutant emissions to a measure of activity level - for example, emissions from a car might be measured as grams of NOx per ; kilometre driven, while an industrial boiler's emissions might be in grams of NOx per kilowatt fired or per kg of fuel input. In general, where source strength data is not available, emissions (E) have to be estimated from a measure of activity - an **activity statistic** (A) and an **emissions factor** (F):

$$E=A\times F$$

An example application of the above formula in establishing a baseline emission inventory for motor vehicles in Delhi is shown in Box 1 below.

The results of similar calculations for each vehicle type and selected 'local' pollutant are shown in Table 2. This preliminary inventory of air ; emissions in Delhi reveals that motor vehicles annually emit 21,530.5 tonnes of $PM_{10}$, 100,030.8 tonnes of HC and 279,382.9 tonnes of CO.

Comprehensive tables of emissions factors for a very wide range of industrial, commercial and domestic activities, have been prepared by the USEPA (USEPA

**Box 1: Baseline Emission Inventory: Example of Transport Sector in Delhi**

Before examining measures to reduce emissions from motor vehicles (as part of a wider urban air pollution programme), a baseline emission inventory must be established against which the performance of the proposed abatement measures may be judged. The specific formula used to estimate air emissions in this example is given below. Basically, estimates of vehicle utilisation, by vehicle type (given in Table 2) are combined with relevant emission factors (given in Table 1).

$$\text{Emissions}\left(\frac{\text{tonnes}}{\text{year}}\right) = \text{emission factor}\left(\frac{\text{g}}{\text{km}}\right) \times \text{total vehicle kilometres}\left(\frac{\text{veh - km}}{\text{year}}\right) \times 10^{-6}\ \frac{\text{tonnes}}{\text{g}}$$

**Table 1: 'Local' Pollutants: Emission Factors by Vehicle Type**

| Vehicle Type | $PM_{10}$ (g/km) | HC (g/km) | CO (g/km) |
|---|---|---|---|
| Light duty gasoline vehicles | 0.25 | 1.50 | 9.50 |
| Taxis | 0.33 | 6.20 | 28.90 |
| Diesel buses | 2.00 | 2.10 | 12.70 |
| Diesel trucks | 2.00 | 2.10 | 12.70 |
| 3-wheel vehicles | 0.50 | 7.65 | 12.25 |
| 2-wheel vehicles | 0.50 | 5.18 | 8.30 |

***Source:*** Various in Xie et al (1998)

Taking three-wheelers for example, annual $PM_{10}$ and CO emissions are respectively given by:

$$1{,}684.4\frac{\text{tonnes}}{\text{year}} = 0.50\frac{\text{g}}{\text{km}} \times 3{,}368.7 \times 10^{6}\ \frac{\text{veh - km}}{\text{year}} \times 10^{-6}\ \frac{\text{tonnes}}{\text{g}} \quad \text{and}$$

$$41{,}266.6\frac{\text{tonnes}}{\text{year}} = 12.25\frac{\text{g}}{\text{km}} \times 3{,}368.7 \times 10^{6}\ \frac{\text{veh - km}}{\text{year}} \times 10^{-6}\ \frac{\text{tonnes}}{\text{g}}.$$

1985), WHO (Economopoulos, 1993), the European Commission (Bouscaren, 1992) and the IPCC (1996).

## 2.2.3 Dispersion Module

Emissions are dispersed and diluted in the atmosphere before they reach people, or other receptor points. The purpose of the *dispersion module* therefore,

**Table 2: Vehicle Utilisation in Delhi in 1996 and Estimates of 'Local' Pollutant Emissions by Vehicle Type**

| Vehicle Type | Total Annual Vehicle kilometres (mn veh-km/yr) | PM'o (tonnes) | HC (tonnes) | CO (tonnes) |
|---|---|---|---|---|
| Light duty gasoline vehicles | 6,739.9 | 1,685.0 | 10,109.9 | 64,029.1 |
| Taxis | 448.5 | 148.0 | 2,780.9 | 12,961.7 |
| Diesel buses | 1,878.4 | 3,756.8 | 3,944.6 | 23,855.7 |
| Diesel trucks | 4,848.0 | 9,696.0 | 10,180.8 | 61.569.6 |
| 3-wheel vehicles | 3,368.7 | 1,684.4 | 25,770.6 | 41,266.6 |
| 2-wheel vehicles | 9,120.5 | 4,560.3 | 47,244.2 | 75.700.2 |
| Total | 26,404.0 | 21,530.5 | 100,030.8 | 279,382.9 |

***Source:*** Delhi Transport Statistics in Xie et al (1998)

is to combine the knowledge of pollutant source strengths - that is the emitted quantities per unit time (from the *emissions module*) -with meteorological data and the knowledge of physical and chemical reactions in the atmosphere, in order to produce estimates of concentrations (air quality) at the receptor point. This allows the relationship between emissions and air quality to be established.

The concentrations of one or more of the pollutants in the air, as a function of time and space, typically requires either some or all of the of the following information:

- *Emission* and *source data*: including source location, source dimensions, height and exit diameter; exhaust gas exit temperature and velocity; mass flow of air pollutant from the source; time dependence of pollutant emission, duration and frequency as well as diurnal and seasonal variations.

**Box 2: Air Quality Assessment - Data Requirements**

The following broad categories of data are required for air quality assessments - that is, the **emissions module** and the **dispersion module:**

- measured concentrations of air pollution from representative sites, for all key pollutants, for the relevant averaging time (hour, day, month, year);
- emission data from all significant sources for all the key pollutants, spatial and temporal various, emission conditions (e.g. stack height, etc.);
- relevant measured meteorological data for the dispersion of emissions;
- topographical data

- *Wind* and *dispersion data:* hourly average wind speed and direction; description of atmospheric turbulence and stability (to estimate dispersion); mixing height: height of inversion layers; ambient air temperature (for plume rise calculations).

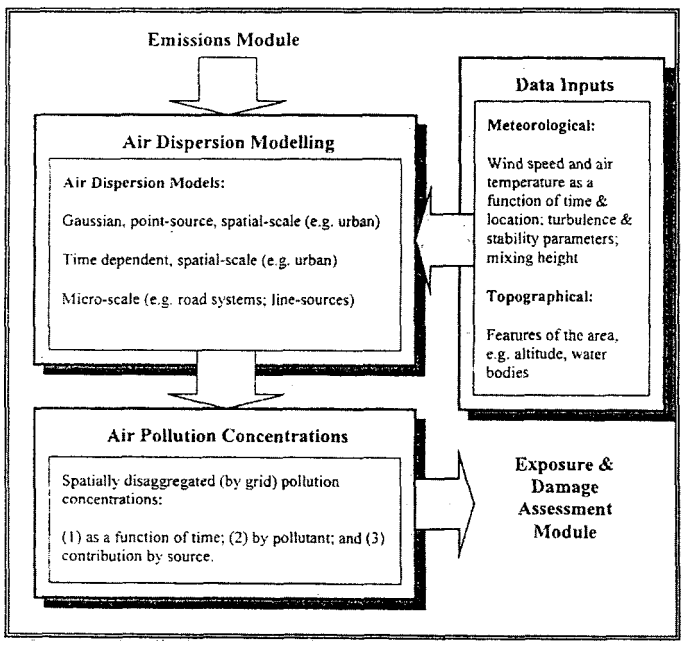
Different models have different degrees of sophistication, and accuracy is rarely better than plus or minus 20-30 percent of the actual value, unless the model is 'tuned' by the use of pollution monitoring data, where available.

# 3 ENVIRONMENTAL DAMAGE ASSESSMENT

## 3.1 Introduction

Air pollution imposes serious health-related social costs in the form of illness **(morbidity)** and premature death **(mortality).** Other physical impacts of air pollution include damage to ecosystems on which humans depend

**Figure 3: Elements of Dispersion Module**




***Source:*** Adapted from Shah, Nagpal and Brandon (1997)

for their livelihood, and damage to the physical infrastructure, as well as dirtying of facades and other physical material. This section considers procedures which can be used to assess air pollution damage, and methods which attempt to place a monetary value on those damages. In particular, we concentrate on placing a dollar value on air pollution damage to human health. Even where all the costs of air pollution cannot be precisely measured, it is still worth producing approximate or partial estimates, since they can at least help improve the case for pollution control measures.

Also, the discussion of damage assessment herein focuses on 'local' impacts (at an urban-scale). This means that regional or global impacts such as acidification, global warming and stratospheric ozone depletion, are not discussed. The impacts of climate change are discussed in a separate Chapter.

## 3.2 Impacts on Human Health

### 3.2.1 General Approach

In general, valuing the cost of pollution-induced morbidity or mortality is a two-step process[3]:

1) Cause-effect functions, which relate the level of air pollution to the incidence of particular health outcomes (types of health impact), must be established, from which changes in the number of additional cases of that health outcome can be estimated.
2) The cost of the pollution-induced health outcome can then be estimated by multiplying the estimated change in the number of cases by the corresponding economic (unit) value.

We will look at each of these steps in turn. The first step is undertaken by the exposure and damage assessment module, the main elements of which are shown in Figure 4.

### 3.2.2 Exposure and Damage Assessment Module

The cause-effect functions we seek relate information on changes in ambient environmental quality for different pollutants to different **health outcomes** (also referred to a **health endpoints**). The principle is that changes in ambient pollution levels for certain pollutants can be statistically related to observed changes in sickness/illness and premature death in a population. Through regression analysis, coefficients are estimated that are then multiplied by changes in ambient pollution concentrations and the population exposed.

In general, the estimated health outcome can be estimated by the : following relationship:

$$\Delta H_i = b_i \, x \, POP_i \, x \, \Delta A$$

where

$\Delta H_i$ = change in population risk of health outcome $i$,

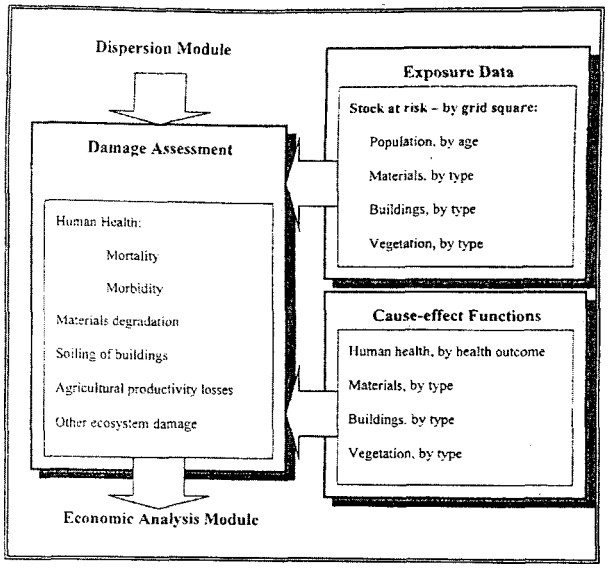
$b_i$ = slope from the cause-effect function for health outcome $i$,

$POP_i$ = population at risk to health impact $i$, and

$\Delta A$ = change in ambient pollutant under consideration.

3 This two-step process is also applicable to the valuation of damage to agricultural crops, buildings and materials.

**Figure 4: Main Elements of the Exposure and Damage Assessment Module**




*Source:* Adapted from Shah, Nagpal and Brandon (1997)

The $POP_i$ variable may include the entire population of a defined area e.g. a city, or it may focus on a particularly vulnerable sub-group. For example, **children**, who are particularly vulnerable to lower respiratory tract infections from PM and lowered Intelligence Quotient ratings from Pb, or **adult males,** who are particularly vulnerable to elevated blood pressure from Pb.

### *Cause-effect (Dose-response) Functions*

Substantial evidence of strong correlations between exposure to ambient air pollution concentrations and health risks has been gathered in developed countries, and for some developing countries (Working Group on Public Health and Fossil Fuel Consumption, 1997; Cropper *et al*, 1997; Ostro, 1994; Ostro *et al*, 1995 and 1996; Schwartz and Dockery, 1992; Dockery and Pope, 1993; and Pope *et al*, 1995). There are three main types of study which have been used to establish causal relationships between air pollutants and health outcomes, these are: (1) ***clinical studies*** based on human volunteers in controlled experiments; (2) ***toxicological studies*** in which effects on animals are extrapolated to humans in risk assessment studies; and (3) epidemiological studies at examine how the incidence of specific health outcomes vary with variations in ambient air pollution levels. Of these studies, the latter are thought to provide the most reliable measure of health effects under normal human exposure conditions. However, epidemiological studies are both expensive and timeconsuming. Most epidemiological studies are either **cross-sectional** studies, or **time series** studies.

**Time series** studies typically correlate daily variation in mortality and morbidity with air pollution concentrations on the preceding days, and hence primarily measure **acute** exposure to pollution. On the plus side, these type of studies do not need to control for a large number of other confounding factors - e.g. age, smoking, occupational exposure, etc. since these population characteristics remain basically unchanged.

> It is estimated, for example, that the average percent change in acute mortality rates per 10 $\mu g/m^3$ change in $PM_{10}$ (i.e. particulate matter less than 10 microns in diameter) ranges from 0.74 (Maddison *et al*, 1997) to 1.23 (Ostro *et al*, 1996). These two figures represent $b_i$ in the above equation

In contrast to time-series studies, **cross-sectional** analysis compares differences in health outcomes across several locations at a selected point in time. In principle, this allows one to capture both the **acute** and **chronic** impacts of exposure. One would therefore expect a more significant correlation between exposure and the incidence of health outcomes by comparison with time series studies.

> Pope *et al* (1995) estimate a 4.2 percent change in 'all-cause' mortality rates per 10 $\mu g/m^3$ change in $PM_{10}$.

We can further distinguish between two types of cross-sectional study (World Bank, 1997):

- **Retrospective (ecological) design** - which correlate variations in pollution levels with health outcomes across various locations at a single point in time. A problem with this type of study however, is whether potentially omitted and confounding factors have been adequately controlled.
- **Prospective (cohort) design** - in which a sample of the population is selected and followed over time. Since these studies work with individual level data, it is possible to control for potentially omitted and confounding factors. Cohort studies thus yield the most reliable results. However, they are relatively expensive to conduct, and as a result, there are few such studies in the literature.

In addition to the studies quoted above, which relate concentrations of ambient particulate matter to mortality, other epidemiological studies relate particulate matter to morbidity health outcomes, including respiratory hospital admissions (RHA), emergency room visits (ERV), restricted activity days (RAD) for adults, asthma attacks, symptom days and chronic bronchitis. Similar studies also exist for other air pollutants, e.g. sulphur

dioxide ($SO_2$), nitrogen oxides ($NO_x$), volatile organic compounds (VOC) and ozone ($D_3$) - see European Commission (1995 and 1998) and Hurley and Donnan (1997).

Table 3 below shows the slope of the cause-effect functions ($b_i$) for changes in $PM_{10}$ levels for selected morbidity health outcomes.

**Table 3: Morbidity Health Outcomes of 10 μg/m³ Change in $PM_{10}$**

| Type of Morbidity | Central Estimate | High Estimate |
|---|---|---|
| RHA1100,000 people | 2.94-12.0 | 15.6 |
| ERV/100,000 people | 235.4 | 342.5 |
| RAD/adult | 0.575 | 0.903 |
| Asthma attacks/per asthmatic | 0.326-0.65 | 2.73 |
| Respiratory symptoms/adult | 1.83 | 2.74 |
| Chronic bronchitis/100,000 adults | 61.2 | 91.8 |

***Source:*** *Adapted from World Bank (1997)*

An important issue which arises with the use of cause-effect functions, is that of **threshold levels.** That is, whether a concentration exists for a particular substance, below which no adverse health effects are measurable. Generally, air quality standards and guidelines for noncarcinogenic compounds are based on such threshold values, thus it could be argued that cause-effect functions should only be applied to that part of the concentration of a pollutant which exceeds the standard or guideline (Wijetilleke and Karunaratne, 1995). Alternatively, for studies of certain substances including all carcinogens, linear or log-linear functions have been produced, and threshold levels are not actually identified. In these cases, the use of threshold values is disputable, and should be included in the interpretation of results (Wijetilleke and Karunaratne, 1995).

### *Carcinogenic Impacts and Unit-risk Factors*

When calculations are being made of health outcomes, the occurrence of cancer should be given special attention. For **carcinogenic** compounds it is only possible to use risk or probabilities, since ethical objections to epidemiological experiments mean that no research results are available on the effects of carcinogenic compounds on human beings. The factors which are used to assess health damage from carcinogenic compounds are: **unit risk factors** which indicates the number of cases of cancer that may arise from a lifetime exposure to 1 μg/m³ of a carcinogenic compound. This unit risk factor is then multiplied by the actual human dose, to give an estimation of the number of potential cases of cancer. This calculation can be carried out using both average and maximum concentrations, and for both the maximum and averaged exposed part of the population (Shah, Nagpal and Brandon, 1997). Unit risk factors which ; have been set by international bodies are given in. It is normal to assume the cancer estimates are additive - that is, where simultaneous exposure to more than one carcinogenic compound occurs, each one should be calculated separately, and then summed.

**Box 3: An Example: Damage Assessment- Human Health**

Suppose that we are interested in estimating the change in acute mortality and ERV associated with a 50 μg/m³ reduction in ambient $PM_{10}$, which may represent the change in concentrations required to meet a local air quality objective.

Acute Mortality:

If 1 million people were exposed to this reduction, and the crude mortality rate of this population is 5 deaths per 1,000 people per year, the estimated health impact would ranges from 185 to 308 fewer deaths per year:

0.74 to 1.23 % per 10 μg/m³ change in $PM_{10}$ * 1/100 *5 deaths per 1,000 persons per year *1/1000 * 50 μg/m³ change in $PM_{10}$ *1/10 * 1,000,000 persons = 185 to 308 deaths per year.

Emergency Room Visits:

For emergency room visits (ERV) the calculation is as follows:

235.4 additional cases per year per 100,000 persons per 10 μg/m³ change in $PM_{10}$ * 1/100.000 * 50 μg/m³ change in $PM_{10}$* 1/10 * 1,000,000 persons = 1,177 fewer ERV per year.

**Table 4: Unit Risk Factors for Selected Carcinogenic Compounds**

| Compound | Unit Risk Factor[1] |
|---|---|
| Arsenic | $4.00 \times 10^{-3}$ |
| Asbestos | $1.00 \times 10^{-5/6}$ |
| Benzene | $4.00 \times 10^{-6}$ |
| Cadmium | $1.80 \times 10^{-3}$ |
| Diesel fuel (high duty) | $1.00 \times 10^{-4}$ |
| Formaldehyde | $1.30 \times 10^{-5}$ |
| PAH | $9.00 \times 10^{-2}$ |

***Source:*** WHO (1997) and USEPA (1990) in Shah, Nagpal and Brandon (1997)

**Notes:**

[1] For lifetime (=70 years) exposure to I μg/m³

***Transferring Cause-effect Functions to Developing Countries***

For many health impact assessments in developing countries there have been no epidemiological studies carried out. In these instances, results from such studies carried out in industrialised countries are typically used. Dose-response functions generated by these studies are applied to ambient pollutant levels to get estimates of selected health effects.

When dose-response estimates from industrialised countries are applied to populations in developing countries, there are a number of assumptions which are often made (Shah, Nagpal and Brandon, 1997)

- Baseline health status is the same in industrialised and developing countries. People in developing countries may be more susceptible to adverse health outcomes due to poorer sanitary conditions, thus it may be right to assume the general level of health is worse. Less healthy individuals are more susceptible to pollution-induced diseases, so a dose-response function for a city like Delhi may be significantly steeper than one for Athens. It is difficult, however, to assess if the dose-response functions will be steeper for a pollutant in a developing country unless they have been estimated specifically for that country.

- Distribution of causes of death are the same as the industrialised countries. A study in Delhi, India, found that less than 20 percent of all deaths were attributable to cardiovascular and respiratory disease, compared to 50 percent in the US (Cropper *et al*, 1997) Thus even if people in Delhi have the same physiological reactions to a reduction in PM as people in the US, the impact on total mortality of a reduction in that pollutant, will be less.

  Most epidemiological studies look at the impact of a change in air pollution on the percentage change in total deaths or total cases of a particular morbidity outcome. Hence, to take cause-effect coefficients from City A and apply them City B, one must know the baseline number of cases of the health outcome in City B, and not just the population of City B. By expressing the cause-effect coefficients on a per person basis we implicitly incorporate the baseline number of cases in City A into the analysis. Given the above, estimated health outcomes in City B will be over/underestimated to the extent that baseline health outcomes differ from City A.

- Levels and duration of exposure. In developing countries, individuals generally spend a greater proportion of their time outside, or in buildings which are not air conditioned. They are therefore exposed to higher concentrations of pollutants over a longer period of time. This is exacerbated by the fact that ambient concentration levels are much higher than in industrial countries, both factors which combined could cause an underestimate of benefits if dose-response estimates are transferred.

Another problem, which arises when using dose-response functions generally, is that they do not account for complex physiological reactions which occur as a result of exposure to several pollutants simultaneously. That is, respiratory illness may be an outcome of simultaneous exposure to PM, $SO_2$ and several other pollutants. Current dose-response functions do not capture such complex causal relationships.

The usefulness of a cause-effect relationship is not only dependant on its relevance and reliability, but also on its transferability to other situations and conditions. This requires information on other factors which may influence the relationship, such as other pollutants, temperature etc. It may not always be possible to actually include the influence of these co-factors in the cause-effect relationships, but conclusions are usually possible in terrns of the direction of their influence, that is, whether it is positive or negative. For example, temperature, age and marital status have been shown to exert a positive, negative and positive influence respectively on the relationship between TSP and work loss days (WLD) (Ostro, 1983).

Despite all this, however, dose-response functions estimated for developing countries such as Chile and India, have been shown to correspond well with those from industrialised countries (see Cropper et al, 1997 and Ostro et al, 1996).

## 3.3 Economic Analysis Module

The economic analysis module comprises to main elements, as shown in Figure 5 below:

1) damage valuation and

2) cost (-effectiveness) analysis of abatement options.

We will consider each in turn. The end result of these types of economic analysis is an 'optimal' control strategy in the form of an action plan, with prioritised abatement measures.

### 3.3.1 Estimating the Unit Cost of Health Outcomes

The assessment of the damage caused to human health by pollution is clearly important, however it is also controversial. Although there is general agreement on the fact that pollution related illness results in financial and non-financial welfare losses, putting a monetary value on this is both uncertain and difficult, indeed estimating the monetary value of the risk of excess mortality is often the most difficult part of health damage assessment. The task is made even harder for developing countries, due to the lack of dose-response functions, or willingness-to-pay data. As noted above, the estimates are essential, however, in order for the costs and benefits of various air pollution abatement strategies to be calculated.

The health impacts of air pollution have been divided into sickness or morbidity, and premature deaths or mortality. Once the physical health impacts have been estimated, a monetary value needs to be placed on them, to give the economic benefits of a reduction in air pollution. The two main approaches economists use to value health outcomes (mortality and morbidity) are based on either:

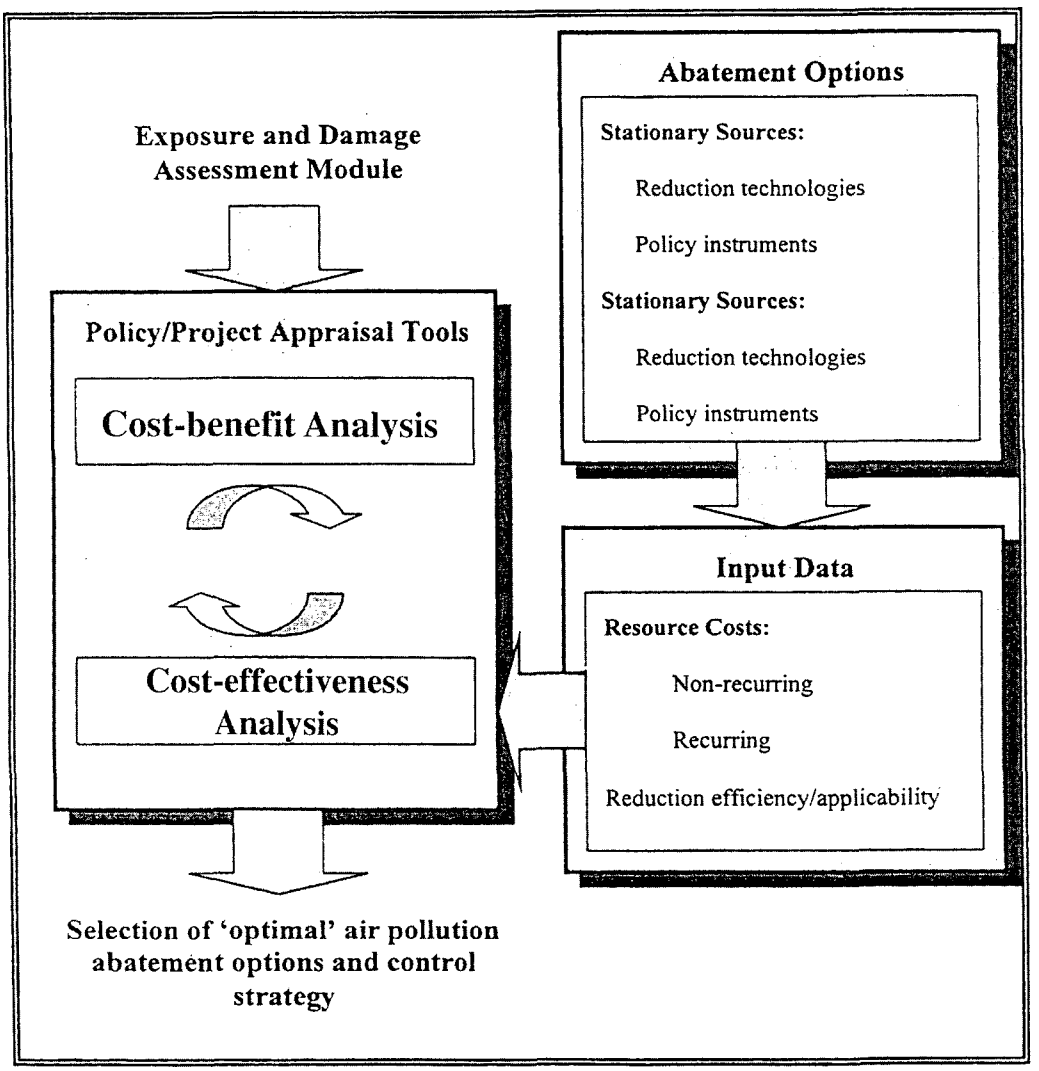
- individual preferences (WTP or WTA compensation) or
- resource or opportunity costs (Freeman, 1993).

Individual WTP/WTA compensation-based estimates are typically derived through **averting expenditure, wage-risk** or **contingent valuation** studies. Measures of the value of a reduction in a health risk valued in these ways are preferable from an economists perspective. However there are few studies which estimate WTP/WTA-based data for developing countnes. In the absence of such studies, more 'direct' measures are commonly used to value morbidity and mortality - these are the **cost of illness** approach for morbidity, and the **human capital approach** for mortality.

*Morbidity Valuation*

**The cost-of-illness** (COI) is the name economists

**Figure 5: Main Elements of the Economic Analysis Module**




*Source: Adapted bom Shah, Nagpal and Brandon (1997)*

**Box 4: Valuation of Morbidity Outcomes Using the COI Approach**

The economic costs of a RHA (albeit, in the high cost US medical care sector) is computed as follows using the COI approach (Ostro, 1992):

| | |
|---|---|
| the average stay | 10.13 days per case |
| the average out-of-pocket expenses | US$ 2,655 per day |
| the lost day wage rate | US$ 125 per day. |

So, each RHA is assumed to cost

US $28,164 per case = (10.13 days per case * US$ 2,655 per day) + (10.13 days per case * US$ 125 per day).

give to the resource/opportunity cost-based approach to estimating the costs of morbidity. With the COI approach, as the name implies, the cost of illness is measured as the sum of (see Box 4 for details)

the direct out-of-pocket expenses resulting from sickness (e.g. medicine, doctor and hospital bills)

+

any associated opportunity costs (e.g. loss of earnings resulting from the sickness).

There are several advantages of the cost of illness approach, including that the necessary data is relatively easy to obtain, and estimates made can be effectively communicated since they are based on costs which people easily understand. In addition, primary data on the duration of illness can be gathered from health centres, hospitals, or departments of health, while the costs of treatment can be gained from local public and private health systems, or estimated based on the observed costs in other countries.

With respect to the latter, treatment costs in developed countries (primarily the US) have been extrapolated for use in studies in other countries by adjusting them downwards by a factor equal to the ratio of average daily wages. Alternatively, US health costs can be adjusted to the ratio of one or more easily measurable health cost, for example the cost of a doctor consultation (see Brandon and Hommann, 1995).

There are basically two ways of estimating WTP/WTA-based measures of morbidity costs. It can be inferred either from an individual's **averting behaviour,** or from an explicit expression of what the individual is WTP/WTA (as compensation) to avoid (accept) the illness **(contingent valuation)**.

- The **averting behaviour** approach is based on the idea that the time and money an individual spends to avoid exposure to air pollution, or to avoid illness, gives an indication of the lower bound value they attach to avoiding illness. The techniques most frequently adopted to avoid exposure to air pollution, include staying inside or in air conditioned environments, using face masks, or moving to suburban or 'cleaner' areas to live. Estimating the various types of averting behaviour however, can be difficult due to a lack of available data.
- The **contingent valuation** approach involves the use of individual survey questionnaires, which are designed in such a way as to elicit individual valuations of goods and services which are not traded in the markets. Individuals are essentially asked how much they would be willing to pay to avoid a certain symptom or illness (see Box 5).

In the European Union (EU), for example, there have been three new projects using the contingent valuation method for valuing morbidity impacts from air pollution, one in Helsinki (Finland), one in Strasbourg (France) and Kehl (Germany), and a 5-country study - looking at Amsterdam, Lisbon, London and surroundings, Oslo and Vigo. The latter represents the most extensive of the three studies.

The WTP results from the 5-country study are given in Table 5.

The CSERGE *et al* (1999) study also provides a list of productivity loss from absenteeism (see Table 6) and health service costs of going to emergency rooms and hospitalisation (see Table 7) in selected EUcountries. These COI measures have to be combined with the individual welfare loss (WTP) estimates presented in

**Box 5: Example CV Question**

The following contingent valuation survey question was asked in Taiwan in order to gain information about an individual's recent illness episode and associated costs (Alberini *et al*, 1995):

*Suppose that you were told that, within the next few days, you would experience a recurrence of the illness episode you have just described for us. What would it be worth to you - i.e. how much would you be willing to pay- to avoid the illness episode entirely.*

*Remember that you are paying to eliminate all your pain and suffering, your medical expenditure, the time you spent visiting the doctor or clinic, your missed work leisure or daily activities. Bear in mind, if you pay to completely avoid being ill this time, you have to give up some other use of this money. For example, you may reduce your expenditures for entertainment or education.*

**Table 5: WTP Values for Selected Health Outcomes (UK£[1997] per episode)**

| | 'Pooled' Data | Netherland | Norway | Portugal | Spain | UK |
|---|---|---|---|---|---|---|
| Hospital | 306 | 283 | 301 | 300 | 426 | 164 |
| Casualty | 158 | 128 | 239 | 185 | 146 | 131 |
| Bed | 97 | 71 | 119 | 88 | 113 | 83 |
| Cough | 27 | 28 | 36 | 28 | 39 | 20 |
| Eyes | 35 | 40 | 31 | 70 | 53 | 14 |
| Stomach | 35 | | | 61 | | 26 |

*Source: CSERGE et al (1999)*

Table 5 to provide a measure of the 'full' economic cost of the selected morbidity outcomes.

For example, for an RAD we can use the BED value, which is 148 (i.e. £97 * 1.52 per £). Since this BED value is to avoid an episode lasting three days, the estimate has to be divided by three (i.e. 148 ÷ 3 = 49). To this should be added the EU average *per diem* productivity loss (i.e. 55). Thus, one RAD can be valued at 104 (NB no health service costs are incurred in this case).

It can be expensive to carry out large scale surveys of this kind, so it is not commonly done. The study cited in Box 5 above, however, did show that for some countries it is valid to use the **benefits transfer** approach, whereby costs estimated for one country, in this case the US, can be adjusted and applied to another, in this case Taiwan. The use of a conversion factor of the ratio of average household income in Taiwan to the same average income in the US, resulted in values which fell within a the 95 percent confidence interval estimated by the WTP survey.

***Mortality Valuation***

As with morbidity outcomes, there are two approaches, which are generally used to value premature mortality: (1) the human capital approach - which values an individual's life according to the net present value of their productivity, and (2) WTP/WTA-based methods - which measures the value which society places on individual distinct from an individual's wage-earning capacity.

When the COI approach is extended to measure the cost of premature death, it is commonly known as the **human capital** (HC) **approach**. The logic underpinning the HC approach goes someing like is: while you are living and healthy your value to society is expressed in terms of the output you generate from working (please note that 'working' is very broadly defined here). The value of your labour, i.e. the wage you earn, is usually assumed to be representative of the value of your working time (and output produced). Based on this, the HC approach assumes that over your entire life the discounted expected value of your future wage stream constitutes the minimum value society attaches to your life at this moment.

Hence, in the case of premature death, society will lose the value of your output. It follows then, that if you die prematurely, the **minimum** cost imposed on society is represented by the present value of your projected future earnings foregone. As such the HC approach considers humans as units of capital that produce goods and services for society.

Some average values of HC-based mortality costs for different age groups in the US are provided in Table 8.

**Table 6: Estimated per diem Productivity Costs from Abenteeism (€[1997])**

| Country | Productivity Cost |
|---|---|
| Belgium | - |
| Germany | 68.46 |
| Greece | 18.06 |
| Spain | 46.03 |
| France | - |
| Ireland | 48.64 |
| Italy | 53.56 |
| Luxembourg | 121.11 |
| Netherlands | 61.34 |
| Austria | - |
| Portugal | 23.30 |
| Finland | - |
| Sweden | 54.07 |
| United Kingdom | 53.35 |
| **EUAverage** | 54.97 |

*Source: CSERGE et al ( 1999) .*

**Table 7: Health Service Costs (local currencies)**

| Country | Emergency Room (cost per visit) | Hospitalisation (cost per inpatient day) |
|---|---|---|
| Belgium | - | BF 8,000 (1995 prices) |
| France | FF 150 (1996 prices) | FF 2,020 (1994 prices) |
| Germany | DM 37.8 (1996 prices) | DM 515.47 (1995 prices) |
| Italy | L 31,075 (1994 prices) | - |
| Netherlands | - | G 706 (1995 prices) |
| Spain | P 14,174 (1995 prices) | - |
| United Kingdom | £ 52.24 (1997 prices) | £187 (1996 prices) |

*Source: CSERGE et al (1999)*

**Table 8: Human Capital and Mortality Costs by Age Group in the US**

| Age Group | Average Life Years Lost (= to Average Life Expectancy) (Years) | Average Mortality Cost (= to Present Value of Average Earnings Stream Foregone) (US$ 1992 Values) |
|---|---|---|
| Under 5 years | 75 | 502,421 |
| 5 - 14 years | 68 | 671,889 |
| 15 - 24 years | 57 | 873,096 |
| 25 - 44 years | 42 | 785,580 |
| 45 - 64 years | 25 | 278,350 |
| 65 years and over | 10 | 22,977 |

*Source: The US Institute for Health and Aging in World Bank (1997)*

The HC approach provides a useful methodology for obtaining country specific mortality costs in the absence of WTP/WTA-based estimates, which do not exist for many developing countries.

In contrast to the approach described above which measures tangible changes in productivity, the WTP approach incorporates many intangible aspects, for example pain, suffering, and discomfort. The valuation of a marginal reduction in the risk of dying from air pollution related disease, is taken as the amount which people are willing to pay to marginally reduce that risk. The WTP for reducing mortality risks can be measured using one of two basic methods:

- **Compensating wage differential** - which assumes that, other things being equal, workers in riskier jobs must be compensated with higher wages than those in safer jobs. For example, a wage premium of US$ 100 may be necessary to get workers to undertake a job with a 1-in-10,000 chance of increased mortality. This yields an implicit 'value of a statistical life' (VSL) of US$ 1 million. Compensating wage studies have produced VSL estimates which range from US$ 1.9 to US$ 10.9 million. Box 6 presents a rather simplistic example of the use of this approach.
- **Contingent valuation** - which involves asking people directly what they would be willing to pay for reduced risk of increased mortality. Contingent valuation studies have estimated VSL at anything from US$ 1.2 to US$ 9.7.

Although ideally wage differential and CV-based estimates from country specific studies should be used, these are rarely available for developing countries, and estimates from industrialised countries are often used (transferred). However, as numerous authors have noted, most compensating wage differential and CV studies measure compensation of mortality risks for individuals who are an average of 40 years of age. Assuming that an individual is productive until 65 years of age, and will live into their 70's, this values about 25 (productive) to 35 (total) lifeyears lost. Deaths from exposure to air pollution typically occur at an age over 65 - at least in developed countries - which implies much fewer life years lost. This therefore suggests that an 'adjustment' is required when applying these estimates to deaths related to air pollution exposure - one way of dealing with this is illustrated in Box 7.

**Box 6: Wage Differential Approach for Bangkok - Approximation**

A World Bank study carried out in 1993 in Thailand considered the value of using 'back-of-envelope' estimates of wage differentials for Bangkok. It was estimated that for the 149,000 construction workers in the city in 1992-3, the wage premium was 15 to 20 percent, compared to wages in less risky industries.

Approximately 200 workers were assumed to die per year on the job, so death risks were estimated to be 13.4 per 10,000. Assuming an average wage of US$ 3,000 per year, construction workers would earn a US$ 450,600 risk premia. The corresponding VSL estimates range between US$ 336,000 and US$ 448,000 per year, i.e. US$ 450 ÷ (13.4 ÷ 10,000) to US$ 600 ÷ (13.4 ÷ 10,000).

**Box 7: Approximate Valuation of Life Years Lost**

A study in Santiago, Chile, which was carried out by the World Bank, estimated an average loss of 12.9 discounted productive years, based on two assumptions: that the distribution of deaths due to air pollution is identical to that due to pneumonia; and that people who die prematurely from air pollution would have lived as long as others in their age group (Sanchez, 1994).

The life years lost are then multiplied by the average wage rate in the country, to give an approximate HC-based value of premature death. For the Chile study, estimates of the value of a premature death from air pollution worked out at US$ 45,000, based on an average annual wage rate of US$ 3,482.

Alternatively, as recommended by a UK government working group on the valuation of the health impacts of air pollution (The Stationary Office, 1999), we could apply adjustment factors to the baseline WTP-based VSL. In recognition of the fact that epidemiological studies only report data on the whole population at risk over 65, the group suggests using an adjustment factor

of 70 percent, weighted to take into account of the average distribution of ages within the over 65 age group in the general population. Further adjustments are made to reflect the life expectancy of the affected individual.

Markandya (1997) directly derives the value of a life year lost (VLYL) from a baseline VSL using the following formula:

*Where a is the age of the person whose VSL has been estimated, Pi is the conditional probability of survival in year i, having survived to year i-l. T is the upper age bound and r is the discount rate.*

Estimates of (male) survival probabilities for the EU population obtained from Eurostat were used to estimate the VLYL for individuals aged from 35 to 45 - using a discount was in the range 0 to 3 percent and a baseline VSL of €$^{2000}$ 3.36 million. The mid-point estimates, which we adopted in the 'ExternE' project for the 'standard' acute VLYL, were €$^{2000}$104,760 (at a O percent discount rate) and €$^{2000}$165,700 (at a 3 percent discount rate). These estimates are then used to value the years of life lost as a result of acute mortality.

As a general rule, when valuing mortality costs in developing countries, we would take locally derived HC-based values as a lower bound estimate (since they are more readily obtained), and transfer[4] WTP/WTA-based values from the EU and US to obtain an upper bound estimate.

It is worth stressing that all valuation exercises only capture partial benefits. For example, valuation of the health effects of air pollution using the techniques above will not capture lifestyle changes, sub-clinical physical and mental discomfort caused, or some of the more chronic intangible effects. An example of the latter is that children might not be allowed to go outside to play due to the poor air quality, and this could affect their quality of life, physical activity and have implications for long term well-being; poor air quality may also cause eye irritations which affect the ability to read.

Many of the existing studies also do not include the costs of illness in calculations of the value of premature mortality from exposure to air pollutants. In reality however, people who die from respiratory diseases often suffer prolong periods of illness, so including the costs of illness would actually give a more accurate estimation of the costs of death from air pollution.

***Combining the Predicted Changes in Health Outcomes with the Economic Unit Values***

The final step is to combine the predicted changes in health outcomes - as derived through cause-effect functions (**exposure** and **damage assessment module**)- with the economic unit values estimated with one of the above outlined techniques, in order to estimate the benefits of reduced levels of air pollution on human health. This is accomplished as follows:

$$C/B_i = EUV_i \times \Delta H_i$$

where

$C/B_i$ = the economic cost (benefit of damage avoided) of associated with the change in health outcome $i$,

$EUV_i$= the economic unit value (or 'price') of health outcome $i$, and

$\Delta H_i$ = the change in population risk of health outcome $i$ .

An example is given in Box 8.

**Box 8: Example Calculation of Mortality Costs Based on the HC Approach**

Previously we estimated the change in **acute** mortality associated with a 50 $\mu g/m^3$ reduction in ambient $PM_{10}$ to range from 185 to 308 fewer deaths per year. In general, the majority of cases of acute mortality associated with exposure to $PM_{10}$ occur in the elderly - i.e. those individuals over the age of 65. Therefore, based on a HC-based mortality cost, every premature death avoided can be valued at US$ 22,977 (Table 8). The total benefit, in terms of mortality costs avoided, of the 50 $\mu g/m^3$ reduction in ambient $PM_{10}$, is thus given by:

US$ 22,977 per premature death avoided * 185 to 308 fewer deaths per year = US$ 4.3 to 7.1 million per year.

## 3.3.2 Other Damage Assessment

Air pollution also has potential adverse effects on buildings, crops and 2 vegetation, and on an area's aesthetic value, thereby possibly affecting e tourism. The direct costs of cleaning soiled buildings is relatively straight forward to assess and value, as is the cost of repairing and maintaining damaged buildings. Less easy to value is the damage to buildings of, historic and cultural significance, since these include not only financial costs, but the probably irreversible losses to the cultural heritage of a city. In addition to losses of intrinsic value inherent in unique cultural monuments or property, there may be adverse impacts on tourism. ' Contingent valuation techniques can be used for individual cases to give; an estimate of the damage to cultural monuments.

[4] After making appropriate adjustments for differences in real income.

### 3.3.3 Cost of Abatement Options

## Introduction

In order to achieve the reductions in air emission that will ultimately yield the environmental and human health benefits eluded to above, abatement ' measures will need to be put in place. In pursuit of an 'optimal' control strategy, it is important to analyse the costs and benefits (emission 'reduction potential) of all technical and policy measures to combat air pollution relevant to the problem at hand. In general, this involves three ' steps:

1 ) Identify potential technical and policy measures to abate air emissions.
2) Identify and assess the costs of each of the identified measures.
3) Assess the effectiveness of each of the identified measures in reducing the target pollutants.

In other words, some form of least-cost or cost-effectiveness analysis (CEA) is required. We covered CEA in a previous Chapter, and therefore will not repeat ourselves herein. Instead, we will present an example of how CEA may be used to identify the most 'cost-effective' technical options to reduce emission of local pollutants from two-stroke three- wheelers in Delhi. This example follows on from that presented in Box 1 above.

The results of the cost analysis presented below could then be combined with a damage valuation assessment to determine the overall net benefit of implementing the identified set of 'least-cost' abatement measures.

### *Example of CEA of Abatement Options*

In this section we examine a number of possible technical measures to control emissions from two-stroke engine three-wheelers in Delhi. Specifically, we determine the cost-effectiveness of the selected set of measures in abating $PM_{10}$ equivalents[5]. This data could then be used to identify the least-cost path for meeting a particular emission reduction target and, thereby, an ambient air quality target for Delhi.

Energy and Environmental Analysis (1998) and Chan and Weaver (1994) examined a number of technical options for controlling emissions from two-stroke three-wheel vehicles. Among the measures examined are vehicle emission standards, inspection and maintenance programmes (I & M), vehicle retrofitting, engine rebuilding, cleaner fuels and lubricating oils, and alternative fuels. The Environmental performance of a selection of these measures, in terms of their effectiveness in reducing baseline emissions of $PM_{10}$, HC and CO, is provided in Table 9. The estimated change in fuel economy associated with each measure is also provided.

Electronic (capacity-discharge) ignition systems, for example, reduce baseline emission of $PM_{10}$, HC and CO by 20, 20 and 10 percent, respectively. The corresponding annual emission savings are given by (assuming that the entire fleet is retrofitted - see Table 2):

$$336.9\frac{\text{tonnes PM10}}{\text{year}} = \left(0.50\frac{\text{g PM10}}{\text{km}} \times 3{,}368.7 \times 10^6\frac{\text{veh - km}}{\text{year}} \times 10^{-6}\frac{\text{tonnes}}{\text{g}}\right) \times 20\%$$

$$5{,}154.1\frac{\text{tonnes HC}}{\text{year}} = \left(7.65\frac{\text{g HC}}{\text{km}} \times 3{,}368.7 \times 10^6\frac{\text{veh - km}}{\text{year}} \times 10^{-6}\frac{\text{tonnes}}{\text{g}}\right) \times 20\% \text{ and}$$

$$4{,}126.7\frac{\text{tonnes CO}}{\text{year}} = \left(12.25\frac{\text{g CO}}{\text{km}} \times 3{,}368.7 \times 10^6\frac{\text{veh - km}}{\text{year}} \times 10^{-6}\frac{\text{tonnes}}{\text{g}}\right) \times 10\%.$$

The annual weighted emission reduction, in terms of $PM_{10}$ equivalents, is thus:

$$605.0\frac{\text{t PM10 eq.}}{\text{year}} = 336.9\frac{\text{t PM10}}{\text{year}} \times 1 + 5{,}154.1\frac{\text{t HC}}{\text{year}} \times 0.04 + 4{,}126.7\frac{\text{t CO}}{\text{year}} \times 0.015.$$

The installation of an electronic ignition requires some design work. The retrofit cost of a typical system to the vehicle manufacturer ranges from $12 to $15; however, the retail price in India could be 2 to 2.5 times as much (World Bank, 1999). Assuming that the required investment expenditure is $35 per vehicle, which may be financed by a five-year loan at an interest rate of 10 percent, the annual capital cost of the electronic ignition system is computed as follows[6]:

$$A_t = C_0\left[\frac{r(1+r)^n}{(1+r)^n - 1}\right] = \$35 \times \left[\frac{0.10(1.10)^5}{(1.10)^5 - 1}\right] = \$35 \times (0.2638) = 9.23\frac{\$}{\text{veh} \cdot \text{yr}} \quad \text{I}$$

The electronic ignition system also improves fuel economy by 2 percent. Assuming that the price of gasoline in Delhi is $0.71 per litre, the ignition system will yield annual savings of about $30 per vehicle per year, i.e.

$$29.23\frac{\$}{\text{veh} \cdot \text{yr}} = 0.71\frac{\$}{\text{litre}} \times \left[\left(42{,}000\frac{\text{km}}{\text{veh} \cdot \text{yr}} \div 20\frac{\text{km}}{\text{litre}}\right) - \left(42{,}000\frac{\text{km}}{\text{veh} \cdot \text{yr}} \div \left(20\frac{\text{km}}{\text{litre}} \times 1.02\right)\right)\right]$$

The total annual incremental cost of the electronic ignition system is thus:

5 See Eskeland (1994) for an explanation of the use and determination of $PM_{10}$ equivalents.

6 Where $A_t$ is the equal annual amount to be received each year over an $n$ year period in order to recover CO with interest at $r$ percent.

**Table 9: Environmental Performance of Selected Control Measures forTwo-stroke Three-wheelers: Percentage Change in Baseline Emission Factors**

| Abatement Measure | $PM_{10}$ (%) | HC (%) | CO (%) | Fuel Economy (%) |
|---|---|---|---|---|
| Baseline emission factors (g/km) | 0.50 | 7.65 | 12.25 | 20km/l |
| Periodic I & M | -20 | -20 | -35 | +5 |
| Electronic ignition | -20 | -20 | -10 | +2 |
| Pumpless lubrication system | -80 | 0 | 0 | +3 |
| Engine rebuilding | -30 | -40 | -SO | +5 |
| Phaseout of old three-wheelers | -100 | -100 | -100 | - |
| Fuel/oil remix | -30 | -10 | 0 | +3 |
| Convert to CNG vehicles | -80 | +10 | -88 | n/a |
| Replace with 4-stroke vehicles | -80 | -90 | -100 | +20 |

*Source: Energy and Environmental Analysis, Inc (1998) and others in Xie et al (1998)*

| | | |
|---|---|---|
| **Annual Capital Cost:** | | |
| Annual recovered capital | +$9.23 | |
| Sub-total | = | +$9.23 |
| **Net Recurring Costs:** | | |
| Energy | $0 | |
| Labour | $0 | |
| Materials | $0 | |
| Offsetting benefits | -$29.23 | |
| Sub-total | = | -$29.23 |
| **Total Annual Cost** | = | -$20.00 |

Retrofitting the entire fleet of two-stroke three-wheelers (80,208 vehicles in 1996) would therefore yield annual incremental savings of just over $1.6 million:

$$-1{,}604{,}160\ \frac{\$}{\text{year}} = -20.00\ \frac{\$}{\text{veh} \cdot \text{yr}} \times 80{,}208\ \text{vehicles}.$$

The cost-effectiveness of the ignition system, as measured by the incremental cost (in this case saving) per unit of weighted emission reduction, is computed as:

$$-2{,}652\ \frac{\$}{\text{t PM10 eq.}} = -1{,}604{,}160\ \frac{\$}{\text{year}} \div 605.0\ \frac{\text{t PM10 eq.}}{\text{year}}.$$

Similar calculations have been performed for each of the emission control measures listed in Table 9. The results are shown in Table 10 below, where measures are ranked in order of cost-effectiveness[7]. From the table, one can easily see that 6 of the 8 measures considered represent so-called 'win-win' measures, i.e. measures which are justified economically in their own right because they yield cost savings to vehicle owners. These cost savings arise because reductions in annual fuel (and maintenance) costs more than offset annualised investment expenditures. Periodic I & M and the phase-out of old three-wheelers on the other hand, places an additional cost burden on vehicle owners of $15.71 and $233.68 per vehicle respectively.

**Table 10: Cost effectiveness of Selected Measures in Abating 'Local' Pollutants**

| Abatement Measure | Percentage of Fleet Affected (%) | Annual Emission Savings (t $PM_{10}$ eq./year) | Net Annual Cost ($ million) | Net Abatement Cost ($/t $PM_{10}$ eq.) |
|---|---|---|---|---|
| Convert to CNG vehicles | 30 | 536.7 | -6.22 | -11.590 |
| Replace with 4-stroke vehicles | 10 | 289.5 | -2.73 | -9,431 |
| Fuel/oil remix | 30 | 182.5 | -0.54 | -2.959 |
| Electronic ignition | 100 | 605.0 | -1.60 | -2,652 |
| Engine rebuilding | 30 | 368.1 | -0.84 | -2.282 |
| Pumpless lubrication system | 50 | 673.7 | -1.39 | -2,063 |
| Periodic I & M | 100 | 759.7 | 1.26 | 1,659 |
| Phaseout of old three-wheelers | 17 | 566.8 | 3.05 | 5,381 |

*Source: Adapted from Xie e'al (1998)*

*Notes:*

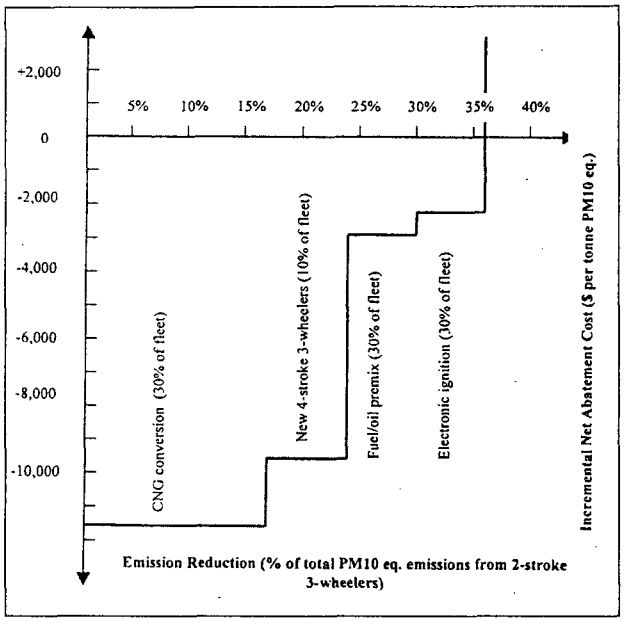
*[1] The percentage of the entire fleet affected by the technology is based on preliminary judgements by Xie e' al (1998). They are therefore subject to change and are purely illustrative.*

Using the results presented in Table 10, one can begin to construct a preliminary abatement cost curve. An example is shown in Figure 6. The curve in the figure indicates that just under 36 percent of the total emission of $PM_{10}$ equivalent from two-stroke engine three-wheelers may be abated at no incremental costs. In this simple example it is assumed that all measures are mutually exclusive and therefore cannot be used in parallel or in sequence (i.e. only one measure is applicable to each vehicle). Total emission savings are therefore likely to be underestimated. With further refinements (e.g. to account for interactions between measures) cost curves like the

[7] It is important to realise that the results in Table 10 do not take into account interactions between measures, e.g. the ordering of measures may effect total applicability and reduction effciency. Consequently, the environmental performance of the listed measures may not be strictly additive. Such interactions need to be addressed before an accurate abatement cost curve can be constructed, from which an effective control programme may be designed.

one shown in Figure 6 may be used to develop 'least-cost' urban air pollution control programmes.

**Figure 6: Preliminary Abatement Cost Curve for Two-stroke Three-wheelers in Delhi**

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# Chapter 7a

# THE ECONOMICS OF FORESTRY

*by*

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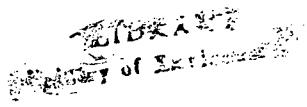


## 1 INTRODUCTION

This chapter aims to explain in economic terms the current trends in global forestry, and to show how economic analysis can be used to identify the appropriate policies with which to prevent excessive deforestation. Section 2 describes recent trends in the coverage of forestry, both in developing and developing countries, and in the quality and density of forests, drawing heavily on an FAO report on the changes in forest statistics.

Section 3 analyses the economic causes of deforestation. These include the fact that discounting tends to encourage the depletion of resources, and it is shown that the higher the discount rate relative to the rate at which a forest grows, the more likely it is that the forest will be depleted over time. Also analysed is the seminal Faustmann model, which illustrates the optimal rotation length in commercial forestry and shows that the higher the discount rate, the shorter the optimal rotation length. This section also provides a model that highlights the problems of open access to forest resources and conflicting uses of forest land, namely timber extraction and animal grazing. Also discussed are the role of government failure in forest depletion, and the fact that the provision of subsidies to forestry and other activities is often associated with excessive deforestation.

Section 4 explores the role of economic analysis in identifying the policies with which to limit deforestation. These include strategies for government management of forests, and for providing subsidies for the environmentally sensitive management of forests by private companies. The section also notes that the global values of forests, in particular tropical forests, mean that there is the role for international financing of forest conservation in developing countries. Section 5 concludes.

## 2 RECENT TRENDS IN GLOBAL FORESTRY: THE STATE OF THE WORLD'S FORESTS.

This section draws on information provided by a 1995 report by the Food and Agriculture Organization of the United Nations (FAO) "Forest Resources Assessment 1990: Global Synthesis." and unless stated otherwise, the data quoted comes from this report. The report constitutes a comprehensive estimate of global forest cover in 1990 and the changes in forest cover between 1980 and 1990.

The report discusses the major indicators for global forestry and its sustainability, namely remaining forest cover, its distribution and condition, and the changes in forest cover and condition over time. While in developed countries forest stocks are stable, and are in many cases increasing, remaining stocks of virgin forest, e.g. in Canada and the USA, continue to be harvested. Moreover, forest stocks are significantly degraded relative to their original condition, in many developed countries being considerably less than half their original levels. Pollution, and in particular acid rain, also causes damage to forest stocks. In developing countries, levels of both deforestation and degradation of remaining stocks are still high.

### 2.1 Changes in the Extent and Distribution of Global Forest Cover

Forest cover is defined by FAO as land with a minimum tree cover (native trees or plantation forests) of 20 percent in developed countries and 10 percent in developing countries. The FAO (1995) report reveals that in 1990 forest and other wooded land covered 5.1 billion hectarest[1], which is about 40 percent of the Earth's land area. During the period from 1980 to 1990 forests and other wooded land area declined by 2 percent, or 100 million hectares. The vast majority of this decline is accounted for by deforestation in tropical countries; forest and other wooded land area declined by 3.6 percent in tropical countries and by less than 0.04 percent in developed countries.

The FAO 1990 Assessment estimates natural forest change only for developing regions, data were not available for developed countries. In developing countries, natural forest cover declined by 8 percent (163 million

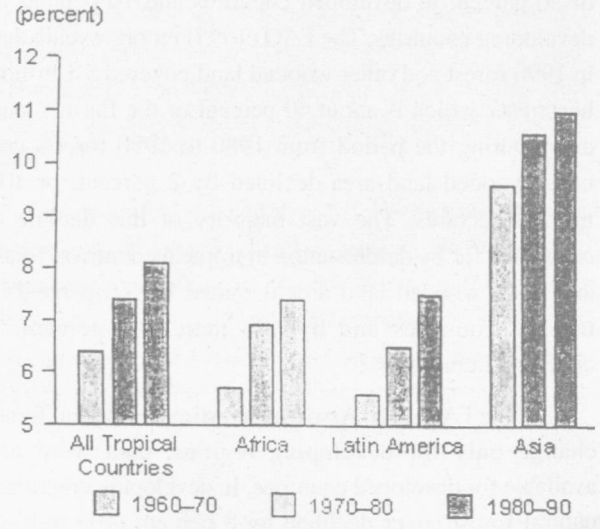
1 of this, 3.4 billion hectares was Forest, and 1.7 billion hectares consisted of other woody vegetation such as open woodland, scrubland and brushland. and areas under shiRing cultivation, according to the FAO Report.

hectares) during the 1980s. On a net basis, however, total forest and other wooded land area in developing countries declined by only 98.7 million hectares. The loss of natural forest cover was offset by new plantation cover totalling 31.9 million hectares and by a 32.1-million hectare increase in other wooded area, which consists mainly of replanted former natural forests.

The aggregate figures for reduction in forest cover thus disguise larger reductions in tropical and virgin forests. Temperate forests, located largely in Northern Canada and the Russian Federation, increased over the 1980's due to increased planting. The USA was the only developed country, which experienced major decreases in forest cover during this period. However, natural temperate forest cover in developing countries did decline by 4.5 percent, the large percentage figures being due to the fact that in many temperate developing countries, centuries of depletion mean most of the original forest cover has already been lost.

Tropical forests (including plantations) constituted roughly 1.8 billion hectares, just over half the world's forest cover, in 1990. Rainforests covered almost 714 million hectares in 1990, according to the World Resources Institute (1994). FAO data reveals them to have declined by 450 million hectares between 1960 and 1990, a decline of almost 39 percent. The total tropical cover in Asia declined by a third during this period, while in Africa and Latin America it declined by around 18 percent. Between 1980 and 1990 8 percent of global natural tropical cover was lost, with the highest rates of loss being in Asia. Figure 1 below summarises these data.

**Figure 1: Estimated rates of Tropical Deforestation 1960-1990. Source: FAO.**




Increases in planted forests appear to compensate in part for the loss of virgin forests, although they tend to be less diverse, thus to be less stable in terms of resistance to pests and climatic changes. During the 1980s plantation cover in developing countries virtually doubled, but compensated for only a fifth of the forest cover lost. However, a tree planting programme in China meant that net change in forest cover in Asia and the Pacific was positive in the 1980's.

## 2.2 Changes in the Condition of Global Forests

The second major indicator of the state of the world's forests is the condition of forests. This can be assessed by several indicators, including the extent to which forests resemble their natural state, the extent to which they are fragmented, and the intensity with which they are cultivated. These indicators are particularly important with respect to tropical forests because losses of biodiversity are not accounted for in the estimates of deforestation reported in section 2.1. Fragmentation can result in patchy habitats that are no longer large enough to support certain species, or can make species more susceptible to predators or changes in climate.

In the case of temperate forests the introduction of new pests, as well as air pollution and intensive logging, have affected the health of many temperate forests. While in Europe, according to McKibben (1995), forested areas have increased since 1950, virgin forests continue to decline, reducing biodiversity still further. Plantations of uniform ages and species are more prone to fire, wind and disease than natural forests.

The FAO report includes estimates of changes in the nature of forest cover, finding that more than 3.5 per cent of the forest cover in 1980 was subject of moderate to severe degradation, where an example of moderate degradation is loss of density, while an example of severe degradation is the practice of short-fallow cultivation.

The model in section 3.2 explains how economic factors, namely open access to forest land and the fact that forest land is often subjected to conflicting uses, cause degradation of forest stocks.

# 3 THE CAUSES OF DEFORESTATION: AN ECONOMIC ANALYSIS

The causes of deforestation can be thought of as having three major causes. The first can be explained with the theory of economically efficient renewable resource depletion, and relates to the effect of discounting when the discount rate is greater than the rate of growth of a renewable resource. The second cause relates to the problem of open- access to forest resources and the tendency of open access resources to be over-harvested.

The third problem relates to government failure, and the fact that many governments encourage the unsustainable depletion of forests by means of subsidies. This section discusses each of these issues in turn.

## 3.1 The efficient depletion of renewable resources and the role of discounting.

Sustainability of forests and other renewable resources is often cited as a feature of an environmentally and economically sustainable economy. Forest are one of the ecological assets that many believe should be accorded a "special status" (Toman and Ashton 1996). By this is meant that ecological capital should not be treated as if it were substitutable for manmade capital, but should be preserved in its own right as being a crucial component of the overall capital stock which we pass on to future generations. Maintenance of stocks of renewable resources, of which forests are an example, is one of Daly's (1990) "operational principles of sustainable development".

However, there are many possible levels at which forest stocks could be maintained in a steady state and preserving any given forest stock almost always implies an opportunity cost. Therefore some criterion of efficiency is required to decide among the possible steady states. The efficient management of a single stand of forestry was identified by Preinrich's (1938) solution of Faustmann's (1849) problem, which calculates the rotation age that maximises the stream of financial benefits from forestry. This framework maximises the value of a given piece of forest land, and therefore maximises the direct financial benefits of forestry, or the net benefits from timber harvesting. In this section we illustrate the problem by describing both the Faustmann framework and the biomass renewable resource model, in which the rate of growth of the forestry stock is assumed to be a decreasing function of the stock of forestry on a given area of land.

### 3.1.1 Maximising the financial return to forestry using a biomass renewable resource model.

This model illustrates the effect of discounting and a forest's growth function on the economically efficient management strategy. This model can be used to describe the efficient management of fisheries, and other renewable resource stocks.

In this very simple version of the biomass renewable resource model, the Owner of the forest is assumed to wish to maximise the net present value of the forest. That is, the objective function is:

$$\underset{R}{Max} \int_{t=o}^{\infty} PR(t)e^{-rt}\,dt$$

Where $P$ is the price at which timber can be sold, $R$ is the timber harvest in each period $t$, and $r$ is the rate of interest on alternative investments. The growth function of the forest is assumed to be:

$$S(t) = g(S(t)) - R(t),$$

Where $S$ is the resource stock and $g$ is the stock growth.

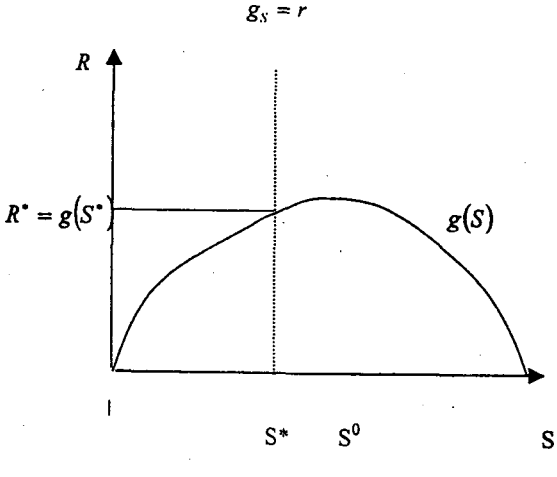
Maximising the objective function subject to the forestry growth function reveals the optimal harvesting plan. The first-order conditions can be used to find the efficient path of timber extraction, leading to the efficient steady state of the forest stock. In particular, they can be used to show that for efficiency, the following condition must hold:

$$\frac{P}{P} = r - g_s$$

This illustrates that any disparity between the marginal productivity of the resource stock and the interest rate must correspond to an increasing timber price. This is the renewable resource version of the Hotelling rule (Hotelling 1932) for efficient resource extraction. It requires that the efficient extraction path deplete the forest stock up to the point at which the marginal productivity of the forest stock equals the interest rate. The level $S^*$ at which $g(S^*) = r$ is the only efficient steady state stock level with a positive stock and extraction rate.

For the case in which timber extraction follows the saddlepath to this steady state, the efficient path of the forest stock and of timber extraction is depicted in Figure 2 below. In the current example, in which the forest owner is assumed to have no market power and thus cannot affect the price of timber, this efficient path involves reaching the steady state as quickly as possible.

**Figure 2: The financially optimal paths of timber extraction and forest stock**



The economically efficient steady state forest stock S* is at the point at which the marginal growth rate of the forest, shown by the slope of the growth function, is equal to the interest or discount rate. The higher the discount rate, the steeper must be the growth function, and thus the lower must be the efficient steady state stock. Figure 2 illustrates the point made by Ackerman (1994) that for the current level of forestry stocks to be managed both (financially) efficiently and sustainably, the return on invested capital must equal the marginal productivity of the forest stock. It is possible in the open economy that the international interest rate might be sufficiently high for this condition never to be met, so that the efficient steady state forest stock is zero.

Thus, this example illustrates the effect of discounting on the economically efficient management strategy for forest resources. The higher the rate of discount, the more likely it is that the efficient strategy will be for the forest stock to be depleted altogether, and the proceeds invested in an alternative asset, and the forest land converted to an alternative use.

### 3.1.2 Maximising the financial return to forestry using the Faustmann model.

The Faustmann model is one of the original and best-known results in forestry economics, and is still used widely in economic analysis of forestry management. The Faustmann framework provides a static result describing the optimal rotation age on a given area of land used for commercial forestry. This is an important feature of forest management, in that the greater the rotation age, the higher the quality of the forest in terms of the habitat it provides for biodiversity. Thus it relates to the condition of forest cover as discussed in section 2.2. The basic Faustmann problem is to maximise the NPV of the flow of net benefits:

$$(F(T)- Z)e^{-rT} + (F(T) - Z)e^{-r2T} + (F(T)- Z)e^{-r3T} + .....$$

Where $F$ is the revenue (price times volume) from the harvest as a function of $T$, the rotation age. $Z$ is replanting costs, and $r$ is the rate of interest. This is an infinite geometric series and can be re-expressed as:

$$\frac{F(T)- Z}{e^{rT} - 1}$$

Maximising equation this expression with respect to the rotation age yields the following condition:

$$\frac{F'(T)}{F(T) - Z} = \frac{re^{rT}}{e^{rT} - 1}$$

$$= \frac{r}{1 - e^{-rT}}$$

This expression states that marginal net revenue, with respect to increasing rotation age, must equal the interest rate inflated by a term reflecting the opportunity cost of each rotation, namely, the following rotation.[2] The higher the interest rate, the lower is the optimal rotation age. As with the biomass renewable resource model, if the interest rate is sufficiently high then commercial forestry will not be viable at all.

## 3.2 Open access and conflicting uses of forestry: ill- defined property rights and deforestation

The FAO study referred to in section 2 provides some information on the root causes of deforestation and the degradation of forests, and in particular tropical forests. To a large extent, this has been related in developing countries to rural population pressure and the simultaneous use of forest land for subsistence farming, including shifting cultivation, grazing, and wood extraction (for fuelwood and building materials). In Asia, increased shifting cultivation as well as the conversion of land to agriculture and plantations have been primary causes of deforestation.

Population pressure and conflicting uses of forest land can combine to result in the degradation and depletion of forests. These uses include the legal extraction of timber, the extraction of non-timber products, the use of forest land for shifting cultivation, the grazing of animals and illegal logging. In this section we present an economic model that illustrates the effect of the simultaneous presence of two of these uses, namely timber extraction and grazing. This shows how the failure of different groups of people to account for the effects of their own activities on other people (externalities) means that forest resources tend to be over-exploited, resulting in outcomes that are both unsustainable and socially inefficient.

The model represents the dynamics of the forest economy as a whole. The objective is to identify an efficient and sustainable steady state of grazing and timber extraction, illustrating how open access means that this outcome is unlikely, and then to suggest how the society might move from the current situation to this state.

Consider a forest that is used to graze animals as well as for timber extraction. This timber extraction could represent either commercial harvesting or the extraction of fuel and building materials by local people. To represent the socially efficient case, we assume that a social planner maximises the present discounted value of the net revenue obtained from timber harvesting and from animal husbandry, using as a discount rate the

2 Note that if the problem were simply to maximise the present value of a single harvest, then the condition would be for the rate of growth of revenue to equal the interest rate.

constant and exogenous interest rate, $r$. This revenue consists at any moment of the revenues from timber sales, $PR(t)$ and the return $D(G(t))$ from the stock of grazing cattle, $G(t)$. The timber growth function, $g$ is a function of the current timber stock, $S(t)$, as well as being a negative function of $G(t)$. The rate of change of the timber stock is equal to this growth, less timber extraction $R(t)$.

The social planner maximises the present value of net revenue over time subject to the growth function of the resource as follows:

$$\underset{R,G,M}{Max} \int_0^{\infty} (PR+D(G))e^{-rt}dt$$

$$s.t.\ S(t) = g(S(t), G(t)) - R(t)$$

The Current Value Hamiltonian, suppressing time arguments, is:

$$H_C = PR + D(G) + \psi_s [g(S, G) - R]$$

With first order conditions for an interior solution:

$$\frac{\partial H}{\partial R} = P - \psi_s = 0 \qquad 1$$

$$\frac{\partial H}{\partial G} = D_G - \psi_s g_G = 0 \qquad 2$$

$$\dot{\psi}_s = r\psi_s - \frac{\partial H}{\partial S} = \psi_s[r - g_s] \qquad 3$$

These equations can be used to show the effect of both controlled and uncontrolled grazing on the efficient forestry management plan. Firstly, equations 1 and 2 can be combined to show that an efficient condition is that:

$$D_G = Pg_G$$

That is, for social efficiency, the marginal return to grazing animals should be equal to the value of the marginal effect of grazing animals on the forestry growth. However, if farmers do not account for this effect, and they are unlikely to do so under open access conditions, they will graze forest land to a greater extent. If farmers act so as to maximise the return to grazing then they will increase grazing to the point at which $D_G = 0$. If individual farmers fail also to account for the impact of their own grazing on the availability of land to other farmers, then the number of animals put onto the land could be even greater. In any case, this model illustrates that without controls on the numbers of grazing arumals, the effect on the forest's ability to provide timber is reduced to a socially inefficient extent, in that the marginal costs of lost timber are greater than the marginal benefits of grazing. This model could be extended to account for the effects of grazing on non-timber forest uses.

A second point revealed by the model is that any use of the forest for grazing, as long as it reduces the marginal productivity of the timber stock, will reduce the efficient steady state of the forest stock. This can be illustrated by reproducing the renewable resource version of the Hotelling rule, which is derived from equations 1 and 3 above:

$$\frac{\dot{p}}{p} = r - g_s$$

If the marginal rate of timber growth is reduced by the presence of grazing, then to reach equality with the interest rate, and thus satisfy the condition for the efficient steady state, the timber stock will have to be reduced to a lower level than would other wise be the case. The higher the level of grazing, the more likely it is that the optimal solution for timber extractors will be simply to deplete the forest stock altogether.

This simple model illustrates the possible effects of ill-defined property rights over forest land, or a lack of regulation of the use of forest land, in terms of inefficient and possibly unsustainable use of forests. Possible solutions to this problems, and to those raised earlier in this section, are discussed in section 4.

## 3.3 Government failure: the role of perverse subsidies in deforestation

The previous two sections have shown that it can be economically efficient, especially in conditions of open access to forest resources, to carry out activities that result in deforestation rather than the sustainable management of forests. However, in addition to these factors, much of the loss of the world's forest resources over recent decades can be attributed to government subsidies, which mean that depletion of forests is greater than it would be without government intervention.

There are many examples of subsidies, by means of which governments encourage exploitation of forests. For example urban dwellers in Brazil have been given credit at negative rates of interest in order to convert forests to farming land. Indeed, almost half of the loss of Brazilian rainforests has been attributed to just four programmes of government subsidy. Indirect subsidies can be just as damaging; often subsidies given to mining companies mean that roads are built, remote areas become accessible and timber extraction becomes viable.

Institutional and legal factors can also play a part in encouraging deforestation. For instance, land tenure laws may require land to be cleared for an individual to be able to claim property rights over it.

Such government subsidies mean that while

private companies and individuals benefit, society as a whole loses out doubly. Not only are the social benefits of forestry being lost, these losses outweighing the private benefits, but also society is paying for them to be destroyed.

# 4 THE ROLE OF ECONOMIC POLICY IN FOREST CONSERVATION: ACCOUNTING FOR THE FULL ECONOMIC VALUE OF FORESTS

This section explores the role of economic policy in forest conservation, and the role of economic analysis in identifying the true total economic values of forests. These include the use of forests for non-timber products and for recreation, the global values of, especially tropical, forests as repositories of biodiversity and the global values of carbon sequestration In this section we look first at the effects on socially efficient forest management strategies of extending the economic models discussed in section 3 to account for the non-timber values of forestry. We then discuss some of the government policies that could be put in place in order to achieve this social optimum. Finally, we discuss the global values of forests and the international policies that can be used so that people using the forests account for these values.

## 4.1 Extending the economic analysis to account for non-market benefits

In almost all countries, forests are a source of the non-market benefits associated with recreation and aesthetic enjoyment. These benefits have economic values, which can be measured using a variety of tools, and accounting for these benefits can be shown to increase both the socially efficient levels of forest stocks, and the socially efficient age to which trees are allowed to grow before being harvested. Forests have a variety of other uses as well; in particular in developing countries they are used directly to provide food and other materials. Examples of these are the collection of fuelwood, fruits, vegetables, mushrooms, spices and nuts, the hunting of wild animals, the collection of medicinal plants, the collection of canes, poles and roofing materials and the use of forest land for pasture.

### 4.1.1 Extending the biomass model for non-timber benefits

First we consider the model of section 3.1.1. We saw that the economically efficient management strategy is to deplete the forest stock up until the point at which:

$$g_s = r$$

That is, where the marginal growth rate of the forest stock is equal to the interest rate. It can be shown that if we include the non-timber benefits described above, which can usually be assumed to increase with the level of the stock, the efficient steady state becomes the one where:

$$g_s + n_s = r$$

Where $n_s$ is the marginal contribution to non-timber benefits of the forest stock.

The implications of this result are as follows. If the government is managing the forest stock for timber revenues, then in order to maximise social benefits, it should harvest the forest less intensively than it would do to maximise solely the financial return from timber. If, on the other hand, the forest is managed privately, then it is economically efficient for the government to provide a subsidy to the forestry company in return for less intensive harvesting of the forest.

### 4.1.2 Extending the Faustmann model for non-timber benefits

In this section we extend the Faustmann model described in section 3.1.2 to include non-market benefits. This allows us to analyse the age structure of forests. Hartman (1976) extended this model by including in the objective function a term describing the non-market benefits of forest stocks. This term takes the form:

$$\int_{t=0}^{T} e^{-rt} h(t)dt \; [1 + e^{-rT} + e^{-r2T} + ....]$$

which is the present discounted value of the non-constant flow of nonmarket benefits, $h(t)$, that accrue over each rotation. In the extension in this section we take a slightly different approach by assuming that a forest management programme consists of many stands, and that the rotations on different stands are staggered so that harvesting is continuous and the stocks of trees of each age class are constant at any one time. Therefore, non-market benefits can be thought of as being simply a function of the rotation age, and not as changing during the rotation, as they would if non-market benefits were being derived from an individual stand.

We thus include non-market benefits as a function, $B(T)$, of the rotation age, so that $B_T > 0$, $B_{TT} < 0$. $B(T)$ is a current benefit flow as opposed to the present discounted values of benefit flows. Therefore, the process must be set up to maximise the sum of ongoing benefits. We achieve this by including the basic financial return from the forestry programme as the interest on the net present value of the programme[3]. Therefore, the maximand becomes:

[3] Maximising $r\frac{F(T)\,Z}{e^{rT}-1}$ would result in exactly the same efficient rotation age as the identified in section 3.1.2.

$$r\,\frac{F(T)-Z1}{e^{rT}-1} + B[T] \qquad (8.4)$$

Differentiating this expression with respect to $T$, the rotation age, and setting the result equal to zero reveals that for total annual benefits to be maximised, the following expression must be satisfied.

$$\frac{F'(T)}{F(T)-Z} = \frac{r}{1-e^{-rT}} - B'(T)\,\frac{e^{rT}-1}{r(F(T)-Z)}$$

This expression illustrates how including the non-market benefits of forestry will extend the optimal rotation age. The first term on the right hand side is the basic Faustmann optimality condition. The second term on the right hand side represents the non-market benefits of forestry. This term is positive and decreasing as the rotation age increases. This, in other words, has a similar effect to reducing the interest rate.

The implications of this result correspond to those of the previous section. If the government manages the forest stock for timber, then to maximise social benefits it should employ a rotation age that is longer than would be suggested by purely financial criteria. Likewise, if the forest is managed by a private company, it is efficient for the government to subsidise a longer rotation age.

### 4.1.3 Further policy implications of the discussion

One of the most obvious, and cost-effective, means by which to discourage the unsustainable depletion of forest resources is to eliminate the damaging subsidies to resource depletion outlined in section 3.3. However, this is often politically difficult since those who receive the subsidies are often politically powerful. There are also transitional effects since certain ventures are not longer viable, with implications for employment. Thus, subsidies must be phased out gradually in such a way as to limit social and economic disruption.

The model of section 3.2 illustrated the problems associated with ill- defined property rights and the resulting tendency for forest resources to be over-used. The role of the government in correcting this problem is two-fold. Firstly, the government can help to regulate the use of the forest for various purposes, including grazing and timber extraction, so that this use is both sustainable and socially optimal. Secondly, property rights can be assigned to those using the forest, in order to give them an incentive to use the resource sustainably and to assist in regulating against illegal use of the forest e.g. for illegal timber extraction.

## 4.2 Accounting for global benefits of forest conservation

The final consideration for the role of policy in forestry conservation concerns the global values of forests, in particular those in developing countries. These values consist mainly of the biodiversity contained within many forests, and of the carbon sequestration values. The models of section 3 could be extended further to include the global values of forest stocks, the result of which would be that these values warrant the conservation of still larger stocks of forests than would be the case if only domestic non-timber values are accounted for. However, the implications for financing are that the international community should provide funds to compensate developing country governments for the additional costs of preserving forest stocks. Some of the mechanisms by which this could be achieved are discussed in the chapter on sustainable development.

The international community in the area of forest preservation has already done much. Although there is not yet a binding international agreement specifically on forest loss, both the Convention on Biological Diversity and the Framework Convention on Climate Change recognise the global values of forest ecosystems. The World Commission on Forests and Sustainable Development (WCFSD) aims to build consensus between stakeholders regarding the dual, and often conflicting roles, of forests in preserving biodiversity and constituting instruments of social and economic development. The Intergovernmental Panel on Forests was created by the WCFSD to promote the implementation of international agreements relating to forests at both national and international levels.

Initiatives to bring these values into the commercial arena are in place, with the International Tropical Timber Organization having produced criteria and guidelines for the sustainable management of tropical forests. In 1994 the International Tropical Timber Agreement was renegotiated and timber producing and consuming countries agreed (although this is a non-binding agreement) to comply with the ITTO's criteria.

## 5 CONCLUSION

This chapter has aimed to show that economic analysis and economic policy can play an important role in the conservation of the world's forests. This role is applicable to both developing and developed countries. Section 2 provided an overview of recent trends in global forestry, highlighting the fact that it is important to account for both the coverage of forestry and the condition of the forest cover. In particular, the condition of forest cover can influence the extent to which it can provide a

habitat for biodiversity, and thus can affect both the global and local value of the forest.

Section 3 introduced economic analysis in order to explain, at least in part, the problems of deforestation. The first section used a biomass renewable resource model, which maximises the present discounted value of the returns to forestry to explain the effect that discounting has on an efficient forestry management strategy. We saw that the higher the discount rate, other things being equal, the more likely it is that the efficient strategy will be to deplete the forest stock altogether and to convert the land to an alternative use. The second section used the well-known Faustmann model to explain the relationship between the discount rate and the efficient rotation age in commercial forestry. The rotation age is an important factor since the higher the rotation age, the better the average condition of the forest and the higher the non-timber values of the forest will be.

Section 3.2 went on to look at the problem of open access to forestry resources, in particular where the forest land is being subjected to different, conflicting, uses, as is often the case in developing countries We saw that where the forest is being used both for timber extraction and for grazing, the failure of farmers to account for the effects of their animals on the growth of the forest leads to the forest being used more intensively than is socially effficient. This makes it more likely that the forest will become degraded or depleted. Finally in this section, we looked at the role of government failure, noting that many instances of forest loss can be attributed to the provision of subsidies that make conversion of forest land economically viable when otherwise it would have been conserved.

Section 4 outlined the role of economic policy in alleviating the problems of deforestation. We saw that if those responsible for managing forest resources account for their non-market benefits, which include their recreational and aesthetic values and the gathering of non-timber products, then it is likely to be efficient to conserve more of the forest, in a better condition, than would otherwise be the case. The policy implications of this are that the government should itself manage the forest so that it is used less intensively, or in the case that the forest is owned privately, provide a subsidy to compensate the owner for the financial costs of managing the forest less intensively. This conclusion was supported by extending both the biomass renewable resource model and the Faustmann model to include the non-market benefits of forests. Finally in this section, we noted that including the global benefits of forest conservation implies a role for international financing of forest conservation.

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# Chapter 7 b

# TOWARDS POLICY RELEVANT ECOSYSTEM SERVICES AND NATURRAL CAPITAL VALUES: RAINFOREST NON-TIMBER PRODUCTS

*by*

*B. M.S. Batagoda, R. Kerry Turner, Robert Tinch and Katrina brown*

## 1. INTRODUCTION

Given the cumulative process of global environmental change, driven by range of socio-economic parameters the world's ecosystems and their services have come under increasingly severe pressure. Components of such systems are being destroyed and degraded on a daily basis (see Table 1). But just how significant in human welfare terms are these losses? How valuable are ecosystem services such as waste assimilation, maintenance of biological diversity, clitnate and ultraviolet radiation regulation, the provision of food, shelter and medicine and many others? One recent study has estimated the current economic value of seventeen ecosystem services on a biosphere-wide basis at between US$ 16 - 54 trillion ($10^{12}$) per year (Costanza et al. 1997).

**Table 1: Selected Ecosystem goods and services and functions**

| | Ecosystem Service/Good | Ecosystem Functions | Examples |
|---|---|---|---|
| 1 | Gas regulation | Regulation of atmospheric chernical composition | $CO_2/O_2$ balance, $O_3$ for UVB protection and SOx levels. |
| 2 | Climate regulation | Regulation of global temperature, precipitation, and other bio-logically medicated climatic processes at global or local levels. | Greenhouse gas regulations, DMS production affecting cloud formation. |
| 3 | Disturbence regulation | Capacitance, damping, and integrity of ecosystem response to environmental fluctuations. | Storm protection, flood control, droutht recovery, and other aspects of habitat response to environmental variability mainly controlled by vegetation structure. |
| 4 | Water regulation | Regulation of hydrological flows. | Provisioning of water for agricultural (e.g. irrigation) or industrial (e.g. milling processes or transportation). |
| 5 | Water supply | Storage and retention of water. | Provisioning of water by wata sheds, reservoirs, and acuifers. |
| 6 | Erosion control and sediment retention | Retention of soil within an ecosystem. | Prevention of loss of soil by wind, runoff, or other removal processes, storage of silt in lakes and wetlands. |
| 7 | Soil formation | Soil fomation process. | Weathering of rock and the accumulation of organic material |
| 8 | Nutrient cycling | Storage, internal cycling, process-ing, and acquisition of nutrients. | Nitrogen fixation, N, P, and other elemental or nutrient cycles. |
| 9 | Waste treatment | Recovery of mobile nutrients and removal or breakdown of excess of xenic nutrients and compounds | Waste treatment, pollution control, detoxification. |

| Ecosystem Service/Good | Ecosystem Functions | Examples |
|---|---|---|
| 10 Pollination | Movement of floral gametes. | Provisioning of pollinators for the reproduction of plant populations. |
| 11 Biologlca1 control | Trophic-dynamic regulations of populations. | Keystone predator control of prey species, reduction of herbivory by top predators. |
| 12 Refugia | Habitat for resident and transient populations. | Nurseries, habitat for migratory species, regional habitats for locally harvested species, or over wintering grounds. |
| 13 Food production | That portion of gross prunary production extractable as food. | Production of fish, game, crops, nuts, fruits by hunting, gathering, subsistence farming, or fishing. |
| 14 Raw materials | That portion of gross primary production extractable as raw materials. | The production of lumber, fuel or fodder. |
| 15 Genetic resources | Sources of unique biological materials and products. | Medicine, products for materials science, genes for resistance to plant pathogens and crop pests, ornarnental species (pets and horticultural varieties of plant). |
| 16 Recreation | Providing opportunities for recreational activities. | Eco-tourism, sport fishing, and other outdoor recreational activities. |
| 17 Cultural | Providing opportunities for non-comrnercial uses. | Aesthetic artistic,educational, spiritual, and/or scientific values of ccosystems. |

*Source : Costanza et. al., (1997)*

Such global aggregate values, while they do serve to raise awareness and stimulate dialogue between scientists, social scientists, policymakers and citizens, should not be taken too literally. Valuing individual ecosystem function services is not a straight forward task, aggregating the values of different functions within an ecosystem and scaling up site specific values to regional or global scales represents a further order of magnitude of complexity and difficulty (Turner, Adger and Brouwer, 1998). Nevertheless, there is still a strong case in favour of meaningful economic valuations of ecosystem services. Many of these services lack market prices and/or are inadequately protected because of ill-defined or absent property rights regimes. The danger is that they will therefore be assigned too little or zero value and weight in policy decisions at local, national and global forums.

The dual challenge will be to quantify in monetary and other terms the value significance of ecosystem services; as well as to formulate mechanisms by which such function-based values can be realistically appropriated by society These marginal value estimates could then play an increasing role in socioeconomic and political decision taking via national income/resource accounting exercises and cost-benefit appraisal procedures.

This paper focuses on tropical rainforest ecosystems and the use value of their non-timber forest products (NTFPs) provision. Table 2 summarises the results of economic valuation studies undertaken in order to estimate the value of tropical forests (both use and non-use values). In a much quoted paper (Peters et al. 1989) thc economic value (net present value) of the fruits and latex harvest from an Amazonian forest was estimated to be US$ 6,330 $ha^{-1}$. Even more significantly, it was also claimed that such an economic return was higher than a number of alternative, but less sustainable, land use options such as sawtimber extraction/production (NPV = US$ 1,000 $ha^{-1}$), timber and pulpwood extraction/ production (NPV = US$ 3,184 $ha^{-1}$) or fully-stocked cattle pasture (NPV = US$ 2,960 $ha^{-1}$). This is a strong conclusion with obvious implications for tropical forest conservation versus development policy.

Some researchers have gone further and have conducted meta analysis studies on the published NTFPs

**Table 2: Forest Products Valuation Studies of Tropical Rainforests**

| Country | NetValue/$/ha/ year | Comment | Reference |
|---|---|---|---|
| **Stock Estimates** | | | |
| Iquitos Peru | 700 gross 420 net | Total stock of know major use woody trees, climbers and habaceous species excluded | Petas, Gentry and Mendolsohn 1989 |
| Sinharaja Sri Lanka | 622 gross 377 net | total stock including woody trees, climbers and herbaceous species | Batagoda 1997 |
| **Potential flow** | | | |
| Ecuador | 200net | Potential flow from managed rainforest | Myers 1988 |
| Iquitos Peru | 20 net | Potential value of 6 species of latex and fruits | Pinedo-Vasques, Zarin and Jipp 1992 |
| Sinharaja Sri Lanka | 186 net | All know use species - estimates consider the regeneration and accessibility | Batagoda 1997 |
| **Actual flow** | | | |
| Amazon Brazil | 4.8 gross | Flora only | Schwartzman 1989 |
| Maya Gautemala | 10 gross | Export earnings of flora | Nations 1992 |
| Westcrn Amazon | 5-16 gross | Flora | Nations 1992 |
| Equitos Peru | 16-22 net | Flora | Padoch and de Jong 1989 |
| Combu Island Brazil | 79 gross | Some flora | Anderson and Ioris 1992 |
| Veracruz Mexico | 116 net | Flora | Alcorn 1989 |
| India | 117-144 gross | Flora | Chopra 1993 |
| Sinharaja Sri Lanka | 13 net | Flora | Gunatilleke Gunatilleke and Abeygunawarden a 1993 |
| Cross River N.P | 16.50 gross | Hunting gathering trapping | Ruitenbeek 1989 |
| Venezuela | 0.75 nct | Caiman harvest | Thorbjarnarson 1991 |
| Sinharaja Sri Lanka | 14 net | All know use flora | Batagoda 1997 |

Table 2: continued

| Country | NetValue/$/ba/ year | Comment | Reference |
|---|---|---|---|
| **Actual fauna flow** | | | |
| Korup N.P | 1.06 net | Forgone hunting | Ruitenbeek 1988 |
| Ituri Zaire | 0.5-3.18 net | Garne hunting of primary forest | Wilkie 1989 |
| Sarawak East Malaysia | 8 net | Wildlife value | Caldecott 1988 |
| Sinhara Sri Lanka | 1.89 net | Hunting | Batagoda 1997 |
| **$CO_2$ sequestration** | | | |
| Indonesia | $5/ton of $CO_2$ | Indonesian forest report | World Bank 1993 |
| Mexico | $ 650 - 3400 per ha | $CO_2$ sequestration benefits for Mexicon forest | Pearce et al. 1993 |
| Tropical forests | $ 20 per ton of $CO_2$ | Every ton of $Co_2$ released from burning of tropical forest contribute to global warming | Turner *et al* 1994 |
| | $ 600-1000 per ha | Costs of conversion of open forests | |
| | $ 2000 3000 per ha | Costs of convereion of closed forests | |
| | $ 4000-4400 per ha | Costs of conversion of primary forest | |
| Rondorna - Brazil | $300 per ha | Upper bound value for tropical forests | Schneider 1992 |
| **Soil Erosion Control** | | | |
| Mexico | $ 2.3 Million per | Soil erosion cost estimate year based on wager purification cost | Adger *et al* 1994 |
| Java - Indonesia | $ 2 - 7 per ha | Soil erosion damage cost | Magrath and Arens 1989 |
| Philippines | $ 17- 28 per ha | Soil erosion damage cost | Cruz et al. 1988 |
| Cameroon | $ 4 per ha year | Soil fertility value | Ruitenbeek 1992 |
| **Hydrologlcal cycle** | | | |
| Carneroon | $ 14 per ha year | Fisheries protection | Ruitenbeek 1992 |
| Cameroon | $ 2 per ha year | Flood control | Ruitenbeek 199 |
| Malaysia | £ 2S per ha year | In terms of conserved agriculture | Kurnari 1994 |
| " | $ 2-3 per ha year | Domestic water benefits and fisheries protection | " |
| " | $ 30 per ba year | Overall watershed protection . | " |
| Mexico | S 0.04 per ha | Watershed protection | Adger *et al.* 1994 |

| Country | NetValue/$/ba/ year | Comment | Reference |
|---|---|---|---|
| **Option Value** | | | |
| Mexico. | $ 6.4 per ha per year | The pharmaccutical option value for Mexican forest | Adga et al. 1994 |
| Global | $ 20 per ha per year | Commercial value for medicinal plants | Pearce and Puroshotharnan 1992 |
| Kenya | $ 89 per respondents | Value of viewing elephants on safari: Kenya | Brown and Hemy 1993 |
| USA | $ 72 per household per year | Option value of Pennsylvanian users to use tropical rainforests | Griffith 1991 |
| **Existence Value** | | | |
| Global | S 1.2 - 64 per annum per household | Exietence value for wild species - based on available studie | Turner *et al.* 1994 |
| | $ 9 - 107 per annum per household | Existence value for scenic and wilderness area | |
| Thailand | $ 8 per annum per person | Willingness to pay of non-user. to protect the park | Kaosard *et al.* 1994 |
| USA | $28 per annum per household | Existence velue of Pennsylvarua citizen for Topecial rainforests | Griffith 1991 |
| Mexico | $ 0.03 - 10.4 per ha | Using debt for naturo swaps and tourism survey of Mexican forests | Adger *et al.* 1994 |
| Global | $ 5 per hectare | Global existence value denved fiom conservation finance for tropical forests | Pearce and Moran 1994 |

values from various countries. Simple averaging of the different estimates gives a world median value of NTFPs of US$ 50 $ha^{-1}$ $yr^{-1}$ (Godoy and Bawa, 1993; Pearce and Moran, 1994; Pearce, 199i).

Unfortunately, aggregation and scaling issues are not straightforward and appropriate guidelines and protocols for carrying out such studies have not been consistently adhered to in the literature (Godoy *et al.*, 1993; Godoy and Lubowski, 1992). The study reported on below has rigorously applied its data collection and analysis to validity and reliability protocols in order to estimate the NTFPs value derived from the Sinharaja rainforest in Sri Lanka. This forest land use has then been compared with alternative land use options, timber harvesting and tea plantation cultivation, in a cost-benefit analysis. The results indicate that previous studies have significantly overestimated NTFPs value, and consequently that biodiversity conservation policy cannot be economically justified (economic efficiency criterion) on the basis of sustainable NTFPs collection alone. Such a strategy does, however, have other wider social benefit for local forest village communities such as income distribution equalisation. Conservation of the forest ecosystem would also generate other uses and non-use values linked to other 'NTFPs-compatible' forest function services - see Table 2. An economic case for such a conservation strategy will have to be based on the multiple services value that a given 'healthy' forest ecosystem can provide sustainably over time.

## 2. Economic Value and Stock and Flow Concepts

The total economic value of a forest can be broadly divided into direct extractive use value, direct non-extractive use value, indirect use value and non-use values. Of these, only the direct extractive use value i.e. timber and NTFPs extraction can be readily converted

**Table 3: Comparison of the Results of Cross-sectional Survey and Longtltudlual Log-book Survey**

| | Cross Sectional Survey | | Log-book Survey | |
|---|---|---|---|---|
| | Per household US$ | Per Hectare US$ | Per Household US$ | Per Hectare US$ |
| Total forest products flow | 367 (191) | 29 (14) | 205 (101) | 16 (9) |
| Total Commmercial NTFPs flow | 205 (101) | 16 (8) | 105 (52) | 8(4) |
| Total Consumptive flow | 162 (101) | 12 (6) | 100 (49) | 8(4) |
| Total food products flow | 299 (148) | 23 (11) | 112 (55) | 8(4) |
| Total medicinal products flow | 5 (2) | 0.37 (0.2) | 44 (22) | 3(1.4) |
| Total building materials flow | 13 (6) | I (0.49) | 18 (18) | 2(1) |
| Total weaving materials flow | 3 (1.4) | 0.2 (0.1) | 9 (4) | 0.7 (0 3) |
| Total fuelwood flow | 48 (23) | 4 (2) | 21 (10) | 2 (1) |

*Note: The cross-sectional survey interviewed 150 randomly chosen households while the log-books survey included 32 household and continued for 9 months from September 1995. The numbers within parenthesise are net values, with costs estimated at 50.6% of gross value.*

into a flow of real income for local people. In developing countries, where most rainforest ecosystems occur, indirect and non-use values of forests that cannot be readily converted into 'real' money have little practical significance, for everyday livelihoods. There are mechanisms such as debt-for-nature swaps, transferable development rights, and carbon offsets trading (see Table 3) to convert the non-market values into real (money) resources, but the practical application of such mecharusms has so far been limited (Pearce, 1997). Therefore a policy of maximising the extractive use value of forests in developing countries is financially and politically difficult to refute. The most important current extractive use of rainforests is the harvesting of timber and NTFPs.

A proper estimation of NTFPs value has to distinguish between four ecological economics concepts: the potential total stock value flow (related to the total standing biomass of products in the forest); the total extractable stock limit (defined as the maximum amount of the total stock inventory that may be sustainably harvested); the potentially extractable output flow value (governed by accessibility constraints and therefore representing practicable harvesting limits); and the actual extracted (harvested) flow value, which is the amount local people really do appropriate under any given economic regime i.e. market price and demand.

These four numbers or ranges should always be clearly distinguished both conceptually and for policy analysis purposes. However, previous studies have utilised all of these different measures while claiming to have quantified and valued NTFPs. For example, Peters et al., (1989) used inventory stock data to value NTFPs, Myers (1988) used potential extractable flow data, while Gunatilake *et al.*, (1993) used the actual flow for the same purpose. Some meta analysis studies seem to have ignored these fundamental conceptual differences in the data and have attempted to find world mean NTFPs value by averaging the results of different studies (Godoy et al., 1993; Pearce and Moran, 1994). Simple averaging of the different estirnates yields a world median value of nontimber forest products of US$ 50 $ha^{-1}\ yr^{-1}$ (Godoy and Bawa, 1993; Pearce, 1997). However, this value has no scientific credibility, because stock and flow data have not been correctly identified and partitioned in the aggregation procedure.

If we are to more precisely estimate global average non-timber forest products value per hectare of forest, at least three separate values: average extractable stock limit value; average potential flow value and average actual flow value should be clearly distinguished and not blended together. On biophysical grounds it is also more meaningful to report average values, at least for the major forests types i.e. tropical rainforests, tropical dry forest and temperate forests in the world. In any case, more often than not, the politically most relevant values are those that emerge at the local or regional levels.

## 3. Sinharaja Forest Study

To demonstrate the application of a conceptually more robust approach to NTFPs valuation, we undertook a broad analysis of the use of such products from the Sinharaja forest in Sri Lanka. The Sinharaja is the least disturbed largest remnant of rainforest in Sri Lanka, extending over 47,000 ha and located at latitudes 6 21' to 6 26'N and longitude 80 21' to 80 34 E The entire forest receives more than 2500 mm of annual rainfall with no dry spells. February, the driest month, produces an average precipitation of 189 mm. Up to an elevation of 900 m, the Sinharaja forest is classified as tropical wet evergreen forest, and above 900 m as sub-montane evergreen forest. Based on the topography, the land is broadly categorised as lower slopes and valleys (150 m - 600 m), middle slopes (over 600 m and below 1000 m) and upper slopes and ridges (above 1000 m). There are 30 relatively isolated villages around the Sinharaja forest with a total of 5000 people in some 900 families. Some villagers have to walk about 7 to 9 miles to reach a market place, school or hospital. For more than 100 years these people have been highly dependent upon the forest and its resources. The Sinharaja forest is therefore an ideal research site for botanical, ethnobotanical and socio-economic studies on tropical rainforests and the services they provide.

This study encompassed estimations of the total inventory stock, the total extractable limit of the inventory stock, the total potential flow and the actual flow of NTFPs. The wild meat flow, the seasonality of NTFPs collection and the biophysical and socio-economic factors influencing the stock and flow of NTFPs were also investigated within the context of a sustainable strategy for NTFPs extraction. The study was carried out on the assumption that NTFPs extraction is, in principle, sustainable and therefore is an economically and environmentally viable option for rainforest biodiversity conservation. On this basis, the study attempted to investigate whether the conservation of the Sinharaja rainforest biodiversity can be economically justified solely in tertns of its non-timber forest products potential.

A range of methods and techniques were deployed including, a botanical inventory survey, an ethnobotanical cross-sectional survey, an ethnobotanical longitudinal survey and an ethnozoological survey. A total of 507 sample points distributed along 12.6 km long representative transects were surveyed using the point-centred quarter method (Causton 1988), a plotless survey, for the botanical inventory. Some 2024 woody trees ≥ 25 cm gbh (girth at breast height) were recorded. A climbers and herbacious species inventory was constructed using 507 sample 25 $m^2$ plots, set up at the same sample point locations used in the woody tree survey. All climbers, saplings and herbaceous species occrring in these plots were recorded. The productive stage of each tree was decided by observing flowers or using local knowledge. The annual yield of each species was estimated by interviewing local people, with the exception of three species whose yield was estimated by counting fruits from a sample of trees. Market prices or marginal shadow prices were used for the economic valuation of the NTFPs. The costs of collection were estimated to be just over 50% of the total value of the given product (Batagoda, 1997).

The ethnobotanical cross-sectional survey used to estimate the actual extracted flow of NTFPs was carned out by interviewing 150 households (17% of total in the area) randomly selected from 12 villages using a structured questionnaire. The survey collected total annual household NTFPs flow data, market prices, costs of collection - labour, capital, processing, and transport and data on socioeconomic background. Thirty two households selected from Kosgulana and Pitakele villages participated in the longitudinal ethnobotanical survey by; entering their daily NTFPs collection in log-books for a period of 9 months from September 1995. The enthnozoological questionnaire survey was carried out by interviewing a sample of 23 households stratified from the 150 households covered in the preliminary survey.

## 4. NTFPs Survey Results

The botanical inventory survey identified 147 species ≥ 15 cm gbh in the Sinharaja forest, of which 96 were found to have some sort of use other than timber for local people. In total, around 74% of the forest species was found to be economically valuable. Based on the intensitity of use, these 96 useful species were classified into major use and minor use. The minor uses were relatively trivial and yielded a low economic value. They were nevertheless included in the analysis in order to capture the full potential economic value of the forest products.

The gross economic flow value of the potential total stock of forest products (based on these 96 species) was found to be US$ 833 $ha^{-1}$ $yr^{-1}$ (net value US$ 440 $ha^{-1}$ $yr^{-1}$) of which the minor use inventory stock accounted for US$ 156 $ha^{-1}$ $yr^{-1}$. The 96 useful species included 33 species that were used as food and 47 species that were used for their medicinal properties. The total food stock flow value was estimated to be US$ 627 $ha^{-1}$ $yr^{-1}$, while the inventory stock flow value of the 47 medicinal species was estimated to be US$ 174 $ha^{-1}$ $yr^{-1}$. The remaining sixteen species were split between six weaving material species and 10 building material species, which were valued at US$ 47 $ha^{-1}$ $yr^{-1}$ and US$ 35 $ha^{-1}$ $yr^{-1}$ respectively. Finally, the total economic value

of the full poential flow of non-timber herbaceous species and climbers was estimated to be US$ 139 $ha^{-1}$ $yr^{-1}$. The gross economic value for commercial forest products was US$ 660 $ha^{-1}$ $yr^{-1}$ and for consumptive products was US$ 173 $ha^{-1}$ $yr^{-1}$ (Batagoda, 1997).

Of the total stock of NTFPs, a certain amount must be left in the forest in order to permit regeneration for sustainable forest resource management. Folloing Peters et al., 1989, in assuming 25% of the total stock is left in the forest for regeneration, the gross economic value of the potentially extractable output flows of forest products was estimated to be US$ 622 $ha^{-1}$ $yr^{-1}$.

However the extractable stock limit of NTFPs is also conditioned by accessibility constraints such as lack of footpaths and the existence of rough terrain. A regression analysis revealed that the higher the travel time required to walk in the forest (higher travel time represents lower accessibility) the lower the total forest products collection. An analysis carried out using a geographic information system (GIS) indicated that some 43% of the Sinharaja forest area remains inaccessible due to difficult terrain and lack of roads. Hence, the gross economic value of the total potential NTFPs flow in the Sinharaja forest, given the accessibility constraints, was estimated to be US$ 377 $ha^{-1}$ $yr^{-1}$ (net value US$ 186 $ha^{-1}$ $yr^{-1}$).

## 4.1 Comparing cross-sectional and log-book surveys

The total gross economic value of the actual forest products flow from the Sinharaja forest as estimated by the cross sectional ethnobotanical survey was US$ 367 per household per year. This is equivalent to US$ 29 per hectare per year. The net total economic value of the actual forest products flow (after allowance for the costs of collection) was estimated to be US$ 181 per household per year, which is equivalent to US$ 14 $ha^{-1}$ $yr^{-1}$'. The costs of collection, (capital, labour, processing and transport costs) was estimated per household per year via a questionnaire survey. The total, per hectare, gross economic values of the commercial and consumptive flow of forest products were found to be US$ 16 $ha^{-1}$ $yr^{-1}$ and US$ 12 $ha^{-1}$ $yr^{-1}$ respectively. The total flows of food forest products and medicinal plants were estimated to be US$ 23 $ha^{-1}$ $yr^{-1}$ and US$ 0.37 $ha^{-1}$ $yr^{-1}$ respectively. The flow of building materials provided a value of US$ $ha^{-1}$ $yr^{-1}$ while the flow of weaving materials was valued at US$ 0.2 $ha^{-1}$ $yr^{-1}$.

The results of the actual forest products flow provided by the longitudinal log book survey were found to be significantly different from those estimated in the cross sectional survey, see Table 3.

The reported food products flow is significantly lower in the log-book survey than in the cross-sectional survey. Conversely, medicinal plants, building materials, and weaving materials values are signiffcantly higher in the logbook survey than in the cross-sectional survey. One reason for this difference is that food collection is carried out over an extensive area on a seasonal basis. Collectors therefore find it easier to recall how much has been collected during the season. when answering cross sectional survey questions. Medicinal plants, building and weaving materials are collected on a smaller scale throughout the year, and collectors often find it more difficult to recall the exact amounts they collected sometime in the past.

## 4.2 Wildmeat flow

The ethnozoological survey of wild animals hunting revealed that the gross economic value of the total wild meat flow from the Sinharaja forest was US$ 31 per household per year, which is equivalent to US$ 2.44 $ha^{-1}$ $yr^{-1}$ (see Table 4). This included US$ 0.97 $ha^{-1}$ $yr^{-1}$ as consumptive wild meat flow and US$ 1.47 $ha^{-1}$ $yr^{-1}$ as commercial wild meat flow. The net wild meat flow was estimated to be US$ 24 per household per year ($ 1.89 $ha^{-1}$ $yr^{-1}$).

## 4.3 Timber potential

The Sinharaja forest contains 48 merchantable size woody tree species, of which 17 are commercially valuable (see Table 5). The annual potential timber harvest from the Sinharaja forest was estimated at 2.6 m3 $ha^{-1}$ $yr^{-1}$ . This estimate was derived from the mean annual volume of timber actually harvested by the Sinharaja Mechanical Logging Project, which was abandoned in 1976. The project actually harvested 51.6 m3 $ha^{-1}$ (Gunatilleke and Gunatilleke, 1983) and if a 20 year logging cycle is assumed, the estimated annual harvest comes out at 2.6 m3 $ha^{-1}$ $yr^{-1}$ . According to the State Timber Corporation of Sri Lanka, the average market price of average timber is RS. 4900 (US$ 89) per cubic metre. The total economic value of the potential timber harvest from the Sinharaja forest was therefore estimated to be US$ 230 $ha^{-1}$ $yr^{-1}$ (gross value) and US$ 115 $ha^{-1}$ $yr^{-1}$ (net value).

## 4.4 Structure and floristic composition of the forest

Both the forest structure and floristic composition are thought to influence the total stock of NTFPs. In order to investigate this relationship, the study area was classified into ten forest types based on phytosociological

**Table 4: Wildmeat flow Values**

| Common Name | Scientifc name | Number of animals killed by (23 house hold) $yr^{-1}$ | Mean Meat per household kg $yr^{-1}$ | Unit Price US$/ kg | Mean meat per household US$ $yr^{-1}$ | Total meat flow US$ $ha^{-1}$ $yr^{-1}$ |
|---|---|---|---|---|---|---|
| Pangolin | *Manis crassicaudata* | 1 | 0.027 | I 09 | 0.03 | 0.0023 |
| Rabbit | Lepus *nigricollis* | 7 | 0.147 | 1.09 | 0.16 | 0.0127 |
| Common palm civet | Paradoxurus *hermaphroditus* | 7 | 0.253 | 1.09 | 0.28 | 0.0220 |
| Grey languor | Presbytis *entellus* | 6 | 0.287 | 1.09 | 0.31 | 0.0248 |
| Mouse deer | Tragulus *meminna* | 7 | . 0.3 | 1.09 | 0.33 | 0.0260 |
| Malabar bandicoot | Bandicota *indica* | 45 | 0.493 | 1.09 | 0.54 | 0.0427 |
| Porcupine | Hystrix indica | 25 | 1.333 | 1.27 | 1.69 | 0.1346 |
| Barlcing deer | *Muintiacus muntjak* | 14 | 2.4 | 1.09 | 2.62 | 0.2080 |
| Sambhur<br>Wild boar | curvus unicolor<br>*Sus scrofa* | 8<br>43 | 3.787<br>14.667 | 1.36<br>1.36 | 4.86<br>20 | 0.3863<br>1.5886 |
| TOTAL | | | 23.693 | | 30.82 | 2.4480 |

characteristics using cluster analysis. The economic valuation analysis revealed that the total NTFPs stock values of these ten forest types (clusters) ranged from US$ 365 to US$ 1697 $ha^{-1}$ $yr^{-1}$ . The NTFPs value map is shown as Figure 1. Regression analysis of the phytosociological characteristics and the total NTFPs flow values of these ten forest types found only a weak (statistically insignificant) correlation between greater species diversity ($R^2$= 0.36), lower species density ($R^2$ = 0.007), lower mean girth size of a forest ($R^2$ = 0.07) and greater total forest product flow value.

## 5. Socio-economic Factors and Harvesting Effort Rates

Geographical factors such as villager accessibility to the forests and to the market place can influence the extraction of the NTFPs. A correlation analysis together with a geographic information system indicated that both accessibility to the forest and accessibility to the market from villages may be positively but weakly correlated with the total NTFPs flow (Batagoda, 1997).

All forest-based households do not extract non-timber forest products on the same scale. Socio-economic characteristics and the ethnobotanical behaviour of villagers may also influence the flow of forest products. The logbook data from 150 households includes data on labour supplied, capital costs, products extracted and so on, allowing the estimation of a production function and a labour supply function.

The data distinguish between male and female labour, so initially these variables were included separately in a linear production function alongside capital and processing costs. The dependent variable was the total value of forest products extracted by a household.

| | Estlmate | SE | Tstat | PValue |
|---|---|---|---|---|
| 1 | -147.719 | 34.9443 | -4.22728 | 0.0000446392 |
| LABOURF | 6.12684 | 1.20161 | 5.09886 | $1.19962x10^{-6}$ |
| LABOURM | 6.23034 | 0.441828 | 14.1013 | 0. |
| CAPITAL | 0.991224 | 0.230345 | 4.30322 | 0.0000331489 |
| PROCCOST | -5.07362 | 1.26869 | -3.9991 | 0.000106849 |

Adjusted $R^2$= 0.748 F = 98.9 (P=0.000) DF=128

**Table 5: Merchantable Timber Species and Their Density**

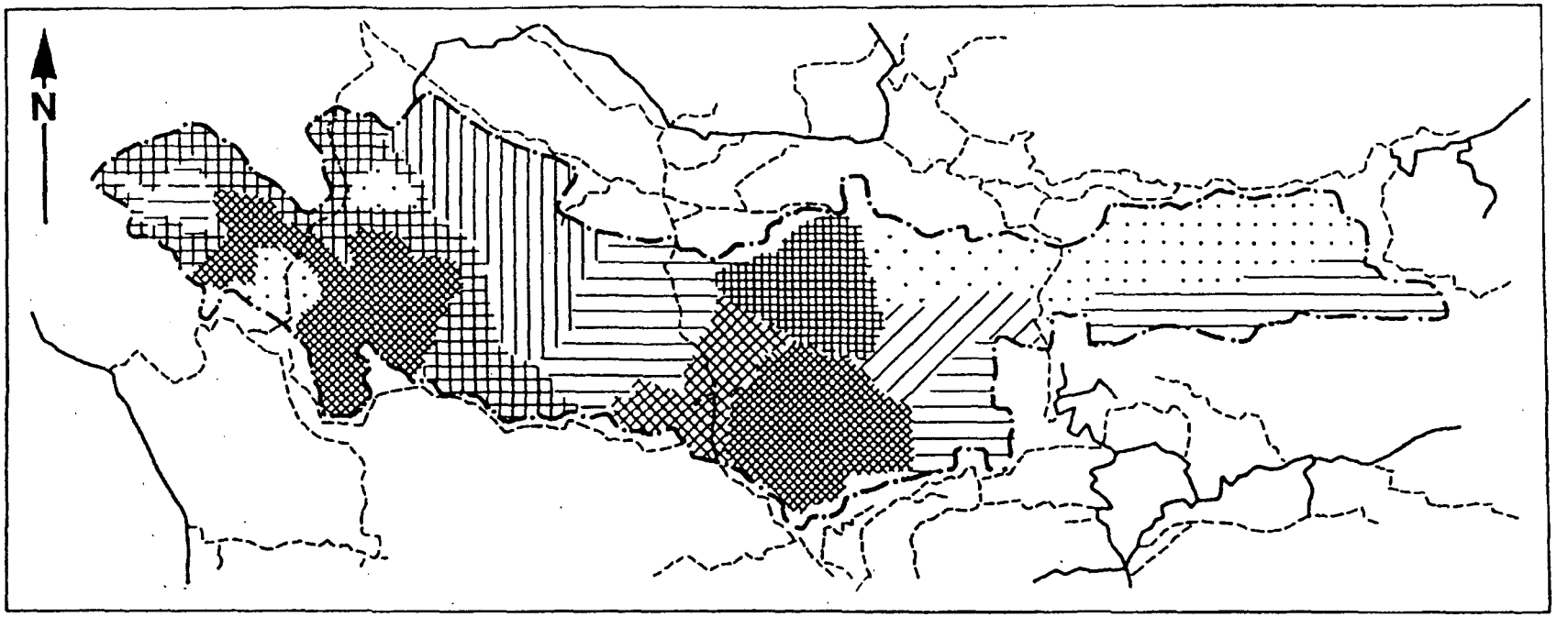
| Local name | Species Name | Density $ha^{-1}$ |
|---|---|---|
| Thiniya | *Doona congestiflora* | 4 |
| Hora | *Dipterocarpus zeylanicus* | 3 |
| Kirihembiliya | *Palaquium grande* | 7 |
| Beraliya | *Coona cordifolia* | 8 |
| Atamba | *Mangifera zeylanica* | 5 |
| Yakahalu | *Doona trapezifolia* | 3 |
| Hampalanda | *Terminalia parriflora* | 0 |
| Na | *Mesua ferrea* | 8 |
| Welipiyenna | *Anisophyllea cinnamomoides* | |
| Keena | *Calophyllum walkeri* | 2 |
| Diyapara | *Wormia triquetra* | 0 |
| Malaboda | *Myristica dactyloides* | 8 |
| Molpedda | *Isonandra lanceolata* | 1 |
| Hedawaka | *Chaetocarpus coriacaus* | 0 |
| Liyan | *Homalium zeylanicum* | 0 |
| Kataboda | *Cullenia ceylanica* | 8 |
| Bu Hora | *Dipterocarpus hispidus* | 1 |
| Total | | 61 |

Where LABOURF and LABOURM are male and female labour respectively, CAPITAL is capital cost and PROCCOST is processing cost. This last is primarily composed of labour costs, and therefore reduces the value gained from a certain level of labour. All the coefficients and the equation overall are highly significant ($P<0.01$), and labour and costs together explain threequarters of the variation in value of forest products extracted However the highly negative constant term is puzzling and may suggest misspecification, which prompted analysis of a Cobb-Douglas relationship.

Male and female labour have almost identical coefficients, which justifies adding these variables in a Cobb-Douglas function: they can not be left separate since neither is essential on its own. The Cobb-Douglas function does not improve the fit, however this is on the basis of a much reduced data set (43 observations) because many households have a zero for either capital or processing cost. The coefficient on the natural logarithm of total labour is significantly greater than unity (1.329, standard error 0.1620) and this is indicative of an increasing return to scale of labour. There are three reasons why this might occur: one is learning by doing, so that those supplying more : labour tend to become more productive, presumably partly through better knowledge of where products are to be found, a second is the reverse - the more productive tend to supply more labour; and the third is that the households supplying more labour may be supplying more individuals per trip, and cooperation may make labour more productive. Various regressions were tried to test these hypotheses, and it seems that both the frst pair and the third play some role, as discussed below.

Appropriately modified Cobb-Douglas regressions were also estimated for the three data sub-sets with one or both cost observation equal to zero. The noteworthy result was that the coefficient on labour was significantly lower for the households with zero capital, suggesting that the presence of capital has a strong effect on labour productivity. This might be expected, since while there is not much capital needed for forest product collection (knives of various sorts being the main kind), these are essential for some forms of collection.

**Figure 1: NTFs Stock Value Distribution in the Sinharaja Forest**



This suggests the strategy of including variables for capital times labour in a linear regression. This has the advantage of keeping all the data together in one set, without the need for arbitrary additions to avoid problems with taking the logarithm of zero, while picking out the effect of capital and processing costs on labour productivity. To capture returns to scale, labour squared was also included. Finally, persons per trip times total labour was included to pick up any co-operation effect.

| | Estimate | SE | Tstat | PValue |
|---|---|---|---|---|
| 1 | 14.785 | 46.7526 | 0.31624 | 0.752365 |
| LABOUR | 1.78067 | 1.09737 | 1.62267 | 0.107263 |
| SQUARELAB | 0.00803391 | 0.00393789 | 2.04016 | 0.0435111 |
| PERSLAB | 0.563905 | 0.203815 | 2.76675 | 0.00655128 |
| CAPLAB | 0.0118965 | 0.00148091 | 8.03325 | $7.10543 \times 10^{-13}$ |
| PROCCOST | -3.35567 | 1.04904 | -3.19879 | 0.00176197 |

AdjustedR-squared=0.831 F-ratio= 125 (P=0.000) DF=121

This regression is an improvement on the initial linear regression. Most of the variation in value is explained, and all the coeffcients are significant at the 5% level, except the constant term which is not significantly different from the expected zero, and the coefficient on labour (P-value 0.11); however it must be recalled that labour is also included in three other variables, in square form and premultiplied by persons and by capital. There is evidence of weakly increasing returns to labour, and of a stronger effect of increasing productivity of labour when more individuals are involved in trips. Capital has a positive influence on labour productivity, while processing costs have a negative effect.

In fact, a further regression separating between male and female labour showed a slightly higher and significant coefficient on male labour, and a slightly lower and insignificant coefficient on female labour, but the difference between the coefficients was not significant, and is probably an artefact of the lower mean level of female labour in the sample. So we do not interpret this as strong evidence for a productivity differential between male and female labour and retain the regression presented. The other coefficients were very stable with respect to this change.

Having constructed a satisfactory production function, it remains to explain the labour supply decision as a function of the various socio-economic household characteristics included in the data set. For this purpose it is important to exclude variables which may proxy labour supply, such as days per year, processing costs or even number of persons participating on trips. However the level of capital ought to influence the labour supply decision, because it changes the productivity of labour. The evidence from the OLS regressions that the coefficients on male and female labour are very similar suggests that there is little sex difference in terms of productivity: data is not available to detect differences in consumption volumes or patterns. Age, education and income levels may all influence exploitation of NTFPs. Distance walked to and through the forest should influence labour supply, since travel is not directly productive. Several dummy variables for attitudes were also available.

Several regressions were conducted to explore these relationships. The most inclusive gives reasonable explanatory power (R-squared = 0.42) and is highly significant overall, but there are a lot of low-significance variables, for example the education variables, with no particular pattern in sign. This could be because these are picking up ages of children as much as anything else. Age howevar does not seem significant, but this could be because the categories available do not give fine resolution: for example the under-18 class may include both unproductive and highly productive individuals.

The best resulting equation overall gives labour supply (days per year) as the function

| | Estimate | SE | Tstat | PValue |
|---|---|---|---|---|
| 1 | 22.5845 | 15.0202 | 1.50361 | 0.135145 |
| FAMSIZE. | 6.13221 | 1.69442 | 3.61907 | 0.000424352 |
| NOEDUCN | -10.5318 | 4.96381 | -2.12171 | 0.0357895 |
| DISTANCE | 17.9968 | 5.66623 | 3.17615 | 0 00187045 |
| SQDISTANCE | - 1.31967 | 0.539164 | -2.44762 | 0.0157362 |
| CAPITAL | 0.181025 | 0.0441001 | 4.10486 | 0.0000715891 |
| PURPOSE | -56.6974 | 7.84051 | -7.23133 | $3.86671 \times 10^{-11}$ |
| POSCHANGE | 14.7424 | 9.49042 | 1.5534 | 0.122797 |
| NEGCHANGE | -13.8082 | 8.61458 | -1.60289 | 0.111423 |

Adjusted R-squared = 0.45. F-ratio = 15.1 (P=0.000) DF=128.

Here FAMSIZE is family size; NOEDUCN is number with no formal education; DISTANCI] and SQDISTANCE are distance to the forest and its square; CAPITAL is the value of capital employed; PURPOSE is a dummy for which 1 = collecting for household, 0 = collecting for household and marketing; POSCHANGE is a dummy for which 1 = respondent perceives increase in NTFP supply; NEGCHANGE is the reverse (the omitted category being those perceiving no change).

All the variables have the expected sign, if it is accepted that EN is largely a proxy for very young children before any formal schooling - both unproductive

and requiring adult attention. Family size (mean 5.8) increases labour supply. Distance has the expected effect that labour supply at first increases with distance and then falls, as the disincentive effect outweighs the increased labour requirement per unit output. Having capital equipment in a household has a strong positive effect as is expected. The purpose variable has a very large effect, with households collecting only for their own use supplying significantly less labour.

The dummy variables for positive and negative change in NTFP supply are interesting: when both are included in the regression, neither is significant, however each is significant if included alone. Clearly to omit one would be a mis-specification so both are included. They have the expected signs households who perceive improving supply conditions supply more labour while those perceiving declining conditions supply less.

Despite the limited number of explanatory variables available, it is possible to explain almost half of the variation in the labour supply of these households, with significant coefficients of the expected sign. There is evidence therefore that the labour supply decisions take into account the various influences on productivity and perceived conditions of supply and demand. For example, while there was disagreement about whether the supply of products was increasing or decreasing (29 increasing, 43 decreasing, 77 no change), the household labour supply decisions were in line with household perceptions.

## 6. Impact on Forest Structure

The extent to which the non-timber forest products can be harvested without adverse effects on the natural population of species is not known. Previous research assumed that non-timber forest products harvesting was sustainable, because it did not disturb the forest as much as logging. The present study a investigated, in a preliminary way, the impact of forest products harvesting on the forest structure and regeneration of species using three sample forest sites: a site close to the villages; a site distant from the villages; and a logged site. The results revealed that species diversity is considerably lower at the adjacent site (34 species) than at the distant site (53 species). By contrast, the density of species over $\geq$ 25 cm gbh has higher at the adjacent site (1133 $ha^{-1}$) than at the distant site (926 $ha^{-1}$). The herbaceous species density was also higher at the adjacent site (29,966 $ha^{-1}$) than at the distant site (19,578 $ha^{-1}$). However, the average basal area per tree was lower at the adjacent site (706 $cm^2$) than at the distant site (816 $cm^2$). A pooled data set has been compiled which will allow for a future more rigorous analysis of clusters of sites (Batagoda 1997).

The analysis of forest structure data indicates that the density of species at the logged site (982 $ha^{-1}$) was lower than at both the adjacent site (1133 $ha^{-1}$) and the distant site (1126 $ha^{-1}$). Sirnilarly, the average basal area per tree which generally indicates the average tree size, was significantly lower at the logged site (489 $cm^2$) than at the adjacent site (706 $cm^2$) and at the distant site (816 $cm^2$). A lower frequency of large trees was found at the logged site than at other two sites. This clearly indicates that the logged forest area had not fully recovered during the 20 years after the cessation of logging operations.

The importance value index is a good indicator of the vegetational importance of species within a stand (Curtis and McIntosh, 1951). The analysis of importance values for the three study sites revealed that the values for major non-timber species were significantly lower at the adjacent site than at the a distant site. For example, the importance value for *Caryota urens* was 1.85 at the adjacent site and 9.95 at the distant site (Batagoda 1997). The importance values for samplings of non-timber species were lower at the adjacent site than at the distant site suggesting a lower regeneration rate at the adjacent site. The weaker adult - sapling relationship for some non-timber species at the adjacent site compared to the distant site, confirmed a lower regeneration rate at the adjacent site. Also a lower frequency of larger trees was found at the adjacent site than at the distant site. This overall species diversity, average basal area per tree, population structure, and regeneration rate are all lower at the adjacent site than at the distant site. This difference may not be solely due to forest products extraction other biophysical factors may also be important. Neverthe less, we can reasonably conclude that there is some prima facie evidence that NTFPs harvesting does adversely effect the forest structure and composition, and that forest products harvesting is not always necessarily sustainable. Much will depend on the selectivity and intensity of the harvesting effort. A survey of local attitudes about the future of the forest products flow revealed that 72% of the local people did not perceive a threat to the future of forest products harvesting. An analysis of the stock-flow relationship indicates that the total actual flow of forest products is less than 5% of the total potential flow and less than 2% of the total stock. None of the non-timber species have been harvested at more than 6% of its total potential flow, except for *Asparagus racemosus* locally known as 'hatawariya'. Some 17% of this species' potential flow has been harvested. This seems to suggest that in practice the prevailing scale and intensity of extraction, as determined by existing economic conditions, is sustainable.

## 7. Aggregation and Scaling Issues

There are substantial difficulties involved in any

meta analysis of the numerous non-timber forest products studies that have been carried out in various countries throughout the world. A major difficulty is that different studies with n ostensibly the same objectives, utilised different stock and/or flows measures. Simple aggregation of their results is therefore strictly precluded. Researchers also used different research methods i.e. cross sectional survey, log book survey and footpath survey, to generate their results.

To make matters even more complex, the biophysical characteristics of study areas are very different. Even though appropriate generic guidelines are followed within a given study site, a comparison of results across biophysical regions may not be meaningful. Most non-timber forest products species are endemic to the country of study. For example, of the 200 non-timber species used in Sri Lanka, 129 are endernic. The benefits and use of these endemic nontimber species are unique to the country, or the region, so that the transfer of such endemic benefits to other regions would be misleading. Also the spatial scale of study areas varies significantly and the generalisation of results derived from a small-scale study area to more extensive areas may not be justifiable. This may be a particular constraint because the analysis of Sinharaja indicates that the species composition and therefore the relevant economic values are far from uniforrnly distributed across the whole forest area

Taking the results only from those studies that are compatible, together with the Sinharaja results, some average NTFPs values (net) per hectare of tropical rainforests were estimated, see Table 6. In cases where estimates were only available in gross value terms, net values were derived assuming the costs of collection to be 50% of the gross value (Batagoda, 1997). When estimate were reported as a range, median values were used.

The estimated economic flow value related to the total stock of all known use species in the Sinharaja was US$ 883 $ha^{-1}$ $yr^{-1}$. The same total value of only some woody non-timber species in the Amazonian forest was estimated by Peters *et al.*, (1989) to be US$ 700 $ha^{-1}$ $yr^{-1}$. but the Peters et al., (1989) study did not include the economic value of herbaceous species and climbers or other minor use values. If all the use species including herbaceous and climbers were included the total values would be similar. It is our contention, however, that the estimated average potential flow value should represent a theoretical upper bound rather than a practical and policy relevant lower bound as claimed by Peters and his colleagues.

It is unrealistic to claim that the total NTFPs stock value is a realistic annual economic flow value. Collectors

**Table 6: Average Non-timber Forest Products Values for Tropical Rainforests**

| Estimate type | Average Value $ha^{-1}$ $yr$ |
|---|---|
| Inventory Stock - Flora | $ 370 net |
| Potential extractable Flor - Flora | $ 135 net |
| Actual Flow - Flora | $ 26 net |
| Actual Flow - Fauna | $ 3 net |

Source: Peters et al. 1989; Pinedo-Vasques et al. 1992; Schwartzman 1989; Nations 1992; Hecht 1992; Padoch and de Jong 1989; Anderson and Ioris 1992; Alcorn 1989; Chopra 1993; Gunatilleke et al. 1993; Ruitenbeek 1989; Thorbjamarson 1991; Ruitenbeek 1988; Wilkie 1989; Caldecott 1987.

do not consider all the potentially available species as real and sustainable economic resources, even though all resources may be used at any given time by them. They use a number of these species only in emergencies, as a last option to survive. The problem with these occasional use species is that we do not know what would happen to their market price if the full potential stock was consistently harvested. There may not be sufficient market demand to meet the supply if the full potential was exploited. No study has attempted to empirically investigate this issue. In the Sinharaja case, less than 5% of the total potential extractable flow is being harvested at present. The present market price however reflects this current supply situation and the likelihood is that the price would certainly go down if the full potential supply was put on the market.

The next question then is why is the actual flow as low as 5% of the total potential extractable flow? Of course the legal restrictions might be one significant reason for this low extraction rate. But it might also be argued that the present actual flow may represent the optimal level (or close to it) of demand for these resources. If this is the case, thc total stock value and total potential flow value have little, or only limited, practical significance. Any economic justiffcation of forest conservation should therefore be related to actual flow value rather than total stock and related annual flow value as suggested by Peters et al., (1989); or potential extractable flow value as suggested by Myers (1989). It is also the case that the actual harvest rate should be sustainable. The Sinharaja study has highlighted the difficulties involved in any sustainability test and fieldwork did discover some significant adverse impacts on the structure and composition of the forest close to the villages around the Sinharaja forest. These impacts could be the result of more extensive extraction of forest products in the past because the current effort rate did not seem to be excessive. More research is required to resolve this issue.

**Table 7: Value in Comparison to Alternative Land Uses**

| Use Option | Gross Economic Value $ha^{-1}$ $yr^{-1}$ | Net Value $ha^{-1}$ $yr^{-1}$ | Net Present Value (NPV) at 8% discount rate $ha^{-1}$ |
|---|---|---|---|
| Potential NTFPs flow | $ 377 | $ 186 | $ 186 (20 years)<br>$ 2275 (50 years) |
| Actual NTFPs flow both flora ($29) and fauna ($2 ) | $ 31 | $ 15 | $ 147 (20 years)<br>$ 183 ( 50 years) |
| Timber potential | $ 230 | $ 115 | $ 1129 (20 years)<br>$1307 (50 Years) |
| Land-clearance value (Cultivatio of tea) | $ 872 | $ 436 | $4281 (20 years) |

*Notes: Comparison was made using only the potential extractable flow and the actual flow; and tea plantation extemality costs not included.*

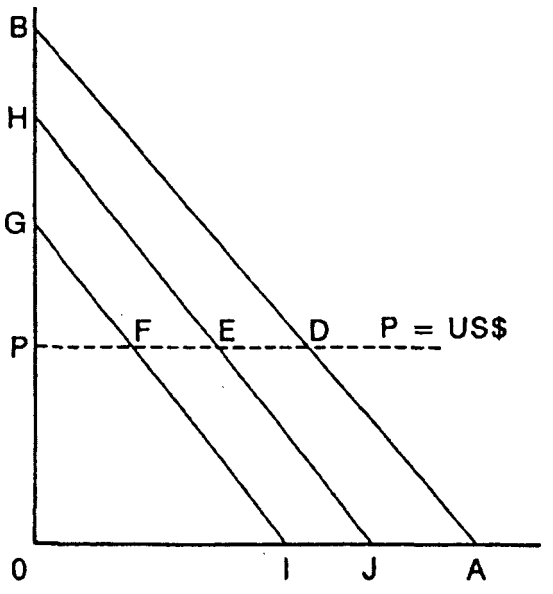
## 8. NTFPs Extraction Versus Other Land Use Options

Some researchers have claimed that NTFPs value itself can exceed both timber use and land-clearance values (Myers, 1988; Peters et al., 1989). This is not a finding that can be supported by the Sinharaja study results. Table 7 presents the economic values of NTFPs extraction in comparison to timber harvest potential value and land-clearance use value (tea cultivation was chosen for this comparison as tea is the popular crop in the area) in the Sinharaja forest. The interviews with farmers indicated that an average income from a hectare of tea in the Sinharaja area is about Rs. 6000 per month and harvesting takes place over a period of eight months per year. Therefore the annual gross income of a - hectare of tea can be estimated at RS. 48,0000 ($ 872). Assuming the costs of tea cultivation is 50% of the total income (based on farmer's responses), the net income of a hectare of tea was estimated to be US$ 436 $ha^{-1}$ $yr^{-1}$.

NTFPs can be collected indefinitely each year if sustainable harvesting practices are adopted. Assuming logging is also carried out in sustainable manner, the periodic revenues from logging follow a 20 year cycle. However, a tea plantation yields an annual revenue over a maximum 20 year period. The net present values (NPV) of all future harvests of NTFPs timber and tea, were estimated by discounting all future revenues at the 8% official Sri Lankan discount rate, over a 20 year and 50 year horizon (NTFPs and timber only). The results given in Table 7 show that the net present values generated by both the timber and the NTFPs harvesting over a 50 year period are lower than the net present value generated by tea plantation during a 20 year period.

If forest land is priced according to the demand for land clearance, the price of one hectare of forest would be at least US$ 436 and the demand curve would lie along the AB line in Figure 2. Then the benefits of land clearance will be PDB. At that price, the demand curve for timber use would be HJ because the net benefits of timber use was US$ 155 $ha^{-1}$ $yr^{-1}$. Therefore the benefits of timber users will be PEH < PDB. Since the benefits of non-timber use was US$ 15 $ha^{-1}$ $yr^{-1}$, the demand curve for non-timber use would lie along the IG line, and the benefits of non-timber users will be PFG < PEH < PDB. If forest conservation is to be economically justified by non-timber forest products potential alone, the benefits of non-timber potential must exceed PDB.

**Figure 2: Demand and Consumer Surpluses**



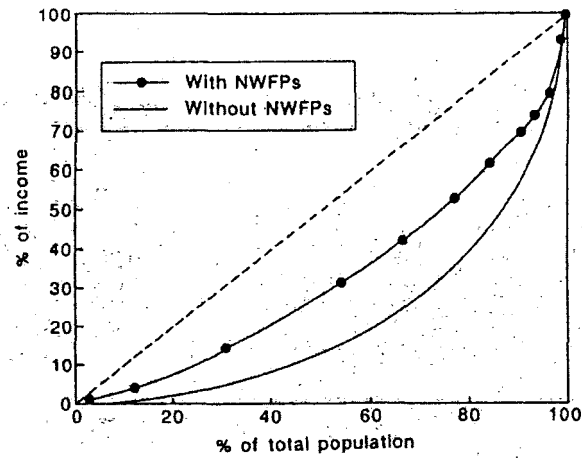
## 9. Income Distribution Effect

Lorenz cune and Gini Coefficients (G) measured with and without a NTFPs contribution were used to measure the income distribution effect (Cowell 1995, 1 Kant 1997). The Lorenz curves estimated with and without NTFPs contributions in the Sinharaja case are shown in Figure 3. The Gini Coefficient (G) was estimated in the following equation, where y = lowest income, $Y_2$= second lowest income, $Y_n$= highest income, and n - total number of households (Cowell 1995).

$$G = \frac{2}{n2y}(Y_{(1)} + 2y_{(2)} + 3y_{(3)} + \ldots + ny_{(n)}) - \frac{n+1}{n}$$

An improvement in the Gini Coefficient was observed (G = 0.29 without NTFPs income and G = 0.29 with NTFPs income) when the income from the total NTFPs was added to the household income of the Sinharaja area indicating that NTFPs harvesting does serve to reduce rural income inequalities.

**Figure3: Lorenz curves with and without contributions of non-wood income**




## 10. Conclusions

The Singharaja Forest analysis indicates that the non-timber extraction value is insufficient on its own to economically and politically justify the conservation of rainforest biodiversity, at least in Sri Lanka and perhaps elsewhere. Of course, the environmental externality costs such as soil erosion, sedimentation, habitat loss etc. associated with tea plantations should also be included in a full economic appraisal of the alternative land use options and this would narrow the gap between the land use options. Further work on the Amazonian forests has also suggested that the best prospects for the sustainable harvesting of plant resources are restricted to oligarchi forests i.e. areas dominated by only one or two tree species (Peters, 1991). But even in these areas social and economic constraints will limit the practicability of "extractive reserves" in Amazonia (Browder, 1991).

Forest biodiversity conservation policy will have to be justified on the basis of a calculus that includes NTFPs flow values, together with other compatible forest function-based flow values (see Table 1). It might also be argued that the conservation strategy encourages other beneficial impacts such as the effect that NTFPs harvesting income has on overall income distribution in local communities. It seems clear that forest biodiversity conservation will have to be justified by a full total economic value (use and non-use values) appraisal, together with other scientific and ethical reasoning and cannot be promoted solely on the basis of non-timber extraction value, as some commentators have in the past claimed.

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12 . **Appendix A: Summary of NTFPs Valuation Study of the Sinharaja Forest of Sri Lanka**

| Local name | Species name | Part used | Density ha$^{-1}$ | Productive density ha$^{-1}$ | Yield per tree kg yr$^{-1}$ | Unit price kg/US$ | Total stock US$ ha$^{-1}$ yr$^{-1}$ | Extractable stock limit US$ ha$^{-1}$ yr$^{-1}$ | Potential flow US$ ha$^{-1}$ yr$^{-1}$ | Actual extracted flow US$ ha$^{-1}$ yr$^{-1}$ | Actual extracted flow US$ Sep | Actual extracted flow US$ Oct | Actual extracted flow US$ Nov | Actual extracted flow US$ Dec | Actual extracted flow US$ Jan | Actual extracted flow US$ Feb | Actual extracted flow US$ Mar | Actual extracted flow US$ Apr | Actual extracted flow US$ May |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| MEDICINE | | | | | | | | | | | | | | | | | | | |
| Angana | *Timonius jambosella* | Leaves | 8.205 | 4.59 | 2.00 | 0.090 | 0.827 | 0.620 | 0.354 | 0.000 | | | | | | | | | |
| Ankenda | *Acronychia pedunculata* | Bark | 8.205 | 6.15 | 2.00 | 0.090 | 1.108 | 0.831 | 0.474 | 0.012 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Aratia | *Ophiorrhiza mungos* | Whole | 42.600 | 42.60 | .20 | 0.182 | 1.549 | 1.162 | 0.662 | | | | | | | | | | |
| Athaketiya | *Xylopia parrifolia* | Leaves | 15.897 | 10.33 | 2.00 | 0.091 | 1.878 | 1.409 | 0.803 | | | | | | | | | | |
| Badulla | *Semecarpus gardneri* | Fruits | 10.769 | 6.68 | 10.00 | 0.090 | 6.009 | 4.507 | 2.569 | | | | | | | | | | |
| Benduru | *Polypodium quercifolium* | Tubers | 3.945 | 1.97 | 4.00 | 0.090 | 0.710 | 0.533 | 0.304 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Bim tal | | Whole | 20.513 | 10.26 | .0012 | 0.180 | 0.002 | 0.002 | 0.001 | | | | | | | | | | |
| Bombu | *Symplocos loha* | Bark | 2.564 | 2.56 | 4.00 | 0.091 | 0.932 | 0.699 | 0.399 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Bulu | *Terminalia bellirica* | Fruits | 1.538 | 1.54 | 10.00 | 0.182 | 2.797 | 2.098 | 1.196 | | | | | | | | | | |
| Dambu | *Syzygium gardneri* | Bark | 1.026 | 1.03 | 5.00 | 0.091 | 0.466 | 0.350 | 0.199 | | | | | | | | | | |
| Dawata | *Carallia brachiata* | Bark | 2.051 | 2.05 | 4.00 | 0.091 | 0.746 | 0.559 | 0.319 | | | | | | | | | | |
| Dorana | *Dipterocarpus glandulosus* | latex | .513 | .51 | 5.00 | 1.455 | 3.729 | 2.797 | 1.594 | 0.091 | | | | | | | | | |
| Illa-patta | *Syzygium cylindricum* | Whole | 8.205 | 5.66 | 2.00 | 0.090 | 1.019 | 0.764 | 0.436 | | | | | | | | | | |
| Iriya | *Horsfieldia irya* | Bark | 1.538 | 1.03 | 4.00 | 0.090 | 0.371 | 0.278 | 0.159 | | | | | | | | | | |
| Ginihiriya | *Exacum trinerva* | Whole | 21.300 | 21.30 | .20 | 0.273 | 1.162 | 0.871 | 0.497 | | | | | | | | | | |
| Gondiwa | | Leaves | 267.456 | 133.73 | .02 | 0.090 | 0.241 | 0.181 | 0.103 | | | | | | | | | | |
| Hatawariya | *Asparagus racemosus* | Whole | 13.41 | 6.71 | .50 | 0.180 | 0.603 | 0.453 | 0.258 | 0.104 | 0.004 | 0.007 | 0.006 | 0.007 | 0.006 | 0.005 | 0.005 | 0.005 | 0.001 |
| Hondala | *Adenia palmata* | Tuber | 0.789 | .39 | 5.00 | 0.090 | 0.178 | 0.133 | 0.076 | | | | | | | | | | |
| Karanda | *Pongamia pinnata* | Bark | 4.615 | 2.58 | 5.00 | 0.091 | 1.175 | 0.881 | 0.502 | | | | | | | | | | |
| Katuimbul | *Salmalia malabarica* | " | .513 | .51 | 5.00 | 0.090 | 0.231 | 0.173 | 0.099 | | | | | | | | | | |
| Kekiriwara | *Axinandra zeylanica* | Whole | 20.512 | 12.31 | 2.00 | 0.090 | 2.215 | 1.661 | 0.947 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Kekuna | *Canarium zeylanicum* | Seeds | 2 | 1 | 15.00 | 0.091 | 1.399 | 1.049 | 0.598 | 0.001 | 0.001 | 0.000 | 0.001 | 0.001 | 0.000 | 0.001 | 0.003 | 0.000 | 0.000 |
| Kirihambiliya | *Palaquium grande* | Seeds | 43.075 | 26.28 | 5.00 | 0.091 | 11.944 | 8.958 | 5.106 | | | | | | | | | | |
| Kokun | *Kokoona zeylanica* | Bark | 3.077 | 3.08 | 5.00 | 0.364 | 5.594 | 4.196 | 2.392 | 0.008 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.008 | 0.000 | 0.000 |
| Kudadawala | *Neolitsea cassia* | Leaves | 1.026 | 1.03 | 5.00 | 0.091 | 0.466 | 0.350 | 0.199 | | | | | | | | | | |
| Kudumirissa | *Thalassia hemprichii* | " | 7.101 | 3.55 | 0.4 | 0.180 | 0.256 | 0.192 | 0.109 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 |
| Madol | *Garcinia echinocarpa* | Seeds | 84.613 | 51.61 | 5.00 | 0.090 | 23.226 | 17.420 | 9.929 | | | | | | | | | | |
| Malaboda | *Myristica dactyloides* | Fruits | 44.101 | 32.63 | 8.00 | 0.090 | 23.497 | 17.623 | 10.045 | | | | | | | | | | |
| Na | *Mesua ferrea* | Leaves | 59.998 | 37.80 | 5.00 | 0.091 | 17.181 | 12.886 | 7.345 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 |
| Na imbul | *Pometia tomentosa* | " | 1.026 | 1.03 | 3.00 | 0.091 | 0.280 | 0.210 | 0.120 | | | | | | | | | | |
| Nawada | *Shorea stipularis* | Bark | 15.384 | 9.69 | 10.00 | 0.090 | 8.723 | 6.542 | 3.729 | 0.040 | 0.010 | 0.003 | 0.007 | 0.001 | 0.000 | 0.002 | 0.013 | 0.006 | 0.000 |
| Nuga | *Ficus benghalensis* | " | .513 | .51 | 10.00 | 0.018 | 0.093 | 0.070 | 0.040 | | | | | | | | | | |
| Pihimbiya | *Filicium decipiens* | Leaves | 1.026 | .51 | 4.00 | 0.091 | 0.186 | 0.140 | 0.080 | | | | | | | | | | |
| Potawel | *Pothos scandens* | " | 87.574 | 43.79 | .05 | 0.090 | 0.197 | 0.148 | 0.084 | 0.003 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Puwak | *Areca catechu* | Fruits | 1 | .51 | 8.00 | 0.364 | 1.492 | 1.119 | 0.638 | | 0.000 | 0.053 | 0.000 | 0.000 | 0.021 | 0.000 | 0.000 | 0.026 | 0.000 |
| Radaliya | *Connarus monocarpus* | " | 168.836 | 84.42 | .20 | 0.090 | 1.520 | 1.140 | 0.650 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Rathkeliya | *Litsea longifolia* | Whole | 4.102 | 2.05 | 3.00 | 0.182 | 1.119 | 0.839 | 0.478 | | | | | | | | | | |
| Rukattana | *Alstonia scholaris* | Bark | 1.026 | .51 | 3.00 | 0.091 | 0.140 | 0.105 | 0.060 | | | | | | | | | | |
| Sadikka | *Myristica fragrande* | Seeds | 3 | 1 | 15.00 | 0.090 | 0.831 | 0.623 | 0.355 | | | | | | | | | | |
| Samadara | *Samadera indica* | Bark | .513 | .51 | 3.00 | 0.180 | 0.277 | 0.208 | 0.118 | | | | | | | | | | |

# Appendix A: Summary of NTFPs Valuation Study of the Sinharaja Forest of Sri Lanka...../continued

| Local name | Species name | Part used | Density $ha^{-1}$ | Productive density $ha^{-1}$ | Yield per tree kg $yr^{-1}$ | Unit price kg/US$ | Total stock US$ $ha^{-1}$ $yr^{-1}$ | Extractable stock limit US$ $ha^{-1}$ $yr^{-1}$ | Potential flow US$ $ha^{-1}$ $yr^{-1}$ | Actual extracted flow US$ $ha^{-1}$ $yr^{-1}$ | Actual extracted flow US$ Sep | Actual extracted flow US$ Oct | Actual extracted flow US$ Nov | Actual extracted flow US$ Dec | Actual extracted flow US$ Jan | Actual extracted flow US$ Feb | Actual extracted flow US$ Mar | Actual extracted flow US$ Apr | Actual extracted flow US$ May |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Foods** | | | | | | | | | | | | | | | | | | | |
| Alamba | *Mangifera zeylanica* | Fruits | 23 | 18 | 80.00 | 0.009 | 12.834 | 9.626 | 5.487 | 0.003 | 0.000 | 0.000 | 0.001 | 0.001 | 0.000 | 0.001 | 0.001 | 0.000 | 0.001 |
| Alubo | *Syzygium makul* | " | 6 | 5 | 40.00 | 0.018 | 3.364 | 2.523 | 1.436 | | | | | | | | | | |
| Beraliya | *Doona cordifolia* | " | 57 | 20.00 | 10.00 | 0.545 | 13.636 | 10.227 | 5.830 | 0.401 | 0.118 | 0.022 | 0.012 | 0.008 | 0.016 | 0.010 | 0.010 | 0.000 | 0.000 |
| Gahiviti | | Leaves | 32 | 8.2 | .50 | 0.090 | 0.369 | 0.277 | 0.158 | 0.001 | | | | | | | | | |
| Gonala | *Diospyros insignis* | Tuber | 2 | .8 | 2.00 | 0.180 | 0.288 | 0.216 | 0.123 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 |
| Gonika | *Hoya ovalifolia* | Leaves | 472.58 | 94.00 | .20 | 0.090 | 1.692 | 1.269 | 0.723 | 0.025 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 |
| Hal | *Vateria copallifera* | Fruits | 3 | .64 | 20.00 | 0.545 | 6.993 | 5.245 | 2.989 | 0.400 | 0.037 | 0.055 | 0.000 | 0.004 | 0.008 | 0.012 | 0.000 | 0.012 | 0.000 |
| Himbutu | *Salacia reticulata* | " | 49 | 12.3 | 3.00 | 0.090 | 3.308 | 2.481 | 1.414 | 0.003 | 0.005 | 0.000 | 0.000 | 0.001 | 0.005 | 0.000 | 0.000 | 0.001 | 0.000 |
| Katuboda | *Cullenia ceylanica* | " | 43 | 29 | 20.00 | 0.009 | 5.286 | 3.965 | 2.260 | | | | | | | | | | |
| Miyanakoku | | Leaves | 2 | 1.8 | .50 | 0.090 | 0.081 | 0.061 | 0.035 | 0.051 | 0.003 | 0.001 | 0.002 | 0.002 | 0.002 | 0.001 | 0.002 | 0.001 | 0.000 |
| Mora | *Euphoria longana* | Fruits | 5 | 2.58 | 10.00 | 0.364 | 9.398 | 7.049 | 4.018 | 0.040 | 0.004 | 0.008 | 0.000 | 0.008 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Mushroom | Fungi | Whole | 6 | 3.0 | 5.00 | 0.730 | 10.950 | 8.213 | 4.681 | 0.766 | 0.216 | 0.078 | 0.055 | 0.009 | 0.108 | 0.149 | 0.074 | 0.299 | 0.144 |
| Kaluaala | *Languas galanga* | Tuber | 2 | 1.8 | 2.00 | 0.090 | 0.324 | 0.243 | 0.139 | | | | | | | | | | |
| Kara | *Canthium coromandelicam* | Leaves | 1 | 1 | 4.00 | 0.273 | 0.559 | 0.420 | 0.239 | 0.008 | | | | | | | | | |
| Katukitul | *Oncosperma fasciculatum* | Pith | 4 | 1.56 | 3.00 | 0.218 | 1.020 | 0.765 | 0.436 | 0.012 | 0.000 | 0.000 | 0.001 | 0.001 | 0.002 | 0.003 | 0.000 | 0.005 | 0.000 |
| Katukenda | *Scolopia acuminata* | Fruits | 5 | 1 | 5.00 | 0.009 | 0.046 | 0.035 | 0.020 | | | | | | | | | | |
| Keena | *Calophyllum walkeri* | " | 39 | 28 | 10.00 | 0.009 | 2.490 | 1.868 | 1.065 | | | | | | | | | | |
| Kitul | *Caryota urens* | Sap | 17 | 4.00 | 450.00 | 0.270 | 486.000 | 364.500 | 207.765 | 19.898 | 0.707 | 0.568 | 0.421 | 0.066 | 0.176 | 0.096 | 0.168 | 0.079 | 0.013 |
| Kohila | *Lasia spinosa* | Tubers | 1 | .4 | 2.00 | 0.180 | 0.144 | 0.108 | 0.062 | 0.085 | 0.005 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.001 | 0.000 | 0.000 |
| Madan | *Syzygium aromaticum* | Fruits | 2 | 1.8 | 2.00 | 0.090 | 0.324 | 0.243 | 0.139 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 | 0.002 |
| Thebu | *Costus speciosus* | Leaves | 18 | 9.2 | 1.00 | 0.090 | 0.828 | 0.621 | 0.354 | 0.074 | 0.006 | 0.002 | 0.004 | 0.022 | 0.007 | 0.005 | 0.017 | 0.001 | 0.000 |
| Tholol | *Trichadenia zeylanica* | Fruits | 6 | 5.05 | 10.00 | 0.100 | 5.046 | 3.784 | 2.157 | 0.023 | 0.011 | 0.011 | 0.002 | 0.002 | 0.000 | 0.000 | 0.004 | 0.000 | 0.005 |
| Wal del | *Artocarpus nobilis* | " | 7 | 3.87 | 10.00 | 0.100 | 3.867 | 2.900 | 1.653 | 0.152 | 0.024 | 0.000 | 0.005 | 0.012 | 0.011 | 0.003 | 0.000 | 0.000 | 0.005 |
| Wal divul | *Hydnocarpus octandra* | " | 16 | 12 | 30.00 | 0.009 | 3.348 | 2.511 | 1.431 | | | | | | | | | | |
| Wal jambu | *Zyzygium firmum* | " | 12 | 6 | 10.00 | 0.009 | 0.532 | 0.399 | 0.227 | | | | | | | | | | |
| Wal Rambutan | | " | 8 | 5 | 15.00 | 0.182 | 12.531 | 9.398 | 5.357 | 0.001 | 0.001 | 0.000 | 0.000 | 0.003 | 0.001 | 0.000 | 0.005 | 0.004 | 0.000 |
| Weralu | *Elaeocarpus serratus* | " | 1 | .26 | 20.00 | 0.182 | 0.932 | 0.699 | 0.399 | 0.006 | 0.005 | 0.011 | 0.000 | 0.003 | 0.010 | 0.001 | 0.001 | 0.000 | 0.000 |
| **Total foods** | | | | | | | 586.191 | 439.643 | 250.597 | 21.950 | 1.142 | 0.758 | 0.502 | 0.141 | 0.345 | 0.283 | 0.284 | 0.401 | 0.170 |
| **SPICES** | | | | | | | | | | | | | | | | | | | |
| Wal Kurundu | *Cinnamomum multiflorum* | Bark | 1 | .05 | .50 | 2.727 | 0.070 | 0.052 | 0.030 | 0.118 | 0.000 | 0.004 | 0.000 | 0.000 | 0.000 | 0.000 | 0.010 | 0.069 | 0.000 |
| Kapuru kurundu | | " | 2 | 1 | .30 | 2.727 | 1.124 | 0.843 | 0.481 | | | | | | | | | | |
| Ensal | *Elettara ensal* | Seeds | 133 | 13.0 | .30 | 3.640 | 14.196 | 10.647 | 6.069 | 0.589 | 0.469 | 0.368 | 0.184 | 0.028 | 0.066 | 0.037 | 0.185 | 0.295 | 0.000 |
| Goraka | *Garcinia cambogia* | Fruits | 17 | 4.00 | 3.00 | 1.800 | 21.600 | 16.200 | 9.234 | 0.848 | 0.066 | 0.082 | 0.082 | 0.000 | 0.000 | 0.000 | 0.123 | 0.082 | 0.033 |
| **Total spices** | | | | | | | 36.990 | 27.743 | 15.813 | 1.556 | 0.534 | 0.454 | 0.266 | 0.028 | 0.066 | 0.037 | 0.318 | 0.446 | 0.033 |

## Appendix A: Summary of NTFPs Valuation Study of the Sinharaja Forest of Sri Lanka...../continued

| Local name | Species name | Part used | Density ha$^{-1}$ | Productive density ha$^{-1}$ | Yield per tree kg yr$^{-1}$ | Unit price kg/US$ | Total stock US$ ha$^{-1}$ yr$^{-1}$ | Extractable stock limit US$ ha$^{-1}$ yr$^{-1}$ | Potential flow US$ ha$^{-1}$ yr$^{-1}$ | Actual extracted flow US$ ha$^{-1}$ yr$^{-1}$ | Actual extracted flow US$ Sep | Actual extracted flow US$ Oct | Actual extracted flow US$ Nov | Actual extracted flow US$ Dec | Actual extracted flow US$ Jan | Actual extracted flow US$ Feb | Actual extracted flow US$ Mar | Actual extracted flow US$ Apr | Actual extracted flow US$ May |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Selaginela | | Leaves | 570.410 | 285.21 | .02 | 0.180 | 1.027 | 0.770 | 0.439 | | | | | | | | | | |
| Thiniya | *Doona congestiflora* Thw. | Resin | 29.230 | 7.00 | 2.00 | 0.727 | 10.182 | 7.636 | 4.353 | 0.128 | 0.077 | 0.064 | 0.028 | 0.074 | 0.121 | 0.037 | 0.066 | 0.312 | 0.074 |
| Walgammiris | *Piper argyrophyllum* | Leaves | 1253.649 | 626.82 | .10 | 0.090 | 5.641 | 4.231 | 2.412 | 0.001 | | | | | | | | | |
| Walinguru | *Zingiber cylindricum* | Tuber | 382.643 | 191.32 | .01 | 0.180 | 0.344 | 0.258 | 0.147 | 0.005 | | | | | | | | | |
| Wallagiriya | | Leaves | 3.156 | 1.58 | .10 | 0.180 | 0.028 | 0.021 | 0.012 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.002 |
| Weniwel | *Coscinium fenestratum* | Vine | 195.660 | 20.00 | 3.00 | 0.182 | 10.909 | 8.182 | 4.664 | 0.035 | 0.009 | 0.009 | 0.029 | 0.019 | 0.036 | 0.017 | 0.047 | 0.029 | 0.005 |
| Yakahaludun | *Shorea trapezifolia* | Resin | 42.563 | 10.00 | 3.00 | 0.727 | 21.818 | 16.364 | 9.327 | 0.128 | 0.106 | 0.060 | 0.037 | 0.087 | 0.363 | 0.124 | 0.207 | 0.255 | 0.124 |
| **Total medicine** | | | | | | | 176.338 | 132.253 | 75.384 | 0.562 | 0.206 | 0.196 | 0.107 | 0.188 | 0.547 | 0.186 | 0.349 | 0.634 | 0.207 |
| WEAVING AND BUILDING MATERIALS | | | | | | | | | | | | | | | | | | | |
| Bambarawel | *Dalbergia pseudo-sissoo* | Vine | 372.39 | 37.00 | 6.00 | 0.009 | 1.998 | 1.499 | 0.854 | 0.329 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Bata | *Ochlandra stridula* | Leaves | 617.75 | 61.00 | 3.00 | 0.055 | 10.065 | 7.549 | 4.303 | 0.081 | 0.000 | 0.010 | 0.033 | 0.018 | 0.015 | 0.012 | 0.043 | 0.039 | 0.001 |
| Beru | *Oplismenue compositus* | " | 1361.74 | 136.00 | 3.00 | 0.055 | 22.440 | 16.830 | 9.593 | 0.077 | 0.010 | 0.010 | 0.024 | 0.028 | 0.030 | 0.043 | 0.031 | 0.043 | 0.013 |
| Kalawel | *Derris uliginosa* | Vine | 11.05 | 2.76 | 4.00 | 0.009 | 0.099 | 0.075 | 0.042 | | | | | | | | | | |
| Kalubambarawel | *Dalbergia pseudo-sissoo* | " | .79 | .20 | 5.00 | 0.009 | 0.009 | 0.007 | 0.004 | | 0.003 | 0.004 | 0.005 | 0.003 | 0.010 | 0.003 | 0.000 | 0.000 | 0.004 |
| Kirindiwel | *Rourea minor* | " | 59.96 | 14.99 | 3.00 | 0.009 | 0.405 | 0.304 | 0.173 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Kiriwel | *Ichnocarpus frutescens* | " | 6.31 | 1.58 | 3.00 | 0.009 | 0.043 | 0.032 | 0.018 | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Kukuluwel | *Calmus radiatus* | " | 50 | 5 | 2.00 | 0.091 | 0.909 | 0.682 | 0.389 | 0.058 | 0.000 | 0.000 | 0.001 | 0.005 | 0.020 | 0.000 | 0.020 | 0.005 | 0.008 |
| Ma bandawel | | " | 2.37 | .59 | 3.00 | 0.009 | 0.016 | 0.012 | 0.007 | | | | | | | | | | |
| Okeyya | *Pandanas ceylanicus* | Leaves | 1 | 1 | 1.00 | 0.091 | 0.091 | 0.068 | 0.039 | | | | | | | | | | |
| Pattikka | | Vine | 68.64 | 17.16 | 2.00 | 0.009 | 0.309 | 0.232 | 0.132 | | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.004 | 0.000 | 0.000 | 0.000 |
| Puswel | *Entada phaseoloides* | " | .513 | .51 | 5.00 | 0.182 | 0.466 | 0.350 | 0.199 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 |
| Walla | *Gyrinops walla* | " | 10.769 | 5.92 | 3.00 | 0.091 | 1.615 | 1.211 | 0.691 | | | | | | | | | | |
| Wewel | *Calamus rotang* | " | 836 | 41 | 3.00 | 0.364 | 44.727 | 33.545 | 19.121 | 0.088 | 0.000 | 0.147 | 0.001 | 0.013 | 0.080 | 0.042 | 0.079 | 0.026 | 0.009 |
| Wiyakeyya | | Leaves | 4 | 1 | 1.00 | 0.091 | 0.091 | 0.068 | 0.039 | | | | | | | | | | |
| Wetake | *Pandanus tectorius* | " | 15 | 3 | 3.00 | 0.091 | 0.818 | 0.614 | 0.350 | 0.031 | | | | | | | | | |
| **Total weaving and building materials** | | | | | | | 84.101 | 63.076 | 35.953 | 0.666 | 0.013 | 0.171 | 0.065 | 0.069 | 0.155 | 0.104 | 0.173 | 0.113 | 0.035 |
| GRAND TOTAL | | | | | | | 883.62 | 662.72 | 377.75 | 24.734 | 1.896 | 1.579 | 0.939 | 0.425 | 1.114 | 0.611 | 1.124 | 1.594 | 0.445 |

Note: Sri lankan Rupees (RS) were converted to US$ using current (1997) exchange rate (1 US$ = RS 55)

# Chapter 8

# WATER RESOUECE MANAGEMENT OF VALUE OF WATER SUPPLY AND QUALITY

*by*

***Richard Boyd***

## 1 INTRODUCTION

Estimates of the economic benefits relating to **water management decisions** are useful for several types of allocation decisions, with probably the most obvious being decisions relating to investments in **structural** approaches to water management. Countries world-wide continue to make investment in water resources one of the most important components of their public infrastructure budgets. Water related investments; in irrigation, hydropower, urban and rural water supply, flood control and sanitation, have been designed to contribute to economic development and public welfare.

Economic evaluation is important because it helps identify whether the population wants the proposed projects to go ahead, and to estimate the degree to which they are willing to pay for the benefits. With prevailing constrained public budgets, conceptually correct and empirically valid estimates of the economic contribution of water in each water using sector are essential for making economically sound investment decisions.

Another class of decisions for which economic values of water are useful are those evaluating **non-structural** or policy options. For example, as the demand for fresh water increases against the finite supply, estimates of the economic value of water are useful in the context of optimal allocation of water between water using purposes and sectors.

In the context of water management decisions, economic evaluation can be used to answer questions such as:

- How are needs to develop added food supplies to be balanced with the wish to preserve watercourses or wetlands for fish and wildlife habitat?
- How are wants for hydroelectric power generation and other instream uses to be balanced against demands for water from cities and farms?
- How much ground water versus how much surface water should be withdrawn to meet current water demands?
- How much treatment should be given to wastes discharged into watercourses?
- Taking another consideration, one related to cost recovery, how much can beneficiaries afford to pay for water supplies?

For each of these issues, estimates of the net economic contribution of the water resource are important for decisions.

Because of the limited role played by market forces in the allocation of water however, market prices on which to base water-related resource allocation decisions are seldom available. In the jargon of the economist, **shadow prices** - reflecting the value of water - must be developed in their place. In recent decades economists have developed a number of techniques for measuring the economic values or benefits associated with non-market allocation. The theoretical foundations of non-market economic valuation of environmental resources has been well developed (see, e.g. Freeman, 1993; and Braden and Kolstad, 1991). Progress with methods for estimating economic benefits in actual cases, is also well advanced (see, e.g. Markandya *et al*, 2000).

Against this background, the purpose of this Chapter is to provide a review, exposition and assessment of the concepts and methods for estimating the economic benefits (value of water) of allocation decision involving water. The aim is to provide a practical, yet sound, set of the main procedures (methods) to help guide the economic analysis of such decisions.

The Module is structured as follows. Section 2 presents a conceptual framework for valuing (changes in) water supply. It begins by outlining the key concepts and terms that you will encounter in water resource valuation. The need for economic valuation in the context of water supply projects and policies is then defined, followed by a description of the main methods for valuing water as an intermediate good and a final consumption good. The quality of water also influences its economic value. Hence, a conceptual framework for valuing changes in water quality is presented in Section 3. The role of valuation within this framework is discussed, and key source of economic (benefit transfer) data identified.

# 2 CONCEPTUAL FRAMEWORK FOR VALUING CHANGES IN WATER SUPPLY

## 2.1 Introduction

As we noted above, in a water resource management context (e.g. strategies for river basins), the principal opportunities for economic welfare enhancement, and hence the need for measures of water value (benefits), are:

1) investments in capturing, storing, delivering and treating new water supplies; and
2) reallocation among water-using sectors.

In this section we explain the specific need for (role of) economic valuation in both the above contexts. This is followed by a review of the main techniques for deriving measures of water value. Before we begin however, it would be useful to outline some of the key concepts and terms that you will encounter.

## 2.2 Some Key Concepts and Deffnitions

It will become apparent during the course of this section that the valuation of water resources requires a multidisciplinary approach. The inherent multidisciplinary nature of water valuation can be problematic, however. One problem area which can cause potential confusion, concerns water valuation terminology, for which there exists no common taxonomy. Moreover, no one set of terms is obviously superior.

### 2.2.1 Water Resources as Natural Assets

Water, whether in aquifers, lakes, rivers, or estuaries, can be considered as a natural asset. As with man-made assets, the value of water resides in its ability to generate flows of services over time. The services provided by water resources, and hence the value or benefit of water to humankind, can be grouped into four broad categories, namely (using the taxonomy suggested by Young, 1996):

- commodity benefits;
- public and private aesthetic and recreational values;
- waste assimilation benefits;
- and dis-benefits or damages.

Each of these categories clearly involves economic considerations, because they are characterised by increasing scarcity, and the associated problems of allocating resources among competing uses.

The **commodity benefits** are those derived from personal drinking, cooking and sanitation, and those contributing to productive activities. What are here called commodity values are distinguished by the fact of being rival in use, meaning that one person's use of a unit of water necessarily precludes use by others of that unit. Commodity uses tend to be private goods or services. Additional distinctions can be made, for example, those types of human uses of water which normally take place away from the natural hydrological system, and which may also be called **extractive** (withdrawal or off-stream) **uses**. Since extractive uses typically involve at least partial depletion or consumption they may further be distinguished as **consumptive uses.**

Other types of economic commodity values associated with water may not require it to leave the natural hydrological system. This group may be labelled **in-situ** (in-stream) **uses** and include hydroelectric power generation and waterways transportation. Since in-situ use often involves little or no physical loss, they are also sometimes called **non-consumptive uses.**

The economic benefits from water for recreation, aesthetics, and fish and wildlife habitat are a second group or type of **in-situ** value of water. However, benefits in this category are closer to the non-rival end of the spectrum. In developing nations, water-based recreational activities are becoming more important for their own citizens, and also often provide a basis for attracting the tourist trade. Significant in-situ values are also found as habitat for wildlife, and fish are the basis for sporting activities.

The economic benefits of **waste disposal** is a third general class of economic benefits of water use. Bodies of water are considered as a sink for carrying away a wide range of residuals from processes of human production and consumption. Water resources are also used for disposing of waste, diluting them, and for some substances, aiding in processing them into less undesirable forms. They are therefore significant for what is called their **assimilative capacity.**

Dis-values (or damages) of water represent an important related classification. Flood waters or excesses of pollutants reduce value. Conversely, reduction in dis-benefits increases human welfare. In such cases, mitigation policies may be assessed by valuing the reduction in damages.

**Non-use values** are also an important consideration in water allocation, and for the economic valuation of water. It is observed in addition to valuing the commodity benefits of water use, that people are willing to pay for environmental services they might neither use nor experience. Voluntary contributions towards preserving an endangered fish species represents an example.

Because of differing conceptual frameworks, an additional useful distinction can be drawn between **intermediate goods** and **final consumption goods.** Intermediate goods are employed to make final products. Intermediate goods represent the largest class of off-stream uses of water. Examples include water for crop irrigation, industrial processing and hydroelectric power generation. Consumption goods are those providing human satisfactions. Residential water is an example of a final consumption good, and recreation and amenity services. The importance of this distinction between intermediate and consumer goods is that the economic theory of profit-maximising producer provides the conceptual framework for the valuation of intermediate goods, while the theory of the individual consumer is the basis for valuing consumer goods.

The above valuation terminology is displayed in Table 1 below. The relationship between the this taxonomy of water resource benefits and standard economic valuation terminology is also depicted.

Table 1 also displays the relationship between the two taxonomies and the **stocks** and **flows** of water resources. The distinction between values associated with a flow or stream of goods/services and values which are associated with stocks or assets which generate those streams, is also important. Use values tend to occur over time, thereby producing flows or streams of value. Stock values, in contrast, represent the value of an asset which yields flows of value over time. The process of discounting allows us to link flow and stock values: a stream of values (costs or benefits) can be converted to a stock value by computing the present value of the flow. Equally, discounting operations can be used to convert stock values into a stream of equal flow values. Accounting for this distinction is important if we are to avoid double-counting or other similar errors when computing the value of water resources. In addition, the presence of flow values necessitates that assumptions are adopted concerning the time horizon of the economic analysis.

**Table 1: Water Valuation Terminology**

| **Physical Terminology** | **Standard Economic Terminology** | **Accounting Terminology** | |
|---|---|---|---|
| | | Stocks | Flows |
| **Commodity Values** (rival in use): | | | |
| Extractive (consumptive): | | | |
| Public water supply (final) | | | * |
| Agricultural uses (inter) | | | * |
| Industrial uses (inter) | | | * |
| Other consumptive uses | | | * |
| *In-situ* (non-consumptive): | | | |
| Hydropower (inter) | **Use Value** | | * |
| Navigation (final) | | | * |
| ***In-situ* Values:** | | | |
| Recreational values (final) | | | * |
| Waste disposal (inter) | | | * |
| Ecological values (final) | | * | * |
| Buffer values (final) | | * | * |
| Existence values (final) | | | * |
| Non-use Value | **Non-use Value** | | * |
| Bequest values (final) | | | * |

*Source: Adapted from NRC (1997)*

## 2.3 The Need for Economic Valuation[1]

### 2.3.1 Appraising Investment in Water Supply

Consider an investment in a water project, which is being assessed for its economic feasibility. In general, the economic feasibility criterion can be written as:

$$PVNB = \sum_t \left[\frac{\sum_i B_{it}}{(1+r)^t}\right] - \sum_t \left[\frac{C_t}{(1+r)^t}\right] - \sum_t \left[\frac{D_{jt}}{(1+r)^t}\right]$$

where

PVNB = The present value net benefits of project or policy

$B_{it}$ The incremental benefit for water use or availability in sector *i* in year *t*.

$C_t$ = The investment expenditure and recurring costs in year *t*.

$D_{jt}$ = The incremental project-induced dis-benefit (foregone benefits or external costs) to sector *j* in year *t*.

*r* = The chosen discount rate.

The acceptability of the water supply project - i.e. the Potential Pareto Improvement criterion to be tested - is given by[2]:

*PVNB > 0?*

In implementing this test, economic valuation is required for the terms $B_{it}$ and $D_{jt}$.

### 2.3.2 Evaluating Proposals to Reallocate Water Among Users

Another likely welfare improvement opportunity is for reallocating water among use sectors. In this case, the analytic question is:

*"can a reallocation from sector i to sector j yield incremental gains to sector j in excess of the foregone benefits in sector i ?"*

The Potential Pareto Improvement hypothesis to be test is given by the following question ( MB is the marginal benefits of water to sector *i* and *j*):

$$\sum_t \frac{MB_i}{(1+r)^t} \; \# \; \sum_t \frac{MB_j}{(1+r)^t} \; ?$$

For, if they were equal, no gains from reallocation would be possible. An example is given in Box 1.

[1] The material in this section is adapted from Young (1996)

[2] Of course, the Potential Pareto Improvement test can also be expressed in the alternative forms of benefit-cost ratios or internal rates of return.

***Appropriate Allocation Variable: Depletion versus Withdrawal***

To assign an economic value to water, it must be expressed as a monetary value per unit water volume or quantity used.

For off-stream uses - the quantity variable can be either the amount **withdrawn** from the river, or the amount **depleted** or **consumed**. Values will differ greatly, depending upon which measure is chosen.

The choice of withdrawal or depletion will depend on the purposes at hand. For the economist interested in predicting user behaviour in response to changing prices, the withdrawal measure is often more appropriate, because that is the measure upon which water users base their allocation decisions. Where necessary, adjustments to express benefits per unit depleted can be subsequently made.

### 2.3.3 Reliability of Supply and Water Quality

Economic analysis of both the above issues requires the estimation of incremental or marginal benefits of changes in water supply or use. This discussion has focussed on measuring benefits of increments or decrements of water supply. The analysis has abstracted from two other dimensions of water supply - **water quality and supply reliability.**

***Benefits of Improved Water Quality***

The quality of water also influences its economic value. Water in natural environments is never perfectly pure. Humankind uses water bodies as sinks for disposal of numerous wastes from production and consumption activities. The extent to which micro-organisms, and dissolved or suspended constituents are present varies greatly, and in sufficiently high concentrations can affect health, and reduce aesthetic values and productivity. Therefore, the content of pollutants, or conversely, the degree to which the water is treated for various uses is important in determining its economic value.

Estimating benefits of improved water quality raises some complex and challenging issues. For the important cases of degradable effluents, the detrimental effects depend on the nature of downstream water uses, the distance downstream, temperature, rates of flow and the quality of receiving waters. Willingness-to-pay (WTP) for water quality improvement is usually assumed to reflect damages to subsequent water users. Benefits are the damages avoided from a given project.

The framework for conceptualising the benefits of water quality improvement can be readily derived by extending the above framework for increments of water

**Box 1: Example - Hypothesis to be Tested When Reallocating Water Among Use Sectors**

Consider a proposal to reallocate water from agriculture to municipal uses. Indirect impacts are expected on the hydropower sector. The Potential Pareto Improvement test can be expressed by developing measurements for two conditions.

The **first condition** is that the benefits (both direct and indirect) to the municipal sector exceed the sum of: foregone direct benefits to the selling sector + foregone indirect benefits to the selling sector + foregone indirect benefits to the hydropower sector). This condition can be written (assuming all benefit and cost expressions are in present value terms):

$$DB+IB>FDB+FIB+TC+CC$$

where

| | | |
|---|---|---|
| *DB* | = | Direct economic benefit (value) to receiving sector. |
| *IB* | = | Economic benefit to indirectly affected sector. |
| *FDB* | = | Foregone direct benefit (value foregone) in source sector. ; |
| *FIB* | = | Foregone benefit in indirectly affected sectors. |
| *TC* | = | Transaction costs. |
| *CC* | = | Conveyance and storage costs. |

The **second condition** is that the direct foregone benefits in irrigated agriculture be the least-cost source of water for the purchasing sector:

$FDB+FIB+TC+CC < AC$

The second condition simply asserts that the sum of direct and indirect foregone economic benefits and the transactions and conveyance costs should be less than the cost of the next best alternative water source ( *AC* ) - otherwise an alternative source should be utilised.

supply. In short, all other factors held constant, an improvement in quality of water for either producers or consumers will shift the demand or marginal benefits curves to the right. The increment in producers' or consumers' surplus - summed over all affected individuals accruing to the change will be the appropriate measure of benefits of an improvement in water quality.

A framework for measuring the benefits of improved water quality (or conversely, the costs of deterioration in water quality) is presented in Section 3 below.

***Benefits if Improved Water-supply Reliability***

The degree of certainty with which supplies are available, in addition to its quantity and quality, is another important factor influencing the willingness-to-pay for water. Domestic, industrial and agricultural water demanders all place a higher value on reliable water supplies than on supplies with high risk of availability.

Increasing reliability comes at increasing costs however, so trade-offs are necessary between cost and risk.

Howe and Smith (1994) developed a model for assessing reliability and apply it to the case of municipal water supply. Of interest here is how they formulate a function reflecting the economic benefits of reliability to compare with costs of achieving reliability. They defined the **'Standard Annual Shortage Event'** (SASE) as a drought of sufficient severity and duration that certain specified restrictions on water use would be put in place. Systems reliability, R, is defined in terms of probability, P, of occurrence of the SASE:

$$R(SASE) \equiv 1 - P(SASE)$$

Next, a **loss function**, *L(SASE)*, is introduced, which represenes the reduction in economic value accruing affected individuals if the SASE were to occur. The desired economic measure, the marginal benefit of improved reliability, is given by the incremental reduction in expected losses - that is

$$\frac{\Delta \exp(L)}{\Delta(R)}$$

In general, valuing reliability has received relatively little attention.

## 2.4 Techniques for Measuring the Value of Water[3]

Previously we made a distinction between **intermediate goods** and **final consumption goods** - intermediate goods are employed to make final products, whereas consumption goods are those providing direct human satisfactions. The importance of this distinction is that the economic theory of profit-maximising producer provides the conceptual framework for the valuation of intermediate goods, while the theory of the individual consumer is the basis for valuing consumer goods.

### 2.4.1 Valuing Water as an Intermediate Good

The most extensive uses of water are for intermediate or producers' goods. Water used for crop irrigation, in industry, or for much of hydro electric or thermal power generation, are examples of intermediate good uses. For intermediate goods, the theory of producer's demand for inputs provides the conceptual basis for valuing the economic welfare implications of increments or decrements to input supplies.

A model for producers' demands for water can be obtained from the theory of a cost- minimising producer:

$$Q_w = f(P_W, P_i, P_a; X; S)$$

where $P_w$ and $P_a$, represent the price of water from the given system and from an alternative source, respectively, $P_i$ represents a vector of prices of inputs (e.g. capital, labour and materials); $X$ stands for the quantity of product to be produced and $S$ represents a vector of other factors, such as technology and climate.

The producers' demand for an input is its **value of marginal product** (VMP). Hence, the measurement problem requires an approximation to VMP. A number of methods of valuing water as an intermediate good are in common use, including:

- the **residual approach,** and
- the **alternative cost** approach.

*The Residual Imputation Approach*

The 'residual' method is the most frequently-used approach to applied shadow pricing of producers' goods. In the residual method, the incremental contribution of each input in the production process is determined.

The typical problem will be to estimate values for a multi-product operation. Often, too, the task is to estimate the value contributed by a partial increment or decrement to the water supply, rather than that for the entire quantity. A method developed for such situations is called the **'change in net income'** approach. This method is a form of what is sometimes called the **'variation of productivity change'.**

The willingness-to-pay for an increment of water is the net producer income associated with that increment. A general multi-product/multi input production function can be written as:

$$f(Y_1, \ldots, Y_m; X_1, \ldots, X_n) = 0$$

Where $Y$ is a vector of outputs of feasible products and $X$ is a vector of production inputs. The net income (denoted by $Z$) from producing a given set of products can be represented by:

$$Z = \sum_{i=1,\ldots,m} (Y_i \times P_{yi}) - \sum_{j=1,\ldots,n} (X_j \times P_{xj})$$

Therefore, the **Change in Net Income** is given by:

$$\Delta Z = Z_1 - Z_0$$

Where the subscripts 'O' and '1' refer to the without project and with project situations, respectively.

**Box 2: Implementation of the Change in Net Income Method - Irrigation Development**

Implementation of the analysis requires the assembly of **farm budgets** for the with and without cases. Farm budget analysis draws on agricultural production expertise, economics and accounting to infer returns under alternative situations. Analysts prepare budgets for the purpose of identifying and valuing the inflows of resources and the corresponding outflows of products within a specified accounting period, usually one year. A budget provides the basis for assessing the impact of a specific plan, in which a clearly specified package of inputs is committed and from which emerge a flow of costs and revenues over the project life (Eidman *et al.* 1994).

*General Use of Residual Imputation*

The residual technique is extremely sensitive to small variations in assumptions about either the nature of the production function or about prices. Biases can be introduced by any of several types of errors or omissions. Therefore, the approach is most suitable for cases where the residual claimant (in this case, water) contributes a significant fraction of the value of output. If an input that should be represented in the production function is omitted, this will permit the contribution of that input to

3 The material in this section is adapted from Young (1996)

be attributed to the residual claimant, thereby overstating the value of water. A similar but lesser bias arises if the amount of input, such as labour, is merely underestimated rather than omitted entirely.

The assumption of residual imputation is not overly restrictive, but care is required to assure that the conditions of production under study are reasonable approximations of the conceptual model. The main issues can be divided into two types:

- those relating to specifications of the production function, and
- those relating to the market (i.e. the pricing of outputs and nonresidual inputs).

*The 'Alternative Cost' Approach*

Another method appropriate to evaluating water-related intermediate good benefits is the **alternative cost approach** (which can also be used for consumption goods). The technique is attractive under the assumption (valid only in certain limited instances) that if a given project of specified output costs less than the next-best project which can achieve the same output, then the cost of the next-best project can be assigned as the benefit to the project under consideration. However the analysis must verify that the higher cost alternative would actually be constructed in the absence of the project under consideration. Put another way, the effective demand must be established for the alternative product.

In the field of water resource planning, the alternative cost approach has been employed for evaluation of many types of benefits, including municipal, industrial, hydroelectric and thermal electric power. However, the approach is subject to particular limitations, and should be used when its applicability is assured.

The alternative cost approach is based on the notion that maximum willingness-to-pay for a supplied good or service is not greater than the cost of providing that good or service via some other process or technology. The alternative should be a substantially different means of producing the same output.

The alternative cost technique is easily misused, and should be applied only with caution. The main limitation is that some alternative could always be conceived which would be more expensive than the project being evaluated, thereby inevitably producing an estimate of positive net benefits. Therefore, the alternative cost method must be supplemented by a study to confirm that the demand for the alternative is sufficient to justify the alternative expenditure.

The alternative cost test is yet another application of discounted cash flow analysis. The present value costs of each alternative are calculated on the basis of commensurate planning period, price level, discount rate and the like.

## 2.4.2 Measuring The Value of Water as a Consumptive Good

Turning next to estimating consumers' willingness-to-pay for water, we essentially value water in two contexts ( or as two types of goods):

- Water as a **private good** - demands for water as a private consumption good are found primarily among the residential water users.
- Water uses as **public goods** - public good demands are for recreation, fish and wildlife habitat and amenity values.

In general, it is difficult to draw the boundaries between these categories.

We will only consider the former here, since the specialist valuation techniques (e.g. travel cost, hedonic pricing, contingent valuation, etc.) for valuing water in its role as a public good were covered in a previous sesslon.

*Measuring WTP for Houselold Water Supply*

During the past decade, a concerted effort has been mounted to formulate and test improved approaches for measuring demand for domestic water in developing countries. A comprehensive discussion of the procedures and findings of this research programrne (on measuring the economic benefits of potable water supply projects in developing countries) is found in Whittington and Swarna (1994). The authors discuss and illustrate the use of contingent valuation methods and cost savings approaches, among others.

In general, there are two components of the economic benefits received by a household from the installation of an improved water supply:

- The first benefit component is the consumer surplus from the increased water purchased at a lower total cost.
- Second is the monetary value of savings in resources used to obtain water prior to the new system (These savings may be from not having to purchase water from vendors, or from not having to boil water.)

An example of quantifying both these types of benefits is provided in Box 4 below.

Whittington (1988) and ADB (1999) provide practical and very useful handbooks to guide the design

**Box 3: Example - Economic Value of Freshwater**

Resources for the Future (RFF), a Washington based research institution, undertook a study in 1995 which describes and presents nearly 500 water value estimates in the continental US from 41 published and unpublished studies performed over several decades (see Frederick *et al*, 1997). The reported values relate to four extractive uses, namely: irrigation (estimates were further subdivided into 22 crop types), industrial processing, thermoelectric power generation, and domestic uses (internal and external) - and four non-extractive uses, namely: hydroelectric power generation, recreation and fish and wildlife (broken down by fishing, wildlife refuges, fishing and whitewater, whitewater and shoreline recreation), navigation and waste disposal[4].

**Table 2: National Freshwater Values by Use**

| Use Category | Average | Median | Minimum | Maximum | Number of Values |
|---|---|---|---|---|---|
| | ($US/acre-R) | ($US/acre-R) | ($US/acre-R) | ($US/acre-R) | |
| **Non -Extractive Uses:** | | | | | |
| Waste disposal | 3 | 1 | 0 | 12 | 23 |
| Recreation/F & W habitat | 48 | 5 | 0 | 2,642 | 211 |
| Navigation | 146 | 10 | 0 | 483 | 7 |
| Hydropower | 25 | 21 | 1 | 113 | 57 |
| **Extractive Uses:** | | | | | |
| Irrigation | 75 | 40 | 0 | 1,228 | 177 |
| Industrial processing | 282 | 132 | 28 | 802 | 7 |
| Thermoelectric power | 34 | 29 | 9 | 63 | 6 |
| Domestic | 194 | 97 | 37 | 573 | 6 |
| **Total Number of Values** | | | | | **494** |

***Source:*** Frederick *et al (1997)*

all 500 individual water value estimates are provided in two appendices to the main report. For comparability, all estimates are expressed in 1994 $US per acre-foot. Since the estimates are generally based on conditions relevant to specific locations, times, and water supply conditions, information on each of these factors, and the methodology used to derive each value and the reference study, is also provided in the appendices. To assist in locating information, water value estimates are organised by geographical region and type of use. Table 2 below presents the national averages and medians of the water values for the four extractive and four non-extractive uses.

[4] The latter relates to the ability of a stream to assimilate wastes without exceeding water-quality standards, which depends on, among other things, the quantity of flow. Thus, releasing water from storage during critical periods can help maintain water quality. The value of water released for these purposes is calculated as either the downstream damages avoided or the waste-treatment costs foregone.

**Box 4: Benefits of a Water Supply Project**

The gross benefits from a water supply project (WSP) are estimated by conveniently apportioning the supply of water into:

- **Non-incremental** - if the output of a WSP replaces the existing supply to users, that output is defined as non-incremental output For example, if the present source of water to consumers is from vendors or wells (with time and effort spent on such use of water), the supply or water from the project which replaces this is to be considered as non-incremental. Non-incremental outputs are valued in terms of their adjusted supply price or opportunity costs.
- **Incremental output** - the supply of water from a WSP that meets additional or induced demand (possibly as a result of availability of higher quality water at lower cost) is to be considered as incremental output. Incremental outputs are valued in terms of their adjusted demand price or WTP.

The following numerical example explains the concepts of valuing incremental and non-incremental output (demand).

A piped WSP is proposed to meet a growing demand for an area from its existing level of 150 $m^3$ per year to 250 $m^3$ per year. The present supply of 150 m per year is met as follows: 25 percent from vendors and 75 percent from household wells, at the prices given in (including the in-house cost I of treating the water to a standard commensurate with that is piped supply.

**Table 3 Data Set for WSP Project**

| Sources | Proportion (%) | Annual Quantity ($m^3$ per year) | Cost ($ per $m^3$) |
|---|---|---|---|
| Private vendors | 25 | 37.5 | 5 |
| Household wells | 75 | 112.5 | 3 |
| Average of supply price | | | 4 |

The proposed WSP is a public scheme and the price of piped water supply is only $1.5/m3, which is lower than the present cost of supply. Due to the higher quality and lower price of piped supplies, the existing supply of water by vendors and households wells will be fully replaced. The concept of gross benefits is illustrated in Figure 1.

The quantity consumed is given by:

$Q_0$ = 150 $m^3$/year = water from vendors and wells

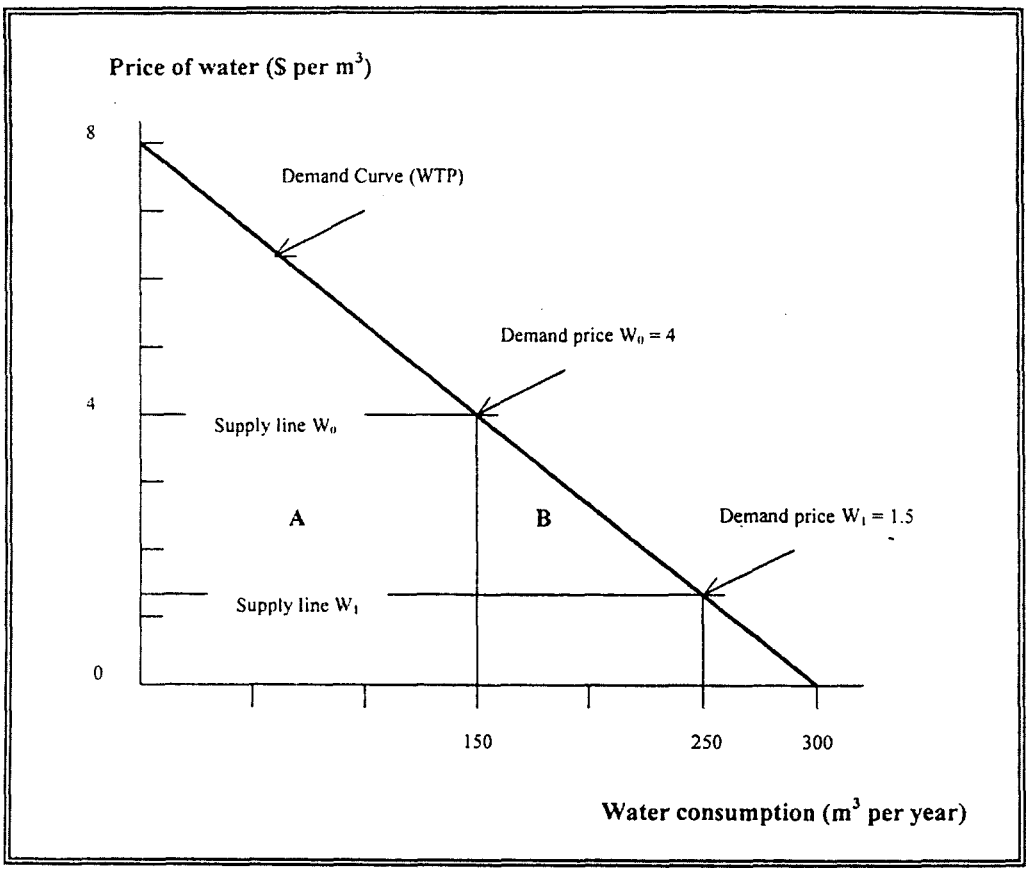
$Q_1$ = 250 $m^3$/year = water from project

The corresponding prices are:

$P_0$ = $4 per $m^3$ = cost of water without project

$P_1$ = $1.5 per $m^3$ = tariffwithproject

**Figure 1: Gross Benefits of WSP**




Therefore, the **non-incremental benefit** as a result of full replacement of existing supply (based on average supply price) is given by:

= Area *A*

= $Q_0 * P_0$

= 150 $m^3$/year * \$4/$m^3$

= \$600 per year

The **incremental benefit** as a result future expansion in water use (based on average demand price) is given by:

= Area B

= $0.5 * (P_0 + P_1) * (Q_1 - Q_0)$

= 0.5 * (4 + 1.5) * (250- 150)

= \$275 per year

Hence, the total **gross** benefits of the WSP are:

= Area A + Area B

= \$600 + \$275

= \$875 per year

The non-incremental part of the gross benefit is based on the average supply price of water in the without-project situation, whereas the incremental part is based on the average demand price of water.

The demand curve may be derived by the contingent valuation method.

**Source:** *ADB (1999)*

and conduct of household surveys of willingness-to-pay in developing countries. The report addresses questionnaire design and implementation of the survey - including sampling, translation of the questionnaire and recruitment and training of enumerators. Appendices provide examples of questionnaires used in various developing countries.

Contingent valuation methods have also been employed to estimate willingness to pay for sanitation services. For example, Whittington et al (1992) studied the demand for improved sanitation services in Ghana.

*Use of the Contingent Valuation Method*

The contingent valuation (CV) method is evolving as the primary means of directly estimating the economic benefits of improved water supply CV studies can provide useful data to inform the following types of decisions in a water supply context (ADB, 1999):

- **Cost-benefit analysis** - provided that households understand all the changes and perceive all the benefits which will result from an improved warter supply, the WTP bids can serve as a measure of the economic benefits of the project.
- **Setting priorities** - if a water agency has a limited budget and must choose between villages or towns to receive a piped water supply, WTP measures can assist in prioritising investments and site selection. For example, sites which show a high WTP for improved water supply are likely to benefit considerably from a new piped water system, and the potential for cost recovery is likely to be high.
- **Choice of service level** - planners in developing countries often assume that a community should be provided with the highest level of service possible, as long as the cost to obtain the water does not exceed 5 percent of household income. It is assumed that as this threshold is not exceeded, households would abandon their existing supply in favour of the improved system. CV surveys can assist in defining the appropriate technology and service level.
- **Tariff design** - water utilities are under increasing pressure to be financially viable and to raise prices they charge for water to reflect better the cost of the service. However, few water utilities in developing countries have adequate information on which to base decisions regarding tariff design. If prices are set too low, revenues will not be sufficient to cover supply costs. In contrast, if prices are set too high, households may not be able to afford to connect to the distribution system, and again revenues will be too high. With WTP data from CV surveys, the relationship between the price of water, the number of households connected, and expected revenues can be estimated.

**Box 5: Example of a Bidding Game**

When the new project starts, and assuming (1) if the piped water quantity is increased to 12 hours supply per day at adequate pressure so that you can get the additional supply of water of good quality and (2) the tariffs are re-fixed at Rs per month, would you want a connection and pay for the bill? (go to the bidding game)

| | | |
|---|---|---|
| 1a. | | "NO", I do not want a connection |
| 1 b. | | "YES", I want a connection |
| If "YES", would you be willing to pay Rs .....per month | | |
| 2. | Rs 400 | If "YES", then stop; if "NO", then go to 3 |
| 3. | Rs 350 | If "YES", then stop; if "NO", then go to 4 |
| 4. | Rs 300 | If "YES", then stop; if "NO", then go to 5 |
| 5. | Rs 250 | If "YES", then stop; if "NO", then go to 6 |
| 6. | Rs 200 | If "YES", then stop; if "NO", then go to 7 |
| 7. | Rs 150 | If "YES", then stop; if "NO", then go to 8 |
| 8. | Rs 100 | If"YES", then stop; if"NO", then go to 9 |
| 9. | Rs 75 | If "YES", then stop; if "NO", then go to 10 |
| 10. | Rs 50 | If"YES", then stop; if"NO", then go to 11 |
| 11. | Rs 25 | If "YES", then stop; if "NO", then explain why |

The selection of the initial amount is important and should reflect realism - in general, the initial amount should not be higher than two times the unit cost of the enhanced level of service.

*Source: ADB (1999)*

**Table 4: Summary of Selected WTP Estimates for Water Supply in Developing Countries**

| Study | Without Project | WTP Estimate (US$ per month or % of househol income) |
|---|---|---|
| CV Kumasi, Ghana, 1993 | No water | $1.5 (or2.1%) |
| Market observations, Ukanda, Kenya, 1989 | Purchase of water from vendors as a % of income - | 9.0% |
| Market observations Onitsha, Nigeria, 1989 | Purchase of water from vendors as a % of income | 8-18% poor households; 2-3% rich households |
| Market observations, Nsukka, Nigeria, 1990 | Purchase of water from vendors as a % of income | 6-10% |
| CV, Haiti, 1990 | No public stand post, no private connection | 1.7-2.1% |
| CV, Weuye, Kenya, 1997 | No private water connection | 1.5-1.9% |
| CV, Pakistan, 1992 | No piped water connection | $0.8-2.26 |
| CV, Kerala, India, 1993 | Improved piped water connection | $0.53 |
| CV, Costa Rica, 1995 | No piped water connection | 7% |
| CV, Nigeria, 1992 | No public taps<br>No private connections | 1.0-1.4%<br>2.0-2.6% |
| CV, Pradesh, India, 1996 | No private connection | $0.58 |

***Source:*** *Markandya, Taylor and Boyd (1999)*

With the CV studies, measures of WTP are basically derived be asking one the following types of question:

- Respondents may be asked a direct, open-ended question, such as "what is the maximum amount of money you would be willing to pay for (e.g. a given level of service)?".
- Respondents are presented with a specific choice which requires a 'yes/no' response, such as "suppose a water distribution line were installed in front of your house, and assuming the connection fee was $...., and that the monthly tariff was $.... (fixed or volumetric charge), would you choose to connect to the next water distribution system?"

It is also possible to combine different questions, and develop bidding games. (see Box 5).

The validity and reliability of WTP estimates from CV survey were presented in an earlier Chapter and are therefore not discussed further here.

Table 4 provides a summary of WTP estimates for (improved) potable water supply in developing countries.

# 3 CONCEPTUAL FRAMEWORK FOR VALUING CHANGES IN WATER QUALITY

## 3.1 Introduction

This section outlines a framework for measuring the benefits of improved water quality (or conversely, the costs of deterioration in water quality)[5] The estimated benefits of improved water quality can then be compared with the costs of achieving the improvement. Several significant pieces of work do provide frameworks, which address aspects of water resource valuation, e.g. determining the total value of groundwater or estimating the total benefits of surface water quality improvements. The most relevant contributions in these areas include Boyle and Bergstrom (1994), US EPA (1995), FWR (1996), EA (1997) and NRC (1997). The framework outlined below combines material from these sources.

[5] It is also possible using the same framework, to estimate the unit damage costs of selected waterborne pollutants.

In short, the framework considered herein is based on an overall economic valuation approach that integrates the hydrological, physical and social components of the valuation problem. Specifically, we link changes in the physical characteristics of the water resource (resulting from changes in the distribution of the pollutant(s) of interest - those targeted for reduction) to changes in the level of services or uses provided (quantity and quality) and finally, to how individuals value the changes in the services or uses. Implementation of the framework requires a multidisciplinary approach. For example, establishing the connections between changes in the quality of the water resource and subsequent changes in the services provided, requires the collaboration of researchers, including hydrologists, engineers, ecologists, and economists. Measuring how individuals value the changes in the service flows or uses is primarily the domain of economists.

The main steps of the integrative framework are shown in Figure 2 below. These steps are outlined in the following sections, under the headings of impact assessment and economic valuation. The former includes Boxes 1 through 8, whereas Boxes 8 and 9 are included under the latter heading. Box 8, "change in the services flows provided", provides the link between the impact assessment and economic valuation components.

## 3.2 Impact Assessment

The impact assessment component of the integrative framework, (Boxes 1 through 8 in Figure 2), basically involves quantifying the impact of the pollutant(s) of interest on the quality of a selected water resource and subsequently on the service flows provided by that water resource.

Of fundamental importance, therefore, is the definition of the change in the condition of the water resource and the ensuing changes in the service flows created by the resource. The first task is to develop a technical definition of the **reference condition** for the selected water resource.

This requires monitoring of the water resource **(Box 1)** in order to establish the current condition of the resource, in terms of both quality and quantity **(Box 2).** (While the focus of this section is essentially water quality, knowledge of water quantity is required as, for example, flow rates in a river will influence water quality.) As a result of this assessment the selected water resource is defined according to the following quality and quantity dimensions:

$$q_l^0 = g\ \{\ C\ (q_l^0\ );\ p^{-0}\ (q_l^0\ )\ \}$$

where

$q_l^0$ = Current quality of water resource.

$q_l^0$ = Current quantity of water resource.

$g\ \{.\}$ = Denotes that current water quality is a function of the parameters listed within.

$p^{-0}\ (q_l^0\ )$ = The current concentration of the pollutant(s) of interest in the water resource.

$C\ (q_l^0\ )$ = A vector of the current concentrations of other contaminants in the water resource.

The next task is to assess how the current quality (and quantity) of water will change as the distribution of the pollutant(s) of interest ( p) is reduced to naturally occurring levels for that particular water body **(Boxes 3 and 4).** This will require some form of hydrological modelling exercise. The complexity of the modelling exercise will vary, depending on whether we simply take "snap-shot" view of today or alternatively, look at the water resource over a prolonged time period. In addition to considering factors such as natural recharge and discharge (evaporation), natural contamination, and current human use of the water resource (extraction and *in-situ*), the latter modelling approach would require predictions of future use patterns and policies regarding the use and protection of the resource.

The hydrological modelling exercise therefore provides estimates of the **baseline** (with $p$ ) water quality and quantity and the new water quality and quantity (without $p$) - that is, respectively[6]:

$$q_l^0 = g\ \{\ C\ (q_l^0\ );\ p^{-0}\ (q_l^0)\ \}\ \textbf{(Box 5)}$$

and

$$q_l^1 = g\ \{\ C\ (q_l^1\ );\ p^{-1}\ (q_l^1)\ \}\ \textbf{(Box 6)}$$

where

$q_l^1$ = New quality of water resource.

$q_l^1$ = New quantity of water resource.

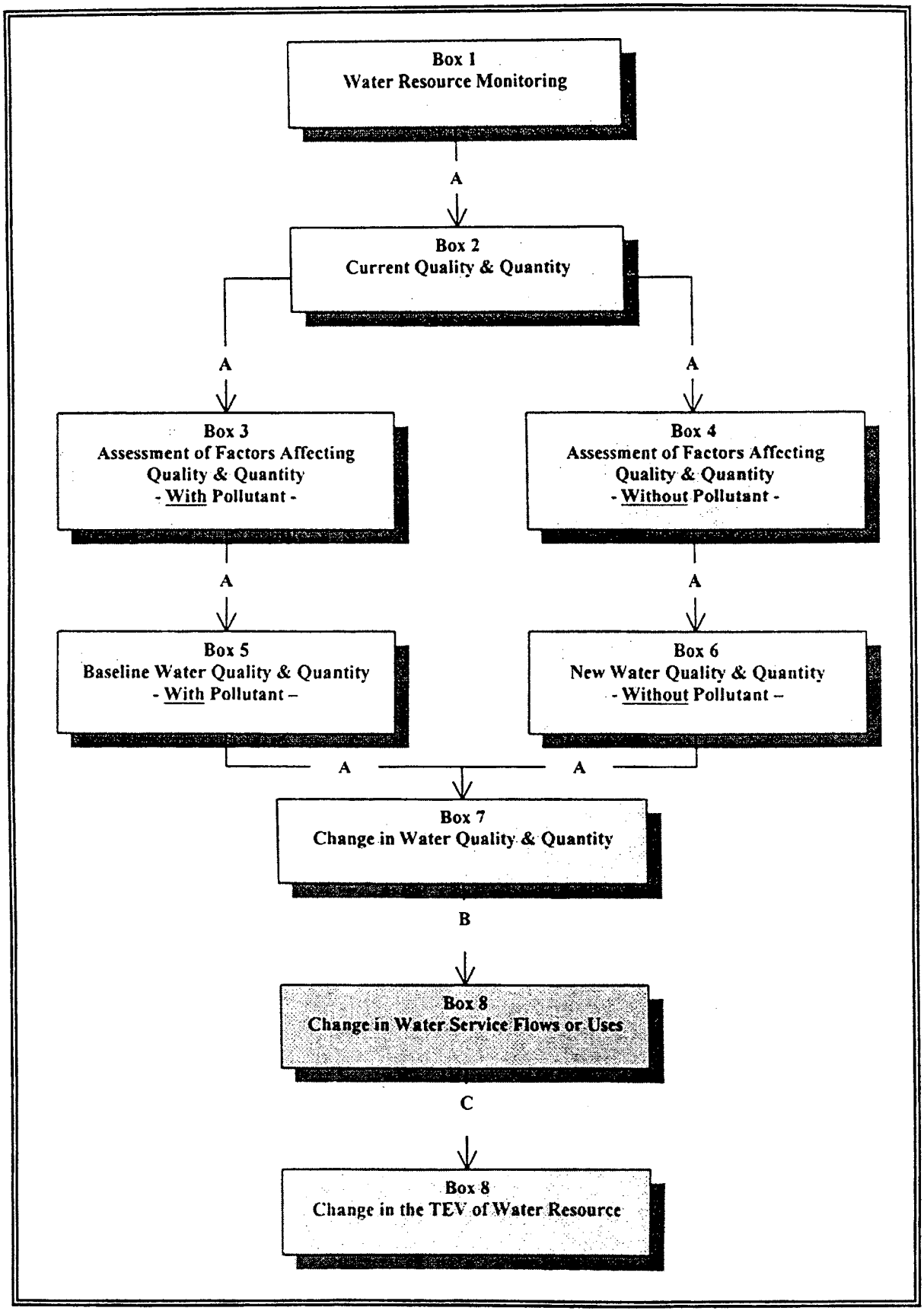
$p^{-1}\ (q_l^1)$ = The naturally occurring concentration of the pollutant(s) of interest in the water resource.

$C\ (q_l^1\ )$ A vector of the new concentrations of other contaminants in the water resource.

Note that if we adopt the simpler "snap-shot" modelling approach, then the current and baseline water quality and quantity are equivalent. In terms of

6 Although, strictly speaking this scenario refers to the pollutant(s) of interest being reduced to naturally occurring levels - and not removed completely from the water resource.

## Figure 2: Integrative Valuation Framework




Source: Adapted from Boyle and Bergstrom (1994), US EPA (1995) and NRC (1997)

simplifying the modelling exercise it is also desirable to assume that $q_l^1$ equals $q_l^0$ and $C(q_l^1)$ equals $C(q_l^0)$. The appropriateness of these simplifying assumptions will need to be established on a case-by-case basis.

Given the estimates of the "baseline" and "new" water resources conditions, it is relatively straightforward to establish the resulting change water quality and quantity **(Box 7).** We define the change in water quality and quantity by:

$$\Delta q_l = q_l^0 - q_l^1 = g\{C(q_l^0); p^{-0}(q_l^0)\} - g\{C(q_l^1), p^{-1}(q_l^1)\}$$

and

$$\Delta q_l = q_l^0 - q_l^1$$

The difference in water quality between the "baseline" and "new" scenarios is essentially defined by the corresponding concentrations of p, which is given by:

$$\Delta p = p^o - p^1$$

To summarise, up to this point in the framework we have modelled a change in water quality ($\Delta q_l$) resulting from a change in the distribution of a particular pollutant(s) of interest ($\Delta p$ ). Moreover, the analysis is specific to a particular type of surface water or groundwater resource. What remains is to examine how the service flows from the resource change in response to $\Delta q_l$, and estimate the subsequent change in the TEV of the resource.

If desired, the resulting estimate of economic benefit can then be normalised to $\Delta p$, yielding a measure of 'average' benefits foregone (i.e. unit damage costs).

The steps and linkages indicated by arrows marked "A" in Figure 2 primarily require input from hydrologists, geologists, engineers, ecologists, soil scientists, and other biological and physical scientists. The modelling of these steps (illustrated by Boxes 1 through 7) represents probably the greatest challenges, a point emphasised in the literature (see, for example, US EPA, 199S and NRC, 1997).

## 3.3 EconomicValuation

The economic valuation component of the integrative framework, which comprises Boxes 8 and 9 in Figure 2, involves estimating the change in **total economic value** (TEV) resulting from the change in the services provided by the selected water resource.

### 3.3.1 Water Services[7]

The first task is therefore to establish what service flows are provided under baseline resource conditions, and what services will be supported under new water quality conditions. In doing this, it is important to recognise that the level of services supported by a water resource are conditional on the presence of substitute and complementary service flows. For example, the abstraction of water for public water supply from one source will depend on the characteristics of other sources. Likewise, the decision by an angler to fish on a particular stretch of river will depend on the availability of other rivers offering similar angling opportunities. Hence, a vector of **substitute** and **complementary** services flows must also be defined.

The purpose of this task is therefore to define:

$$S^0 = f\{q_l^0;\ q_l^0 \mid S_s^0\}$$

and

$$S^1 = f\{q_l^1;\ q_l^1 \mid S_s^0\}$$

where

$S^0$ = The vector of service flows supported by the water resource under baseline quality and quantity conditions.

$S^1$ = The vector of service flows supported by the water resource under new quality and quantity conditions.

$S_s^0$ = The vector of substitute and complementary service flows, which are assumed not to change.

The change in water services **(Box 8)** is thus given by:

$$\Delta S = S^0 - S^1$$

The mapping of changes in water quality conditions to changes in service flows, indicated by the arrow marked "B" in Figure 2, is primarily facilitated by scientists and engineers.

Broad categories of the services provided by water resources which give rise to economic value were provided in Table 1. Table 5 and Table 6 below provide a more detailed list of the types of services generated by groundwater and surface water, respectively.

However, both types of water resource should not be considered in isolation. Groundwater, for example, essentially serves two functions: the first function is the storage of a water stock; and the second function is to discharge to surface waters. Through discharging to surface waters, groundwater thus indirectly contributes to the services provided by those surface waters. A change in the quality of an aquifer may therefore also induce changes in the services generated by any surface waters that it recharges. It should also be noted that surface waters may also serve to recharge groundwater stocks.

[7] Note that this task could just as easily been included under the heading of Impact Assessment.

**Table 5: Groundwater Services Likely to be Affected by Quality Changes[1]**

| Water Services (or Uses) Provided | Potential Impacts on Value |
|---|---|
| Drinking (public) water supplies | Change in availability of potable water<br>Change in production costs<br>Change in human health or health risks |
| Other public water supplies (landscape/gardening) | Change in cost of maintaining property<br>Change in human health or health risks |
| Agriculture - crop irrigation | Change in value of crop or production costs<br>Change in human health or health risks |
| Agriculture - livestock watering | Change in value of livestock products or production costs<br>Change in human health or health risks |
| Industrial use - food processing | Change in value of food products or production costs<br>Change in human health or health risks |
| Industrial use - manufacturing processes | Change in value of manufactured good or production costs |
| Power generation - cooling water | Change in cost of electricity generation |
| Transport and treatment of wastes and other by-products of human activity | Change in waste disposal costs<br>Change in human health or health risks<br>Change in ecological health or health risks |
| Improved water quality through the support of living organisms | Change in ecological health or health risks<br>Change in human health or health risks |
| Provision of non-use services (e.g. existence services) | Change in personal utility or satisfaction |

***Source:*** *Adapted from Boyle and Bergstrom (1994), US EPA (1995), FWR (1996 and NRC (1997).*

***Notes:***

[1] Due to the interrelationships between groundwater and service waters, changes in groundwater quality (and quantity) may also impact on the services provided by surface waters (see Table 6).

**Table 6: Surface Water Services Likely to be Affected by Quality Changes[1]**

| Water Services (or Uses) Provided | Potential Impacts on Value |
|---|---|
| Drinking (public) water supplies | Change in availability of potable water<br>Change in production costs<br>Change in human health or health risks |
| Other public water supplies (landscape/gardening) | Change in cost of maintaining property<br>Change in human health or health risks |
| Agriculture - crop irrigation | Change in value of crop or production costs<br>Change in human health or health risks |
| Agriculture - livestock watering | Change in value of livestock products or production costs<br>Change in human health or health risks |
| Industrial use- food processing | Change in value of food products or production costs<br>Change in human health or health risks |
| Industrial use - manufacturing processes | Change in value of manufactured good or production costs |
| Power generation - cooling water | Change in cost of electricity generation |
| Transport and treatment of wastes and other by-products of human activity | Change in waste disposal costs<br>Change in human health or health risks<br>Change in ecological health or health risks |
| Recreational activities - swimming (bathing), boating, fishing, informal, etc. | Change in the quality or quantity of recreational activities<br>Change in human health or health risks |
| Aquaculture - commercial fisheries | Change in value of harvest or harvesting costs<br>Change in human health or health risks |
| Aquaculture - fish farming | Change in value of harvest or harvesting costs<br>Change in human health or health risks |
| Aquaculture - shell fisheries | Change in value of harvest or harvesting costs<br>Change in hurnan health or health risks |
| Visual amenity | Change in value of commercial or private property |
| Support of on-site and off-site fish, wildlife and plants | Change in ecological health or healthrisks<br>Change in quality or quantity of habitat |
| Provision of non-use services (e.g. existence services) | Change in personal utility or satisfaction |

***Source:*** *Adapted from Boyle and Bergstrom ( 1994), US EPA ( 1995), FWR ( 1996 and NRC ( 1997).*

**Notes:**

[1] Due to the interrelationships between surface waters and groundwater, changes in surface water quality (and quantity) may also impact on the services provided by groundwater (see Table 5).

Hence, surface waters also indirectly 2 contribute to groundwater services. As a result of these interrelationships, the services listed in Table 5 and Table 6 are not totally independent. This means that, when estimating the economic value arising from changes in water quality, these interrelationships should not be ignored.

The share of groundwater services attributable to surface water quality, and vice versa, is however, often extremely difficult to quantify. It is necessary to model the physical linkages between the two water resources such that the marginal contribution of one resource to the other can be measured. Failure to take this into account will result in estimates of changes in service flows and, in turn, economic values, being underestimated.

### 3.3.2 Valuing the Changes in Water Service Flows

In order to move towards the ultimate goal of estimating changes in the TEV of the selected water resource, we need to identify the impacts on (changes in) economic activities resulting from changes in the water services. Column 2 in Table 5 and Table 6 provides potential impacts on economic activities for each of the main services listed. For example, in Table 6 the potential impact of a change in commercial fishery services is a change in harvest values (or harvesting costs) and a change in human health or health risks (through the ingestion of contaminated fish).

Measuring changes in economic activities, e.g. health risks, requires careful consideration of changes in the incidence of mortality and morbidity, and acute, chronic and cancerous forms of illnesses[8]. This, in turn, requires the input of health professionals. Ultimately, what is required to quantify links between $\Delta p$, $\Delta S$ and changes in economic activities are cause-effect models (or dose-response functions). These cause-effect models are ideally required to measure changes in all service flows, not just human health risks. Once all changes in water services have been quantified **(Box 8)**, the final task is to assign monetary values to these service changes **(Box 9)**. This involves capturing the change in TEV associated with the total change in the water service vector S- that is:

$$\Delta TEV = v\{\Delta S | S_s^0$$

where

$\Delta TEV$ = The change in the TEV of the selected water resource.

$v\{.\}$ = Denotes the change in TEV is a function of the parameters listed within.

Alternatively, the change in the TEV of the water resource could be estimated as:

$$\Delta TEV = TEV^0 - TEV^1$$

where

$$TEV^0 = v\{ S^0 | S_s^0 \text{ and } TEV^1 = v\{ S^0 | S_s^0 .$$

The valuation of changes in individual services should be based on changes in consumer and producer welfare. Moreover, the desired measure of economic value in this context is given by an individual's maximum willingness-to-pay (WTP)[9].

A number of valuation techniques exist for measuring the welfare changes associated with changes in water services, including change in output approaches, production cost (savings) approaches, cost-of-illness approaches, hedonic analysis, travel cost methods and contingent valuation methods. These techniques, which were discussed in an earlier Chapter, are listed in Table 7 below- organised according to extractive and in-situ services. This list is not exhaustive. Moreover, it should be treated with caution. In some cases, several techniques can be used to value the same service flow. While this permits an analyst to validate benefit estimates, it also raises the potential for double-counting the benefits when attempting to arrive at a total value estimate for a particular service. An overview of the existing valuation literature in the context of valuing changes in water quality is provided in Table 8 below.

8 The US EPA's Offce of Groundwater and Drinking Water, as part of its efforts to implement amendments to the 1996 Safe Drinking Water Act (SDWA), formed the National Drinking Water Advisory Council's Benefits Working Group. This group was charged to review those quantifiable benefits that could be considered when developing drinking water regulations and provide recommendations to the EPA on which benefits should be evaluated. The working group recommended that the EPA should focus its benefits analysis efforts primarily on assessing effects on human health (US Office of Groundwater and Drinking Water, 1999). Moreover, the EPA should devote substantial efforts to better understand the health effects of water contaminants, including the types of effects, their severity, and affected sensitive sub-populations. It was also recommended that the EPA consider the benefits of commercial water treatment cost reductions and the benefits due to source water protection (e.g. ecological benefiS and non-use benefits). To these ends the EPA has initiated a series of research activities, which are due to report next year.

9 The consensus among economists is to use what are referred to as Hicksian measures of compensating or equivalent surplus. However, the commonly used valuation techniques that utilise market, or choice, based data yield estimates of Marshallian surplus - with income held constant rather than utility. Therefore, in valuing environmental goods/services we tend not to work with the conceptually desired measures of welfare change, but this does not imply they are not appropriate measures of economic value.

**Table 7: Water Functions/Services and Applicable Valuation Techniques**

| Water Function/Service Flow | Applicable Valuation Technique[1] |
|---|---|
| **A. Extractive values:** | |
| 1. Drinking water | |
| a) Treatment | Change in input / output - production costs; Cost-based approaches |
| b) Human health | Cost-of-illness; Hedonic analysis; Contingent valuation; Cost-based approaches |
| 2. Agriculture water use | |
| a) Treatment | Change in input/output - production costs |
| b) Productivity/output | Change in input/output - production costs |
| c) Human health | As above - ingestion |
| 3. Industrial water use | Change in input/output - production costs |
| **B. *In-situ* Values:** | |
| 1. Recreation values (in-stream; angling; informal; boating; bathing; etc.) | Travel cost; Contingent valuation; As above for human health - bathing |
| 2. Commercial fishing/shell fishing | |
| b) Productivity/output | As above |
| c) Human health | As above - ingestion |
| 3. Amenity values | Hedonic property price; Contingent valuation |
| 4. Ecological values | Cost-based approaches; Travel cost; Hedonic analysis; Contingent valuation |
| 5. Existence/bequest value | Contingent valuation |
| 6. Buffer value | Change in input/output - production costs; Cost-based approaches; Contingent valuation |

*Source: Adapted from Freeman (1993), NRC (1999) and US EPA (1995) Notes:*

**Notes:**

[1] Although not listed, benefit transfer is applicable in most cases.

**Table 8: Usefulness of Existing Valuation Literature**

| Water Function/Service Flow | Existing Valuation Studies | | Direct Relevance of Studies[2] | Feasibility of Primary Study[3] | Benefit Transfer Required[4] |
|---|---|---|---|---|---|
| | Groundwater | Surface Waters | | | |
| A. Extractive Values: | | | | | |
| 1. Drinking water | | | | | |
| a) Treatment | Y | Y | H | H | N |
| b) Human health | Y | Y | M | L | Y |
| 2. Agriculture water use | | | | | |
| a) Treatment | Y | Y | H | H | N |
| b) Productivity/output | ? | Y | L | M | P |
| c) Human health | ? | N | - | L | Y |
| 3. Industrial water use | ? | Y | M | H | N |
| **B. In-situ Values** | | | | | |
| 1. Aquaculture | | | | | |
| a) Productivity/output | N/A | Y | L | M | P |
| b) Human health | N/A | Y | L | L | Y |
| 2. Recreation values (all forms) | N/A | Y | M | L | Y |
| 3. Amenity values | N/A | Y | M | L | Y |
| 4. Ecologicalvalues | ? | Y | L | L | Y |
| 5. Existence bequest values | ? | Y | L | L | Y |
| 6. Buffer values | Y | N/A | L | M | P |

**Notes:**

1 Refers to whether valuation data exists for each of the groundwater and surface water service flows listed - Y = yes; N = no; ? = possibly; N/A = service flow is not applicable.

2 Refers to whether the valuation data (and studies) are directly relevant to the integrated framework, e.g. do they relate to measurable changes in water quality - H = high relevance; M = medium relevance; L s low relevance.

3 Indicates how feasible it would be to conduct a site-specific primary study for the particular service flow, e.g. depending on costs, time and data availability - H = high feasibility; M = medium feasibility; L = low feasibility.

4 Indicates whether benefit transfer may be required for valuing the particular service flow - Y = yes, very likely; P = possibly required; N = not necessary, primary study feasible.

The estimated change in the TEV of the water resource resulting from an improvement in water quality can then be compared with the costs of achieving the improvement.

If unit damage costs are required, a further task is to normalise the change in the TEV of the water resource to the change in concentration of the pollutant(s) of interest. This will require the latter to be converted to mass units using appropriate flow (quantity) data. In this case, the output of the integrative framework is a measure of the average benefits foregone (or damage costs) associated per unit mass of the targeted pollutant, given by:

$$AC_{\bar{p}} = \frac{\Delta TEV}{\Delta \bar{p}}$$

where

$AC_p$ = The average damage cost (benefits forgone) per unit mass of pollutant $\bar{p}$

The mapping of changes in water services to changes in the TEV of the selected water resource, indicated by the arrow marked "C" in Figure 2, is primarily facilitated by economists, building on the scientific analyses which links Boxes 1 through 7.

A word of caution, throughout the integrative framework we are dealing with imperfect data regarding, for example, the quality (and quantity) of water and the actual change in services. In all cases the results are uncertain, with associated probabilities of occurrence. This uncertainty also extends to the economic valuation of the estimated changes in water services. We are therefore always working with expected, as opposed to deterministic, results. Consequently, appropriate statistical techniques for dealing with uncertainty must be incorporated within the framework.

### 3.3.3 Overview of Integrative Framework

The methodological framework outlined above is based on a total economic valuation approach which integrates changes in the physical characteristics (quality and quantity) of water resources with changes in the level of services or uses of the resource, and ultimately, with how individuals value the changes in the services or uses. Application of the framework therefore requires that progress is made on two fronts:

- **Impact Assessment** - formal links between the pollutant(s) of interest and the biophysical condition of the water resource must be established **(Boxes 1- 7).**
- **Economic Valuation** - these links must be developed in a manner that facilitates their economic valuation **(Boxes 8 - 9).**

Finally, the framework **(Boxes 1- 9)** is inherently multidisciplinary, and requires interaction and collaboration between economists and other scientists (e.g. hydrologists, ecologists, biologists, health professionals, engineers, etc.).

## 3.4 Data Sources

Several good sources of (water) valuation data are available - which may be used within the integrated framework discussed above - two of which have been specifically developed with benefit transfer in mind. Both these databases are both outlined below.

### 3.4.1 The Environmental Valuation Reference Inventory

The Environmental Valuation Reference Inventory (EVRI) *"is a webbased infobase that can help policy analysts use the benefit transfer approach to estimate economic values for changes in environmental goods, services or human health"* (Environment Canada, 1998). The EVRI was essentially created in response to the special 1992 issue of Water Resources Research, which focused exclusively on benefit transfer. In the year that followed the release of this issue Environment Canada organised a workshop in Ottawa to examine the key issues relating to benefit transfer. This workshop was preceded with the development of a structure for a preliminary benefit transfer database. Further workshops were held to critique early versions of the database, and its structure was gradually completed over the next few years. During the same period, work began on capturing valuation data from primary studies. The EVRI was eventually adapted to the Internet and the supporting software finalised, subject to a final expert review, in early 1998.

Unfortunately, the EVRI cannot be accessed without a subscription, but a "tour" is available on its web-page, which illustrates the key features of the database[10].

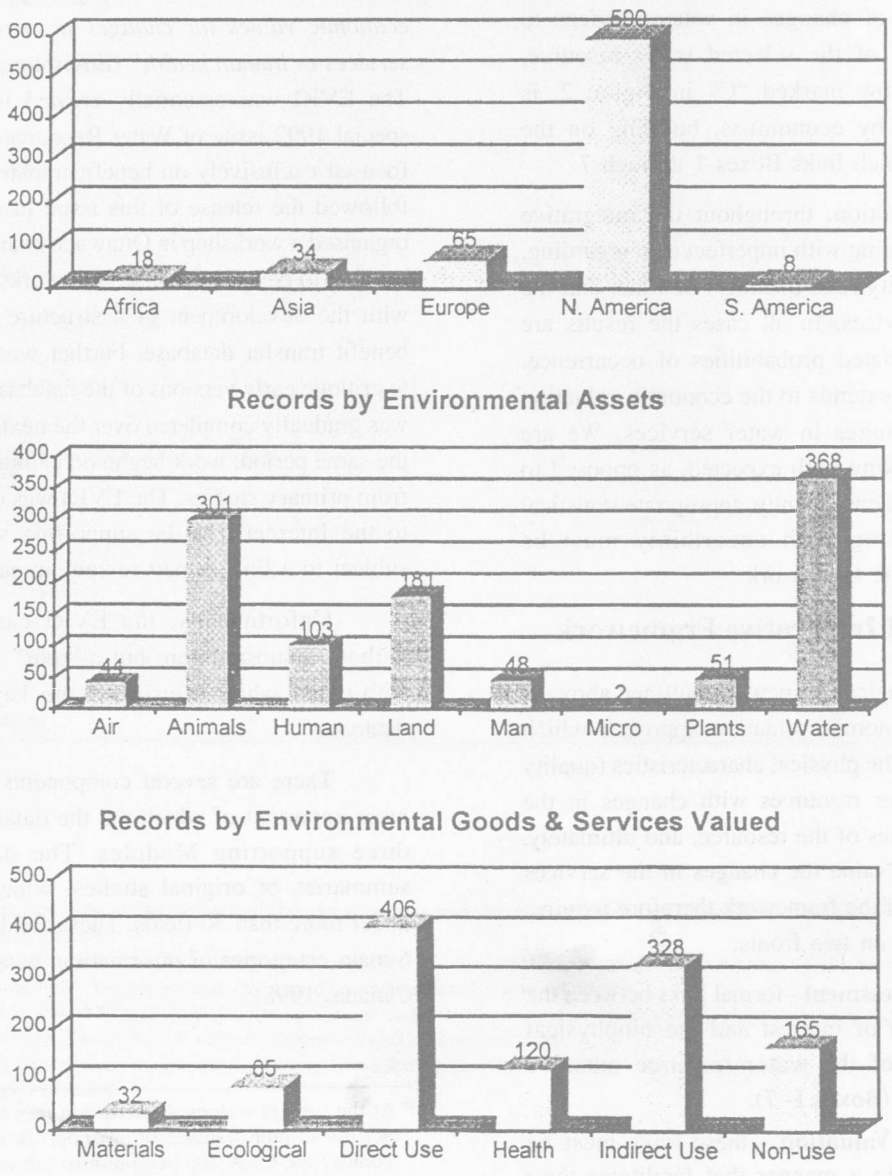
There are several components to the EVRI, the most important of which are the database itself and the three supporting Modules. The database contains summaries of original studies, which are categorised under more than 30 fields. These fields are grouped into 6 main categories of information, namely (Environment Canada, 1998):

[10] At the time of writing subscription fees are as follows: (1) SC 900 per 12-month subscription or per 100 log-ons, \ whichever comes first; (2) $C 600 per 6-month subscription or per 50 log-ons, whichever comes first; and (3) $C 200 ,per 1-month subscription or per 20 log-ons, whichever comes first.

- Study reference - basic bibliographic data.
- Study area and population characteristics - data about the location of the study along with population and site data.
- Environmental focus of the study - text and keyword fields that describe the environmental asset being valued, the stressors on the environment, and the specific purpose of the study.
- Study methods - technical data on the actual study, along with the specific techniques that were used to arrive at the results.
- Estimated values - the dollar values that are presented in the study as well as the specific units of measurement.
- Abstract and alternative language summaries - an abstract of the study available in English, French and Spanish.

Figure 3 below summarises the information currently available in the EVRI, organised into the four main categories used to search for data. Clearly, a great number of the records held by the EVRI concern the valuation of **water resources.** Moreover, an extremely useful characteristic of the database is that it holds benefit function data, in addition to average unit value data. Hence, the EVRI facilitates the application of both approaches to benefit transfer (discussed in the earlier I Chapter on valuation). Of the three specialised modules, the Searching and Screening Modules facilitate the use of the information contained in the database. The Capturing Module, in contrast, permits one to enter study

**Figure 3: Summary of Records Currently Available in the EVRI**




**Source:** Environment Canada (1998)

**Table 9: Contents of Register Relating to Water Resources**

| Impact | Valuation Technique | Data Entered[2] |
|---|---|---|
| **Coasts and Estuaries:** | | |
| Coastal protection schemes | Contingent Valuation | Yes |
| | Implicit Valuation[1] | Yes |
| Habitat protection | Contingent Valuation | Yes |
| | Implicit Valuation | No |
| Recreation | Contingent Valuation | Yes |
| | Travel Cost | No |
| | Implicit Valuation | Yes |
| Navigation | Contingent Valuation | Yes |
| Remediation | Implicit Valuation | Yes |
| **Fresh Water:** | | |
| Angling | Contingent Valuation | Yes |
| | Travel Cost | No |
| | Implicit Valuation | Yes |
| Recreation | Contingent Valuation | Yes |
| | Travel Cost | No |
| | Implicit Valuation | No |
| Navigation - recreation | Contingent Valuation | Yes |
| | Travel Cost | No |
| | Implicit Valuation | No |
| Navigation - commercial | Contingent Valuation | Yes |
| | Travel Cost | No |
| | Implicit Valuation | Yes |
| Extinction of species | Contingent Valuation | Yes |
| Raw materials | Contingent Valuation | Yes |
| | Implicit Valuation | Yes |
| Flood protectiori | Contingent Valuation | Yes |
| | Local cost data | No |
| | Implicit Valuation | Yes |
| Remediation | Contingent Valuation | Yes |
| | Local cost data | No |
| | Implicit Valuation | Yes |

***Source:*** *Metroeconomica & NETCEN (1998)*

**Notes:**

1 Implicit valuations refer to approaches which derive values from decisions made by regulators or policy makers.
2 Where values exist and have been entered in the Register they are denoted with a "Yes", cases where values are likely to exist, but have not included in this version of the Register, are denoted with a No".

findings into the database, and is consequently only available to suitably qualified subscription holders upon invitation. An obvious advantage of this module is that the records of the database are continually being up-dated.

As the name suggests, the **Searching Module** allows the user to identify potential studies in one of two ways: (1) a general text-based search; or (2) a more directed search based on "protocol" which is consistent with the literature on benefit transfer. The purpose of the **Screening Module** is to permit the selection of a subset of studies from the larger set identified by the Searching Module. Basically, this module adds another level of sophistication to the search and selection process. Further details of these modules and the EVRI in general may be found on the web-page.

### 3.4.2 The Environment Agency's Registry of Environmental Values

In early 1998, the Environment Agency (EA) contracted Metroeconomica and NETCEN to undertake the first part of a programme of work to develop a database of economic values[11]. The project sought to develop a user friendly database (the so-called Register of Environmental Values) and to populate this Register with economic valuation data relevant to the EA's numerous functions. The overall aim was to provide Agency staff with a single source of accessible valuation data that they could use in a range of decision-making contexts.

The general structure of the Register comprises four levels, of increasing detail:

- The selection of a desired environmental sector, e.g. freshwater, coasts, estuaries, etc.
- The selection of a desired environmental impact or asset within the selected sector, e.g. coastal protection scheme - impact on recreational values.
- Summary data on the economic value of selected impact(s) or asset(s), e.g. £7.85 per (local) visit, £10.85 per (part-day) visit, etc.
- Reference data on the primary study, e.g. title of study, year of study, literature reference, etc.[12].

As the primary purpose of the Register is essentially to facilitate benefit transfer, specifically "benefit value" transfer, it also contains fields which provide summary statements relating to: (1) the robustness (i.e. validity and bias) of the valuation technique employed; (2) the physical uniqueness of the study (e.g. site-specificity), (3) what statistical data is provided, and (4) the overall transferability of the data (e.g. low, medium,high). In terms of data on water resources, Table 9 below surmmarises the contents of the Register in its present version.

[11] See Metroeconomica and NETCEN (1998).

[12] The aim here is to facilitate transparency in the system, assist users in assessing the transferability of estimates, and to permit validation of the data entered into the system.

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# Chapter 9

# SUSTANIABLE TOURISM

*by*

*Akistair Hunt*

## 1 INTRODUCTION

As one of the fastest growing sectors of the economy, tourism has a key contribution to make to sustainable development. The growth in numbers of international tourists is shown in Table 1.

**Table 1: International Tourist arrivals by Region (millions) (Source WTO 1996).**

| Region | 1985 | 1990 | 1995 |
|---|---|---|---|
| Africa | 9.7 | 15.1 | 18.7 |
| Americas | 66.4 | 93.6 | 110.6 |
| East Asia/Pacific | 30.8 | 53.1 | 83.0 |
| Europe | 213.8 | 286.7 | 333.3 |
| Middle East | 6.2 | 7.6 | 11.1 |
| South Asia | 2.5 | 3.2 | 4.3 |

The size and importance of the tourism industry is also reflected in terms of its' value. In 1995 tourism receipts accounted for 30% of the value of world trade in the service sector and almost 8% of the total value of world trade.

Sustainable tourism can be seen as a balance of five factors: economic growth, subjective well-being of the local population, unspoilt nature/protection of resources, healthy culture and, finally, optimum satisfaction of guest requirements. Environmentally and socially compatible tourism would mean maximising the positive relationships between these factors while keeping the negative repercussion, particularly on nature and culture, to a minimum. This review of the subject gives particular emphasis to the means by which the negative environmental repercussions of tourism can be minimised. The first section examines how environmental concerns can be accounted for in deciding whether to develop a tourist site. It focuses on one, fast growing type of tourism - eco-tourism - but the principles outlined are common to any tourist development being considered. The second section illustrates how a tax instrument could be devised to internalise the environmental effects of a tourist development.

## 2 THE ECONOMICS OF NATURE TOURISM[1]

Consider a tropical rain forest somewhere in South Asia. Though it is presently inaccessible, an extension of a national highway will soon open up the area. Various potential users of the resource become interested in the possibilities. A *campesino* considers the area's potential for agricultural development and sees dollar signs. A logger looks at timber resources and also sees money to be made. A nature tourism operator reaches the same conclusion. The government, on the other hand, sees a major quandary.

Each potential user of this previously inaccessible resource seeks monetary benefits from its exploitation and use. Their different visions, however, are likely to conflict. Some uses will preclude others, though certain combinations of uses can coexist.

Governments face the difficult decision of how best to use these natural areas. Should they be preserved intact? Should they be exploited for short-term profits? Should they be converted to another use such as agriculture? Should tourism be encouraged? How should decisions be made?

Keeping the natural area relatively intact, thus ensuring the survival of the plants and animals it contains, providing opportunities for recreation and tourism, and maintaining the other benefits such as watershed protection may sound reasonable to someone who has no economic stake in the alternatives.

However, as mentioned earlier, the logger may also have plans for the same site. These alternative uses of natural areas often appear extremely attractive in the short term. Many tropical forests, for example, contain large amounts of valuable timber. Other areas can be

1 This section is based on a paper by Paul B. Sherman and John A. Dixon entitled *The Economics of Nature Tourism: Determining if it pays in Nature Tourism*, edited by Tensie Whelan. 1991 Island Press. Washington.

converted to uses such as agriculture, grazing, or tree corps. In many cases, a traditional economic analysis would find that some form of development would provide greater financial returns than the modest direct returns from maintaining an area in its natural state, even if the latter supports ecological functions and a small-scale nature tourism industry. As a result, there is often substantial pressure to convert and exploit natural areas.

In addition to pressure on natural areas from proposed development projects, in many countries an even greater threat is gradual encroachment and resource extraction by nearby residents. These patterns of resource use, whether non-sustainable (a logger clearing steep slopes for annual crop production) or sustainable (collection of various minor forest products in wooded areas), are not easily stopped by the mere creation of a protected area. The economic pressures on the resource users continue and, barring use of extreme regulatory measures, must be taken into account if effective protection is to be provided.

Thus economics, and the quest for financial returns, will most likely drive decision-making about the use of natural areas (though politics may also be a factor). The government must arbitrate and make decisions that will allow some uses and prohibit others. When deciding whether to allow the forest or other natural resources to be used for nature tourism, the government should ask the following type of economically driven questions.

Does nature tourism pay? That is, are the benefits from nature tourism greater than the costs?

Which benefits are forgone if the site is not developed for alternative uses? What are the total costs of protecting a site for nature tourism?

Who owns the resource? Is it publicly or privately owned?

What approach should be taken in carrying out an economic analysis? Should the decision-making criterion emphasize public gain or private financial benefit?

What are the total benefits from nature tourism, including benefits that may not be translated directly into tourist dollars. Can these benefits be identified and quantified?

Is nature tourism an economically efficient way to generate income and help maintain and conserve natural areas?

In reality, these economic questions must be framed at two levels: financial and social. We need to know if nature tourism will pay as a business venture, and if it will pay as a social investment, particularly if governments must protect natural areas to support it.

A financial analysis of the various alternative uses of an undeveloped natural area alone is often misleading. Such an analysis is designed only to examine costs and benefits as measured by market prices - it leaves out many important factors that are not bought or sold. Many of the benefits of conserving natural areas are difficult to measure (e.g., bio-diversity, watershed protection, filtering of pollutants). These benefits are not exchanged in markets and, consequently, the value of conserving, rather than developing, an area is often underestimated in a financial analysis. This leads to a bias toward development and exploitative use of an area, with the end result being that fewer natural areas are protected than would be the case if all of the benefits of conservation were acknowledged.

A social welfare analysis, on the other hand, will account for the social and intangible benefits and costs of an area, including conservation. This type of analysis can distinguish between private, financial benefits accruing to individuals or firms and public, social benefits (what economists call "social welfare benefits"). It facilitates the decision-making process by providing three categories that describe the nature and scale of the economic benefits of the site and who receives them. The three categories are: privately beneficial, socially beneficial, or undetermined benefits.

## 2.1 PRIVATELY BENEFICIAL

In some areas, the economic benefits directly obtainable by individuals, groups or firms are larger than the associated costs or the benefits of alternative uses. In these cases, the individual will provide the "service" (i.e. recreation a protection of a natural area) without government intervention. Examples of such cases are not uncommon, but the areas tend to be small and the nature of the service provided rather specific.

Privately run recreational areas such as campgrounds, ski resorts, or game reserves, for example often keep limited portions of an area in its natural state, in order to keep customers. Outstanding areas such as the Galapagos Islands or Yosemite National Park, though currently administered by national governments, are also examples of privately beneficial areas.

Some natural areas, currently unprotected, may be considered so important that private individuals or groups feel strongly enough to purchase them from their current owners. Conservation groups such as the Nature Conservancy in the United States have begun acquiring important natural areas threatened by development. These groups pool donations from their members to acquire development rights or to buy areas that might not be protected otherwise. The contributors to such private

conservation efforts perceive the benefits to outweigh the costs. In addition, the sites often become nature tourism destinations for the group's members.

## 2.2 SOCIALLY BENEFICIAL

In this more common scenario, the net benefits to society at large are positive, but one individual or privately owned concern cannot capture all the benefits effectively and therefore is not willing to provide protection or preservation. Protection of upper watershed areas, for example, may be justified by their effects on the water supply and water quality for a downstream area, more than for their tourism receipts. In East Africa, government support of wildlife parks is usually socially beneficial in terms of attracting tourists and the wide range of associated tourist expenditures both within and outside protected areas.

Many nature tourism destinations fall in the socially beneficial category. Since these areas may not be capable to generating direct revenues greater than their costs, some government support may be needed to maintain them.

## 2.3 UNDETERMINED BENEFITS

In some cases, it may be difficult to determine whether the net benefits of maintaining a natural area for tourism and other uses are positive or negative. The costs of protection may be known, but the benefits may be diffuse or difficult to measure. This is especially true of wilderness areas or remote locations where nature tourism may be sporadic or of very low intensity. Governments may well decide to protect some of these areas, but at what cost and to what extent? These are issues that need to be addressed.

Let us return to our initial scenario where the logger and the nature tourism operator are eyeing the same piece of property. They will assess the economic benefits and costs differently, but will use a typical financial analysis model. It is the government's responsibility to conduct a social welfare analysis that includes impacts of alternative development options, regardless of whether or not these impacts occur on-site or elsewhere in the economy.

Below we demonstrate how to perform and implement a social welfare analysis for ecotourism projects. Our thesis is that ecotourism needs to be considered both as a business (and therefore subject to a financial profit/loss analysis) and as a type of resource use that helps ensure other, long-term social goals (such as protection of natural areas, biodiversity, or retaining options for the future).

## 2.4 BENEFITS AND COSTS OF NATURE TOURISM

In performing a social welfare analysis of nature tourism, we will need to weigh its costs and benefits. If the net benefits to society (total benefits minus total costs) are greater than zero, then nature tourism is a potentially desirable form of land use. However, even if these net benefits are positive, this does not imply that nature tourism is the best use; the net benefits of nature tourism must then be compared with the net benefits of other alternative forms of land use. Ideally, each piece of land should be used according to its "highest and best use", that is, the use that generates the greatest net benefits to society. In broad terms, there are three additional alternatives that should normally be considered: leaving the land in its natural state but not allowing nature tourism; developing resort tourism; or developing the

**Table 2 Benefits that accompany Nature Tourism**

| | |
|---|---|
| 1 | Watershed values |
| | Erosion control |
| | Local flood reduction |
| | Regulation of stream flows |
| 2 | Ecological processes |
| | Fixing and cycling of nutrients |
| | Soil formation |
| | Circulation and cleansing of air and water |
| | Global life support |
| 3 | Biodiversity |
| | Gene resources |
| | Species protection |
| | Ecosystem diversity |
| | Evolutionary processes |
| 4 | Education and research |
| 5 | Consumptive benefits |
| | Timber |
| | Wildlife products |
| | Non-timber forest products (e.g. edible plants, herbs, medicines, rattan, building materials, rubber) |
| 6 | Non-consumptive benefits |
| | Aesthetic |
| | Spiritual |
| | Cultural/historical |
| | Existence value |
| 7 | Future values |
| | Option value |
| | Quasi-option value |

***Source:*** *Dixon and Sherman 1990.*

land for other uses such as agriculture or housing that would normally preclude tourism.

### 2.4.1 Benefits

Table 2 shows the types of benefits that may accompany nature tourism: watershed protection, ecosystem preservation, biodiversity, education and research, consumption, non-consumptive benefits, and future values.

While some benefits of nature tourism may be relatively easy to value (e.g. tourist receipts), others such as biodiversity, non-consumptive benefits, and future values are more difficult to analyse (Conrad 1980). Still, much can be done either to directly value these benefits or at least to provide a framework where choices can be made that explicitly consider qualitative benefits.

Benefits from nature tourism can be analysed in several ways. Specific examples follow:

**Spatial dimension.** Benefits can be divided into local, regional, national, or global benefits. They may include employment opportunities, new markets for locally produced goods, and indirect benefits such as improved infrastructure associated with tourism development.

Regional benefits fall into the same categories, though their overall importance may be less due to the larger size of the region as compared with the local area. For example, creation of fifty jobs may be significant locally but relatively unimportant from a regional perspective.

The national benefits of tourism may include tax revenues collected from visitors, the additional foreign exchange earnings from international visitors to the country, and any capital investment from either domestic or foreign sources that might otherwise have been spent outside the country's borders. They may be larger or smaller than the local/regional benefits. If a new nature tourism development serves as a substitute for another, pre-existing site within the country, there may be no additional net benefits for the nation associated with the new site. Only to the extent that the new site attracts new visitors or increases the average length of stay will the benefits associated with the new site truly add to national benefits.

The global benefits of nature tourism stem from benefits such as watershed protection, ecosystem support, biodiversity, and consumption. Since nature tourism frequently requires the preservation of an area in a relatively pristine state, other benefits associated with this protection arise. For example, nature tourism may be consistent with species and habitat protection (though not necessarily so). There may also be existence and option values generated throughout the world. One problem with these global benefits, however, is that they are freely provided - there is no obligation on the part of recipients to compensate the country of origin.

**Private versus social benefits.** The broad range of benefits associated with nature tourism can be divided further into private and social benefits. The private benefits associated with nature tourism are the financial returns received by those providing tourism services. Tour operators, transportation companies, lodging operators, food and equipment suppliers - all these groups receive financial remuneration for services provided. These profits are what drive private sector interest in nature tourism.

Social benefits can be described broadly as any gains in social welfare, either direct or indirect, associated with nature tourism. Such gains may be either financial in nature or non-monetary, such as ecological benefits. A number of the benefits listed in Table 2, for example, are primarily social benefits, and future values. Other benefits such as watershed protection and education/research are partly private, partly social benefits.

While government officials may be primarily interested in the amount of expenditures and employment opportunities generated, they also should be aware of the larger picture - that is, the social benefits associated with nature tourism. Most private sector interests will be interested only in financial aspects and specifically the profits they are able to earn. Non-profit organizations and educational groups often fall in between - they may seek to earn at least a nominal profit but also may consider other social benefits associated with nature tourism.

**Primary versus secondary benefits.** Benefits can be divided into primary and secondary tourism-related expenditures. Primary expenditures are direct purchases by tourists of goods and services. Secondary expenditures (sometimes called "indirect benefits") occur when the recipients of the primary expenditures spend the money they receive from tourists. For example, when a native tour guide is paid and I then spends his salary on food and housing, these expenditures are secondary expenditures. These secondary expenditures often work their way through the system many times, creating what is termed a "multiplier effect" - the initial primary expenditures are multiplied as the money is spent and then spent again.

The extent of the multiplier effect depends on who is the recipient of the direct expenditures. If the bulk of the primary expenditures is for imported goods or services provided by foreigners, who then take the money is re-spent and the multiplier will be small. If the

primary expenditures go to local entrepreneurs, who spend the money locally, and which then gets spent again locally, the multiplier will be much larger.

In each round of expenditures, the money re-spent is less than the amount spent in the previous round - some of the money will be saved, some will go to pay taxes, some will leave the area. The money that is not re-spent is known as "leakage" - it leaks out of the local economy.

The multiplier effect can also be applied to employment. The employment multiplier looks at how many jobs are indirectly created for each job directly created by the tourist operation.

## 2.4.2 Costs

As with benefits, costs can be categorized in a number of ways and will be viewed differently from different perspectives. The following presents one potentially useful categorization of costs.

**Direct costs.** Direct costs are financial outlays associated with the establishment and maintenance of a nature tourism site. These costs may be born either by the government sector, the tour operator, or, as is often the case, split between the two. In many cases, the highest single cost is developing access to the site. By its very nature, most forms of nature tourism take place off the beaten path. While remoteness is a draw for many people, access must be easy enough not to discourage potential visitors. Depending on the site and the activity involved, access may require upgrading or development of roads, airstrips, or boat docking facilities. Since these facilities may provide benefits in addition to access to nature tourism sites, governments may be willing to contribute to their cost.

Other facilities for tourists can be developed and provided either by governments or private operators. Lodging and food service are more amenable to private development than multipurpose items such as roads. Regardless of who develops the facilities, however, it is critical that ample attention be paid to handling the wastes such as trash and sewage. The full costs of dealing with these items must be considered at the development stage. Similarly, the costs of maintaining and repairing all facilities must be accounted for.

**Indirect costs.** Certain types of nature tourism may be associated with damages indirectly caused by the existence of the nature tourism industry. The most common example of these indirect costs are damages caused by wildlife, either inside or beyond the boundary of the tourist enterprise. These damages may include crops trampled or eaten, as well as harm to people, livestock or materials. Examples include predation of livestock in Africa, and problems with tigers in India. These indirect costs can create local resentment of plans to expand areas and associated nature tourism activities.

Private operators usually will not consider these indirect costs in their financial analysis of nature tourism. Governments, on the other hand, should anticipate such problems and develop means of compensating those adversely affected. Though not compelled to compensate for such damages, governments should realize that community attitudes towards the success of nature tourism will be much more positive if nearby residents are reimbursed for any damages they suffer. The costs of such compensation programs should be considered a cost of doing business, and included in a social welfare analysis.

**Opportunity costs.** The opportunity costs of nature tourism are the benefits that society or individuals must give up if nature tourism precludes other uses of an area. They may include forgone resources from the area (such as timber, animals, edible plants), as well as any resources that could have been developed through more intensive exploitation or conversion to some alternative use. The magnitude of the opportunity costs will depend on the type of nature tourism and the potential alternative uses of the area. If not profitable alternative exists for the area, and then the opportunity costs may be very low or even non-existent.

Not all opportunity costs will be apparent in a standard financial analysis. Nature tourism may involve restrictions on local use of resources; for example, hunting is banned in many national parks, which means that local residents can no longer count on those resources for sustenance or livelihood. The value of products no longer available is also an opportunity cost and should be included in the analysis. As with the case of indirect costs, local residents should be compensated for any losses they suffer as a result of establishing a nature tourism site.

## 2.4.3 Comparing Costs and Benefits

The exercise of comparing costs and benefits differs depending on whether one is using a financial or social welfare analysis. In the first case, it may try to determine the profitability of a nature tourism enterprise. Alternatively, a social welfare analysis may seek to determine whether nature tourism is socially desirable. The latter is much broader in scope, and often more diffficult. While there are well established methods of financial analysis based on market prices, social welfare analysis must consider a broader range of benefits than just financial returns. All effects on individuals - whether

monetary, environmental, cultural, or otherwise must be included. Since many of these effects have no market prices, the task of placing values on them is a complex one, too complex to develop here. Interested readers can find details on valuing environmental effects in Dixon and Sherman (1990).

**Financial analysis.** A private operator considering development of a nature tourism site or co-ordinating trips to an established site should perform a financial analysis to determine the profitability of the proposed investment. The financial analysis begins with a list of the expected costs and benefits that will occur in each year. Capital costs can be entered in the year they are spent, spread over the life of the project (amortized) if internal funds are used, or spread over the loan period if costs are included, along with expected changes over time. The interest rate used will be either the market rate of interest (if funds are borrowed) or if internal funds are used, the opportunity cost of capital (the returns that these funds could generate if invested in another project). Revenues will be based on projections of number of visitors multiplied by the revenue per visitor.

Governments may also choose to perform a financial analysis when considering nature tourism as a form of land use. This will show whether nature tourism will pay for itself, or whether a subsidy will be needed. In some situations, subsidies will be warranted - if, for example, nature tourism will achieve other national goals, such as increasing foreign exchange receipts or developing employment opportunities in low-income regions.

**Social welfare analysis.** From a national perspective, the economic analysis should consider the social benefits, and social costs, of the proposed activity. The most common form of social welfare analysis is benefit-cost analysis (BCA). Like financial analysis, BCA involves the evaluation of a stream of benefits and costs over some chosen period of time. Here, however, the focus is not just on financial costs and benefits but on the social welfare of the community as well. The prices used are not always market prices, instead, what might be called "economic efficiency prices" are used. Distortions such as those caused by taxes, subsidies, or regulatory effects are removed so that prices reflect true resource commitments to society.

BCA may also include effects that are ignored in a financial analysis. If tourism is accompanied by some undesirable environmental effects, the value of these effects is also considered. This is true whether these effects occur at the tourism site or some distance away from it.

Overall, there are a number of differences between a social BCA (sometimes referred to as a SBCA) and a financial analysis. Whereas a financial analysis includes, subsidies, taxes, and interest payments, an SBCA considers these as transfer payments. A transfer payment is simply a transfer of resources between two different units within the economy; for example, income tax is a transfer from the individual to the government. Transfer payments change the distribution of income but do not change the overall amount of income. Hence, a BCA does not include them in evaluation overall social welfare changes.

A financial analysis usually uses market borrowing rates to determine the discount rate; a SBCA uses a social discount rate that is usually lower and reflects a number of societal decisions. Prices may be similarly adjusted. Details of the differences between financial and SBCA can be found in and Dixon and Hufschmidt (1986). These references also discuss decision criteria such as net present value (NPV), internal rate of return (IRR), and benefit-cost ratio (BCR).

While relatively straightforward in theory, preparing a BCA for a proposed nature tourism project is often difficult in practice. Evaluating the environmental and cultural effects, both positive and negative, associated with nature tourism is particularly problematic. In most cases, at least a portion of these effects will remain unquantified. Therefore, the quantified net benefits will often be less than the actual total benefits from nature tourism.

If the quantifiable benefits alone are greater than the measured costs (and also greater than the net benefits from alternative uses of the site), nature tourism is a viable option. When the quantifiable benefits associated with tourism are less than the costs, however, or when the net benefits are positive but less than the benefits associated with another alternative use, the decision-making process becomes more difficult. In both these cases, governments must decide whether the non-quantified benefits associated with nature tourism justify either a subsidy or forgoing the additional benefits associated with the alternative land uses.

The siting of major infrastructure facilities, such as dams, ports, or airports, may have potential direct or indirect impacts on areas used for or suitable for nature tourism. The social BCA of such projects must examine all the impacts as well as other possible alternatives. (The suggested project is assumed to pass a financial analysis - e.g., positive net benefits - or else it would not have been proposed.)

A classic case in the United States occurred in Hell's Canyon. In this case, a series of dams would have flooded Hell's Canyon and disrupted the last wild river

in the area, a popular recreation site for residents and visitors. In an innovative study, Krutilla and Fisher examined a number of aspects of the preservation-versus development alternatives. In the end, the dams were not built and the canyon was left intact. This type of analysis is sometimes referred to as "opportunity cost analysis". Here, the analyst compares the net economic benefits of alternative uses. While the net economic benefits of development usually can be monetized relatively precisely (in the case of Hell's Canyon, these were power generation benefits), nature tourism and the associated ecological protection often involve many nonquantifiable benefits. The net monetary benefits of development are first compared with the quantifiable benefits from nature tourism. If the former are larger than the latter, then the analyst looks at how large the non-quantifiable benefits would have to be to outweigh the total benefits of development. If it appears that the nonquantifiable benefits will be at least this large, then nature tourism would be the preferred option.

In addition to environmental and cultural effects, other non-quantifiable effects include job creation and foreign exchange impacts. While these may be desirable, they are not easily converted to dollar terms. Moreover, these effects must be compared with other possibilities for the area - while creation of twenty jobs in a nature tourism enterprise might seem beneficial, it is not truly so if some alternative development would provide forty jobs. As discussed earlier, the benefits of nature tourism must be weighed against the best alternative use - only the difference between the two alternatives is actually a benefit.

In some cases, a natural area may have been given protected status already, and the analysis is being performed only to determine whether or not nature tourism should be allowed. In this case, the analysis is simpler, since development alternatives need not be evaluated. First, the financial costs and benefits of nature tourism are estimated. The net financial benefits are then compared with other effects, such as environmental impacts (which should also be monetized to determine if social welfare will be improved by allowing tourism.)

### 2.4.4 Distribution of Benefits and Costs

The potential profitability of a nature tourism enterprise is only one of many factors used to determine whether or not it is desirable. While profitability may be the most important factor in a private operator's decision, governments may be more interested in how the benefits and costs will be distributed. If nature tourism will provide large profits to a foreign-owned company but adversely affect local residents, it may not be in the country's best interests to allow it. On the other hand, a locally owned company that makes an effort to include nearby residents may be much more desirable, even if the enterprise generates little or no profit.

Consider a publicly owned site suitable for outdoor recreation and believed to be capable of supporting a profitable tourism operation. Three alternatives exist: selling the site to a private operator; leasing the use of the site to a private operator; or establishing a government-owned and operated enterprise.

If the government sells the site, it loses a great deal of control over what type of development and management occurs. The buyer will probably attempt to maximize profits, possibly at the expense of social benefits. On the other hand, if the government leases the rights to use the site, it can establish conditions under which the leaseholder must operate. Under this set-up, the government can mandate conditions that are more socially beneficial by, for example, establishing a minimum percentage of jobs that must go to local residents or limiting the allowable number of visitors per year.

Governments can also choose to operate the tourism enterprise alone. While governments often may be less efficient that the private sector, this option gives the government the greatest leeway in operating the tourism enterprise in a manner that maximises local benefits. Governments must make a policy decision on this issue does the increase in local benefits outweigh the reduction in profits?

Private ownership of nature tourism sites may be a viable option in some cases. While governments may have less oversight authority, a private owner will recognize that it is in his own self-interest to maintain the site to ensure its continued attraction to visitors. Monteverde Reserve in Costa Rica and the Community Baboon Sanctuary in Belize are but two examples of successful privately owned nature tourism destinations (Boo 1990).

Ideally, nature tourism should be beneficial for everyone involved - tourism operators receive profits, governments receive tax revenues and foreign exchange, visitors enjoy their experience, and local residents receive jobs and increased income from visitor expenditures. Too often, however, this is not the case for the latter. Unlike everyone else whose participation is voluntary, local residents may be unwilling participants and are the ones who lose.

Opening an area to tourism often means a dramatic change in lifestyle for local residents. Depending on their previous level of isolation, village life may be totally disrupted by drastic cultural changes. The introduction of relatively wealthy visitors may cause large price

increases and shortages of certain goods in local markets. While this may mean profits for a few, the general population often suffers ill effects. In some cases, markets in drugs and prostitution, and adoption of other undesirable practices, may develop.

Such negative impacts need not always occur, however. With proper foresight, many can be avoided or minimised. Given the importance of this issue, special attention must be given to ways of maximizing the local benefits; this topic is discussed more fully in the next section.

In addition to the effects on local people, the effects on the local environment must also be considered. These can arise either directly, as a result of tourism and the additional people using the area, or indirectly, from increased access associated with tourism development. Erosion from hiking trails, deforestation for firewood, increased trash and sewage, clearing for accommodation facilities - these consequences and others can be severe in the absence of adequate regulatory and enforcement efforts by the authority responsible. And if a new road is built to ease access for tourists, for example, it also may make it easier for people to move into the area, with subsequent clearing of forests for farmland. Governments should take measures to ensure that this secondary development does not unduly threaten the area.

The adverse effects of these impacts are often felt hardest at the local level. If tourism is not to "destroy" itself, proper planning and management are critical at an early stage.

Nature tourism can also bring about positive influences on the environment. Revenues collected through entrance fees, hotel taxes, sales tax, or other means can provide much needed funds for natural area protection and management. Properly allocated, these funds can be used to improve an area and minimize any negative environmental effects associated with tourism.

The distribution of benefits and costs can be considered within an economic analysis in two ways. The analysis could include only those benefits and costs that remain in the country (or region or village). This would eliminate operations that primarily benefit outside interests at the expense of local residents. Alternatively, weights can be placed on benefits and costs that accrue to different groups. For example, benefits that accrue to local residents in poor villages might be multiplied by a factor of two, while benefits to wealthy developers might be weighted by only a fraction. Similarly, costs that adversely affect local residents might be given greater weight than adversely affecting the wealthy. It is also possible to set a constraint on the allowable distribution of benefits. This involves setting targets that establish a minimum acceptable distribution of benefits among a designated low-income group or area. Only projects that will provide a certain percentage of benefits to member of this group are given consideration under this scenario.

## 2.5 NATURE TOURISM AND ECONOMIC DEVELOPMENT

Properly implemented, nature tourism can integration conservation and rural development by helping to protect valuable natural areas by providing revenues for planning and management, stimulating economic development through tourism expenditures, and providing jobs and markets for local goods.

Nature tourism has the potential to help economic development at both the local and the national level. Depending on the scale of the nature tourism industry relative to the size of the local economy, the effect on the local level can be anywhere from minimal to substantial. At the national level, nature tourism is likely to have less impact, but it still may have significant influence in countries with small economies or where the potential size of the industry is very large. In Kenya, for example, the safari industry generates foreign exchange earnings of some $350 million to $400 million per year and is Kenya's largest source of foreign exchange.

In this section, we will first look at how national and local governments can maximize the revenues they receive from nature tourism. Then we will discuss how to maximize benefits for local residents.

### 2.5.1 Maximising Government Revenues

While nature tourism has the potential to provide substantial benefits to countries with outstanding nature tourism sites, this will not always be the case. Too often the majority of benefits accrue to the tour operator and little remains in-country. Boo (1990) cites a World Bank study that estimates that over one-half of gross tourism revenues in the developing countries leak back to developed countries. This is hypothesized to be even higher in the least-developed countries, where most of the goods used by tourists are imported. Nevertheless, there are a number of mechanisms governments can put in place to increase the benefits their country receives from nature tourism.

**User fees.** The easiest method of capturing benefits from nature tourism is to charge a fee to use the area. Though many countries already charge small fees at cultural sites and in national parks, few countries have instituted fee schedules that reflect consumers' willingness to pay. While a small, token payment is clearly better

than no fee at all, there is not reason for a country, especially a developing country to subsidize the cost of foreigners' visits.

Developing countries should consider adopting a two-tier fee system, with a lower charge for domestic residents and a higher charge for international visitors. Some countries have already instituted such a system: China, for example, uses a two tiered fee structure for most cultural and historic sites. Given the expense of international travel, even a relatively high fee of US $10 or more per day would probably have a negligible effect on the total number of visitors. This is especially true for unique areas that can handle only a limited number of visitors. In the Mountain Gorilla Project (MGP) in Rwanda, for example, foreigners are charged an entrance fee of $170 per day and yet demand has remained strong. It has been noted that this is among the highest such fee charged anywhere in the world and may be near the upper limit of visitor willingness-to-pay.

User fees help to support the Saba Marine Park in the Netherland Antilles. Since the main attraction of the park is its scuba diving and snorkelling, divers are charged $1 per dive, paid through the dive boat operators, to support conservation activities. This modest fee provides valuable revenue and is unnoticed in the overall costs of diving.

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Fees for government-owed accommodations near nature tourism sites should be priced at levels comparable to privately owned accommodations. Camping fees could also be set on a two-tier system as suggested for entrance fees. At present, many national parks charge very low accommodation or camping fees, resulting in excess demand for these facilities and insufficient funds for operation and maintenance. Businesslike behaviour can be as beneficial to public operations as it is to private ones.

**Concession fees.** In addition to charging fees directly to visitors, fees can also be charged to individuals or firms who provide services to these visitors. This would include licensing of concessions for food, lodging, transportation, guide services, and retail stores. By auctioning or leasing the rights to operate such concessions, governments can control the types of development in and nearby nature tourism sites and simultaneously raise revenues to help maintain the area. Governments also can impose conditions on concession leases to address other objectives such as hiring local employees or selling locally produced goods.

**Royalties.** Establishment of royalty systems on activities and products in tourist areas is another potential source of revenue. For example, permission for books, photos, or films to be made at tourism sites could be exchanged for some percentage of the revenues made on these items. In the Saba Marine Park, sales of T-shirts and guidebooks are a major source of revenue. Such souvenir sales, either direct or via licensing, can be major revenue producers.

**Tax policies.** Governments can enact tax policies to increase the revenues they receive from nature tourism. Perhaps the most common type of tax is a hotel room tax, which is also relatively popular among residents since it falls primarily on visitors. Hotel room taxes of 5 to 10 per cent are found in many areas.

Special taxes also can be enacted near popular tourist sites. In pre-war Cambodia, for example, the famous complex of ruins associated with Angkor Wat was maintained by the government but was completely open to visitors without any formal payment. This enhanced the visitors' enjoyment of the site and allowed casual exploration. The government, however, collected a special tax on all hotel rooms in the nearby town of Siemreap to support its conservation and preservation efforts. Since virtually all visitors to Angkor stayed in these hotels (and the ruins were the main reason for people coming to the town), this was an effective and unobtrusive means of revenue collection.

Other forms of taxes include sales or excise taxes on tourist-related goods and services. These might be levied on food bought in restaurants specialized equipment, and tour guide services.

**Donation programs.** Governments can take advantage of the increasing international awareness of the problems faced by important natural areas by establishing and promoting donation programs. Such programs can be geared toward both tourists and non-tourists. For tourists, guides can point out the problems of protecting the area, and encourage donations to help alleviate these pressures. To reach non-tourists (and potential tourists), governments might joint with conservation groups in a campaign to raise funds from interested individuals.

## 2.5.2 Maximizing Local Benefits

One of the critical issues concerning nature tourism is its impact on local residents, and especially rural villagers, in developing countries. Since much of the growth in nature tourism will take place in such areas, it is important that steps are taken at an earlier stage to ensure that local residents benefit from the tourist industry.

The most direct way of benefiting local communities is to employ as many residents as possible in tourism-related services. This includes jobs in

restaurants, accomrnodation facilities, and as guides. Other employment possibilities include construction activities, helping to build trails, providing daily maintenance, and retail sales. If local workers do not possess the skills needed, training programs should be considered before bringing in workers from other areas.

Use of locally produced goods will also benefit the community. Governments and/or NGOs can help farmers grow crops and livestock to supply tourist facilities. Promotion of local handicrafts also provides income-earning opportunities.

Local residents also will benefit if a portion of fees collected from nature tourism is earmarked for them. This is especially important if local residents have had to give up use of an area to ensure its continued existence for tourism. For example, in many countries, collection of firewood, food, timber, or other products is not allowed in national parks. This loss of income may be devastating to people already living at or near subsistence levels.

Using a portion of the fees collected to compensate local residents provides a means of offsetting these losses. In the Chitwan national Park in Nepal, for example conservation of this important rhino habitat is promoted by allowing villagers to harvest elephant grass periodically, thereby helping to meet their needs for income from this valuable thatch material while discouraging illegal harvest of park resources.

Compensation can be provided in a number of ways. One possibility is to develop alternative supplies of the resource outside the tourist destination. Woodlots for firewood, captive breeding for wildlife, and farms or plantations for plant species are examples.

Developing a substitute for the lost resource is another form of compensation. If the resource was used for good, for example, a different food crop could be substituted. If it was used to generate income, other types of income-generating activities can be used to offset losses to local residents.

Fees collected from nature tourism also can be used for community development activities. Construction of schools, sanitation facilities, electricity, water systems, and health clinics are potential forms of compensation. Residents must be made aware, however, that the provision of these facilities are, at least in part, compensation for losses associated with tourism.

Compensation is also warranted in cases where there are indirect costs to local residents from nature tourism, for example, damage from wildlife. One example of compensation is the case of traditional Masai herders and Kenya's Amboseli National Park. Both the Masai cattle and the area's wildlife depend on water and pasturelands located within and outside the park; the needs and range of both cattle and wildlife change during the year depending on the amount of rainfall and pasture availability. Restricting wildlife to the park's boundaries and excluding all cattle would result in a decreased population of both.

A compromise solution between the local Masai and the park authorities resulted in substantial economic gains to both parties. The solution included payment of a grazing compensation to the Masai to cover their livestock losses to wildlife migrants. The net monetary gain to the park from use of Masai lands is about $500,000 per year and the benefits from the park to the Masai result in an income 85 per cent greater than from cattle herding alone. (There remain, however, significant conflicts between the park and the Masai).

Schemes such as those described in this section are vital if nature tourism is to benefit, rather than hurt, local communities. They also help to discourage activities that may damage tourism by providing alternatives. An International Union for the Conservation of Nature (IUCN) report discusses a number of such schemes that have been successful at both benefiting local communities and protecting natural resources.

## 2.6 EXAMPLES OF ECONOMIC ANALYSES OF NATURE TOURISM

### 2.6.1 Khao Yai National Park, Thailand

Khao Yai, Thailand's first national park, is located about 160 kilometres north-east of Bangkok. Covering 2,168 square kilometres, Khao Yai has been one of Thailand's most popular parks since its establishment in 1962 and is one of ten ASEAN Heritage Parks and Reserves (NPD 1986).

Khao Yai provides a number of benefits both to the surrounding region and to the nation. It is a premier tourist destination in the region, with between 250,000 and 400,000 visits per year. Since it contains most of the remaining forest in the area, it is of critical importance for wildlife and also profoundly affects the hydrology of the region. Four river basins have their headwaters in Khao Yai, and two major reservoirs are dependent on water from the park.

In addition to being the oldest national park in Thailand, Khao Yai is also one of the most popular and well-developed parks for recreation. Located approximately three hours away from Bangkok by car, Khao Yai attracts large numbers of both Thais and foreigners. Visits to Khao Yai more than tripled between 1977 and 1987.

A recent survey of Khao Yai visitors designed by the authors and members of the World Wife Fund for nature Beneficial Use Project, and undertaken by the latter group between March and May 1988, found that the site was visited mainly as a nature tourism destination. For foreigners visiting the park, more than 62 per cent stated that wildlife viewing was one of their three main reasons for coming to Khao Yai. This was followed by scenery (58 per cent), relaxation (43 per cent), and hiking (41 percent).

Thai nationals overwhelmingly said that enjoying the scenery (54 per cent) was their main reason for visiting Khao Yai. (Note that relaxation was not given as a separate choice in the Thai-language version of the survey, so percentages are not directly comparable). Adding the percentage of people indicating any specific activity as one of their top three reasons for coming to Khao Yai showed that viewing the scenery was still number one (86 per cent), followed by seeing the waterfalls (58 per cent), wildlife viewing (36 per cent), picnicking (29 per cent), and overnight camping (25 per cent).

**Financial benefits.** Both the National Parks Division (NPD) and Tourist Authority of Thailand (TAT) operate lodging facilities in Khao Yai. Revenues from NPD operated accommodations were almost 1.5 million baht in 1987 (approximately 26 baht equals US $1). Dobias et al. (1988) report that the TAT income from lodging in 1987 was almost 5 million baht, while TAT-run restaurants received 4.2 million baht in income. TAT also received 400,000 baht from golf course fees, 318,000 baht from their souvenir shop, and 230,000 baht from night time excursions to view wildlife with spotlights. Thus, TAT's gross income was more than 10 million baht in 1987, while its expenditures during that year were approximately 3.3 million baht. While these figures do not include prior capital expenditures to build facilities, it is nonetheless clear that TAT's operations are profitable. Unfortunately, all profits from TAT's operations go to TAT and not to the NPD and, therefore they do not contribute to the management and preservation of the park.

Gate fees from admission to the park in recent years ranged from 1.6 to 2.4 million baht per year. Adding the gate fees and NPD-operated accommodation charges, tourism directly contributed approximately 3 million baht in 1987. In addition, NPD received 150,000 baht in concession fees from the four restaurants/food stalls within park boundaries.

The Beneficial Use Project has generated some interesting data on the expenditures of both Thai and foreign visitors to the park. In general, foreign visitors spend more per person than do Thai visitors. Based on data from organized bus tours, average daily per person expenses for foreign visitors range from 500 to 800 baht, of which the formal admission fee is less than 1 per cent. Clearly, gate receipts are only a very small fraction of people's willingness-to-pay to visit Khao Yai.

With more than 400,000 visitors per year, the total expenditures generated by Khao Yai tourism are large - from 40 million to 200 million baht. These expenditures, of course, are not an economic measure of the value of the park. To determine the true economic (i.e., social welfare) gain from visiting Khao Yai, we would need to measure consumer's surplus, that is, the maximum willingness-to-pay over and above the actual cash costs of visiting Khao Yai. This amount could be estimated by carrying out a travel-cost study, and approach widely used to value the non-priced benefits enjoyed by visitors to parks and other recreational areas. By carefully controlling for origin, visitor background, and other variables, the pattern of recreational use of a park provides the data from which a demand curve and, in turn, consumer's surplus can be estimated.

In sum, the financial contribution of tourism is already substantial and can be expected to increase in the future. Bangkok is near Khao Yai and as incomes rise and fewer alternative open areas remain, Khao Yai will become increasingly valuable. Foreign tourism could also increase with improved facilities and promotion. Furthermore, virtually all Khao Yai tourism activities are restricted to a very small part of the park accessible from the one north-south road. More than 90 per cent of the park is completely undeveloped and inaccessible, other than on foot. Future expansion of facilities is likely. VALUE: Tourism-related expenditures are 100 to 200 million baht (roughly US $4 to $8 million) per year, and estimates of consumer's surplus (an economic measure) are from 10 to 25 million baht per year.

**Biodiversity/ecological benefits.** Maintaining Khao Yai as a national park for nature tourism and other uses provides benefits by protecting biological diversity and maintaining ecological processes. Khao Yai's rich diversity of plants and animals makes it an important reserve for many species. Although most famous for its elephants, numerous other species contribute to its biological diversity. In addition to the pure "existence value" of species diversity, it also provides a powerful pull for tourists. We are not able, however, to place a monetary value on many aspects of the current and future values of the benefits of maintaining biodiversity. VALUE: Undetermined. Expenditures on research and education related to species in Khao Yai total 1 to 2 million baht per year. Option/existence value based on Khao Yai's role as an elephant sanctuary is estimated at more than 120 million baht per year (see Dixon and Sherman 1990 for more information on how this figure was determined).

**Watershed protection.** Khao Yai provides important watershed benefits in terms of the quantity, quality, and timing of water flows. The reservoirs located downstream depend on Khao Yai's watershed protection function. Maintaining Khao Yai in its current state for nature tourism and other uses will preserve these benefits as well. VALUE: Can be calculated but undetermined at present.

**Management costs.** The present annual management budget for Khao Yai is about 3.4 million baht. Implementation of the Khao Yai Management Plant (NPD 1986) to meet protection, interpretation, and development goals will result in increased annual budgets and large capital expenditures in the next few years. With its large area and closely settled borders, greater effort is needed to support programs that help improve the standard of living of nearby residents, thereby reducing their dependence on illegal and unsustainable uses of the park. COST: Current government management costs are 3 to 4 million baht per year but will rise significantly over the next few years.

**Opportunity costs.** A variety of development benefits are lost because of protection. Foremost are water resource development, timber harvesting, and agriculture. The potential economic benefits from agriculture appear to be relatively small and high extraction costs for timber limit its profitability. Precise estimates of these opportunity costs require more data. Impacts on tourism, biodiversity, and ecological processes, if these activities were allowed, may be large.

Another major category of opportunity costs is the loss of income to local villagers due to prohibitions on the gathering and harvesting of plants and animals in the park. Note that the two categories are not cumulative since development of park resources would also result in a loss of opportunity to collect plants and animals. VALUE: A rough "guesstimate" of the reduction in villager-derived income from park resources is 27 million baht per year, though this amount would probably not be sustainable and would result in significant damage to highly valued species (Dixon and Sherman, 1990)

Overall, Khao Yai is a good example of a protected area that fits the socially beneficial category. It provides recreational, wildlife habitat, and watershed benefits that are quantifiable in physical, and in some cases economic, terms. It also provides less tangible benefits in terms of preservation of forest cover and associated biological diversity. Without government intervention, however, such a large area could not exist. The benefits are too diffuse and the financial returns from preservation would be outweighed by the direct benefits from exploitation of Khao Yai's timber, land and animal resources.

**Management issues.** Many areas just inside the park are heavily degraded. These areas should be made into buffer zones and managed to provide benefits to nearby villagers. Programs should be developed in these areas to promote production of plants that are currently being poached within the park or to establish other opportunities to supplement villager incomes. These programs could be paid for, at least partially, with a percentage of tourism revenues. Once established, penalties for poaching beyond the buffer zone should be strictly enforced. However, limited hunting of certain species could be allowed in the buffer zones. Such a policy would have to be accompanied by a clear demarcation of park boundaries.

Certain tourist activities could also have secondary benefits. Development of organized multiday hikes could provide employment opportunities for local villagers as guides and support staff. One program of this type has already begun at Ban Sap Tai village under the auspices of a WWF project (Dobias et al. 1988). These hikes could also be accompanied by guards who would help patrol forest areas currently not guarded effectively.

In 1987, fees from concessions, accommodations, and entrance were almost equal to the budget allocated to Khao Yai (3.18 million baht versus 3.38 million baht respectively). If the NPD were allowed to take over facilities currently run by the Tourist Authority of Thailand (TAT), it is likely that Khao Yai could more than pay for itself with direct revenues from tourism.

The NPD should also consider establishing a two-tier fee system. Current entrance fees, though reasonable for Thais, are extremely low by foreign standards. Fees probably could be raised to ten times their current levels without significantly reducing the number of foreign visitors.

### 2.6.2 Logging and Nature Tourism in the Philippines

Sometimes the threats to nature tourism are direct and readily measured - poaching of elephants for their ivory or encroachment of agricultural fields into protected areas are two examples. In other cases, a nature tourism industry may be harmed by actions that take place at some distance but have effects that are transmitted through the environment. These types of impacts can be just as costly.

One example of the latter, ecosystem-linked impact is occurring in the Bacuit Bay area of northern Palawan, an island in the Philippines. Long famous for its beautiful scenery, abundant fish life, and clear water, Bacuit Bay and the small town of El Nido were largely

undeveloped until recently. Access was difficult and costly.

Prior to 1979, there was little organised tourism in the Bacuit Bay/El Nido region. Then a Philippine-Japanese joint venture set up a scuba diving resort on a small island at the mouth of the bay. This first resort catered to groups of Japanese divers with smaller numbers of other local and foreign divers. The success of this operation led to the establishment of a second resort, located on an adjacent island, in 1984. Both resorts are selling a fragile, rare commodity: clear, warm waters, abundant coral and fish life, and spectacular scenery. The resorts charge top dollar for their services and have a major interest in preserving the bay's ecosystem.

Local artisan fisherman share these concerns. The two groups have been working together to protect the bay's coral and fish population to their mutual benefit. Both groups also had an incentive to cooperate with the marine police to enforce the trawler ban in nearshore areas.

A third industry in the area, however, has caused a major resource use conflict. Most of the Bacuit Bay watershed is forested and a portion falls within a concession granted to a major logging firm. In this steep watershed with highly erosive soils, logging results in substantial erosion of that soil, which is deposited directly into the bay. The sediment kills the coral, thereby reducing biomass production and the dependent fishery, and clouds the water. Both the diving and fishing industry suffer as a result and incur major economic losses. A detailed ecological-economic analysis of the three industries examined the financial implications of continued logging versus a logging ban. Continued logging would result in a loss of a substantial part of the bay's coral ecosystem and the eventual closing of the dive resorts. Fish catch would also be reduced. A logging ban would avoid these costs but would deprive the firm of income from the timber located in the bay's watershed.

Gross revenues were examined for all three industries. A logging ban was estimated to produce over $75 million in revenues over a ten-year period from a thriving tourism ($47 million) and fishing ($28 million) industry. Logging revenue would be zero. Continued logging, on the other hand, would generate $13 million in logging revenues over the same ten years but would result in major decreases in tourism revenue (to $8 million) and fishery income (to $13 million). The "cost" of continued logging, therefore, was about $40 million in gross revenue over the ten-year period.

The biggest loser would be the nature tourism diving business - not only would planned expansion plans be dropped, but the major resorts would close. Divers would not want to incur the expense and inconvenience of coming to El Nido if the diving were not superb - other sites are available in the Philippines and elsewhere.

The Bacuit Bay/El Nido case is still evolving. Because of political factors, attempts to ban logging have failed to date, but a marine park is being set up in the bay. Whether or not less erosive logging practices can be introduced remains to be seen. In the meantime, the bay ecosystem suffers and tourism is hurt.

## 2.7 THE BOTTOM LINE

Earlier, we posed a series of questions about the economics of nature to and posed a leading question about "determining if it pays". It is thought that nature tourism has the potential to be privately profitable as well as socially beneficial. It has the desirable attribute of allowing both conservation and economic development objectives to be met simultaneously.

Nature tourism is not, however, the solution to all conservation problems. Some protected areas cannot sustain any direct use; others may yield larger social benefits when developed for other forms of tourism. Nature tourism will usually be suitable when areas fall between these two extremes. This includes fragile ecosystems that can accommodate limited numbers of visitors but cannot sustain high use levels; for examples, certain coral reef ecosystems or moist tropical forests may not be able to sustain intensive visitor use. This so-called low-impact tourism has important conservation and protection benefits as compared with resort tourism, which is usually more intrusive, even if it yields greater financial benefits.

In other cases, demand, not carrying capacity, may be the limiting factor. Nature tourism is suitable for places that are very remote or difficult to reach, a characteristic that often translates into fairly high costs per visit. Safaris in Africa, or cruises to Alaska or the Galapagos, come to mind.

Nature tourism may also be a desirable alternative when investment funds are limited. Nature tourism frequently uses simpler facilities and has less expensive and less intrusive infrastructure. Thus, it may be practical in cases where funds for largescale development are not available.

Economic analysis of alternatives is helpful in identifying likely benefits and costs, both private and social, of development options. In some cases, large-scale resort tourism will be the preferred option, in others, nature tourism, with its associated environmental and conservation benefits, may be better. In still other cases, more traditional development or strict protection with no

tourism will be the optimal choice. An economic analysis as outlined above can help clarify the issues, and help determine the potential effects of different alternatives.

## 3 THE USE OF ECONOMIC INSTRUMENTS IN SUSTAINABLE TOURISM

As disposal income grows throughout the world, the demand to visit tourist sites will only increase. Since the number of such sites is fixed, the pressure will be felt even more acutely than it is today.

In addressing these problems we need a multiplicity of instruments. Certainly there is an important role for punitive sanctions against offenders to laws protecting natural sites and cultural monuments. Moreover there is a literature on how such sanctions should be designed and operated[2]. We do not, however, deal with such regulations here. Instead we look at the use of market based instruments, such as charges and permits, to achieve a socially optimal solution. Essentially the problem is one of externalities; tourists visiting a site impose negative externalities in terms of damage to the natural and constructed fabric of the place as well as imposing congestion type costs on each other (at least after some critical number has been exceeded). This much is well understood and the basic principles of the use of market-based instruments in the regulation of such environmental externalities has been studies extensively (see, for example Tietenberg, 1997 for an excellent review). The main problems that remain are to design the use of such instruments in a practical context, taking account of the market structures that exist and setting the values for the instruments on the basis of adequate knowledge about the external costs. This is where the effort has to be focussed.

This part of the paper outlines what kind of instruments can be used, how the market structure is a relevant factor in determining the levels at which the controls are set, and the key parameters that determine the levels of the market based instruments. As will become clear, there are many gaps in our knowledge that need to be filled if we are to be successful in controlling tourism in a way that puts this important industry onto a sustainable development path.

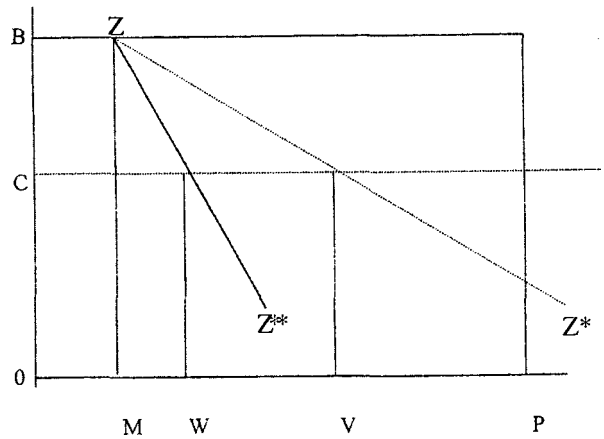
### 3.1 External Costs in Tourism and the Structure of the Tourist Industry

The external costs of tourism have been discussed above and a few estimates exist of impacts of tourism to coral and other materials, but there are virtually no estimates of the monetary values associated with these physical impacts. For analytical purposes, what is needed is not just the average damage per visitor, but the **marginal** damages per visitor and how these marginal damages change with the number of visitors.

Below, some plausible values are provided, but this is clearly an area where further empirical work is needed.

The congestion caused has, likewise, not been assessed in any serious empirical way. The demand functions for tourism have been estimated (e.g. Crouch and Shaw, 1992), but such demand functions do not look at how the willingness to pay (WTP) for a visit is a function of the number of visitors. In terms of Figure 1, the WTP for a group of **identical** visitors OP, assuming that some critical number is not exceeded is given OB.

**Figure 1: Congestion Effects of Tourism**




The marginal cost per visit is OC. Each visitor will compare that marginal cost with the WTP as given by the line ZZ*. This results in a number of visitors equal to OV. However, the marginal visitor creates congestion effects on all other visitors, resulting in an **additional or marginal value** as depicted by the line ZZ**, which is below ZZ*. The socially optimal number of visitors is OW, but the free access equilibrium will result in a number equal to OV. The potential pool of visitors is OP.

Remarkably the literature on tourism does not contain serious estimates of this congestion effect. To be sure, there are estimates of the price demand elasticity of visits to sites using the travel cost method, but these estimates do not separate out the decline in the WTP due to the fact that people with a lower WTP are visiting the site (a factor we have eliminated in Figure 1), and the fact that the WTP of any one visitor declines with the number of visitors. If we are to develop tools for sustainable tourism it is precisely these kinds of data and analysis that are needed.

What kind of market structure is most appropriate to the tourism industry? This will vary according to the

[2] For a review of the literature on the use of sanctions and their impact on behaviour see, Cullis and Jones, 1997.

problem being analysed. One model is that of 'open access' as presented in Figure 1. Individuals or small tour operators deliver visitors to a site, which is not controlled in any other respect with respect to numbers. This will apply to locations such as 'free' beaches, pilgrimage sites, and some public monuments. Even with a site fee, to cover the local costs of maintaining the site, the problem of congestion is, of course, not addressed. The latter needs **specific controls** or market based instruments to address the congestion issue.

A second kind of model is one where a single operator, or a small number of competing operators, provide access to a site. Typically, in such cases there are significant 'set up' costs and the industry is characterised by unit costs declining with the number of visitors, or increasing returns to scale. Such operator(s) will not take account of either the congestion dimension or of other external costs. However, by reducing output below the competitive level, they will make some move towards the social equilibrium. If there is only one operator, the industry will be a monopoly and the producer will maximise profits by equating marginal revenue to marginal cost. If the industry is open in the sense that anyone can provide the service, then it will be characterised by monopolistic competition, a structure where each firm faces a downward sloping demand curve and maximises profits in the same way as a monopolist, but in the long run ends up making zero profits.

With monopolistic competition, some degree of product differentiation is expected. One way this could manifest itself is through 'packages' in which different complementary products are offered, such as different grades of hotels, meals, guides. Another is through competing sites, such as ski resorts or beach resorts, varying in the degree of congestion. In the latter case, as will become clear later, some other aspect of quality variation is also required to fully determine the equilibrium.

In the case of controlled access equilibria and a monopolistic industry, numbers will be below those that would exist with a competitive equilibrium, although the latter does not always make sense if the industry is one of increasing returns to scale. In any event, the monopolistic equilibria will deviate from the social optimum. It is not possible to say, in general, whether the industry will result in too few visitors or too many. In the next section we look at some plausible values for the parameters and compare the equilibria.

### 3.1.1 Some Numerical Simulations

***Congestion and Open Access***

In this section we look at some simple models for visits to a site where congestion and external costs are present, and see how the market and social equilibria compare The first case is that of congestion and open access. Assume that the demand for visits is form a uniform population, with each person willing to pay ECU100 for a site visit if the number of visitors does not exceed 500 on the day of the visit. Beyond that level, the WTP declines and the function is given by:

$$WTP = A - BN \qquad N > 1000 \qquad (1)$$

$$A > O, B > 0$$

The key question is how does the value of B relate to the cost of a visit? As we noted there are no real estimates of this congestion parameter. Consequently we have to make some assumptions. In Table 3 below we have taken 'elasticities' of congestion ranging from -0.1 to -5. The potential number of visitors is taken as 10,000, all of whom have a WTP of ECU100. When the number exceeds 500, a congestion effect comes into force. The elasticity of the congestion effects tells us by how many percent the WTP declines when there is a One percent increase in the number of visitors. Since the demand function is taken to be linear, the elasticity will vary with the number of visitors. The elasticities in Table 3 are calculated for WTP = 100 and N = 500[3].

The figures in Table 3 show that the more inelastic the WTP with respect to the number of visitors, the greater the difference between the social and private optima. Furthermore, the greater is the difference between the marginal cost of a visit and the WTP of the population, the greater will be the gap between the private and socially optimum number of visitors. Finally, the greater is the number of visitors at which the congestion starts to occur, the larger will be both the free access and social equilibria.

Once external costs are brought into the picture, the divergence between the private and social optima is even greater. Formally it is like raising the marginal private cost in Table 3; the higher the cost, the smaller a percentage of the total number of potential visitors is the number of socially optimal visitors. Incorporating the external cost does not change the free access solution, and so the gap between that solution and the optimum gets larger.

What kind of instrument would one use to bring about the social optimum? Recall that the problem arises because of the open access nature of the site being visited. It may be possible for the government to regulate visitors through a permit system, but that is often impracticable.

[3] It is easy to show that the free access equilibria is given by N(FA) = (A-MC)/B and the optimal position is given by N(OPT) = A*MC/2B, where MC is the marginal cost and A and B are as given in equation (1). A* is the intercept ofthe marginal value curve going through N = 500 and WTP = 100.

**Table 3: Relationship Between Private and Social Equilibria with Congestion Costs**

| Elasticity of WTP | Marginal Private Cost as % of WTP | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | 20 | | 50 | | 75 | | 90 | |
| wrt N | Free Access | Optimal | Free Access | Optimal | Free Access | Optimal | Free Access | Optimal |
| -0.1 | 4500 | 2500.0 | 3000 | 1750 | 1750 | 1125 | 1000 | 750 |
| -0.25 | 2100 | 1300.0 | 1500 | 1000 | 1000 | 750 | 700 | 600 |
| -0.50 | 1300 | 900.0 | 1000 | 750 | 750 | 625 | 600 | 550 |
| -1.00 | 900 | 700.0 | 750 | 625 | 625 | 563 | 550 | 525 |
| -2.00 | 700 | 600.0 | 625 | 563 | 563 | 531 | 525 | 513 |
| -3.00 | 633 | 566.7 | 583 | 542 | 542 | 521 | 517 | 508 |
| 4.00 | 600 | 550.0 | 563 | 531 | 531 | 516 | 513 | 506 |
| -5 .00 | 580 | 540.0 | 550 | 525 | 525 | 513 | 510 | 505 |

Potential number of visitors with WTP = 100 is 10000

WTP declines when number of visitors exceeds 500

WTP function is assumed linear.

Elasticity is calculated for WTP =10O, N=500.

Many sites have too many points of access to be able to control entry. For such cases checks on visitors at the site to ensure they have a permit may be required. In any event, if a licensing system is introduced, it is highly desirable to have licences subject to trade or auction. By that means those with the highest WTP can visit the site, thereby ensuring the greatest benefits. If the state wants to encourage students and others with low income to visit the site, it can issue free permits to such groups. If they then choose to sell them, however, it is difficult to rationalise why such sales should be banned.

An alternative system of control is to offer a franchise for the regulation of the site to one or more operators, who then have an incentive to internalise the congestion costs. Responsibility for access is then that of the operator and no fee is charged for the privilege. The government, may, however, need to impose a charge per visitor to cover the external costs, and/or a tax on profits to recoup some of the profits created by the franchise. The following section considers equilibrium with such a system of a control.

***Equilibria With Monopoly***

Consider the following simple market structure. There is a single producer with a cost function

$$C = S + DN - EN^2 \qquad (2)$$

$S > O, D > O, E > 0$

The producer faces the demand function (1) given above. For simplicity, however, we assume that congestion sets from N = 0. Simple profit maximisation yields a number of visitors of $N_M$ of

$$N_M = (A-D)/2.(B-E) \qquad (3)$$

The second order conditions for a maximum require that

$E<B$

And an interior maximum requires that

$A>D$

Let us assume, further, that the optimal number of visitors under the monopoly is 1000. It is not difficult to show that, at the profit maximising position, the elasticity of the demand function as given in the previous section, must be numerically less than one.

In order to present the simulations we need to give numerical values to a few more variables. These are as follows:

Elasticity of WTP with respect to N at N = 1000, P given by equation (1). We assume values of the elasticity of between -0.11 to -0.67.

Fixed or Set Up Costs (S). We assume that these range from 0.5 percent to 5 percent of the turnover of the monopolist

Cost parameters (D) and (E). Given (3), A, B and $N_M$, there is only one degree of freedom to determine either D or E. Recall also that E < B. Hence the range of values for E are limited. We assume values of 0.4B and 0.8B.

External costs per visitor. We look for the values of the external costs per visitor that will represent the maximum value of the external cost beyond which the socially optimal number of visitors is zero.

Table 4 below gives the solutions for the different values of the parameters. The model has been calibrated so that the optimal output for a monopolist is 1000 visitors in each case. **The corresponding socially optimal output is the same as the monopolist's output.** The reason is that the monopolist is taking account of the marginal value from each visit, because that is also the marginal revenue to him. Hence the two equilibria are identical. This simple property will not hold once we bring in visitors with different WTP for a visit. Then the monopolist will exploit the differences in WTP and produce too low a number of visitors.

The table provides some interesting insights. First the fact that the monopolist acts as a social maximiser is unusual. It arises, as noted above, because of the fact that the marginal revenue and the marginal value of an extra visitor are identical. Second, the range of external costs that can be accommodated within the model is rather small. For example, in the first scenario the maximum value of the external cost is only 12 percent of the WTP. It can rise to as much as 48 percent but it does depend on the precise values of the parameters B and E. Although linearity accentuates these switches, it is clear that the parameter values for the slopes of the marginal cost curve and the demand curve are critical. The policy implication is that the costs and the taxes have to be estimated with great care if the market based instrument is to get correct solution. There is little room for error in setting the tax. With other parameter values the scope for manoeuvre is a little greater but not that much. It should be noted that the upper limit of the external costs varies from 11 to 110 percent of the unit costs for a monopolist delivering 1000 visitors.

### 3.1.2 Further Developments

The model presented so far has not looked at competition between sites and differences in the WTP between visitors. If there are several sites competing for the visitors, and not all are controlled by the same authority, there will be a disincentive to internalise external costs, on the grounds that you will lose visitors to the uncontrolled sites. It would be interesting to have some empirical estimates of how much substitution there is between sites (presumably as a function of distance) depending on the presence of taxes or other instruments. Such estimates are sadly lacking.

One way round the problem would be to have an agreement among countries within a region to impose charges based on external costs and not to compete on this issue. Within the EU, for example, such an agreement should be possible. Details of how the external costs are assessed will have to be worked out, but this is not an impossible task. Further international co-operation is more diffficult but perhaps less necessary, given that the greatest competition is between sites that are close together.

Another way in which the model is complicated is by competing sites offering different services. It is possible, and casual empiricism supports this, for some

**Table 4: Output for a Monopolist and Socially Optimal Output**

| Elasticity of WTP WRT Price | Value of E | Monopolist Output | Social Optimal Output with Zero External Cost | Value of Ext. Cost at Which Optimal N is Zero |
|---|---|---|---|---|
| -0.11 | 0.004 | 1000 | 1000 | 12 |
| -0.11 | 0.008 | 1000(*) | 1000(*) | 4 |
| -0.25 | 0.008 | 1000 | 1000 | 24 |
| -0.25 | 0.016 | 1000 | 1000 | 8 |
| -0.43 | 0.012 | 1000 | 1000 | 36 |
| -0.43 | 0.024 | 1000 | 1000 | 12 |
| -0.67 | 0.016 | 1000 | 1000 | 48 |
| -0.67 | 0.032 | 1000 | 1000 | 16 |

**Notes_ The Value of A (the max WTP for a visit) is ECU 100 throughout.**
**The value of B is given by $\eta=1/\{1/10B)-1\}$ where $\eta$ is the price elasticity of WTP with respect to N.**
**(*) The output is 1000 only if the set up costs are less than 2.7 percent of total revenue. Otherwise it is zero.**

sites to offer a 'low cost/high congestion' package and others a 'high cost/low congestion package'. The above model of monopolistic competition is the most suitable for analysing such combinations. It is easy to bring in different WTPs for visits as a function of, for example, income and for sites to compete in terms of congestion levels. That alone, however, is not enough to characterise an equilibrium. Sites have to offer some other differentiating factors if the profit maximising and zero profits conditions are to be simultaneously satisfied. These could include different levels of hotel accommodation, restaurants etc. Work on such models is being undertaken by the present author. It suggests that, separating equilibria with different levels of congestion will emerge. Hence the congestion dimension can largely be handled through franchises, but the external costs will be different in different sites and differential charges will be needed, with higher charges in the more congested sites, which goes against any equity objective. Furthermore, if a site is to have many operators, a higher level decision will be needed to determine the overall level of congestion.

## 4 CONCLUSIONS

This paper has looked at the question of sustainable tourism in terms of its congestion and external cost impacts. With unregulated access to tourist sites, both of these result in levels of development that are unsustainable and socially sub- optimal. In order to determine the optimal numbers of visitors, however, we need information on the sources of sub-optimality that is quantitative. Much of this information is lacking.

The congestion issue can be addressed through a monopoly being provided control in the operation of a site. This will result in restrictions that are optimal with respect to the congestion dimension but will also generate high profits. Some way of clawing back these profits will be needed - through special taxes or through the auctioning of franchises.

In some cases an industrial structure based on monopoly is not feasible and monopolistic competition is more appropriate. This structure has been analysed and found to generate equlibria that do not deviate too much from the monopolists in terms of total numbers of visitors, but with a lower prices and a zero excess profit level. Hence such a structure may be acceptable as a second best solution, but this should be investigated further.

Internalising the external costs require data on the levels of these costs. Preliminary results suggest that this information has to be rather precise as the solutions are sensitive to the chosen values of the charges.

Competition between sites has not been analysed in this paper, but is being studied. Preliminary results suggest that sites will offer different levels of congestion, as well as other quality differences, but that dealing with the external cost dimension will be more difficult. Some agreement to impose such costs will be required if the competitive model is to operate between sites under different jurisdictions. Furthermore the full implications of different external costs and therefore different charges need to be studies in greater detail. Indeed, the robustness of the results presented here to models that include: multiple visits, multiple stages in the delivery of tourist services, each with a different industrial structure, and visits with multiple objectives is essential. There is much work to be done before we can achieve a tourism industry that is fully consistent with the goals of sustainability. But it is clear that economic instruments have a role in that strategy, along with other methods of regulation.

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# Chapter 10

# SUSTAINABLE AGRICULTURE

*by*

*Anil Markandya*

## 1 INTRODUCTION

### 1.1 What is sustainable agriculture?

There are literally thousands of papers written about sustainable agriculture, and what constitutes sustainable agricultural policies and practices. What is notably missing from most of them, however, is a clear definition of the term sustainable agriculture. This is not because of a lack of awareness of the importance of defining terms on the part of the authors (who are extremely knowledgeable about the subject), but because the definition is elusive and hard to pin down. Let us consider an agricultural community that uses land in a variety of ways, producing crops, rearing livestock and drawing on forest and aquatic resources. All these activities are carried out to produce goods for personal consumption as well as sale in markets. Some individuals from that community may work in urban areas and not be directly dependent on the land, but provide income and resources that influence the way the land is used.

For such a community what activities would sustainable agriculture consist of? One could ask whether the present land use practices could be continued *ad infinitum*. Of course these practices are not static even now; they are continuously changing and evolving and the question of what constitutes "present practices" needs to be defined in a dynamic context. One interpretation could be to ask whether the way in which land use and agricultural production is changing will continue to support the agricultural community for the foreseeable future. Will agricultural output and land availability be sufficient to maintain (and perhaps improve) the standard of living of the population that is likely to draw on those resources for its livelihood? This requires some assessment of what technologies will be available in the future, how the productivity of the land will respond to these technologies, and how the population of that community will grow and divide itself between urban and agricultural activities. The questions of technology raise issues about available capital inputs (tractors, mechanical devices etc.), as well as the developments in, and availability of, seeds and new crop and animal varieties. The questions of population cannot be answered without some assessment of the labour market options in urban and rural markets and some knowledge of the factors that determine the fertility of the groups concerned.

Thus is becomes clear quite quickly that the concept of sustainability is not a simple one. It is not sensible to speak of "sustainability" of a community in a static sense as all social systems are in a continuous state of flux. One can ask whether that process of change is likely to be along a path that will provide the current inhabitants of that community, and their descendants, with living standards, or more generally welfare, at a level that is **non-decreasing over time.** Some of the more theoretical discussions of sustainable development take this as a working definition. Although it is extremely diffcult to operationalise, we can make it the basis of identifying some activities and changes that look as if they will cause problems to future generations. For example, if forest land is being cleared to accommodate the needs for agricultural expansion to provide for an increasing population, we may ascertain that the loss of forest services is resulting in a decline in the living standards of the present population, let alone that of future generations. Thus present agricultural practices are non-sustainable. Moreover, the problem is not going to be for future generations way down the line; the **present** practices are going to cause difficulties for those who will have to live off that land in the next 20 or so years.

The Food and Agricultural Organization of the United Nations (FAO) has been grappling with this set of issues for some time. In 1988 its council formulated a definition of sustainable development especially as it related to agriculture as follows:

> "The management and conservation of the natural resource base, and the orientation of technological and institutional change in such a manner as to ensure the attainment and continued satisfaction of human needs for present and future generations. Such sustainable development (in the agricultural, forestry and fisheries sectors) conserves land, water, plant and animal genetic resources, is

environmentally non-degrading, technically appropriate, economically viable and socially acceptable".

Few would disagree with the above definition. The key issue is what kind of policies are needed to achieve the goals described in that definition. Markandya (1991) provides a framework for the formulation of such policies. In the paper referred to, he argues that the pursuit of sustainable development can be translated into policies that (a) encourage **efficiency** in the use of resources, so that the maximum output is obtained from a given input of man made and natural resources; (b) promote **equity,** at least to the extent of eliminating the extreme poverty that results in, and is abetted by, high rates of resource degradation, and (c) promote policies that encourage resilience and allow agro-ecological systems to adapt to changing external circumstances, such as droughts, floods etc. For all these objectives the supporting policies fall in three broad areas. The first is to **value** resources correctly, basing these values on the services that they provide. This valuation allows one to make the right decisions regarding the pricing of these resources as well as the allocation of investments that make use of, or conserve, such resources. The second is to set the **right legal and social framework,** so that decisions based on the full social costs of different actions can be implemented. The third is to **monitor** changes in sectoral and national activities and to develop indicators of sustainability based on those changes, so that policy makers can be properly informed when making their decisions.

In looking at agricultural sustainability within the framework of this definition, we can work at the 'macro' level and at the 'micro' level. The macro level looks at the trends in agricultural output and consumption and makes predictions of what will happen in the future. If declines in output are predicted, or if some regions are likely to face declines in output per caput, then the issue of sustainability arises. Section 1.2 below reports on the main trends in world production and consumption of food and changes in major regions of the world.

Even if the macroeconomic picture is one that is reasonably optimistic, the situation in vulnerable communities may not be so positive. Loss of soil nutrients, increased population and poverty may become worse, making the practice of agriculture more environmentally damaging, the local ecosystem degraded and the development path unsustainable. In Section 1.3 we look at communities where such problems are present and where, on an assumption of continued trends, they are likely to get worse.

In both these cases, there is nothing fatalistic or predetermined about the forecasts. The situation **can** be improved and a path that is unsustainable can be converted to one that is sustainable. Section 2 of this paper looks at such policies, dividing them into pricing policies, investment policies, macroeconomic policies and policies for trade in agricultural products Section 3 concludes the paper.

## 1.2 Assessing sustainability in global terms

The recent excellent review of world agriculture carried out by the FAO (Alexandratos, 1995), has looked at trends in world production and consumption of food to 2010. It is interesting that looking 15 years ahead is about as much as anyone can manage with any certainty. The FAO report recognises that we can only make some vague statement of what the situation will be like in 2040. There is a message in this, even if we worry about sustainability as a long-term phenomenon, there is little we can say about the situation beyond 20 years. If we keep on a sustainable path over that period, future sustainability will have to be addressed on a rolling, ongoing basis.

What can we say about the 'near future'? The most important actor is the momentum of population growth. From a figure of 5.3 billion in 1990, the world's population will grow to around 7 billion. Moreover there will be quite disparate growth in different regions. Sub-Saharan Africa has the highest growth (2.9 percent). East Asia has continued but lower growth (1.2 percent), and Europe has a slightly declining population. Overall the growth in population is estimated at 1.4 percent per annum. At the same time, agricultural output is estimated to increase at 1.8 percent per annum, which is much slower than in the last few decades. Thus in aggregate terms per caput food output will increase very slightly, and at a much lower rate than in the last decades of the 20th century. By itself, however, this would not indicate that the situation is an unsustainable one. There is no need for *per capcut* food production to continue to grow for a path to be sustainable. As people get richer their total consumption of food in physical terms increases very little, although the value goes up because of increased processing, marketing and distribution.

The problem in sustainability terms lies in the regional variations in output that are predicted. Although the developing countries as a group are expected to increase food *per caput* from nearly 2500 calories today to just over 2700 calories in 2010, the region of sub-Saharan Africa will remain at present levels, or even possibly below present levels. This would not matter so much if the region was increasing the production of other goods, which it could export so as to import food from

agriculture surplus regions. Unfortunately this is not predicted to be the case either and so, for this region, agricultural development cannot be regarded as sustainable.

The other major features of the projections to the year 2010 are:

i. The developed regions of the world will become bigger exporters of cereals (Europe moving from being a net importer to being a net exporter), and developing countries will become bigger importers. As a group they will switch from being net exporters to being net importers.

ii. Livestock output will grow substantially in developing countries and slightly in developed countries. Part of the increased imports of cereals into the developing world will be as feed for the livestock.

iii. The amount of exports of agricultural products that developing countries will have to make to buy a given amount of the imports from the developed countries will tend to increase, at least for those countries that rely on exports of products such as cocoa, coffee, cotton, tea, tobacco etc.

iv. The increase in production in the developing world will come about partly as a result of increased yields per hectare, and partly as a result of more land coming under cultivation. The increased yields will require maintenance of existing systems of irrigation, and the expansion of the system of irrigation by 23mn. ha or 19 percent of the present level of 760 mn. ha. Most of this will be in South Asia. It will also require increased levels of nutrient application (fertiliser) and increased applications of pesticides. The expansion of land areas under cultivation are estimated at 90mn. ha. or about 12% of the current level. Most of the expansion will be in sub-Saharan Africa and Latin America/Caribbean.

v. The expansion of land under cultivation will naturally be on more marginal land, and some of it will be from clearance of forest. It is diffcult to estimate how much will come from forest clearance. In the 69 countries that account for 96 percent of all tropical forest resources, even if all the extra land came from forest clearance it would amount to 4.2 mn. ha. per annum, or about 0.25 percent of total forest area. This can be compared with current rates of deforestation of around 15 mn. ha. per annum. In practice the impact of extensification of agriculture on forests is likely to be less than that.

vi. Land with agricultural potential will also be needed for non-agricultural purposes. Best estimates indicate that about 20 mn. ha. will be taken up in urban development.

vii. Fertiliser use rates in developing countries are currently about 62kg/ha, about half those of developed countries. These averages, however, mask very wide differences within countries, particularly in the developing world. The increase has positive and negative impacts. On the one hand it increases production and prevents nutrient mining of the soil. On the other hand it increases problems of agricultural run-off and associated damages to water sources.

viii. Although traditional methods of plant protection remain important in developing countries, the use of pesticides has increased and will continue to grow. The rate of growth is, however, difficult to predict and will certainly depend on the policies adopted to encourage or discourage the use of such chemicals. Use in developed countries may even decline, if the right policies are put in place.

What do these predictions say about the sustainability of agriculture? At one level they are encouraging in that they point to a picture that, while not always positive, is one that suggests the problems to be manageable. Up to the year 2010 we can say that agriculture will develop, with output increasing and, as far as one can judge, this will be achieved without causing serious damage to the eco-systems on which these activities are dependent.

The major problems in sustainable agriculture will be localised. The FAO report notes the difficulties that will arise in freshwater supplies, degradation of soils, water contamination and climate change. These are discussed below as part of the assessment of sustainability in local terms.

## 1.3 Assessing sustainability in local terms

The pattern of development in world agriculture contains a number of regional variations that could be unsustainable, if the right measures are not undertaken. Some of the major ones are discussed below.

***Freshwater supplies.***

The constraint on agriculture in many places will

not be land, but water. As economic development takes place the demand for water for non agricultural uses increases substantially. If the goals of increased yields and agricultural output are to be achieved, increases in water efficiency will be essential. This implies better pricing of water, something that is discussed later in this paper (Section 2.2). Water misallocation arises in many areas, not only drylands but also regions where irrigation systems have been put in place.

***Degradation of soils***

This is estimated to affect some 1.2 bn. ha. of land worldwide. Among the causes, deforestation and overgrazing are estimated to be responsible for about one-third each, with the rest due to the mismanagement of arable land. Each of these can, however, be addressed, but particularly the last two. Overgrazing needs policies on land use, macroeconomic management and trade (Sections 2.2 to 2.5). Mismanagement occurs when soils are mined for nutrients and when irrigation systems are improperly used and applied, resulting in salinization. Nutrient mining is particularly a problem in sub-Saharan Africa, where fertilizer applications are too low. Salinization is a problem in Asia, where much of the development in irrigation has taken place. Corrective measures include measures to increase nutrient application, better pricing of water, better management of irrigation systems and more extension services.

***Desertification***

Desertification is broadly land degradation in dryland areas. Although such systems are more resilient than was thought, pressures brought about by expanding agriculture into such areas can result in irreversible damage The actions needed are those that impinge on land use, and can be economic and as well as of a 'command and control' type.

***Water contamination***

This arises from applications of fertilizers and pesticides as well as from effluents from livestock and fish farms. As pesticide application demand increases, it is important to use pricing policies to prevent overuse (see Section 2.2 for an example). Going beyond that, increases in prices to reflect social costs can reduce damages even further. The same applies to other forms of contamination, where 'good practices' can reduce contamination substantially, but the right incentives have to be in place.

***Climate change***

The eventual impacts of climate change are uncertain, but the evidence points to the regions that are presently vulnerable being adversely affected in the future (sub-Saharan Africa). Other countries that will be seriously affected will be low-lying areas where inundation is likely. Appropriate measures include mitigation and adaptation and these have to be designed and implemented well in advance of the onset of the climate change.

The above discusses the main issues in the agricultural sector, how they create problems of unsustainability and how they can be addressed. The world can also be divided into agro-ecological zones and the sustainability issues analysed from that perspective. The regions are: pastoral systems, tropical forestry systems, irrigated lands and marginal lands. These are discussed in greater detail in Markandya (1991).

***Pastoral***

The main problems in pastoral systems is land degradation from overgrazing, development of subsistence and mechanized agriculture, and from the loss of forest cover due to fuelwood use. At the social level the strategy has to be to reduce population pressure on the land through the creation of jobs outside of agriculture or migration to areas with higher potential. At the technical level, the policy has to be to develop, with the farmers, low cost measures to conserve soil and water, and increase efficiency of livestock. At the same time reduced pressure on wood resources through the development of alternative fuels, such as biogas will e also help. Finally at the economic level the incentives for overstocking have to be reduced by removing some of the subsidies that encourage expansion of livestock.

***Tropical Forestry***

The loss of forest for agriculture is a serious concern. Although we have shown that to 2010 the pressures on forests from agriculture are not likely to increase much, these pressures can and should be reduced. At the technical level alternatives to shifting cultivation and the bush fallow systems have to be developed. These will include encouraging migration to other areas and physical restrictions on land development in areas of high natural value. At the economic level, incentives for deforestation through subsidies for land clearance and opening up of forest for agriculture have to be eliminated.

***Irrigated Agriculture***

We have already identified the problems with irrigation systems: charges that are too low and maintenance levels that are inadequate. These need to be

remedied, although doing so can be politically difficult. Other measures for increased efficiency include: involving farmers in the design of the systems, improving training in water use and improving the design of drainage facilities.

*Marginal Lands*

Marginal lands are characterised not only by low quality and productivity but also be greater production risk. Changes in farming practices such as monocropping or productivity-increasing activities may render them even more vulnerable unless they are properly adapted. Such lands include parts of sub-Saharan Africa, as well as the semi-arid lands, uplands, swamplands and converted forest lands of Asia and Latin America.

Such lands are not only farmed by poor farmers and do have some potential for increased production on a sustainable basis. For highly, erodible soils, a shift out of annual cropping into livestock based system may be appropriate. In other cases returning to traditional systems with development of new technologies and crop varieties can produce positive results. As in other areas, however, the importance of reducing population pressure on the land through migration and off-farm activities remains an important policy option.

# 2 POLICIES FOR SUSTAINABLE AGRICULTURE: SOME EXAMPLES

## 2.1 Policies for sustainable agriculture

The debate on what is and what is not sustainable agriculture is ongoing and not likely to be resolved in the near future. However, the need for policies that encourage, or at the very least are consistent with, sustainable agriculture is great. Hence it is not surprising that considerable attention has been paid to identifying the appropriate policies for sustainable development in general and for sustainable agriculture in particular.

The remainder of this section discusses a number of areas where policies for the agricultural sector are resulting in non-sustainable agricultural practices. It also looks at what kind of policies could produce a more sustainable development of that sector. These policies are divided into (a) pricing policies for the agricultural sector, (b) investment decisions for that sector (c) macroeconomic policies that impinge on the agricultural sector, and (d) trade related policies.

## 2.2 Agricultural pricing and sustainability

In the previous section we stressed the importance of valuing the environmental impacts of the use of different resources. This is important because it allows us to price these resources at their full social cost. Without such pricing some resources are overused and others under used. In practice there are many examples of **underpricing** of key natural resources and of the incorrect pricing of agricultural inputs and outputs.

In many countries, both developed and developing, inputs into agriculture are subsidised. Water is a key example, although fertilizer and pesticide subsidies are also prevalent. The articles by De Shazo and Panayotou (1993, paper attached) give examples of agricultural subsidies. This case study, based on a fictitious country (but taken substantially from the Indonesian experience), deals, with the consequences of ignoring environmental and ecological factors in the economic analysis of rice production in the nation. The fictitious Asian country of Riceland had, over a number of years, heavily subsidised the use of pesticides by rice farmers in an effort to give a big boost to rice production. They hoped to go quickly from being the world's largest importers of rice, to being autonomous in the production of rice. However, a mysterious increase in the incidence of the stem borer, in addition to other pests, finally alerted the authorities to the existence of a problem.

The heavy use of pesticides over the preceding years had actually precipitated this crisis. The cause of the crisis is attributed to the following reasons: the susceptibility of genetically altered rice varieties to the stem borer, the stem borer's increasing tolerance to pesticides due to overuse and the destruction of the natural predators of the stem borer due also to pesticide overuse.

In addition to this outbreak of stem borers, the country was losing valuable foreign exchange due to the heavy subsidising of importintensive pesticides. Yet another negative aspect of the pesticide use was that, for a large number of vegetables tested, greatly exceeded the acceptable level of pesticide residue and a number of people were reported to be severely ill or even to have died due to pesticide poisoning.

The proposed solution to this problem involved the use of a system called Integrated Pest Control (IPC) by its inventors. It replaced the exclusive use of chemical pesticides with biologic, genetic and cultural pest control tactics. Specifically the IPC pest control strategy required: 1) the conservation of natural predators, 2) the development of stem borer resistant rice varieties, 3) careful planting patterns and crop rotation, and 4) the use of a narrow spectrum of pesticides in a manner that targeted specific pests. To encourage the selective and need-driven use of pesticides, prices should reflect their true costs — subsidies were removed. The general

strategy is for farmers to use less pesticides and more information.

The experiment using IPC was carried out and a benefit-cost framework was employed to assess the program. The analysis showed that the IPC program was an outstanding success. The benefits of the program outweighed the costs by a ratio of 2.02 to 1.0. The paper seems to provide a strong case for including environmental and ecological factors in economic analysis: "good economics is good ecology and vice versa".

The above is a good example of what can be achieved with proper pricing of agricultural inputs. Since this is a case of a move from unsustainable to sustainable agriculture that represents a win-win situation, one may ask why do the subsidies persist? There are many reasons for this, notably: vested political interests, the lack of knowledge on the impacts of overuse and limited knowledge about alternatives. These barriers can be overcome but it needs a concerted effort and a political will to do so. We develop this example in the case study that follows this Chapter.

The subsidies on water are more complex. There is no question that underpricing of water results in overuse by those who have the first or easiest access to it. That in turn has been responsible for some of the waterlogging and salinisation problems, as well as the shortages of water in those areas from which it is drawn, and in areas that are the ends of the water delivery systems. The low levels of charges result in under maintenance of the systems, resulting in large losses and further damage to the soils. If water were properly priced many of these problems could be eliminated. The main argument against such pricing is that the poor farmers could not afford them. Water allocation has become so politicized that it is often difficult to discuss these issues in a market context. However, some market solutions that could work. One would be to allocate the water rights to the farmers, so that they would be able to sell them and use the proceeds to buy water or other inputs as appropriate. There would then be a strong incentive to conserve the resource as it has an exchange value. Combining water pricing at its social cost with a system of water rights could be an important step towards sustainable agriculture, and indeed some regions and countries are trying just that (recent attempts have been made in California and Australia).

Often governments keep producer prices for agricultural products arfticially low, so that urban consumers can afford them. This results in an under producffon of those products, and lower income levels for farm households. Faced with low incomes and poverty such households may be forced to exploit their resource base in an unsustainable way. Furthermore it may account for levels of migraffon to the cities that create the urban environmental problems with which we are so familiar.

Apart from the subsidies discussed above there are several others that impact on agriculture in a negaffve way. These include tax deferments, such as those in Brazil that resulted in forest clearance for unsustainable cattle ranching (these were subsequently withdrawn). In OECD countries there are often price support schemes and deficiency payment schemes that stimulate farm production and hence the use of variable inputs such as chemicals, creating more stress on the environmental than would otherwise have occurred. Although in theory an increase in the output price can result in either an increase or a decrease in input use at the farm level, in practice, because commodity-specific policies alter the relative prices of crops that can be grown in rotation - and in some cases even discourage crop rotations which can lead to an increased use of chemical inputs to maintain soil productivity (Runge, 1994, paper attached). Hence it is not the increase in production that is harmful to the environmental but the way it is brought about.

In some cases policies designed to protect land from agricultural use can have damaging impacts on the environment. A case in point is the 'set aside' policy which restricts arable or pasture land from being used as such. The effect of this, however, could be to increase pressure on existing land and to cultivate it more intensively. To the extent that this requires higher levels of inputs, the policy will have damaging side effects. The lesson here is that measures taken in pursuit of one objective often have impacts on other objectives, and the design of the best policy needs a careful evaluation of all such impacts. This applies particularly to price distortions and subsidies at the macroeconomic level, introduced for completely different objectives. These are discussed below in section 2.4.

All the case considered so far are ones where the prices of resources are underpriced, **below even the direct costs of supply.** The prices of pesticides are often below the costs of production and the price of water is below the cost of delivery. However, this does not account for the environmental damage that may be done by the use of that input. A full social cost pricing of pesticides, for example, would include the marginal value of the damage done by the application through their impacts on water, soil, health of affected group etc. Of course, we need to know what that damage is if we are to practice full social cost pricing. Markandya (1991) discusses some of the methods by which such estimates may be obtained. Although full social cost pricing is rare, even in developed countries, there are some movements in that direcffon. Charges are now imposed in Austria, Norway and Sweden

on fertilisers and pesticides to capture some of these environmental or "external" effects. In Norway the pesticide taxes amounted to about 13 per cent of the purchase price. In Malaysia the government has used taxes on palm oil effluent to encourage less polluting technologies and to meet environmental quality standards in river systems where the effluent was being disposed off. The revenues collected from such taxes can be used to fmance environmental education programs, training in sustainable agricultural practices etc. Alternatively they could be used to reduce the taxes collected from other activities, in which the act of taxation creates distortions rather than removing them.

Just as environmental taxes of these kinds provide penalty incentives (or sticks) to the polluters, environmental subsidies can be used to provide carrots. In many developed countries farmers are offered incentives to adopt non-polluting agriculture. Examples are "set aside" polices where the government pays for land not to be used especially for crops that cause soil erosion (US and several EU countries have such policies). Although these incentives can work and move us toward sustainable agriculture, they are less attractive than taxes. One reason is that subsidies have to be paid out of taxes collected on other activities and the collection is costly in economic terms. A second is that there is more scope for misuse and system abuse with subsidies than there is with taxes. Finally the subsidy approach is inconsistent with the "polluter pay principle" under which the rights to a clean environment are implicitly vested with the people rathar than with the polluters.

## 2.3 Investment in sustainable agriculture

The valuation of the environmental impacts of agriculture can help us in making the right decisions regarding investments in that sector. Often these impacts are ignored, so that the clearance of wetlands or forest areas for agriculture are evaluated in terms of the beneffts of the increased agricultural output, without taking account of the loss of the services provided by the land in its present use. The techniques for valuing the environmental impacts of different natural resources are discussed in Markandya, 1991. We are interested in obtaining the **total economic value** of the resource, including its present use value and future use value. (See Chapter 3). The underlying basis for valuation is the willingness to pay for an environmental improvement, or the willingness to accept payment for a deterioration in the environment. Although there are many shortcomings for this individualistic philosophy, it is consistent with a free market economic ideology and is better than valuing resources on the basis of government preferences or the preferences of a dictator.

Methods for obtaining this are divided into 'direct' and 'indirect'. The direct methods elicit values on the basis of observed impacts of the resource on other markets (e.g. the impact of cleaner air or improved irrigation on land values). This method is called the hedonic method of valuation. A related method that is applicable to land developments is called the travel cost method and is based on the time and money expenditures incurred by individuals to obtain a particular resource. For example, people spend time to get to a park or recreational centre and spend money to enjoy the amenities. From a careful study of such expenditures we can value the resources associated with the centres.

A third method of valuation uses questionnaires which mimic the kinds of choices individuals might face if there were a market for the resources in question. This is referred to as the contingent valuation method and has been developed into a sophisticated method for eliciting a wide range of environmental values. Applications of each of these methods are provided in the paper by Markandya (1991).

Finally we can obtain values of environmental goods by looking at the physical impacts of a change in the environment, and then valuing that impact. An example would be to estimate the change in crop yields resulting from an increase in low level ozone concentrations and then value the losses. Such a method can best be used when the relationship between the physical impacts and the goods in question are well known and when the goods have market prices. For many agricultural impacts we can use this method of valuation which is referred to as the dose response method or the indirect valuation method.

One factor of importance in making such valuations is the discount rate. If the rate is too high, projects with high environmental costs or benefits in the distant future tend to be under-costed or undervalued. If the rate is too low on the other hand, capital projects such as dams, which have high up front costs get selected. The choice of the discount rate is discussed in Article 3. It is argued that a rate lower than the one typically used in developing countries should be used to value **all** social investment projects. Rates of return required for investment in these countries range from 10-15 percent in "real" terms (i.e. after allowing for inflation), whereas a rate of 3-6 percent is more appropriate for evaluating investments. This should however, be combined with as complete a valuation of the environmental impacts in monetary terms.

Although the goal should be to value environmental impacts in monetary terms, this is not always possible. In some cases the impacts are too

uncertain for a single value to capture the range of potential damages. In other cases the environmental impact could be described in terms of violating a "safe minimum standard" or some threshold value of some environmental quality variable. For both these situations sustainability could be practised by imposing a constraint on all investments, that they do not breach the safe minimum standards, or that they do not result in environmental disasters if some "worst case scenarios" prevail. The practice of carrying out environmental impact assessments as part of the investment appraisal process is designed to ensure that such cases do not occur and that the project design is modified to prevent them from happening

The appraisal of projects related to irrigation, rangeland management, wasteland development, clearance of wetlands etc., all of which have been carried out as part of the agricultural development of a country, would be better designed and less likely to cause environmental damages and unsustainable development if the evaluation process included both the valuation methodologies described above as well as the use of sustainability criteria based on threshold effects and safe minimum standards.

## 2.4 Macroeconomic policies and sustainable agriculture[1]

Sustainable agriculture requires not only the right prices and incentives at the microeconomic level but also at the macroeconomic or national level. The key variables here are taxes on incomes from natural resources including land, rates of exchange, rates of interest both for borrowing and for lending, and tariff and trade policies. The internaftional dimensions are discussed in the next section (2.5).

Domestic policies that influence the real rate of interest can affect decisions on holdings of cattle and other livestock. When savings placed in financial markets offer lower returns, farmers may increase holdings of livestock until the rates of return on the two assets are equalized. This can affect pasture and rangeland and result in rates of use in excess of the carrying capacity. Lending rates of interest can influence the level of investment in land improvement, drainage etc. If rates are raised for short term macroeconomic stabilization reasons, the effects on the long term viability of the agricultural sector should not be ignored.

Fiscal policies that require reductions in the level of government spending often result in cuts in research and extension services that are so important for a sustainable agricultural policy. The long term goals of such a policy are sacrificed to the shorter term necessity of balancing the government budgets. In studies on the links between macroeconomic policy and the environment in Mexico, the Ivory Coast and Thailand, it was persistently recorded that, in times of macroeconomic difficulty the "environmental" budgets were cut disproportionately. The FAO (1995, article attached) provides details of macroeconomic policies and the ways they impact on sustainable agriculture.

How should one deal with these linkages between macroeconomic variables and sustainable development? Again, taking the discussion from FAO (1995) a number of lessons emerge from the recent experience of both developed and developing countries. First, good macroeconomic management is good for the environment. It is when macroeconomic imbalances occur and budgets are cut, interest rates raised etc, that long term goals such as sustainable development suffer disproporffonately. Second, when cuts have to be made and policies reformed it may be helpful to have additional instruments in the reform package that deal with environmental and sustainability issues. For example, if interest rates need to be raised and credit limited to contain inflation, projects and acffvities that contribute to long term sustainability could continue to be financed from institutions that had access to long term funds at stable rates of interest. Other instruments which, if existed, would "decouple" macroeconomic and environmental or sustainable policies include: environmental charges, better-defined property rights regimes, agricultural land zoning laws etc. Third, sustainable development will be less susceptible to the vagaries of macroeconomic policy if it is funded from non-budgetary sources. Examples would be to fund irrigation projects from a local tax on the increased value of the land that is affected. Such a use of the "beneficiary pay principle:" can be applied in many cases and can take much of the financing of sustainable development out of the public sector.

## 2.5 Trade policies and sustainable agriculture

Trade regimes and trade policies can have a major impact on the use of land and on the sustainability of the agriculture that is practised. A devaluation of the currency will result in an increase in producttion of export oriented crops. Depending on what these crops are, and on what property regimes are in force, the result can be an increase in soil erosion, as erosive crops are planted on marginal lands, or a reduction in soil erosion as the increase in demand is for perennials with good soil retention properties. Even this, however, is simplified. If the

[1] Macroeconomic policies and the environment are also discussed in Chapter 2.

perennials such as rubber are grown on marginal land, or by clearing forests, (as was the case in Thailand), the expansion of exports can be environmentally damaging. In general, where property rights are well defined, the environmental effects of different crops are, to a large extent, internalised. The parties responsible for growing those crops have an interest in maintaining the long-term productivity of their asset. Changes in exchange rates and tariffs will still impact on the environment but the effects will be much less than they would be with open access property regimes. Furthermore the damages will be less a matter of public concern

The paper by Runge (1994, paper attached) shows how the environmental impacts of government policy interventions in agriculture revolve around the commodity composition of agricultural trade and the increasing use of water, fertilisers and chemical inputs. Developed countries have become specialised in growing certain crops due to subsidies aimed at sheltering farmers from market risks. These have lead to surpluses that necessitate import barriers for cheaper products, which in turn have led to environmentally unsound practices (homogenous crops, reduced rotations, increased irrigation, fertiliser and chemical use). In developing countries, farmers have incentives to cultivate crops on marginal soils and deforest environmentally valuable land. Government policies to increase yields lead to subsidies on fertilizers and chemical inputs, leading to degraded soil and water, and to human health problems.

Developing countries ('The South') can only be expected to make changes in environmental regulations for agriculture if they are allowed access to industrial countries ('The North') markets. For developing countries, if "decoupling" (separating price subsidies from specific crops) emerged from the trade liberalisation process, or domestic price pressures, a large impact could be made on improving the environment, but only with improved environmental regulations aimed at decreasing damages to soil and water from agriculture. In developing countries, the first step is to develop policies in agriculture, but policies are expensive to undertake, and income growth is a prerequisite to trade reform.

Several case studies indicate, however, that policy reform has not only a "liberalisation effect", but also an "environmental effect" on welfare Multilateral decoupling, as called for in the Uruguay Round, has a beneficial effect on farm income, which gives incentives for sustainable agronomic practices. Global models of trade liberalisation indicate widespread welfare benefits, creating the wherewithal for targeted environmental improvements in developing countries. The dividend from trade reform in developing countries is much greater than any compensation likely to be paid from environmental improvements.

Conversely, trade preferences, such as those offered by the European Union (EU) influence land use in many developing countries. The above market support prices available to beef exported from Botswana to the EU has been cited as one reason for excessive rangeland degradation in that country, as cattle holdings are increased. The access for tapioca as animal feedstock from Thailand to the EU is believed to have resulted in significant deforestation in that country. The export of low grade subsidised beef **from** the EU to West Africa has been criticised as resulting in a breakdown of traditional cattle markets in the region, and thereby causing hardship to the communities that made their livelihood in those markets.

In other cases the impacts are less clear. Preferential access to the EU under the Lome Convention is provided to bananas and sugar. The exporting countries that are hugely affected by the protocols are the Caribbean islands, Fiji and Mauritius. A change in the protocols, as currently being reviewed, would have major impacts in all sectors of the economy for these producers. The analysis of these impacts requires the use of inter-sectoral models (sometimes called general equilibrium models). It also requires a realistic assessment of the likely patterns of land use with and without the trade regimes. Such models can play an important part in the design of policy changes in the trade sector. Certainly the process of transition from a preferential trade regime to a more open one should take account of the costs of transition, which can be significant.

As with all macroeconomic policies, the trade regimes will have a less serious impact on the environment if the appropriate environmental policy instruments are in place. The article by Runge, (1994) reemphasises this point and identifies some of the policies that are appropriate in a trade context. He also points out, importantly, that it would be a mistake if the process of trade liberalisation were to be slowed down because of environmental considerations. The argument is that the economic gains from trade liberalization to both developed and developing countries are so large that most environmental damage could be compensated for out of the trade-related benefits. Although this point is well taken, it does not mean that trade liberalisation may not result in environmental damages. Furthermore, these damages could be substantially avoided without reducing most of the benefits of trade.

The principles that are identified by Runge as ones that should guide the application of specific policy instruments are the following:

1. Domestic agriculture and trade targets are likely to be inefficient due to political manipulation,

ineffectiveness or protectionism. They may be justified if aimed at environmental targets instead of a disguise of tariff or subsidy to trade,

2. Domestic agricultural and trade targets should aim to minimise distortions of market signals, while remaining environmentally neutral. If environmental problems remain, they should be addressed directly. The separation of these targets does not imply that environmental objectives are a *sine qua non* for welfare improvements from trade liberalisation,
3. Environmental policies should aim to minimise distortions in the use of natural resources due to market failure, while remaining trade neutral;
4. The coordination of policies of many countries can be made easier if their targets are similar (minimising trade distortions and market failures). This is unlikely, so equivalent rather than strictly harmonised instrumental approaches are more practical.

Protection of special interests stand in the way of integrating world markets. An equal requirement for global market integration lies in staunching the flow of "bads" and "disservices" and the environmental consequences of market failure through policies, which acknowledge and account for these costs.

## 3 CONCLUSIONS

There is a widespread concern in the environmental community that present agricultural practices are unsustainable, that we are using the land in such a way that damages the prospects of future generations to maintain output at a level that meets their needs. The worry is with regard to both modern agriculture with its intensive external inputs and with regard to traditional agriculture with its low yields and dwindling soil resources. This paper has reviewed the most recent evidence on agricultural trends and concludes that the situation is not as dire as is sometimes made out Agricultural output will continue to expand, albeit as a declining rate. The situation in developed countries is a matter of concern with regard to the damages done to the environment and to resources such as water, but is not one of declining output in agriculture. With the right policies the present practices can be made even more sustainable, leaving future generations with undamaged resources and an agricultural base that is as productive, if not more productive than the present base.

In developing countries the need is to mix development in intensive and extensive agriculture. Both are needed, each having its appropriate place in the development of agriculture in these countries. In general increased application of fertiliser will be beneficial. So will some increases in pesticides, although these need to be better controlled and combined with more benign methods of protection. In some areas, however, expansion of agriculture is damaging to the natural environment and the losses may be irreversible. Here restrictions to expansion are needed. With the right policy tools and with general economic development, the pressure on the land can be reduced and sustainability in agriculture achieved in these countries.

What are the right policies? Most important of all is right pricing. This should begin with removing environmentally damaging subsidies and go on to social cost pricing and social cost based practices, so that the external costs of agriculture are borne by the farmers. It will take time, and care will be needed to protect the interests of poor communities, but it can be done. In some cases changes are needed to land titles, in others community based management is required.

A key tool for designing the right policies is the valuation of the natural environment. We need this to know what the environmental costs are, so that they can be internalised. Methods of valuing the environment are being developed and improved all the time. They should now be more used in the design of instruments for environmental protection.

Other instruments that impinge on agriculture are macroeconomic and trade policies. Here we need to ensure that these policies are not implemented without consideration for the environment. Nevertheless, it is better to address the environmental objectives with environmental tools and let the macroeconomic and trade policies address the objectives of growth and economic development. As countries become wealthier, so they can devote more to protection of the environment. In the long term sustainability, especially in developing countries therefore needs this development to take place.

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# Chapter 11

# TRADE AND THE ENVIRONMENT

*by*

*Anil Markandya*

## 1 INTRODUCTION

There has been a considerable increase in interest on the linkages between trade and the environment; both in seeing how changing trade regimes affect the environment, and in the question of how stricter environmental regulations (as well as a general increase in environmental awareness and concern) affect trade. This has led to several papers and books, which have helped improve our understanding of the subject. Frequently, however, one finds that the analysis is limited because there is an information gap. *In theory* many environmental impacts could result from, for example, the liberalisation of trade. But how likely, and how quantitatively important are these impacts in practice? In many cases this is simply not known. There is general agreement therefore that more *empirical research* is required.

We can divide the issues into two broad groups. One deals with the impacts of changes in trading rules and regimes on the environment; and the second with the effects of changes in environmental regulations (and other measures deriving from a concern with the environment), on the international trade prospects of developing countries and transition economies. The first group of issues has particularly come to the forefront as more and more developing countries and transition economies are moving toward liberal trade regimes. The shift derives partly from an increased conviction that a country's economic grow, and the standard of living of its citizens, are enhanced as a result of such policies; and partly the pressure that such countries have come under from the international financial institutions (IMF, World Bank) as well as the GATT, to liberalise their trade regimes. The measures involved consist of some or all of the following: removal of non-tariff barriers to trade, reductions in tariff rates, reductions in subsidies to exporters, and competitive devaluations.

Have these policies affected the environment negatively, as in principle they might? For example, a removal of an export tax on forest products, and/or a fall in the real exchange rate of a country that produces such products could result in increased logging and, indeed, there are some studies that show this. On the other hand, there are also studies that show that the more open an economy is, the less environmentally damaging are its economic activities. (see below for examples of both cases). The question of interest here is whether these particular countries have observed a link between trade liberalisation and environmental degradation and/or improvement. It is examined in Section two.

The second set of issues relates to changes in environmental regulations in a formal sense as well as through the growing public concern for a better environment. How do they affect the trade of a developing country or economy in transition? There are a number of questions that need to be addressed here:

(a) Are the stricter environmental regulations in the trading partners of a particular country leading to higher costs for the exporters, as well as a reduced volume of trade?

(b) What impacts are the stricter environmental regulations in the developing countries and economies in transition having on their exports and imports?

(c) Following on from (a) and (b) above, there is the question of what impacts, if any, result from differences in environmental standards between richer and poorer countries. In particular is there any tendency for the more polluting and dirtier industries to relocate in less well-off countries, thereby avoiding the higher costs of operating under stricter environmental regulations?

(d) Are programmes instituted on a voluntary basis in developed countries (such as the eco-labelling programme) having a detrimental impact on the exports of the countries being studied, or are they offering new market opportunities to those countries?

(e) There are several international treaties that have been acceded to in recent years that have significant trade implications. In the countries concemed, what impacts have they had on the trade flows? These treaties include the Convention on Trade in Endangered Species (CITES), the Convention on the Transboundary Movement of Hazardous Wastes

(the Basel Convention), the Protocol on the Phase Out of Ozone Depleting Substances (the Montreal Protocol), and many others.

This set of questions is dealt with in Section three. For each question some background on the current state of thinking on the issue is provided. This is followed by a review of the findings in the eleven country studies cited above. The countries are generally reviewed alphabetically. Section four concludes the Chapter.

The discussion in this chapter draws on many studies, but particularly it reviews some recent work carried out under the auspices of UNCTAD (Jha, Markandya and Vossenaar, 1999). The countries covered in this programme include Brazil, China, Colombia, Costa Rica, India, Malaysia, Philippines, Poland, Thailand, Turkey and Zimbabwe.

## 2 HAVE THE CHANGES IN THE TRADE REGIMES HAD AN IMPACT ON THE ENVIRONMENT?

In assessing the impacts of changes in the terms under which trade is carried out it is important to recognise that observed increased levels of pollution are not themselves evidence that the policy is undesirable. If the liberalisation policy is successful there will be an increase in economic activity, which may well result in some increase in environmental damage[1]. The important policy point is to ensure that the cost of such damage is kept to a minimum, for any given increase in economic activity.

The above is important and should be kept in mind. Nevertheless it is instructive to know whether trade liberalisation has resulted in increased environmental damage, and what the extent of that damage is. Previous studies provide conflicting evidence on this question. One study in the Philippines (Boyd et. al, (1993) seems to indicate that the removal of tariffs as existing in 1988 would result in substantial increases in deforestation. Similar results are derived from another study of the Philippines (Cruz and Repetto (1993). At the same time a study by Birdsall and Wheeler, of 25 Latin American countries over the period 1960-1988, showed that "openness" of the economy was a significant factor in explaining pollution intensity (pollution levels per unit of GDP). Increased openness was found to imply *reduced* intensity. (Birdsall and Wheeler (1992).

[1] The relationship between the level of development and the quality of the environment is now formalised in the so-called "Environmental Kuznets Curve". This postulates that the environment deteriorates in the early stages of development but then starts to improve once a certain level of living standard has been attained. (Seldon and Song, 1994).

The case studies carried out in the UNCTAD programme also shed some little light on this question.

In the case of **Brazil** the picture is of an economy which had made some little move towards trade liberalisation but had seen a trend for more environmentally harmful exports. Trade and investment policies since the 1970s were based on reducing the impacts of external factors on the economy, and included import restrictions, export promotion, and openness to FDI. Exports increased rapidly (and erratically after 1985), particularly in the livestock, forestry and mining sectors, putting a heavy pressure on the exploitation of natural and energy resources. Exports of manufactured products also expanded rapidly over this period however they are concentrated in products with relatively low levels of technology, with pollution intensive industries accounting for a particularly high share of total exports of manufactured products compared to other developing countries. Furthemmore, unlike OECD and developing countries, this share has increased form 21.6% in 1975 to 44.4%- in 1990. In general exports are concentrated in sectors with declining participation in world 'trade, suggesting that Brazil's intemational competitiveness may be fragile. Hence, although the paper does not say much about trade liberalisation it does provide the message that, in the case of Brazil, a development policy based on import substitution and controlled trade did not protect the environment; in fact the level of resource exploitation increased, as one suspects did the degradation of the environment.

The **China** study gives a mixed message. Since the move to market reforms and increased openness, it declares that growth and trade may have been associated with a partial improvement in the environment. New technologies are less harmful and polluting than the ones they have been replacing. At the same time the sheer scale of expansion, especially in the small scale industries (township factories) has been creating serious ecological problems in certain areas *through loss of biodiversity and depletion of natural resources.* The structure of trade is also believed to have improved as far as natural resources are concerned, with the share of primary products declining from 54 per cent in 1978 to 23 percent in 1992. The latter, however, may not reflect less primary product use in total, but simply more domestic processing of primary products which are then exported in other categories.

**The Colombia case study** identifies the following products as ones where trade liberalisation could potentially affect the environment: cotton products, leather tanning, printing, woven and final textile products, wood products and refining of petroleum. This is based on the competitiveness of the industry. The study reports two

indices of industrial contamination from 1974 to 1992. Both of these ***fell*** during the first period of liberalisation (1979-81) and ***rose*** during the second period (1990-1992). "This suggests that industrial pollution exhibits a behaviour independent from foreign trade policy" (Gaviria et.al (1994)). The growth of contaminating industries during the whole of this period has more to do with conditions of domestic development than trade factors. On the import side similar conclusions are drawn. There does not appear to be any strong correlation between liberalisation of the economy and increases in contaminating industry imports. In other words, the domestic contaminating industries did not face a significant increase in competition as a result of the liberalisation moves.

Two exceptions to the above conclusions are cited. One is the impact on the basic iron and steel metallurgy industry which was traditionally overprotected and heavily polluting. As liberalisation took place, imports increased substantially and domestic production declined. This should have resulted in less environmental damage domestically. The second is the impact of liberalisation on the domestic recycling of paper and packaging material. Colombia has started to import large quantities of paper waste from the US, especially New York. It is of better quality and very competitively priced (partly as a result of subisdies from firms in New York transpoting it for free and paying a fee to the reciting agent). Consequently it has reduced the demand for domestic recycling and increased the amount of paper that goes into domestic landfills. In other words, liberalisation has helped solve "other countries trash problems, to the detriment of Colombian environment and society". (Gaviria et. al (op. cit.).

The **Costa Rica** study identified the period 1985 to 1990 as one of export promotion with a slow opening ofthe economy, and the period since 1991 as one of a more rapid opening of the economy. There was found to be a recent trend towards the development of environmentally harmful industries. However this is not considered to be related to economic and e export promotion policies. Although overall industrial pollution has not increased more than real production, further research is recommended for sectors showing a higher growth in their pollution indicators, such as, foods, beverages, and tobacco; paper, printing and publishing; and chemical industries. Case studies were undertaken on three environmentally sensitive major export sectors, bananas, coffee and forestry. Although details are given of the environmental effects of increased production in these sectors, there is no indication of the extent to which these effects are attributable to changes in the trade regime.

The **Indian case study** cites the increase in exports following a more export-oriented policy, as raising some environmental problems in selected industries, particularly tanning and shellfish. At the same time it acknowledges that solutions exist through the adoption of appropriate technologies and the use of the Polluter Pay Principle. The imposition of trade restrictions to resolve these problems would be inefficient, and perhaps also ineffective.

The **Malaysia case study** points out that the country's open and export oriented economy has resulted in exports increasing from 50% of GDP in 1970 to 81% in 1990. There have also been fundamental changes in the composition of exports over this period, with rubber and tin significantly declining in importance, and crude petroleum, palm oil and timber becoming the most important export commodities. Exports of manufactured products have dramatically increased, accounting for 17% e of total export value in 1975 and 70% in 1993. Environmental problems have been associated with trade liberalisation and the study gives two major examples of how these have been alleviated through domestic environmental policies.

In the case of palm oil, which was the country's worst source of water pollution in 1975, the introduction of environmental quality regulations allowed high rates of export growth to be achieved at the same time as significant environmental improvements. In the case of the electronics industry, which has rapidly expanded to account for 30% of total export value in 1993, potentially serious environmental problems have been successfully controlled through environmental quality regulation, with high rates of compliance attributed to multinational companies installing emission control equipment. Toxic wastes remain the most significant problem and lack of treatment facilities have resulted in accumulations at industrial sites and the export of such waste, however a central waste treatment facility is soon to be operational. Thus, the study points to a situation where a strategy of liberalisation combined with a proper environmental controls can achieve export and economic growth as well as a protected environment.

**The Philippines study** is not forward looking, as the two other studies of that country cited above are, but evaluates the impact of past trade restrictive policies on the environment. The ban on the export of logs from that country, which has been in place in form or another since 1976 is shown to have resulted in wasteful investment in wood processing, with high wastage rates for wood. This, combined with restrictions on imports of logs for the domestic furniture industry, has resulted in increased deforestation. Thus a trade restrictive measure is shown to have been harmful to the environment.

In the case of fisheries the reductions in import tariffs (cut from 78 percent to 40 percent in 1980) certainly resulted in a sharp increase in imports, which helped reduce pressure on the domestic resource base. The role of increased imports in this regard is considered particularly important as domestic fish stocks are dwindling and there is pressure on these resources through increased population and poverty.

On the export side, total volume and value have increased consistently since 1985. The main category for which this has been the case is crustaceans. These accounted for 91 percent of exports in 1990, compared with 43 percent in 1980. The increased production of crustaceans has had negative impacts on wetlands where aquaculture is practised, giving an example where liberalisation has resulted in some environmental damage.

Thus, while the two examples point to some environmental problems resulting from the trade liberalisation, it also points to some advantages (e.g. the increased availability of fish imports). In overall terms the Philippines study concludes that exports have shifted from natural resource intensive sectors, including agriculture, fisheries, and mineral based products, to human resource intensive sectors and lower pollution intensity during the last decade. This has coincided with the period of trade liberalisation and reform.

**The Polish case study** focuses on the market reforms that have been central to the economic changes in that country during this period of transition. In assessing the trade and environmental impacts, it does not try to distinguish between these reforms and trade liberalisation, something, which would be very difficult to do. As these reforms take place, the economy is shifting from energy-intensive, heavy engineering activities to a mix of output that is more like that of developed Western countries. Given that it is starting from a industrial structure that is particularly energy and pollution intensive, this shift will take some time. Over the last few years, as prices were liberalised internally and as trade regimes have shifted from a centralised, partially barter system, to one of decentralised commercial trade, Poland has seen a drop in the exports of electrical and engineering goods and an increase in that of "light goods" (clothing, knitting and leather), wood and paper, and metallurgical goods. At the same time imports of almost all items have increased, particularly chemicals, wood and paper, food items and agricultural products.

The environmental impacts of the trade shifts are difficult to analyse. A preliminary assessment shows that the items whose exports have fallen most (electrical and engineering goods) were not particularly energy or pollution intensive. Among the items where exports have risen, the metallurgical industry is energy and pollution intensive; and the wood paper industry destructive of forest resources which are being used unsustainably. The light industry sector is, by and large, low in terms of domestic pollution and energy use (most of the raw materials imported). Thus the picture is mixed, with the increase in metallurgical exports being particularly damaging to the environment.

**Thailand** has been a high-growth export-oriented economy since the mid 1980s. However the case study found 'few systematic effects' of export production on the Thai environment. Rather pollution was mainly a domestic issue caused by rapid growth. The study did identify some environmental effects of export production. Mainly, diversification of ' exports has reduced deforestation and soil erosion due to agriculture, but led to new environmental problems such as water pollution caused by textile production, deforestation due to timber production, use of ozone depleting substances, and creation of hazardous wastes by the electronics industry. The study concludes, however, that it is diffcult to isolate the impact of production for export from production for the domestic market, and that the impact of export growth may well be positive if it has provided resources for environmental measures. Since the early 1990s Thailand has changed its trade policy from export promotion to trade liberalisation. The report states that the effects of this have still to be evaluated.

The **Turkey case study** looks at the iron and steel industry which has been among the most important industrial exports since the switch to an export oriented trade strategy. The products and processing facilities located in the Iskenderun Bay area have transformed that area from one of considerable natural beauty to a heavily polluted industrial environment. Similar developments are noted in Izmit Bay, Istanbul and Izmir regions. ' The study also describes some of the serious environmental problems in the trade-oriented agricultural sectors, especially those associated with cotton production and its use of pesticides.

In the **Zimbabwe case study** little is said about trade liberalisation and its impacts on the environment. It is noted that the "removal of tariff and conventional quantitative restrictions as a means of liberalising trade is occurring simultaneously with imposition of restrictions based on environmental concerns" (Nkomo et. al (1994), Chapter 14 of this book). Much of the case study is concerned with the impacts of these environmental regulations in developing countries and their impacts on Zimbabwe's trade (see below).

In conclusion one can say that these studies do not provide emphatic evidence that trade liberalisation has systematically hurt the environment. There are specific situations in which environmental pressures have

been reduced and others which it has been increased. Where it has been increased, some of the case studies have identified policies that, if effected, could mitigate the damage without losing the benefits of the trade liberalisation. In fact one could argue that in all cases where negative impacts have been identified, some policies of environmental protection, that were acceptable from a trade point of view, could have been introduced to reduce that impact.

## 3 WHAT ARE THE TRADE IMPACTS OF CHANGING ENVIRONMENTAL REGULATIONS AND PRACTICES?

As countries become more concerned about environmental degradation they are taking a number of measures to reduce, or at least keep under control the damage caused to the environment by the production and consumption activities of society. The instruments used to achieve these objectives include: (a) internalisation of environmental costs and (b) increased awareness and information programs that assist individuals in making ecologically sound decisions. In both cases, there is the concern that the measures undertaken will have a detrimental impact on trade, especially on the exports of developing countries and economies in transition. Should that prove to be the case, the prospects for economic growth in these countries could be seriously damaged, thereby making sustainable development in the Nor dependent on economic stagnation (or at least reduced growth) in the South. Related to this is a suspicion, especially among countries of the South, that some of the measures being proposed under the guise of sustainable development are in fact designed to restrict access to the markets of the North for the developing countries and economies in transition.

At the same time, developing and transition economies themselves need to take similar measures to make their development paths more sustainable. There is a tension within these countries between those who wish to enact stricter environmental regulations and those who argue that economic growth is a priority. In so far as there is a conflict between these objectives, one way in which it may arise is through a reduced competitiveness of these countries in world markets. Hence it is important to review the evidence on the effects of environmental regulations on competitiveness, employment and growth in the countries being studied.

This section examines the five questions identified in section two, that relate to different dimensions of the link between changes in environmental regulations and changes in trade.

### 3.1 Are stricter environmental regulations in developed countries leading to higher costs and lower volumes for exporters in case study countries?

The regulations governing trade place certain restrictions on the ways in which one country can impose its standards, environmental or otherwise, on its trading partners. In principle each country can set its own environmental standards but, unless there are transboundary pollution issues, it can not impose them on its trading partners. Although this is clear enough in principle, many issues arise in practice. In order to ensure that goods classified under a particular commodity classification are in fact the same commodity, product standards have been set up, both at the international as well as at the national level. The GATT made a clear distinction between standards that define products and those that refer to processes. A country may define its own product standards but it must apply them without discrimination and should wherever possible seek consultation before introducing changes[2]. However, it cannot apply process standards on its trading partners (Article III). An example of a product standard would be specifications about a particular kind of steel. A process standard would state how that steel was produced. A similar distinction is being elaborated by the successor to the GATT — the WTO.

The distinction between product and process standards, however, is not easy to maintain. By specifying certain products in a very narrow way, a country can more or less determine *how* certain goods are produced. Moreover, in some cases it is impossible for an importing country to ensure that product standards have been met without inspecting the production facilities. Food processing is a case in point and the European Union, for example, inspects meat and drug production facilities in countries such as India and Zimbabwe (see below) before approving exports to its region. This gives rise to a direct involvement in the process of production.

In the existing literature it is generally acknowledged that developing countries have had to adjust their production processes in response to changing environmental requirements in developed countries. Measures such as changing pesticide residue levels permitted in foodstuffs, changing emissions standards for machines, and changing packaging requirements for all commodities have placed a burden on the exporters who are subject to these requirements. What is less clear, however, is whether these measures have had any significant impact on trade.

2 Under the prospective Uruguay Round Agreement, greater emphasis is being placed on internationally determined product standards.

The **Brazil** case study suggests that the country is vulnerable to external environmental requirements for three reasons. First, the high natural resource and energy intensity of exports; second, the large share of exports of homogeneous products, with little product differentiation potential, competing on the basis of price, and third, the high share of exports to markets where environmental requirements are stringent, for example, exports of food products, timber and timber products, paper, textiles and footwear are principally destined for OECD countries. An estimated 25-30% of exports to OECD countries are in sectors where environmental requirements are already emerging.

A case study of the pulp and paper industry concludes that it is particularly concerned with recycling requirements and the use of voluntary instruments. However, the large companies surveyed had generally been able to comply with such requirements "without significant adverse effects on their competitiveness" (Veiga et al., 1995). The iron and steel industry is said to be less effected by external requirements and the EU, where requirements tend to be most stringent, is only a relatively small export market for Brazil.

The study concludes that since sectors tend to be heterogeneous in terms of firm size and level of technology, the competitiveness effects of environmental regulations tend to be differentiated even within sectors. SMEs tend to have greater difficulties with compliance than large firms.

The **China case study** takes the view that foreign laws and regulations have led to some loss of exports but does not cite specific cases. At the same time it does give several examples of the adoption of environmentally sound technologies (EST) (partly in response to foreign requirements of product quality) that have resulted in reduced wastes *and* increased profits for the enterprises operating them. These include a cement plant in Quju, a valve plant in Tanjing and many others. The study definitely takes the view that the adoption of EST is inevitable for companies involved in the export sector and that those that lag behind will be uncompetitive in the international economy.

In the **Colombian case study,** a survey of exporting firms was conducted. It showed that most industries had not perceived or experienced major effects of international environmental standards. 63 percent said they had not perceived pressures from international regulations and 16 percent said they had. Part of this is due to Colombia's trade pattern. The European market is not the most important one for Colombia, Germany, for example, which has the most stringent standards, accounts for 2.4 and 7.6 percent of non-traditional and traditional exports respectively[3].

The most significant problem exporting firms face with regard to environmental regulations in developed countries is the lack of sufficient up-to-date information. Colombian exporters of tropical fruit to Europe faced this problem with regard to the German Packaging Ordinance. Confusion also exists about the Green Dot Program and changes in packaging requirement legislation

One case where foreign, environmentally driven, regulations had a significant impact on Colombia was the US Tuna fish embargo. That market accounted for 32 percent of Colombia's exports of tuna and the ban resulted in a loss of between $20mn. and $32mn. It is also argued that the "dolphin safe" method reduces the volume of captured tuna.

Less cases of actual export restrictions were found in the **Costa Rica** case study than in other national studies. In the survey of 39 firms there was little evidence of external environmental regulations effecting trade, with compliance costs in the majority of cases under 2% of total operating costs. Some small enterprises had difficulties adapting to new norms, but only one firm, in the fishing sector, had experienced a decrease in competitive advantage due to the cost of technological changes necessitated by external environmental measures. Nevertheless, external environmental regulations are perceived as a potential threat due to Costa Rica's heavy dependence on export products based on natural resources. It is estimated that 40% of the total value of exports are potentially susceptible to environrnental regulations in their market destinations, and concern remains over the country's lack of institutional capacity to respond to such restrictions.

The study gives several examples of the vulnerability of Costa Rican exports to environmental protection measures in developed countries, although these have not had significant impacts on total export values. The examples include, first, the 1995 US requirement to use "Turtle Exclusion Devices" in the shrimp industry which was expected to cause a loss of about $2 million per year in foreign currency for the country. Second, the secondary effects on tuna canning companies of a US tuna embargo placed on Mexico aimed at dolphin protection. The effects have not been quantied but are believed to be significant.

The **Indian case study** looked at exports from two important sectors (leather and shellfish) and

[3] Traditional exports, representing 44 percent of total exports are oil, coffee, coal and ferronickel. Non-traditional exports are fruits, flowers, fish, textiles, leather, chemicals and plastic, and resin.

concluded that the increasingly stringent export standards have certainly raised the costs of production, especially for the leather sector, where costs using the more environmentally friendly methods are nearly three times as high. However, nothing is said about whether the producers can recover the higher cost with higher prices, thus leaving profitability unchanged.

Particular concern has been expressed about the German Packaging Ordinance, which requires the use of packaging materials not readily available in India. Other industries that have been affected by foreign regulations are motor vehicles and pharmaceuticals. A factor that affects exports in a big way is obtaining information about changes in regulations and the fact that they vary from country to country.

The higher costs resulting from these regulation will impact most seriously on small producers, who will need assistance to install the equipment required to meet the production requirements. However, there is only anecdotal evidence that overall exports are suffering as a result of these requirements. On the other hand there are indications that industry is flexible and able to adjust to the requirements without too much diffficulty. Another Indian study (Achanta et al. (1994)), looked at the technologies required to meet changing environmental standards in the importing countries. They found that exporters had managed to acquire the necessary technology, often with assistance from the importers. Joint ventures were one way in which the more sophisticated products obtained the relevant technology.

The **Malaysia study** states that the country's export sectors that are vulnerable to external environmental requirements include timber, textiles, air conditioners and electronics. The study focuses on timber exports which fell from 20.4 million $m^3$ in 1990 to 9.3 million $m^3$ in 1993. Bans or restrictions on the use of tropical timber have been implemented by a number of developed countries. Although sawlogs have remained largely unaffected, sawn timber exports have been affected by external restrictions, notably in the Netherlands. Although export earnings from timber have fallen drastically the study does not isolate the effect of external regulation because the reduction is partially due to domestic policies.

The **Philippine study** concludes that foreign regulations have not *per se* posed a serious constrain to Philippine exports. This is based on a limited number of interviews with company managers. They believe that stricter foreign regulations will have the greatest impact on the prices of raw materials and research and development costs. Investment in pollution control devices for domestic reasons will be required in all sectors in the future, but especially in the livestock production and manufacturing sectors, such as printing, non-ferrous metals, food processing and pottery. The additional costs of meeting these and the requirements of foreign importers are not deemed to be large, and should be affordable for most producers (see Section 4.2 below).

The sector where future regulation could seriously impact on exports is "sustainable forestry management" (Intal et al., 1997, Chapter 10 of this book). This is likely to impact on the country's furniture exports, as sources of local sustainable timber plantations are very limited.

The **Polish case study** focuses mainly on the stricter standards that have to be met if it is to become a member of the EU. The costs of meeting EU standards is considerable; according to one estimate it amounts to $30bn.

Given that the country has set a target of achieving membership within 10-12 years, it is likely that many local producers will not be able to meet the standards and therefore be unable to export (or even produce for the home market). At the same time, the decision to join the EU, and the time period over which to phase in the stricter standards are taken by the Polish government and cannot be said to have been imposed from outside. In this sense, the impacts of stricter standards here are different from those of other countries examined in this report. It is also worth noting that the cost of meeting the standards is not fixed but a function of the policies implemented to bring about the higher standards. The use of market based instruments could help to substantially reduce these costs.

The **Thailand case study** states that, although some exports are concentrated on specific markets making them vulnerable to external regulations — for example 70% of fruit exports go to the EU — in general Thai exports have been flexible enough to adapt to such measures, often behaving pro-actively. An estimated 42.2% of total exports were in products identified as sensitive to extemal regulations. Two product groups, fish, tuna, and shrimp; and textiles and footwear, account for 80% of all sensitive products.

In the case of fish, tuna and shrimp exports, these have been the subject of a number of potentially damaging external regulations. However in two in-depth interviews cited in the study, producers have been able to adapt so that they have hardly been affected by these regulations. First, the Japanese Anti-additive Import regulation of 1991 threatened Thai exports of shrimp and shellfish. By adapting production methods exports have, however, continued to grow. Second, tuna exports, which amounted to US$ 5 billion in 1994 making Thailand the worlds largest exporter of canned tuna, have been affected by US restrictions, Again, Thai producers have changed

their sources to comply with these restrictions. A French ban on fishery product imports for health and safety reasons in 1994 has not enabled such adjustment, though an indication of the effect on export values is not given.

In 1995 the textiles industry, which after electronics is the biggest Thai exporter, was faced with a German ban of Azo dyestuffs. Dying firms had to switch to substitutes at a cost increase of 5-20%. Smaller firms were found to have had difficulties absorbing the higher costs.

In the **Turkish case study** a survey of major exporters in Turkey was conducted. It was found that environmental requirements in the markets to which they exported were virtually never a problem. The exporters accepted the standards as inevitable and often worked closely with their trading partners, who would sometimes specify the processes to be adopted. The necessary technology was mostly available in Turkey and the volume of exports was unaffected by the imposition of stricter regulations in the European Union, including the new packaging directives. One issue raised in the survey was that of different environmental standards. As with India, exporters found some loss of economies of scale in meeting the requirements of different markets.

The **Zimbabwe study** cites a number of examples where foreign standards are hampering its export requirements. One is ostrich production. It feels that the restrictions on exports of the live birds or the meat to the EU and Australia (including costly blood tests and quarantine regulations) are unnecessarily strict, having been designed in part to make Zimbabwe exports less competitive. A second is phytosanitary standards for the export of beef to the EU, where European importers have to inspect all produce before it leaves the country of origin. This imposes higher costs on Zimbabwean exporters and may dissuade some producers of beef to look for export markets. A third is the German packaging ordinance, where there is some concern about the cost and availability of recyclable materials, as well as difficulties in submitting packaging for evaluation and certification. To assist exporters, producers associations are running courses on packaging technology. A fourth is the restrictions on the textile industry where certain drying and sizing processes are not being allowed. As a result it is necessary to change and improve the processing technology, which is costly. A fifth is footwear, where leather tanned using PCB is banned in the international markets. This has necessitated a shift to different chemicals and processing techniques.

For all these examples, however, (with the exception of ostrich production) it is not clear whether the international requirements have had a significant impact on levels of exports.

To conclude, the evidence from these case studies on how foreign environmental regulations impact on a developing country or transition economy is mixed. Many of the larger exporting countries claim that the effects have been small, and in most cases manageable for the exporters. In several cases the adoption of the stricter standards not only decreases environmental damage, it increases efficiency and profits for firms.

The exceptions to this favourable picture are:

(a) smaller producers are more affected by the increasing number and variety of regulations. This, in turn, has implications for future economic growth, and for competition inside the country itself.;

(b) several studies point to the difficulty in firms simply keeping themselves informed of the changes in regulations. Lack of information is resulting in reductions in exports, and

(c) some countries believe that the regulations are cumbersome, complex and at least partly designed to make exports uncompetitive. They find the costs of meeting them have affected exports. This is particularly true for Zimbabwe.

## 3.2 What impacts are stricter environmental regulations in the case study countries themselves having on trade?

Relatively little information is available on the economic impacts of domestically generated environmental regulations in developing countries. Industrialists frequently claim that the imposition of stricter standards will result in loss of competitiveness, employment and growth but such claims are largely unsubstantiated[4]. Furthermore one should not ignore the costs of environmental degradation on the business community, in terms of work days lost, congestion etc.

There are several studies that look at the same issues, but for developed countries, principally the US (see Dean (1992) for a survey). In general their findings are:

(a) the cost of pollution abatement measures in the US have been only a small proportion of industry's costs (around 1.5 per cent);

(b) the percentage decrease in output attributable to

[4] One study which has looked at the effects on a developing country's exports of higher environmental costs is Low (1992). He examined the effects of an pollution abatement and control expenditure tax (PACE) on Mexico, raising its costs to the level of the US. The results obtained show that such a measure would have very modest trade effects, amounting to at most 2 percent of Mexico's export revenues.

environmental control costs in the US has averaged less than one percent (the exceptions are industries such as petroleum),

(c) when allowance is made for general equilibrium effects, the cost impacts are lower and more evenly spread throughout the economy;

(d) if abatement costs were to raise prices by one percent in the US, one study estimated the impact to be a reduction in exports of 2.7 percent, with sectoral impacts varying from virtually zero (special industry machinery), to 7 percent (copper) (Robison (1988)). Another study, however, using a different methodology, concludes that the impact on exports is negligible.

These results can be useful in evaluating similar measures in developing countries but some caution is warranted. First the costs of environmental regulations as identified in the above studies (which are incremental costs) may be underestimated. They do not include work place, health and safety protection costs (Chapman, (1991)). Also, developing countries are starting from a lower base in terms of pollution abatement capital. Second, the effective response of industry requires it to be able to finance the capital expenditures and to respond flexibly to changes in relative prices. Both these factors are more difficult in developing countries.

In contrast to the negative impacts of environmental regulation there are also some positive effects to consider. There will be some growth of a domestic pollution abatement industry. Even if much of the technology is imported, various studies have shown that adaptation to local conditions is almost always necessary. This will generate some economic activity. Then there is the view that stricter regulations will benefit the more innovative and enterprising firms (Porter (1991)). This may be because they are able to find ways of meeting the standards at lower cost, or because they can respond more quickly and more effectively.

In the case studies there is little to suggest that, apart from Poland, stricter locally imposed standards have had a major impact on trade.

In the case of **Brazil** environmental policy was first established in 1981 but little evidence is presented in the case study about the overall effect on trade. A principal example given is that of legislation introduced in the early 1990s in the State of Minas Gerais prescribing that by 1998 all charcoal consumed must originate from non-natural or sustainably managed forests. Charcoal consumption accounts for up to 70 % of the production costs of pig iron which is a major Brazilian export, and therefore, this legislation had significant trade effects. However, with the help of fiscal incentives and programmes to support the financing of forest plantations these standards have been successfully introduced and trade effects have been negligible. Interestingly, the regulation that is most concerning the iron and steel industry is that related to the imports of scrap. It is argued that this restriction could result in a decline in the competitiveness effects of domestic production, as supply prices for inputs will rise.

In the **China study**, it is noted that the country only started introducing environmental measures in the 1970s and there is still a wide gap in pollution control between itself and developed countries (perhaps as much as 20-30 years). Where enterprises have made changes to production processes they have generally benefited both in terms of the environment and in terms of foreign exchange earned. Examples quoted include the Zhuzhou hard metal alloy factory which was a large extremely dirty factory with extensive environmental damage. After a major attempt to improve its environmental performance, the situation improved substantially in that regard, as well as in terms of exports, which exceed $800mn. The paper gives a few other examples of a similar nature, and one can assume that they are intended to show what is believed to be a general picture.

In the **Colombia study** a survey of about 300 exporting firms is reported where the question of how domestic environmental standards has affected competitiveness was posed. 29 percent responded that complying with national environmental standards had affected their competitiveness positively, and only 4 percent though that it had a negative affect. When asked whether the recent creation of the Ministry of the Environment and the consolidation of a national environmental policy would hinder or benefit them, 52 percent said they thought it would affect them positively, 8 percent thought it would affect them negatively and 38 percent did not know. Similar percentages emerge for the question of whether these trends will affect their international competitiveness.

The **Costa Rica study** explains that a considerable effort has been made in recent years to update the institutional framework regarding the environment. Most actions have been in the form of domestic governmental regulations, particularly in the cases of banana, coffee and some industrial products. In the case of bananas, the real cost of adapting to domestic environmental measures is hard to determine. Some rough estimates suggest, however, that these are no more than 4% of total costs. In the coffee sector a three stage plan has been developed to address the problem of water pollution with the estimated costs of investments needed for compliance by coffee processing plants given as around 5.5% of total cost of the plant. There is no indication of the effects

these costs have had on trade, although the two products given as examples are the major traditional exports, which continue to have a strong export performance.

The **Indian case study** focuses almost entirely on foreign environmental requirements. However, there are some examples of how the costs of domestic compliance differ from those of international compliance. For the leather sector, price increases in leather and leather products resulting from domestic pollution abatement rules was estimated at 1.5 percent. Even this small average amount would, however, have an impact on small producers, as there is a substantial fixed cost component to it. But compared to the costs of meeting international standards it is tiny; the increase for international products is 300 percent, presumably most of which is recovered from a higher sale price. For shellfish the costs of an effluent treatment plant for the fish processing industry, introduced for domestic reasons, would have a very small cost (0.2 percent of turnover).

The **Malaysia study** details the substantial developments in domestic environmental policy since the 1970s. However there is no overall view on how these have affected trade. The timber case study states that domestic regulations on sustainable forest management under the National Conservation Strategy, has caused a reduction in logging areas and timber production, but the impact of this on trade is difficult to isolate from external and other domestic policies. As stated earlier, the palm oil and electronic industries are examples of where high rates of export growth have been achieved at the same time as the implementation of domestic environmental quality regulations.

The **Philippine study** is the one that looks at this question most systematically. It computes first the share of abatement cost to meet domestic standards as a percentage of total sector cost for all the major sectors. The only ones where this cost exceeds two percent are: public administration and defence, poultry and poultry products, forestry, agricultural crops, and livestock and livestock production. It is less then one percent for 51 of the 61 sectors. Next it calculates the private competitiveness of different sectors with different levels of environmental cost of stricter local standards being imposed on them. If the costs that are imposed are as given above (i.e. generally less than one percent and almost always less than two percent) the ratio of domestic private costs to earnings (which is a measure of private profitability, with values less than one indicating that exporting is profitable) remains less than one for all sectors, on average except inland fishing. However, it is possible, and likely, that marginal firms (those with costs above the average) will be affected in selected sectors. These include aquaculture, and fish processing. Unfortunately no calculations are presented for agroprocessing, forestry products, livestock and poultry. Apart from these sectors, it appears that local environmental standards would not have a significant impact on overall sectoral competitiveness.

The **Poland study** makes the most strong case for export losses resulting from changes in domestic environment and environment-related regulations. Many of Poland's exports are characterised by a high level of raw materials and energy. As part of the market reforms prices of both raw materials and energy have risen sharply in Poland. These reforms have also impacted positively on the environment, but they have contributed to a loss of international competitiveness. One special case of this is the export of electricity, which Poland exports to the Czech republic and Germany. Its price advantage is derived partly from the fact that it does not have to capture up the sulphur emissions. As flue gas desulphurisation is introduced, export of electricity will probably become uncompetitive (costs of generation will rise by as much as 40 percent). Similar factors apply to coal mining, where stricter ecological conditions on saline water discharges are making production uncompetitive (costs will rise by 10-15 percent). In the case of coal it is estimated that, if all current environmental standards are met, exports in year 2000 will fall from around 25mn. tons to 7-10mn. tons. The increase in electricity prices is also conjectured to affect the export of the following energy intensive goods: fertilisers, plastics, organic chemicals, metal products and building materials. At the same time, however, some of the environmentally sound technologies will increase efficiency. For example, with steel it is estimated that the market restructuring will reduce capacity be a large amount and reduce environmental damages by 70-80 percent. Those plants that are left will be more efficient, with production costs down by $20-25/ton. Finally, another positive aspect of the stricter environmental regulations is the development of a pollution control industry that Poland, being a leader in environmental abatement among the economies in transition, hopes to export to other transition economies.

The **Thailand study** does not give details of the effects on trade of domestic environmental regulations. However, in the context of thirty years of rapid industrialisation environmental legislation such as the National Environmental Quality Act (1992) which applies the polluter- pays-principle, and the Factories Act (1992) which can levy fines for non- compliance with environmental standards has only been introduced relatively recently. Even this has been relatively ineffectual. No details are given about the effects of these regulations on exporters' costs and the value of exports.

The **Turkey case study** focuses almost exclusively on the demands of foreign markets. In the domestic market enterprises interviewed made the interesting point that stricter national standards would actually benefit them because: (a) they were currently facing different standards in the home and export markets and this was adding to their costs and (b) producers in other developing countries were selling, in the Turkish market, goods that met the lower standards, thus making it difficult for local producers to compete with their higher standard goods.

The **Zimbabwe study** has little to say about local standards and competitiveness, but makes the point that environmental regulations in the country are poorly enforced. Thus large enterprises effectively avoid any impact on competitiveness by simply evading the regulation or obtaining exemption permits which are easily granted for exporters.

## 3.3 Is the migration of "dirty" industries to less strictly regulated countries a problem?

The decision of where to locate a production facility involves a number of factors, including the cost of labour, access to markets, social and political conditions in the country concerned, the infrastructure facilities in the country concerned and the regulatory framework. The last includes environmental regulations but they are only part of the set of regulations. A lot of evidence exists to the effect that investors look not only at *current* regulations but also at the stability of the regulatory framework (how frequently governments change the rules).

To evaluate whether firms locate in countries to take advantage of lax environmental regulations it is necessary to look carefully at the quantitative importance of the different factors that determine location decisions. Studies of multinational corporations have shown that these decisions are most influenced by such factors as labour costs, access to markets and the existence of a developed industrial base (Wheeler and Mody (1992)). Factors such as environmental regulations and corporate tax rates emerge as less important. At the same time, however, study of majority-owned affiliates of OECD-based companies in developing countries shows that those involved in pollution intensive industries did increase their investment slightly faster than did all manufacturing industries (Jaffe et al. (1993)). This statistical evidence offers only weak support of the pollution-migration hypothesis as it does not point to any significant change in investment patterns. What is observed could be explained by other factors such as the changing structural pattern of demand in the developing countries themselves.

More recent studies have supported these findings. Dean (1990) in a comprehensive survey of studies published up to 1990 concludes that there is little evidence of industrial relocation because of different environmental regulations. Grossman and Krueger (1992) analyse the *maquiladora* program (which permitted US firms to locate on the Mexican side of the US-Mexico border on advantageous terms) and find that pollution abatement costs were not a significant determinant of the trade generated by the program.

Even when pollution intensive industries do locate in developing countries, they do not adopt a minimalist approach in terms of meeting environmental regulations. Often corporate policy dictates the use of the same technology and pollution controls in all foreign countries where plants are located as in the home country (subject to, of course, meeting the local standards where the latter are more strict). This is partly in response to the public image of which they are very conscious, and partly in recognition of the fact that regulations are almost certain to become more strict in the developing countries, and pre-emptive action may well be cost effective (Jaffe et al., op. cit.).

The case studies undertaken here say little or nothing about this issue. Only the ones from China, Colombia and Thailand address the question. The **China study** states that some overseas enterprises did locate in China due to stricter regulation regarding the environment in developed countries, particularly in the leather goods, paper, smelted products, chemicals and pharmaceutical industries. However, there is no specific evidence provided to support this assertion.

The **Colombia study** supports the conclusions of the main studies in this area. First Colombia has relatively strict local pollution and environmental standards. Second, it is not particularly well situated for multinational companies to select it for location; Central America or South East Asia would be preferable locations from which to take advantage of lax standards. Of the 20 multinational companies surveyed none appeared to have established themselves in Colombia for environmental reasons and most were already meeting OECD standards.

The **Thailand study** states that there are some indications that foreign companies are using the country as a "pollution haven" and shifting ODS consuming production there to take advantage of its Annex A status. The significant increases in ODS imports between 1986 and 1991, and the fact that 97% of Thai production using solvents is undertaken by Japanese, European and US firms or is part of a joint venture with Thai firms, is taken as evidence for this trend. In recognition of this situation the Thai Department of Industrial Works has

entered into agreements with the US Environmental Protection Agency and the Japanese Ministry of International Trade and Industry, which encourage Transnational Corporations (TNCs) to halt the use of solvents in their Thai operations on the same timetable as their domestic operations.

To conclude, it appears that industrial location, for reasons of different environmental standards, is either not perceived as a relevant issue, and therefore not investigated; or is not found to be significant effect when actually investigated. The one exception is the case of Thailand where, as a result of special privileges for Thailand under the Montreal Protocol, some TNCs have located their operations inside the country. But the government has acted to control this situation and the impacts should be temporary.

## 3.4 What impacts are non-official pressures for more ecologically friendly goods having on the exports of the developing countries and economies in transition?

Many of the recent actions taken in developed countries do not have an official status, or may only have government support but are not backed by laws. Examples of such measures are the demand by individual importers for specific processes in the exporting factories, even to the extent of demanding changes in social conditions. There are no government-to-government rules about such demands but they are nevertheless real and it hard to see how they could be effectively legislated against. If an importer only wants to trade with an exporting company that meets its standards of acceptability in production, and the exporter is willing to meet those terms, there is little that governments or trade bodies can do to prevent it. Nevertheless it is important to find out how significant such pressures are on producers in developing and transition economies, and what their impact on costs and exports has been.

A related set of pressures arises from the various programmes to label goods as environmentally friendly — the so-called eco-labelling programmes. This is a major topic that is currently being studied in some depth. There are a number of different labelling schemes that come under this general title. They include labels provided range from single criteria labels (recyclability, degradability etc.), to others based on a life cycle assessment of the products environmental assessment. A number of developed countries, as well as some developing countries, have such schemes. In all cases so far they are voluntary schemes, with the extent of government involvement varying from active participation to some administrative support and encouragement. As sources of information to consumers about the products, and as encouragement to be more ecologically aware they are a positive development. However, there is the danger that they could become barriers to trade. Rege (1993) argues that labels will be regarded as causing non-tariff barriers to trade if:

(a) the criteria on which they are awarded are not based on objective or scientific considerations, or fail to take into account adequately the production processes in other countries;

(b) procedures for verification are unnecessarily strict, making it almost impossible for an outsider to obtain the label; and

(c) the system is adopted for a product that is almost entirely imported and the right to grant a label rests with the importing country.

Reviews of eco-labelling systems in a number of countries (Jha et al. (1993); Iba (1993); Pearson (1993)) conclude that:

(a) although foreign products have access to the schemes, they find it more diffficult to obtain them, owing to lack of information and a proportionately higher cost of obtaining the label;

(b) within the countries in which they are awarded, the labels have not always had an identifiable impact in terms of sales of products under the relevant categories. In the Blue Angel scheme in Germany, for some commodities such as low pollutant coatings, the share of labelled products is significant. However, that may have been the case even without the label. Certainly "eco-friendly" products are known to be attractive to some consumers, who are willing in some cases to pay a higher prices for them,

(c) the trade impacts on developing countries are also difficult to measure. Many of the products covered by the labels are not exported by developing countries. If labels are provided covering products that these countries do export the impacts will be significant. Certainly the above objections that such labels could be seen as technical barriers to trade would have to be faced. One such case was the attempted Austrian legislation that products made from tropical timber should be marked as such in the market. This was objected to by the ASEAN countries on the grounds that it discriminated against the imported product against the domestically produced one (the "like product rule" under the GATT),

(d) developing countries are introducing similar schemes in part to promote the exports of their products, and in part to promote environmentally friendly products at home. The success of the schemes to date has been very limited.

The Impacts of eco-labels on prices have been assessed for a number of products. Table 1 summarises the main findings. They show that a small percentage of the population is willing to pay about 10-15% above the price of non-labelled products. This is important in designing policy for such products and is the subject of our case study.

Eco-labelling could emerge as a major barrier to trade if foreign producers cannot have access to the labels on reasonable terms. This will require the recognition of "good practices" in the producing countries, even if these practices are not the same as those under which the label is granted in the home country. One way might be to offer reciprocal recognition to countries that provide eco-labels on the basis of agreed criteria.

There is also some evidence that developing countries may actually turn the system to their advantage by marketing their own "eco-friendly" products more aggressively. The development of "green food" labels in China, "green cotton" in India and the possible use of natural fibres such as jute in packaging are cases in point (see below). To be successful this will require co-operation with the importing countries, to define the appropriate labels and even to modify the importers own labelling schemes (see also Scholz and Wiemann (1993)).

Almost all the country studies reviewed here have something to say on this set of issues. The **Brazil case study** states that Brazilian industry may be more vulnerable to voluntary requirements, related to inputs and PPMs than to mandatory product regulations. The study highlights the potential effects of eco-labels on exports and states that several of the product categories for which criteria are being set in the EU eco-labelling programme are of export interest to Brazil. In the case of pulp and paper, the emphasis of the EU programme on recycling could have significant trade effects since competitiveness is largely based on quick growing eucalyptus while recycling has not been well developed. In the case of textiles, producers consider that it would be difficult to comply with EU criteria limiting use of pesticides and chemicals in cotton growing. In the case of footwear, the study found that although compliance costs would be high, producers would in general be in a position to comply with most eco-labelling requirements, and market shares would not be effected.

At the time of writing there were two proposals for Brazilian eco-labelling programmes. The Brazilian Technical standards Association has proposed a 'Green Seal Programme', and the Brazilian Forest Development Society has proposed the Forest Raw Material Origin Certificate. Both these were facing difficulties over the response from domestic producers and exporters.

In the **China case study** it is stated that the country has instituted its own "green sign" system which is in effect an eco-labelling system. It has also launched a "green food" programme in which the label is awarded to products on the basis of low pollution in production and transportation, high nutrition value, and recyclability of the packaging. A life cycle method of evaluation is used for these labels, of which 384 have been awarded. The current volume of trade in these products is not given although plans for future production are high, as is the current demand for these products.

The study makes no comment on whether foreign private/voluntary schemes of environmental control have had any impact on China's exports. One can only reiterate the previous conclusion that, in general, China sees the need to meet foreign standards as beneficial to itself, as well as being feasible in terms of the technology requirements. It states, "In a word the influence of correct environmental protection laws and regulations on Chinese foreign trade growth is positive...Its negative influence is temporary." (Lu et al. (1993)).

In the **Colombia case study** exporters were asked whether they felt that they faced increased competition in foreign markets from producers who had eco-labels. 85 percent of them said they did no face such competition and only 13 percent, or 8 firms said that they had dealt with competing firms with such seals. 66 percent said they did not think that the implementation of eco-labels in international markets posed a threat to the competitiveness of their products. So far only 10 percent of firms have applied for, or already possess an international eco-label. However, 84 percent of the firms responding said they would be applying and 81 percent said they would be willing to take part in an educational programme for domestic producers.

Overall, the results do not suggest that eco-labelling is a serious issue in Colombia. This may be because the majority of Colombian exports go to the US, Central and South American markets where eco-label schemes are non-existent or weak.

In the **Costa Rica study** no specific details of the effects of external nonofficial pressures for ecologically friendly products are given. The survey of producers indicates that there is general awareness that environmental requirements will play a key role in future market access. Some sectors have been proactive in this regard, for example, in 1992 the PRO-O.K. seal started

to be used in the country. It offers certification to banana producers interested in environmentally sound production. Estimates of production costs of certified bananas are about 2% higher than non certified, however none of the participating firms have as yet reported that they receive a price premium. In the case of forestry only the largest projects are certified by the Green Cross seal, and producers feel that forest certification is costly and not profitable for smaller firms.

In **India,** the government has launched an eco-label scheme for products that, when properly used and disposed off, reduce the damage to the environment. It is essentially a promotional device and is not very popular with industry at present. The study also reports interviews which show that Indian exporters are not interested in subscribing to international eco-label schemes. They are "not confident of the promotional aspects of [such] schemes for their products. For example in garments most exporters were of the opinion that fashion was of primary concern among the buyers and environmental considerations were of secondary importance even in Germany at present." (Parikh et al (1993)). At the same time it is acknowledged that in future the situation may change, and some exporters have complained that marketing certain products to Germany is becoming more diffficult because of the "green dot" label relating to packaging of the products.

India has recognised the profitability of some environmentally products and is promoting them in a number of ways. One of the most important is organically grown cotton (or "green cotton). The federal government has identified 1000 ha. of land in two states for producing organic cotton on a pilot scale. Of course this is only a start (India has 8mn. ha. under cotton) but a promising one. The benefits are not only to the producers, who can receive a 20 percent price premium over cotton grown using chemical, but also to the country where more than half the pesticide consumption is accounted for by cotton production. India is also beginning to grow some naturally coloured cotton, thus avoiding the use of dyes. This fetches a very high premium in foreign markets. Hybrid seeds for this, however, are still being locally developed, as India was unsuccessful in obtaining them from the US where such cotton is now being grown. (Financial Times, 16.6.94).

A similar promotion is taking place for jute and its products, which lost much of their market due to competition from synthetic products but are now seen as eco-friendly. If terms can be agreed for their use in packaging to the European markets, they may have a big increase in demand. Other products where exporters recognise the benefits of environmentally friendly production are food products and leather.

The **Malaysia case study** states that mandatory and voluntary labelling requirements have had an adverse effect on exports but does not give an estimate of the size of this impact. The country has officially opposed the use of unilateral trade measures for environmental purposes including ecolabelling. However it also sees some benefits, for example, in the use of timber certification as a marketing tool for access to green markets. In the case of timber trade, the study states that as a result of responses such as: market substitution, development of higher value-added products, changes to production methods and the use of certification itself, unilateral measures (including non-official pressures) in export markets have not significantly affected exports.

The **Philippine study** makes no comment that eco-labelling schemes have had a detrimental impact on its exports. At the same time it identifies ; certain products that are being developed specifically for environmentally conscious consumers. These include hand made paper charcoal briquettes and fertiliser/pesticide free agro-products.

Although it has no specific comments about eco-labelling schemes as such, the **Poland study** is less optimistic about the possibility of taking advantage of "green consumerism" in the West but at the same time acknowledges that Poland will have to respond to such changes. It states "The market for ecological products and so called 'health foods' in EFTA ; and EU countries expands quickly but...it is subject to a very strong competition and exceptionally strict and full observed regulations. We emphasise this fact because all the programs of ecological agriculture development in Poland are to a significant extent linked to the expansion of 'green product exports." (Fiedor et al. (1994), Chapter 11 in this book). At the same time it notes that Poland does have favourable conditions for eco-agriculture because of its big rural population, low pesticide use and low labour costs. In this regard it notes that the current trend of manufacturers of some pesticides that are banned in developed countries ; to aggressively market their products in Poland is proving an obstacle to the development of eco-agriculture.

In the case of the **Thailand study** the impact of eco-labelling schemes is said to have been negligible so far in the key markets of North America, East Asia and Europe. The majority of eco-label criteria developed did not currently apply to the main export products. The study notes, however, that in the case of one eco-label developed in Germany to apply to textiles, Thai producers are not keen to comply as the market niche is not large enough to justify changes in product standards. If pressed to comply, some technologically less advanced textile producers said that they would prefer to switch to alternative

**Table 1: Price Premiums for Eco-labels Reported in the Literature**

| Country | Estimated Premium | Source |
|---|---|---|
| United Kingdom | 13% price premium for certified tropical timber products. | Haji Gazali and Simula *op. cit.* |
| United Kingdom | 33% of the surveyed public willing to pay an average of 13% premium for sustainably produced timber. | Survey carried out by MORI and WWF. Quoted in Simula, (1997) in Zarrilli *et al.*(1997). |
| United Kingdom | Cafédirect has a price premium of 5-10 % over commercial premium brands. | Bird and Hughes (1997). |
| United Kingdom | Of consumers surveyed [...] premiums suggested (for ethical products) ranged between 10 and 18 pence. | Mintel marketing Intelligence (1994), quoted in Bird and Hughes *op. cit.* |
| The Netherlands | Price for "GEA" labelled flowers is about 30% higher. | Verbruggen, H. *et al.* in Zarrilli *op. cit.* |
| Europe | Consumers willing to pay a 5-15% price premium for sustainable timber products. | Varangis, *et al.* (1993). |
| United States | 34% of consumers surveyed willing to pay a 6-10% price premium for sustainable wood | Winterhalter and Cassels, (1993). |
| United States | 68% of consumers surveyed willing to pay a 1-15% premium. | Ibid. |
| United States | 75% of consumers surveyed willing to pay 1-5% price premium | Survey by Gerstman and Meyer. Quoted in Crossley *et al.* in Zarrilli *op. cit.* |
| United States | WTP for environmental /health attributes between 5-7% | Ravensway, and Hoehn, (1991). |
| United States | 60% of consumers willing to pay 10% (or higher) premium for some 'green' products. | Quoted in Morris, (1997). |
| Canada | 64% of respondents willing to pay a 10% premium for a product bearing the Ecologo. | Hickling Corporation public survey (1993). Quoted in Guevara *et al.* in Zarrilli *op. cit.* |
| Singapore | 5 % premium for the Singapore Green Label | Jha *et al.* (1993). |

Source: Galarraga and Markandya, 2000

markets. The affitude of footwear producers to eco-labelling is said to be more positive because newer and more advanced technology is used. For example, producers stopped using PCPs because this was requested by export markets. Various "Green Labels" have been introduced in Thailand but these are still at an early stage of development and the report suggests that a national scheme based on life cycle analysis is needed to help producers to comply with developments in external eco-labelling.

Another area of difficulty for Thai producers has been packaging requirements. For example, the fisheries and frozen products processing sector has been affected by the German "Green Dot" scheme, although the compliance costs have not been prohibitive.

In **Turkey** a group of institutions together award the "friend of water and the environment" label to approved companies and products. Most companies regard this eco-label as a marketing device. They have not applied for or been granted any foreign eco-labels. Certainly foreign companies have actively been influencing production processes in Turkey, but local manufacturers are not opposed to this.

The **Zimbabwe study** does not much about non-official schemes that promote environmentally friendly products in the West. It simply notes that eco-labelling schemes will mean that local companies will have to invest in new capital and technology. With these changes they can compete anywhere but the problem is the difficulty in obtaining the necessary capital and servicing the high interest costs.

In conclusion these studies are not strongly negative about voluntary schemes that encourage more ecologically friendly products. Many producers do not feel that they have (as yet) affected their exports significantly. A number of others see such schemes are opportunities to develop new products. Only a few cases are cited where they have had a detrimental impact. One should caution, however, that these studies are based on past experience, covering a period when eco-labelling schemes were only just getting started. It may be that views about them will change as they become more widespread.

## 3.5 What impact have international treaties had on the trade of developing countries and economies in transition?

Whereas environmental standards can properly be set at the national level for impacts that are confined to national boundaries, the same does not apply when the domain of the impact is international, or even global. In the case of such issues as greenhouse gases, ozone depleting substances (ODSs) or the conservation of threatened species international agreements are essential. These mandate specific actions to address the environmental problems that arise, and specify how the responsibility for the actions is to be divided up between countries.

Frequently, the required actions include some form of restriction on trade. For ODSs the original agreement in 1985 included trade sanctions against non-members that violated the Protocol condition banning trade in ODSs between members and non-members. The latter was intended as an incentive to join the protocol and in fact proved to be an important factor in encouraging some countries to become parties. It is important to note, however, that the sanctions have never been imposed and, according to some commentators, it would be against the GATT to impose them (Sorsa (1992)). The same applies to any restrictions on trade in products not containing ODSs but made using them.

Restrictions in trade are central to other international agreements, such as the Basel convention on trade in hazardous substances, and CITES, the convention on trade in endangered species. Although such treaties can override GATT rules in so far as the contracting parties waive their GATT rights voluntarily, the same does not apply to non-members to the treaties. Also, members who were to face sanctions for not complying could take issue with the use of these instruments. There is no experience with any such cases before the GATT, which could approve the use of sanctions in connection with global environmental treaties, as long as it was treated as last resort measure, with all other options having been exhausted (Sorsa (1992)). The situation with respect to WTO, the successor to GATT, is still unclear as regards such treaties.

The studies under review indicate that these countries have suffered some loss of trade as a result of such treaties. Under the Montreal Protocol, as ODSs are being phased out, developing countries that had a large export of products using such chemicals (such as refrigerators) are finding that the markets are being squeezed, as buyers want to shift to versions based on substitutes.

In the case of **Brazil,** the study states that the value of exports to OECD countries of products that are potentially affected by ODS phase out policies represented US$ 726 million in 1989 but declined by 45% between 1989 and 1992. The decline of exports of manufactured goods as a whole to the OECD over the same period was 7%. Exports of products which may contain CFCs to developing countries increased quickly in the early 1990s. However, this was commensurate with export growth in manufacturing products as a whole. No

data are given on the overall changes in value of ODS related trade since 1989. The study also gives the example of the effect of the Basel convention on trade in scrap metal. The iron and steel industry imports scrap to make up for fluctuations in domestic supply. Although imports currently only account for a small proportion of total supply of scrap, trade restrictions could adversely effect competitiveness in the event of shortages of scrap.

The **China case study** quotes a decline in the volume of exports of refrigerators from that country of 58 percent between 1988 and 1991. Similar declines were noted for other products using ODSs. It is as a consequence of this decline that China has stepped up its phase out program and hopes to phase out ODSs faster than would be required for developing countries under the Protocol[5].

The **Colombian study** states that the impact of the Montreal Protocol on international competitiveness is not clear. While some firms will not be able to afford the new technology, there are funds available from the Parties to the Protocol to assist in the transition. If the climate change convention results in a carbon tax, or if fossil fuel consumption is restricted in international markets in other ways, Colombia will lose out, as a net exporter of oil and coal. An increase in the price of fossil fuels based on their carbon content would impact on exports of stone, glass and ceramics.

The **Costa Rica study** concludes that in general the application of Multilateral Environmental Agreements (MEAs) have been perceived as generators of new opportunities. Thus the Convention on Climate Change has been declared a priority for the national development strategy. In the case of the Montreal Protocol the study states that it is difficult to quantify the effect this has had on the trade of CFCs, although the overall cost of reconversion resulting from the Protocol required by the country is expected to be no more than $5 million. The use of CFCs and halogens in aerosols has already been prohibited, but this is thought to have had little effect on trade due to the availability of substitutes. Products requiring refrigeration, such as bananas, fruit and meat, represent an important part of the country's exports and could potentially be affected by the phase out of ODSs.

Although there are no precise estimates of the amount of trade in Costa Rica in products covered by CITES, it is known to be an insignificant percentage of total exports and therefore the commercial effect of its application is thought to be limited. The study gives the example of orchid dealers who have complained that the 5% tax on all imports and exports of wildlife species reduces their competitiveness though this is not quantified.

No data exist on the commercial effects of the Basel Convention in Costa Rica, however those consulted by the study suggested that these would be low.

The **Indian study** simply notes in passing that international agreements such as the Montreal Protocol will result in many export sectors being severely affected as a result of the need to adapt to the new standards.

The **Malaysia study** concludes that ODS phase-out as a result of the Montreal Protocol has had relatively little effect on Malaysia's trade and competitiveness. This is due to close co-operation between government and industry in the national phase-out strategy. Malaysia's production of refrigerators and air conditioners have not be greatly affected by cost increases. In spite of an accelerated CFC phase-out in developed countries, Malaysian exports of air conditioners have increased sharply as it had invested in the capacity to produce such equipment that ran without CFCs.

Malaysia acceded to the Basel Convention in 1993 but no view is given on the actual or predicted effects of this treaty on exports. The country acceded to CITES in 1977. It is concerned that some countries are using conservation measures in this treaty as disguised trade barriers.

The **Philippine study** notes that most of the successful global agreements such as the Montreal Protocol, CITES etc., affect commodities that are not significant Philippine exports. One potential treaty it is concerned is that on trade in tropical timber. Such an agreement would impact significantly on the country's exports of furniture as local sources of sustainable plantations are very few and unable to expand quickly due to the long gestation period for hardwood plantations. As far as regulating open-sea resources is concerned, the 1989 Wellington Convention for the Prohibition of Fishing with Long Driftnets and the UN Moratorium on Driftnet Fishing is poorly monitored and enforced. In fact better enforcement would benefit the Philippines as driftnet fishing is not a major method in Philippine fishing.

The **Poland study** states that it is premature to estimate the influence of most international treaties on Polish trade as of this time (its participation in most treaties dates from 1990). Nevertheless it notes that both the Montreal Protocol and the Climate change convention can be expected to have an impact on Poland's foreign trade, the first through the increased cost of the technology that replaces ODSs and the second because of the loss of export markets for fossil fuels, particularly coal.

5 The reduction in exports of refrigerators could also be due to the very rapidly growing domestic market. Nevertheless, it does appear that the loss of export opportunities had some impact.

The Basel Convention that controls the transboundary movement of hazardous wastes has major implications for Poland. On the one hand it has resulted in a major reduction in the importation of wastes into Poland, where some of them (e.g. scrap paper) are often useful inputs into production processes. On the other hand there are other products such as scrap metal where restrictions might benefit Poland, which used to lose a lot of this much need material though exports. Finally Poland is losing out from this treaty through the loss of transport business (Poland is a corridor for the shipment of materials from Russia and Ukraine).

Another case where an international treaty has had negative impacts is over fisheries. With the establishment of 200 mile exclusive economic zones, Poland was forced out of traditional North Sea fishing areas. Since then some new areas have been made available but overall the country is in danger of losing its fishery export industry.

There are also a number of regional treaties that will influence its structure of industrial and agricultural production and hence its foreign trade. The convention for the protection of the marine environment in the Baltic sea will result in increased fees for water intake and sewerage discharge, thus affecting the exports of water-intensive goods such as artificial fibres, cellulose and paper, coke, sulphuric acid etc. At the same time it will have to import a large number of water recycling systems. On the transboundary movement of air pollution, Poland, Germany and the then

**Czechoslovakia** concluded the so-called Black Triangle agreement relating to the improvement of the environment in the three countries. This has resulted in reduced power station sulphur emissions, the costs of which are reflected in higher prices and less export of electricity from Poland. Finally Poland has concluded agreements on nature protection with the Ukraine, Slovakia, Byelorussia and Russia. These combine restrictions on use of nature reserves with the development of alternative activities that could promote trade between Poland its neighbours.

In the **Thailand case study**, producers are said to have been only marginally affected so far by MEAs. As the country qualifies as an Article 5 country of the Montreal Protocol (which permits phase out by 2008 instead of a much earlier period for most developed countries) ODS phase out has been slow, with consumption of the most utilised ODSs increasing three-fold between 1986 and 1991. Therefore no negative effects on trade have yet been observed. The study warns, however, that if phase-out is undertaken at the last minute, additional compliance cost and the rising costs of CFCs could adversely effect producers.

Thailand became a signatory to the Basel Convention in 1992 and is in the process of applying regulations to the trade in hazardous wastes. Toxic waste imports have increased considerably in recent years but are predicted to reduce drastically as regulations come into force. Detailed global figures for waste trade are difficult to obtain due to widespread illegal trafficking.

Thailand's commitment to CITES has affected the major trade of orchid plants, with one whole family of orchids now prohibited from exportation. Artificially propagated species have partly replaced wild orchids in many export markets. Trade in crocodile and crocodile products have also been affected by CITES but the exact impact of such restrictions is generally hard to determine pardy due to the level of illegal trade.

The **Zimbabwe study** identifies trade benefits arising from the Bamako convention on the which restricts the import of wastes into Africa and controls the transboundary movement of wastes with in Africa. Although clearly restrictive of trade this is seen as beneficial as far as Zimbabwe is concerned, because it safeguards against serious environmental hazards. On the other hand the study sees the convention on trade in endangered species (CITES) as damaging to the country's interests. This is primarily because of the ban on trade in ivory, which hurts Zimbabwe, where elephants are not endangered and where it could gain significantly from trade in ivory were it to be permitted. In fact the revenues from such trade could provide much needed resources for managing elephant habitats. Zimbabwe's position on this issue is of course well known and a major source of disagreement within CITES.

The tentative conclusion therefore appears to be that these international conventions certainly will restrict or limit trade in some form or another.

In some cases the restrictions can be significant and countries may wish to seek some compensation or exemption for the losses suffered on that basis. These treaties are voluntarily entered into and it may be possible for groups of countries with similar impacts to negotiate suitable terms. To some extent this is what was done in the Montreal Protocol, and the same could be done in other treaties, such as the Basel Convention. In any event, it will be important to know what these trade impacts are, prior to carrying out such negotiations.

## 4 CONCLUSIONS

In spite of the fact that data have been limited and the conclusions often negative, these eleven studies have provided a lot of useful information on trade environment links. In this synthesis chapter these have been divided into those that arise as a result of trade measures

and those that arise as a result of environmental measures.

On the trade-environment link the focus has been on the impacts of the various recent attempts at trade liberalisation. Past studies have shown such measures could both harm the environment as well as to protect it. The studies reviewed here did not provide any emphatic evidence that trade liberalisation has systematically hurt the environment. There are specific situations in which environmental pressures have been reduced and others in which it has been increased. Where it has been increased, some of the case studies have identified policies that, if effected, could mitigate the damage without losing the benefits of the trade liberalisation. In fact one could argue that in all cases where negative impacts have been I identified, some policies of environmental protection, that were acceptable from a trade point of view, could have been introduced to reduce that impact.

On the environment-trade link five questions were posed. The first was whether environmental regulations in developed countries were hurting the exports of the countries under study. There is much talk of such effects but little hard evidence. These case studies looked for such evidence and found some situations where such pressures were significant. This was particularly true for the smaller countries. The larger exporting countries claimed that the effects generally have been small, and in most cases manageable for the exporters. In several cases the adoption of the stricter standards not only decreases environmental damage, it increases efficiency and profits for firms. The situations where problems have been identified are: (a) where exporters are small producers, in which case they are affected by the increasing number and variety of regulations; (b) situations where there have been a large number of changes in the regulations, where a lack of information is a serious problem; and (c) where there is a perception that the rules are unreasonably cumbersome and complex and at least partly designed to make exports uncompetitive.

The second question was whether domestic environmental regulations were hurting exports or encouraging imports. Apart from Poland, there was little evidence to suggest that they were. Most country studies concluded that the costs of environmental protection could be met without seriously affecting competitiveness. In some cases there was even a gain in competitiveness as a result of imposing such regulations. The Poland case is rather special, as it entailed a very large number of changes in energy prices, which were beneficial to the environment but undoubtedly damaging for exports, which previously had had access to very low cost raw materials and energy.

The third question was whether there was any evidence that polluting industries were migrating to countries with less strict environmental standards. Only three studies offered any comments on this but they support the general finding that such migration is not a common problem. There are special cases where it does occur, however, and where action is needed to control it.

The fourth question was whether consumer-led movements in developed countries, such as eco-labelling were having major impact on the exports of the countries being studied. In general the studies were not particularly negative about voluntary schemes that encourage more ecologically friendly products. Many producers did not feel that they had (as yet) affected their exports significantly. A number of others saw such schemes are *opportunities* to develop new products. Only a few cases were cited where they have had a detrimental impact. One should caution, however, that these studies are based on past experience, covering a period when eco-labelling schemes were only just getting started. It may be that views about them will change as they become more widespread.

The final question was about the effects of international treaties on trade. Such treaties were found to have had significant trade impacts in some cases. In other cases, countries managed to meet the conditions of the treaty and still have their exports in related fields grow. In yet other cases, there is concern that the treaties are having serious detrimental effects on trade and, indirectly on national development. This is an area where changes may be necessary, to reconcile such national interests with those of the international community. One way in which this reconciliation can be made is to compensate those who are seriously adversely affected by the measures.

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# Chapter 12

# CLIMATE CHANGE IMPACTS AND POLICY

*by*

*Alistair Hunt*

## 1 INTRODUCTION

This training material provides a broad overview of the climate change problems emphasising the assessment of greenhouse gas limitation strategies. Climate change is characterised as a global environmental problem connected to the total atmospheric concentration of greenhouse gases. The main focus of policy strategies is identified to be related to greenhouse gas emissions and to climate change impacts. Greenhouse gas limitation policies relate to the first of these areas, while adaptation policies aims at mitigating climate change damages. The establishment of global climate change policies, involves balancing cost and benefits of efforts in different parts of the world and this involves a number of specific complexities outlined in the paper. A summary of basic country study steps and analytical structure suggested by methodological guidelines for greenhouse gas limitation studies are given, and this leads up to a definition of direct and indirect cost and benefit concepts and assessment approaches to be used in national studies.

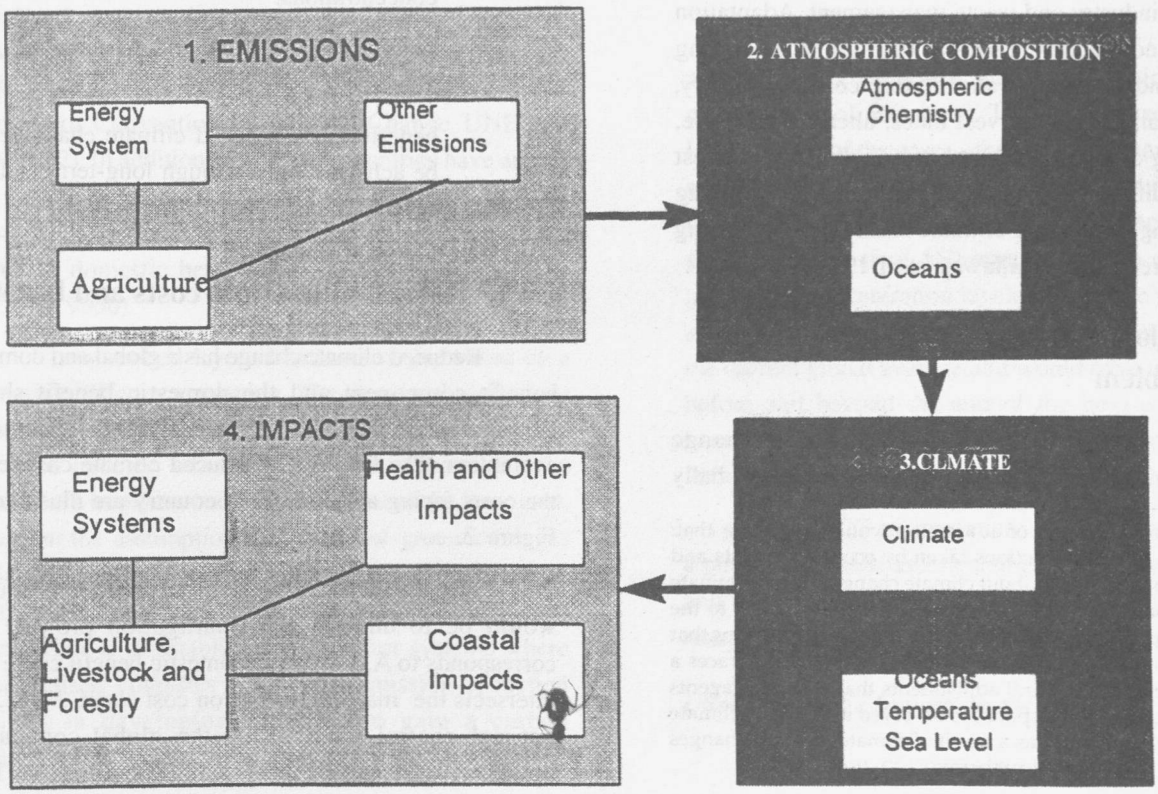
The paper is based on material developed by Halsnees et al. (1998), Markandya (1998), Sathaye and Makundi (1998) and IPCC.

### 1.1 Characterisation of the climate change problem

Climate change is a global environmental problem connected to the total atmospheric concentration of greenhouse gases. This concentration has been built up through greenhouse gas emissions in the recent centuries many as a consequence of $CO_2$ 'emissions from industrialised countries. Climate change is therefore perceived in many developing countries as primarily a responsibility of industrialised countries.

Climate change can be characterised by a system of events from emissions to impacts. One way to categorise these events is depicted in Figure 1:

**Figure 1 schematic overview of climate change and its impacts.**



Emissions of greenhouse gases result from human activities in the energy, agriculture, and forestry sectors, and from industry and waste management. Energy production from fossil fuels leads to the release of carbon dioxide and fugitive methane emissions. Forestry sector biomass decomposition and burning mostly releases carbon dioxide and methane, and agricultural systems are largely responsible for the release of methane and, to a smaller extent, nitrous oxide from paddy fields, livestock, and soil management. The emissions of carbon dioxide from human activities and natural sources increase its concentration in the atmosphere, depending on the extent to which oceans and terrestrial sinks remove the gas. These sinks do not remove other greenhouse gases, however, and their concentration depends on chemical reactions and decomposition in the atmosphere. Climate and sea level are partly determined by the atmospheric concentration of greenhouse gases and atmospheric interactions with the oceans, which act as enormous heat reservoirs. Finally, changes in climate and sea level have impacts on various human and natural systems as indicated in Figure 1, each with significant social and economic implications.

Climate change policies strategies feature prominently in categories 1 and 4, which deal with *emissions* of greenhouse gases and with the *impacts* on natural and human-made systems respectively. Policies either can relate to category 1 Emissions and try to limit these, or can relate to category 4 Impacts and try to reduce these by adaptation strategies. Greenhouse gas emission limitation policies apply to all the above mentioned sources in the energy sector, forestry, agriculture, industry and waste management. Adaptation policies include such measures as building or modifying sea walls and reservoirs; abandoning coastal property, adjusting planting and harvest dates, altering input use, or switching crops/species in the agricultural and forest sectors, modifying reservoir operating rules, constructing early warning systems for climatic hazards, and adjusting insurance premiums[1] (Callaway et al 1998).

## 1.2 The global climate change policy problem

The establishment of global climate change policies aim at establishing a scheme that is globally cost effective and includes a "fair" burden sharing. The policy scheme has to encourage collaboration between very unequal parties with regard to:

- wealth
- past and present GHG emissions
- mitigation costs
- damages of climate change
- capability to implement climate change policies

The establishment of the policy is furthermore complicated by the large uncertainties and long term character of the climate change problem. A summary of the specific complexities of the global climate change policy problem is given in Box 1.

**Box 1 Characterisation of the climate change policy problem**

The environmental control problem is related to a public good:

- non-rivalry of consumption
- benefits of control are non-excludable.

The control problem becomes complex due to the specific character of climate change:

- impacts are related to total accumulated GHG concentrations over the next century regardless of the location of the emission source.
- long time lags - up to 50 years - between emission reductions and atmospheric concentrations.
- large uncertainties related to the extent of climate change and the nature of its effects.
- benefits from reduced climate change can be achieved only through long-term global collaboration.

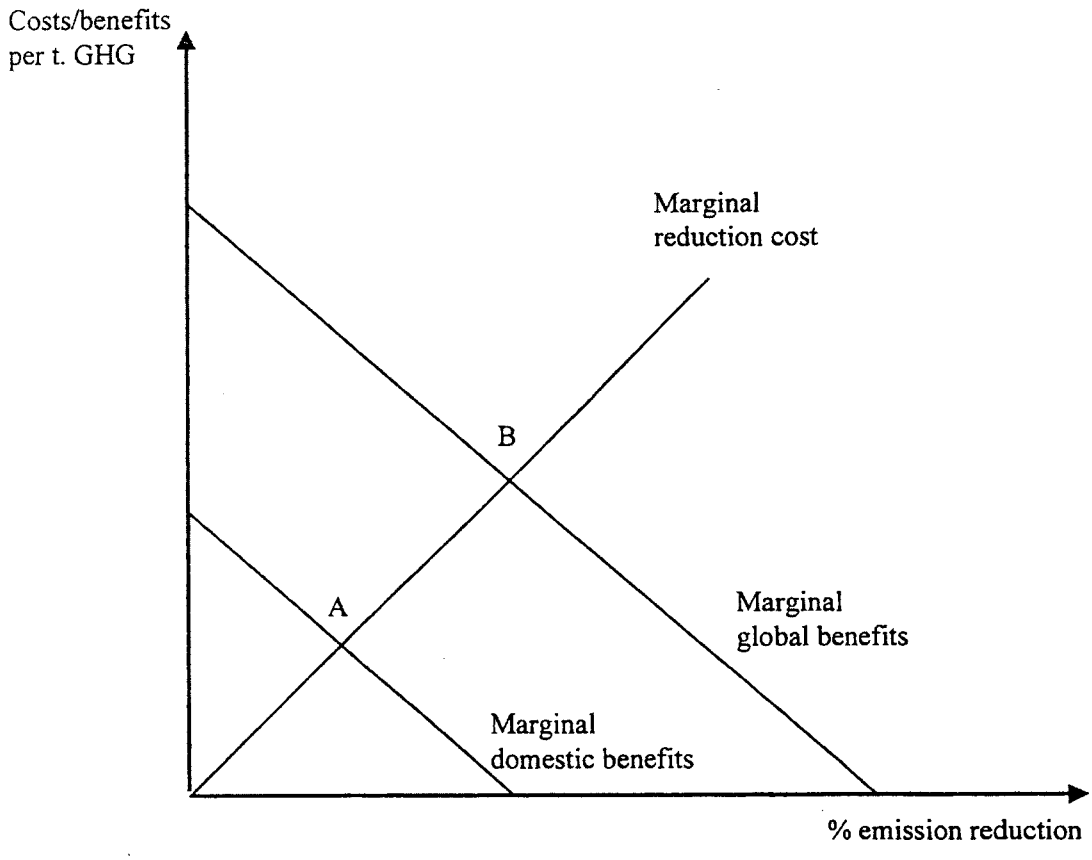
### 1.2.1 Global and national costs and benefits

Reduced climate change has a global and domestic benefit component and the domestic benefit should therefore in principle be included in the national project evaluation. The benefits of reduced climate change and the costs facing and individual country are illustrated in Figure 2.

The rational action for the individual country would be to undertake the mitigation projects that corresponds to A, where the domestic benefit curve $MB_i$ intersects the marginal reduction cost curve, MAC. The optimal control point facing the global community, however, is B, where the global benefit curve $MB_G$

1 An economic definition of adaptation would be stating that: adaptation consists of actions taken by economic agents and government to (1) learn about climate change and disseminate this information, and (2) reallocate resources to adjust to the impacts of climah change. This definition includes actions that take the form of projects and policies, but it also embraces a wide range of behavioural adjustments that economic agents undertake directly, in response to observed or expected climate impacts, and indirectly, as a result of climate-induced changes in relative input and output prices.

**Figure 2 Marginal reduction cost and benefit curves.**




intersects the MAC. The global community therefore has an interest in supporting project implementation in the area from A to B. A financial mechanism to support such project implementation is included in the United Nations Framework Convention on Climate Change UNFCCC (UN, 1992). In addition, developing countries have argued on an ethical basis argued that they should be compensated for all project implementation costs including domestic benefits of avoided climate change (IPCC, 1 996b).

The so-called Kyoto Mechanisms are based on a somewhat different reasoning. The focus primarily here is on global cost-effectiveness issues in greenhouse gas limitation policies, which by definition requires that the policies are implemented where they have the least cost. Based on the assumption that marginal greenhouse gas limitation costs are smaller in developing countries than in industrialised countries, the aim of the Kyoto mechanisms is to establish carbon offset systems where industrialised countries implement emission reduction projects in developing countries to gain a carbon credit.

## 1.2.2 Climate Change Damages

It is impossible to prevent further increases in atmospheric greenhouse gas concentrations from the present level of 360 ppmv. This is a consequence of the long lifetime of the gases - the most important greenhouse gas $CO_2$ has for example a lifetime of over 100 years. If the atmospheric concentration is to remain below a stabilisation level of 550 ppmv (equal to a doubling of prehistoric concentration levels), the future global annual average emissions cannot, during the next century, exceed the current global average and would have to much lower before and beyond the end of the next century. It has been assessed by IPCC that stabilisation at the ' current 360 ppmv level will be impossible (IPCC 1996a).

The IPCC at present are tending towards more of a physically-based assessment of relative vulnerabilities and a world picture on relative vulnerabilities that has a reasonable consensus is presented below. (See table 1)

***Climate Change Impacts in Sri Lanka***

The main climate change impacts in Sri Lanka

**Table 1 The Relative Vulnerability of Different Sectors and Regions**

| | Africa | Temp. Asia | Trop Asia | Austral | Europe | N. Americ | S. Americ | Polar Reg | SIS |
|---|---|---|---|---|---|---|---|---|---|
| Ecosystems | LOW | HIGH | AV | LOW | AV | HIGH | LOW | HIGH | AV |
| Food/ Fibre | HIGH | LOW | HIGH | AV | LOW | LOW | AV | AV | LOW |
| Water resources | HIGH | AV | HIGH | AV | AV | AV | AV | LOW | HIGH |
| Coastl Zone | AV | LOW | HIGH | AV | AV | AV | AV | LOW | HIGH |
| Human Health | HIGH | AV | HIGH | LOW | LOW | LOW | HIGH | AV | HIGH |
| Ranking | HIGH | AV | HIGH | LOW | LOW | AV | AV | AV | HIGH |

**Table 2: Economic Costs of Agricultural Change in Sri Lanka**

| Sub seceor | Change in production | Price Rs./kg | Value Rs. Mn | |
|---|---|---|---|---|
| Rice | 2538 mt | 20.9 | -1591.0 1 | -2651.0 |
| Tea | 223 mt | 65.72 | 0.0 | +153.0 |
| Rubber | 114 mt | 22.93 | -52.3 | -104.6 |
| Coconut | 2523 mt | 3.63 | -366.3 | -549.5 |
| Coarse Grain<br>crops | 8.6 mt<br>7.2 mt | 15.0<br>15.3 | -132.0<br>. | -165.0<br>. |
| Vegetables | 20.5 mt | 10.0 | -205.0 | -287.0 |
| Overall agriculture | 76488 Rs. Mn. | - | -2606.7 | -3909.0 |
| Overall economy | 283,794 Rs. Mn. | -<br>, | -3728.3 | -4971.1 |

***Source:*** *ADB. 199 4. Climate Change in Asia: Sri Lanka Country Report, p85 Manila: Asian Development Bank*

are likely to be in the agricultural and forestry sectors, in water resources and as a consequence of changes in sea level. These impacts are described briefly below.

Impacts on the agricultural sector

An increase in the amplitude of rainfall extremes will lead to more frequent floods which in turn will result in increased soil erosion, landslides and damage to agriculture through flooding of low-lying areas. Temperature and moisture changes will also force farmers to change the existing cropping patterns. The ADB assessment (ADB, 1994) provided an economic assessment of the likely impacts of climate change on Sri Lankan agriculture in 2010 and this is given in Table 2.

Impacts on the forestry sector

The boundaries of existing forests are predicted to change significantly by 2070 and this will have impacts on timber production, biological diversity, recreational opportunities and watersheds. The hydrological cycles influenced by forests would also be affected. Forest fires may also increase and the rise in temperature could increase the insect population and disease incidence that damage forests. The present loss of mangroves could be aggravated by climate impacts such as increase in rainfall due to increased intensity of monsoon circulation and increased flooding from storm surges.

Impacts on the water resources

Climate change would change the timing, extent, and distribution of precipitation and run-off which is the source of renewable water supply. However, existing models do not adequately simulate how hydrology and water supplies would be affected. It is certain, however, that climate change will increase the overall uncertainties associated with water management and supply.

Impacts due to the changes in sea level

The ADB country study focused on the coastal

region extending from Waikkal to Galle since it is one of the most populated and developed coastal regions in the country. The projected sea level rise is by about 30cm by 2010 and a further 10cm by 2070. The erosion already evident on many of the country's beaches is likely to significantly increase with accelerated sea level rise. Since the tidal range is low widespread destruction of coastal wetlands can also be expected. Sea level rise would, also have advrse effects on infrastructure facilities such as ports, harbours and coast protection structures. The costs are summarised in Table 3.

**Table 3 Projected costs of Sea Level rise in Sri Lanka**

| Economic Activity | Estimated cost/Rs. Mn. | |
|---|---|---|
| | Capital | Income |
| Tourism | 186.00 | 321.00 |
| Land loss | 1,242.00 | |
| Organised industries | | 572.00 |
| Small industries | | 6.00 |
| Rice | | 64.50 |
| Coconut | | 83.00 |
| Housing | 62.00 | |
| Assets of fishermen | 18.00 | |
| Roads | 42.88 | |
| Mangroves and wetlands | | 226.78 |
| Total | 1550.88 | 1272.83 |

*Source: ADB. 1994. Climate Change in Asia: Sri Lanka Country Report, p82. Manila: Asian Development Bank*

However, a recent attempt (Tol et. al., 2000) has been made to derive a monetary valuation of global climate change damages, using the FUND2.0 integrated assessment model. Below are given key results together with an outline of how the principal methodological issues have been dealt with.

## 1.2.3 Conventions for Valuation and Reporting

### *Aggregation*

Aggregation of the impacts of climate change valued at national or regional prices may be objectionable because, for example, a life lost in Bangladesh counts less than a life lost in Germany. Altering values, however, would undermine the theoretical foundations of monetisation and yield inconsistencies between national and international environmental policies. Therefore, the aggregation of monetised impacts is changed, using so-called equity weights. Four alternatives are reported here:

EU impacts only, with EU values ('EU only');
1, plus impacts in other regions with local values ('regional values');
1, plus impacts in other regions with globally averaged values ('world average');
1, plus impacts in other regions with EU values ('EU values').

The alternatives are ordered in terms of estimated severity of the marginal cost estimates. Alternative 1 ignores impacts outside the EU. Alternative 2 is based on the expressed willingness-to-pay of people outside of the EU, but suffers from the welfare-economic inconsistencies induced by aggregating money measures over people with very different incomes. Alternative 3 argues from the perspective of a benevolent world leader. Alternative 4 argues from the moral perspective of a European decision maker with a more expansive world-view. We reject alternative 2 as internally inconsistent (but do report the outcomes), and use alternative 3 - in between alternatives 1 and 4 - for our central estimates below.

Table 4 presents the results of FUND2.0 if mortality is valued not on the basis of a 'value of a statistical life' (200 times per capita income) but rather at a 'value of a life year lost' or VLYL, (10 times per capita income, per year). Cardiovascular and respiratory diseases, for which relatively little life-time is lost, dominate in the rich regions, so VLYL methodology leads to substantially lower marginal costs than does the VOSL methodology. The gap closes if more weight is placed on developing countries, where vector-borne diseases, for which more lifetime is lost, dominate. Note that the values are highly sensitive to the discount rate, going from a rate of 1% to 3%reduces costs by a factor of more than 2. Higher rates than 3% would result in very low damages.

Despite their short lifetime, the marginal costs of nitrogen and sulphur ernissions are relatively large, primarily because they are not much discounted. Most of the world damages arise from sea level rise, followed by extreme weather and malaria. In terms of regions the EU receives only 0.5% of total damages. The biggest damages are in South and South East Asia, Africa and Latin America.

**Table 4 Marginal Cost Results**

| | Minimum[b] | Low[c] | Central Estimate[d] | High[c] | Maximum[b] |
|---|---|---|---|---|---|
| C ($/Ton) | 0.30 | 5.2 | 8.9 | 15.2 | 60.1 |
| N2O ($/Ton) | 24.3 | 440.2 | 748.3 | 1272.1 | 5242.1 |
| CH4($/Ton) | 1.9 | 28.2 | 44.9 | 71.5 | 257.0 |
| N ($/Kg) | -5.5 | 198.2 | 337.0 | 527.9 | 1270.2 |
| S ($/Kg) | -35.8 | -16.6 | -9.8 | -5.8 | 0.0 |

Notes:

[a] Emissions are in the period 2000-2009. Costs are discounted to 2000.

[b] Minima and Maxima given by different discount rate/aggregation combinations

[c] High and low approximately span the 67% confidence interval.

[d] Private Rate of Time Preference equals 1% here. Values are world averages

[e] Model is FUND2.0. Time horizon is 2100. Scenario is IS92a. Mortality risks are valued based on the value of a life year lost. Note that the marginal costs of carbon dioxide are here expressed as per tonne of CO2, rather than per tonne of carbon.

The results presented here should not be taken as final estimates. The impacts covered by the models used are only a fraction (of unknown size) of all climate change impacts. Particularly, large-scale disruptions, such as a breakdown of North Atlantic Deep Water formation or a collapse of the West-Antarctic Ice Sheet, are excluded from the analysis. The methodologies to estimate climate change impacts in a different future remain weak. Adaptation is not included in its full complexity. Valuation of impacts is still troublesome, particularly for nature and health. Our knowledge of atmospheric chemistry and climate has substantial gaps. The estimates reflect our current best knowledge. We should also note, in spite of the enormous effort attached to making these monetary estimates, they do not have wide acceptance in the IPCC community. WGII of the the 3rd IPCC Assessment and has not gone down the road of making a monetary estimate of damages.

In conclusion it can be said that climate change primarily is a consequence of industrialised countries past and present emissions, but damages primarily will occur in DC's. Following that, the costs of greenhouse gas limitation policies will be high in industrialised countries while the benefits of control will be relatively small in this region.

## 1.2.4 Adaptation

An adaptation is an adjustment in the unit of analysis. While adaptations occur in both natural and human systems, there are some fundamental distinctions. In unmanaged natural systems adaptation is autonomous and reactive, and is the means by which species and communities respond to changed conditions. Human system adaptation can be motivated by private or public interest (i.e. who adapts?) Private decisionmakers include individuals, households, businesses and corporations, while public interests are served by governments at all levels. This distinction is fundamental to initiatives by public agencies to promote adaptations, the roles of public and private participants are distinct. Table 5 below shows examples of types of adaptation differentiated according to timing, natural or human systems, and public or private decision makers. The important role of adaptation policies is being increasingly recognised, and why the search for more information about the costs and benefits of such policies is accelerating. This is especially so because of growing evidence that damage due to climate change, for example in agriculture, is highly sensitive to the availability of low-cost adaptation measures.

Potential adaptation strategies can be analysed in terms of three key questions: a) what effects to adapt to, b) how to adapt, and c) when to adapt. The development of impact analysis reported in the previous section is the starting point to answer the first question. To answer the second question, an increased research effort has been put into the identification of adaptation options. It should be noted that little research to date has taken this subject further by looking at the dynamics of adaptation in human systems, the processes of adaptation decision-making, the conditions which stimulate or constrain adaptation, and the role of non-climatic effects. This is commented on briefly below. The response to the third question is helped by cost-benefit analysis which can guidance on optimal timing issues, as well as a broader appreciation of likely growth in adaptive capacity over time.

There are serious limitations in existing evaluations of adaptation options. Economic benefits and cost are important, though not the only, criteria. In any

**Table 5 Types of Adaptation to Climate Change with Examples**

| | ANTICIPATORY | REACTIVE |
|---|---|---|
| **NATURAL SYSTEMS** | | changes in length of season.<br>changes in ecosystem composition<br>wetland migration |
| **HUMAN SYSTEMS (Private)** | purchase of insurance<br>construction of houses on stilts<br>redesign of oil-rigs | changes in farm practices<br>changes in insurance premiums<br>purchase of air-conditioning |
| **HUMAN SYSTEMS (Public)** | early warning systems<br>new building codes, design standards<br>incentives for relocation | compensatory payments, subsidies<br>enforcement of building codes<br>beach nourishment |

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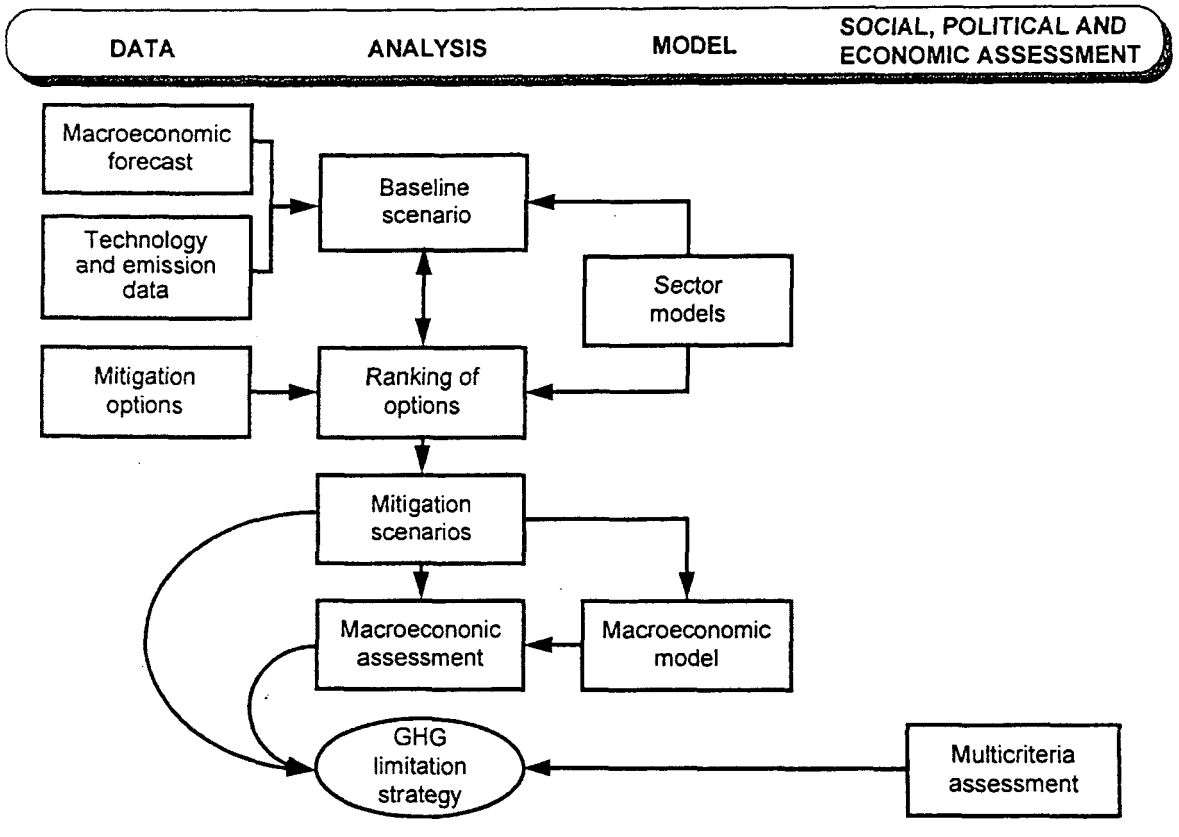
**Figure 3 Common analytical structure of the country studies.**



case appropriate methodologies are only now being properly developed. There is also little research to date on the roles and responsibilities in adaptation of individuals, communities, corporations, private and public institutions, governments and international institutions.

Given the scope and variety of specific adaptation options across sectors, individuals, communities and locations, and given the variety of participants - both private and public - involved in most adaptation initiatives, it is probably infeasible to systematically evaluate lists of particular adaptation measures. Improving and applying knowledge on the constraints and opportunities for enhancing adaptive capacity is necessary for reducing the vulnerabilities.

### 1.2.5 Basic country study approach for greenhouse gas limitation studies

Methodological guidelines for national greenhouse gas limitation studies have been developed by the UNEP Collaborating Centre on Energy and Environment (Halsnaes et al. 1998). The guidelines recommend studies to follow a co~runon analytical structure and comrnon steps. This analytical structure is shown in figure 3.

The common steps in this analytical structure are:

1. **Comprehensive evaluation of national social and economic development framework for climate change mitigation.**

- Comprehensive description of national framework for CC mitigation including: base year statistics on GDP structure, social conditions, energy balance, aggregate GHG inventory, major land use activities, population.
- Evaluation of main national economic and social national development trends and the GHG emissions that are expected to occur as a result of economic development.
- Overview of other climate change studies including impact, adaptation, inventory and mitigation studies.

**2. Baseline scenario projection**

- 10-15 year baseline scenario projection for CO2 emissions from energy consumption and land use activities.
- 30-40 year baseline evaluation of main development trends.

**3. Mitigation scenario(s) projection(s)**

- Identification of mitigation options related to the most important future sources and sinks sectors.
- Assessment of reduction potential and cost of mitigation scenarios.
- Integration of GHG reductions and costs across measures and sectors, through construction of GHG mitigation marginal cost curves.

**4. Macroeconomic assessment**

- Qualitative description of main macroeconomic impacts of national climate change mitigation strategies.
- Assessment of key macroeconomic parameters.

**5. Implementation issues**

- Identification of main implementation requirements including: financial support, technologies, institutional capacity building, regulation policies and further improvements of the national decision framework.

These country study steps can be conducted at many different levels of sophistication ranging from a broad description of main development trends and statistics to a formalised modelling at sector and macroeconomic level.

### 1.2.6 Cost concepts in relation to GHG mitigation Introduction

Actions taken to abate GHG emissions or to increase the size of carbon sinks will generally divert resources from other alternative uses. The purpose of the current methodological framework is to estimate the value of the resources which society must give up when an action is taken to mitigate the effects of climate change, as opposed to not taking that action. These resources are measured in relation to a "no action" reference, or "base case" which represent a scenario in which the economy follows its normal development path, without any policies to mitigate the effects of climate change. In this way the cost concept applied is *incremental costs.*

The cost assessment should ideally consider all changes in resources demanded and supplied by a given mitigation project or strategy in relation to a specific non-policy case. The assessment as far as possible should include all resource components and implementation costs. In many cases some of the cost components will be negative, meaning that there are benefits to be gained by undertaking a mitigation action. Both the benefits and the costs of a mitigation action should be included in the estimation. In some cases, the sum of all the benefits and costs associated with a mitigation action will be negative, meaning that society benefits from undertaking the mitigation action.

**Box 2. Cost assessment approaches.**

*Cost benefit analysis*

The basic idea is to measure all negative and positive project impacts in the form of monetary costs and benefits. Market prices are used as the basic valuation as long as markets can be assumed to reflect "real" resource scarcities. It is in other cases recommended using shadow prices. Shadow prices are meant to reflect prices that would occur in a "perfect" market.

*Cost effectiveness analysis*[2]

A special sort of cost benefit analysis where all costs of a portfolio of projects are assessed in relation to a policy goal. The policy goal in this case represents the benefits of the projects and all other impacts are measured as positive or negative costs (negative costs, with the exception of the benefits of the policy goal, will correspond to benefits of the policy). The policy goal can for example be a specified goal of emission reductions for GHG's. The result of the analysis can then be expressed as the costs ($/ton) of GHG emissions reduction.

*Multiattribute analysis*

The basic idea of the multiattribute analysis is to define a framework fo integrating different decision parameters and values in a quantitative analysis without assigning monetary values to all parameters. Examples of parameters that can be controversial and very difficult to measure in monetary values are human health impacts, equity, and irreversible environmental damages.

Mitigation projects will have a large number of different impacts including economic impacts, and social and environmental impacts. Some of these impacts are traditionally measured as economic impacts while others, for example environmental impacts, in some cases are measured as damage costs and in other cases measured as physical impacts. Monetary assessments, following that, will be used together with physical impact measures in a comprehensive decision making framework.

The assessment of economic impacts should be done in monetary units where prices are used to value the different impacts. The prices can for example be market prices or monetary estimates of environmental externalities. Physical environmental impacts can be measured in a large number of different units - these impacts will by nature not be directly comparable with the monetary impact assessments.

The aim of this section is to define the main economic impacts to be assessed in the formalised cost assessment. These impacts should, as already been said, be supplemented with the assessment of other important decision parameters. The section defines the key concepts to be assessed starting with definition of cost concepts, expanding with broader impacts like environmental externalities and sustainability indicators.

The current framework focuses on the assessment of direct and indirect costs measured at project or sector level.

## 1.2.7 Cost assessment approach

The cost concepts are defined on the basis of traditional cost-benefit analysis as applied in international guidelines for project assessment. The aim of the cost-benefit analysis is to measure the project impacts in comparable units. The term costs is used here to denote negative impacts while benefits denote positive impacts. Benefits, following that, can also sometimes be denoted as negative costs.

It is important to note that social cost-benefit analysis is not a technique, but an approach that provides a rational framework for project choice on the basis of specified national objectives and values. The aim is in this way to integrate the national cost assessment in a broader national decision framework for climate change mitigation.

Social cost-benefit analysis can be carried out in different ways depending on the assumptions applied to the impacts considered. A further development of the cost benefit analysis is the multiattribute-analysis where monetized costs and benefits as well as other quantitative impacts are considered in an integrated objective function. Box 2 gives an overview of approaches.

Climate change mitigation assessment involves a comparison of broad range of costs, benefits, environmental, social impacts and the benefits of reduced climate change. The assessment of benefits of reduced climate change is by nature difficult and uncertain. It is therefore suggested not to include monetary climate change damage estimates but to focus on GHG emission reduction targets (see a more detailed discussion on the issues in the section Measuring the benefits of reduced climate change). Following that it is recommended to conduct a cost-effectiveness analysis (defined in Box 2) where the costs of meeting alternative emission reduction targets are assessed. The current cost assessment framework structures how such a cost effectiveness analysis can be conducted.

2 The term cost-effectiveness analysis is sometimes used in more narrow way, where only the financial costs and no indirect positive and negative costs - of a private agent in meeting a specific policy goal is considered.

## 1.2.8 Outline of the main cost concepts

One can distinguish the assessment of social- and private costs and benefits. Social costs reflects all costs to society including private costs and externalities, while private costs include only the costs faced by the private sector. Another often used concept is financial costs which measures expenditures, or outlays, of money seen from the perspective of an implementing entity.

The current methodological framework has been defined with a main focus on social costs.

In this section, a discussion of the full economic cost concept is provided. This is then employed to show the kind of adjustments that need to be made to private costs, in order to derive the economic costs.

Economic opportunity costs or economic costs

The key idea behind an economic cost of something (call it X) is the value of the scarce resources that have been used in producing X. That in turn is measured in terms of the value of the next best thing which could have been produced with the same resources and is called **economic opportunity costs.** This notion of cost may differ from a narrow notion of cost. For example, take the case of sequestering carbon by growing trees on a tract of public land. In estimating the costs of such a programme, what should be taken as the cost of the land? In some cases no "cost" is attached, because the land is not rented out and no money actually flows from the project implementers to the owner (the State in this case). This, however, is ignoring the opportunity costs. The cost of the land is to be measured in terms of the value of the output that would have been received from that land had it not been used for forestry. Such values may be direct (e.g. agricultural output), and/or indirect (e.g. recreational use).

Often a resource is used and there is a financial flow associated with it. Working with the same example, the government may have leased the land to a farmer, who keeps livestock on it. If it is used for forestry the government often does not demand payment from the forestry authority. In that situation the "opportunity cost" might be interpreted as a loss of revenue to the government. Although that is an opportunity cost to the government it is incorrect to take it as the *economic opportunity costs.* The reason is that the price of the original lease may not be equal to the opportunity cost of that land. Even assuming that the highest value use is livestock, the value of the land is the net income from livestock grazing, after deducting all expenses. Frequently the leases are for much less than that, so the opportunity cost is not equal to the financial flow to the government[3].

The key points of note with regard to opportunity cost are the following:

- there may be an *economic opportunity cost* to the use of a resource even if there are no financial flows associated with that use;
- if there are any financial flows, the *economic opportunity costs* rnay or rnay not be equal to the value of those flows.

In designing mitigation cost strategies the objective is to minimise the *economic opportunity costs* of the prograrnme. *Economic opportunity cost* is sometimes called just the economic cost and is closely related to *social cost* and are in this context used interchangeably. It is also related to the concept of *shadow price*, both of which are discussed below. For a more complete discussion of these concepts see Markandya, Halsnaes & Milborrow (1998).

External cost, private costs and social costs

The term *external* cost is used to define the costs arising from any human activity that are not accounted for in the market system. For example, emissions of particulates from a power station affect the health of people in the vicinity but there is no market for such impacts. Hence, such a phenomenon is referred to as an *externality,* and the costs it imposes are referred to as the *external costs.* These external costs are distinct from the costs that the emitters of the particulates do take into account when determining their outputs (e.g. prices of fuel, labour, transportation and energy). Categories of costs influencing an individual's decision-making are referred to as *private costs.* The total costs to society is made up of both the *external costs* and the *private costs* and together they are defined as *social cost.*

*Social Cost = External Cost + Private Cost*

Estimation of mitigation costs necessitates working with social costs[4]. Often, however, the data will only provide information on the private costs. In these situations a correction has to be made for the missing costs.

Shadow prices

The above discussion concluded that the proper

3 It can be shown that, under competitive markets with no taxes, the market price-based costs will be equal to the economic opportunity costs. This is important in the estimation of opportunity costs from market data.

4 Where the pricing of commercial goods is such that it includes both the private cost and the external cost (i.e. it is based social cost) it is referred to as full cost pricing.

cost to consider in GHG projects is one based on *economic opportunity cost*. As noted above, where markets operate competitively and efficiently, the prices will reflect the opportunity costs and can be used to estimate the correct costs. In many instances, however, this will not be the case, and some correction will need to be made. The corrected market price, which should be equal to the *economic opportunity cost* of the resource, is called the *shadow price*. One important case of this is related to the change in status from "unemployed" to "employed". A low shadow price can be assigned to unemployed while the market salary rate can be used for employed. The true opportunity cost of employment is then the difference between the two. For example, if a project uses labour that is paid a wage of $20 a day but the benefits of employment are $8, then the true *economic opportunity cost* of that labour is only $12, and the *shadow price* of labour is $12.

Adjustments to market prices to obtain shadow prices will be needed when:

- there are distortionary taxes and subsidies, so market prices deviate from economic opportunity costs;
- there are monopolies and other market imperfections making the market price higher or lower than the shadow price.

The simplest way to correct for such distortions, *where the resources are tradable*, is to take the international prices of the resources. Assuming well functioning markets, these prices are seen to be "optimal". If a good is exported, for example, the export price can be taken, or where it is imported the import price can be taken. These prices should then be corrected for taxes and subsidies (i.e. the former should be deducted and the latter added).

Where the good is not traded, the shadow price should be calculated on the basis of the cost of producing the good with the inputs being valued at their economic opportunity cost. A method for doing this has been developed by Little & Mirrlees (1969), Ray (1984), and Squire & van der Tak (1975), and subsequently used by several researchers to estimate shadow prices in a number of developing countries.[5]

When applying this framework to a project (e.g. wind powered irrigation for increased agricultural yields) three important shadow prices are typically required. They are the prices of:

- Capital
- Labour
- Foreign exchange

For these, detailed analysis of the relevant sectors are required. The prices of labour are dealt with in more detail in the report The Indirect Costs and Benefits of Greenhouse Gas Limitation (Markandya, 1998). For capital and foreign exchange, the analyst carrying out a GHG estimation is advised to obtain the relevant values from economists who have worked on the sectors concerned. The World Bank and other bodies involved with Global Environment Facility (GEF) projects appraise projects in : most developing countries and would have a set of values that are used, these can presumably be accessed from their databases (as can the shadow prices for many inputs and outputs). For example, in the case of India, the Institute of Economic Growth in New Delhi has recommended a coefficient value of 1.4 for capital, materials and equipment but a value of one for foreign exchange. The value of 1.4 indicates that capital is 40 percent more scarce than its market price would suggest, so that when estimating the costs of the project the capital value should be increased by that amount. A value of one for foreign exchange implies that the exchange rate is in market equilibrium and there is no need to make any other adjustments in going from domestic to foreign prices and vice-versa.

### 1.2.9 Cost assessment levels

The costs can be assessed at project, sector and macroeconomic level. The assessment at *project level* considers an individual project assuming that this project is an isolated implementation without affecting any other part of the economy. The project assessment also assumes that the project is small and marginal implying that the ; implementation creates no changes in factor prices or final product prices. The assessment at *sector level* considers a case where a number of mitigation projects are implemented in one specific sector. Technical interdependencies between projects in that sector and impacts on production inputs and final products of that sector are to be included, but the macroeconomic development and other economic sectors are assumed exogenous. The *macroeconomic assessment*, finally, considers the full socioeconomic impacts of implementing mitigation strategies in one or more sectors, and the interaction of the different sectors and the economy.

The economic impacts of implementing a climate change mitigation project will imply that the society gives up alternative resource use and final consumption. The "value" of these sacrifices depends on the supply and demand in the related factor and final goods markets. A

5 The difference between using international or national based prices are often referred to as first and second best shadow pricing corresponding to a first and second best situation.

The benefits of reduced climate change are as already said very difficult to assess and a more operational approach is therefore to perform a cost-effectiveness analysis where the costs and benefits of project implementation are ranked according to a common metric representing the benefits of reduced climate impact. It will, in the following, be outlined how GHG emission reduction goals can be derived as a simplified case of a climate change damage function.

Sathaye, Norgaard & Makundi (1993) suggest the shadow value of reduced climate change (*VC*) to be determined as:

$$VC = \int_0^{\infty} P_C(t) e^{-rt} C_0 e^{-at} dt$$

where $P_c(t)$ is the shadow price of avoided carbon at time *t*, $C_o$ is a single year emission reduction, *r* is the discount rate and, *a* is the atmospheric decay rate of carbon.

The shadow price of avoided carbon $P_C$ (t) represent the damage function of climate change which due to large uncertainties cannot be estimated at present. It is therefore only possible to discuss the theoretical implications of different assumptions about this damage function.

If it is assumed that the emission reductions of a project, $C_0$ are small compared with the total atmospheric carbon stock, then $P_c(t)$ is constant for the project. The shadow price of avoided carbon $P_C(t)$ will probably vary over time as a function of the stock of atmospheric carbon. It is however difficult to determine that relation, and $P_c(t)$ will therefore for simplicity be assumed constant, $P_c$. Given this assumption and assuming constant *r* and *a*, *VC* can be written as:

$$VC = P_c \frac{C_o}{r+a} \quad \textbf{Eq. 1}$$

A project is worth pursuing, if *VC* is equal to or greater than the costs of implementing the project. A cost-effectiveness analysis can, following that, rank the projects on the criteria of *VC* versus costs. With the simplified assumption of a constant $P_c$, the ranking criteria then is reduced to a constant factor that converts future carbon emission reductions to a net present value.

It should be noted that projects will typically imply carbon reductions in multiple years. The total net present value of carbon emission reductions achieved in the total lifetime ofthe project *(VCC)* can for discrete time steps be calculated as: ;

$$VCC = \frac{P_c}{r+a} \sum_{t=0}^{T} \frac{C_t}{(1+r)^t} \quad \textbf{Eq. 2}$$

where $C_t$ is the carbon emissions at the time t, and T is the time period where emission reductions occur.

## 1.2.12 A framework for integrating quantitative and qualitative impact assessments

The two previous sections discussed the application of cost concepts to traditional costs and benefits and to a wider range of impacts that should be included in either the cost assessment or an over all evaluation of a mitigation action. In many cases, there will be important impacts from a project that either can not be valued in the cost assessment, due to lack of data, or should not be included in the cost assessment for sound economic accounting reasons. In the final analysis, it is important to integrate all of the cost and non-cost elements into an over all framework that can be used to ; asses all of the impacts of a mitigation action.

The information collected on the impacts of a GHG limitation project or programme needs to be summarised so that different projects and programmes can be compared.

There are three kinds of information to be summarised. These are:

1. Quantitative information in money terms
2. Quantitative information in physical units
3. Qualitative information

The same impacts can, of course, be classified in all three categories, so that data on ; reductions in fossil fuel emissions can be quantified in money terms, reported in terms of tonnes of pollutants, and in terms of quantitative impacts on eco-systems etc. In preparing summary indicators it is important not to count the same information twice, so that it is unduly weighted in the final selection criteria.

## 1.2.13 Quantitative monetary data on the project

***The cost effectiveness criterion***

For programmes that estimate the cost of achieving a certain reduction in GHGs the main criterion is normally cost of GHG removed.

The cost-effectiveness criterion involves a comparison of final cost flows and GHG emission reduction occurring at different points in time. The cost flows can be compared in a net present value, $NPV_c$.

$$NPV_c = \sum_{t=0}^{T} \frac{C_t}{(1+i)^t} \quad \textit{Eq. 3}$$

where i is the interest rate and $C_t$ is the cost at time, *t*.

The GHG emission reductions occur at different points in time in the same ways as the costs. Therefore the time specific value of these reductions have a major implication for the calculated emission reduction costs. There is a high uncertainty about climate change damages and it is therefore diffficult to assign a time specific value to emission reductions. It is therefore suggested to use a simplified approach where the GHG reductions are discounted with the same discount rate as used in the above specified $NPV_C$ formula. The net present value of emissions reduction ($NPV_e$) can then be calculated as:

$$NPV_E = \sum_{t=0}^{T} \frac{E_t}{(1+i)^t} \quad \textit{Eq. 4}$$

The costs can also be represented as levelised costs, where the annual costs as well as GHG emission reductions - are transformed to constant annual flows over the lifetime of the investment. Mitigation projects that imply constant annual emission reductions can be directly compared with levelised cost at a given point in time. The total levelised cost, $C_o$ of a project can be calculated with the following formula:

$$C_o = NPV_c \frac{i}{1-(1+i)^{-t}} \quad \textit{Eq. 5}$$

and the levelised GHG emission reduction can similar be calculated as:

$$E_o = NPV_E \frac{i}{1-(1+i)^{-t}} \quad \textit{Eq. 6}$$

The full economic costs of a project (in so far as they can be monetised) and not just the direct financial costs, measuring the cost effectiveness can be formulated as:

$$C_{full} = {}^{C}/_{E} \quad \textit{Eq. 7}$$

Where *C* and *E* can either both in net present values (as defined in Eq. 3 and 4) or the levelised costs (as defined in Eq. 5 and 6).

The full economic costs, $C_{full}$, is distinguished from the direct $C_{fin}$ of the project and which will be discussed below. Note that for $C_{full}$, all costs are economic costs, as described in Economic opportunity costs or economic cost. The values of $C_{full}$ will depend on the precise value attached to the different components of costs and, as noted earlier, these costs are uncertain, with ranges of values rather than a single value. In view of this, it is important to present a range of such values and to indicate the impacts from which the uncertainty arises. Related to that, it will be useful to present a more detailed table of the components of the costs by time period, so that the policymaker can draw on this information should it be considered necessary.

***Choice of discount rates***

The debate on discount rates is a long standing one (IPCC, 1996b). As that reports notes, there are two approaches to discounting an ethical approach based on what rates of discount should be applied, and a descriptive approach based on what rates of discount people actually apply in their day-to-day decisions.

The ethical approach suggests that a discount rate that reflects the preferences of society to investments in long term sustainability impacts associated with climate change mitigation be used. This discount rate, according to theoretical arguments, should be the so-called social rate of time preference. It can also in a more pragmatic way be argued that the discount rate should be a political choice - this approach often talks about "the social rate of discount". The descriptive approach argues that the marginal rate of return on capital is the appropriate discount rate.

The former leads to relatively low rates of discount (around 3 percent in real terms[7]) and the latter to relatively higher rates (in some case very high rates of 20 percent and above). The arguments for either approach are unlikely to be resolved, given that they have been going on since well before climate change was an issue. Normally the cost effectiveness values are calculated for more than one rate and the results presented to provide the policy-maker with some guidance on how sensitive the results are to the choice of discount rate. The sensitivity is certainly there, at high rates energy projects with long gestation periods become unattractive compared to those with a shorter period. For the purposes of the broader analysis, it is recommended that a central real rate of 3 percent is applied and a sensitivity analysis is carried out for real rates of 1 percent and 10 percent.

In addition to discounting future costs and benefits there is the further issue of whether or not future emission reductions should be discounted when compared to present reductions. The justification for discounting is

7 The real rate of discount is the market rate net of inflation. Thus if a market has a discount rate of 12% and inflation is 8% then the real rate is 4%.

that emission reductions in terms of reduced impacts have a time specific value. The choice of the appropriate rate, however, remains an unresolved issue and, again, taking a range of plausible values is the only solution. This technical basis for this argument is further outlined in the section Measuring the benefits of reduced climate change.

One point perhaps which should be noted relates to the use of low discount rates for appraising GHG programmes in developing countries, where capital is scarce and market rates of discount are very high. This low real rate for mitigation programmes can be justified on the ethical grounds mentioned above. The scarcity of capital, on the other hand, can be dealt with by having a shadow price for capital that is greater than one, as discussed in the section on Shadow prices.

### 1.2.14 Quantitative non-monetary information

Quantitative information in non-monetary units will be available for:

1. Employment impacts,
2. Income gains and losses of different groups,
3. Associated environmental changes,
4. Sustainability indicators of the share of energy devived from renewable sources, now and at the end of the planning period.
5. Macroeconomic impacts on GDP, trade and sectoral changes in GDP,

In addition, some of the other sustainability indicators may be quantified, although that is not certain.

Some of this information will have been converted into monetary units, namely (1)(3). There are two ways of integrating this information with the monetary information. One is to calculate the $C_{full}$ value, which excludes the costs associated with (1)-(3) and then present the cost information as well as the information on (1)-(5) in table form. As with the values of $C_{full}$, there will be ranges of values for $C_{fin}$ and the items (1)-(5). The second is to report the $C_{full}$ value, which include the costs attached to (1)-(3), and then add the information from (4) and (5) in a new table. Both are important and should be carried out. Once the data have been presented, a further summary statistic can be developed based on weights for the different components of the project, both monetary and non-monetary. The weights can be derived through discussions with policy makers, or through reviews of related policy decisions. An evaluation of the project will then constitute a single value (or a range of values, each associated with different estimates of the impacts) summarising its overall impact. This will enable comparison with other projects in the same and related GHG fields. This method is called a multi-attribute analysis,

*Qualitative information*

Qualitative information on impacts is important and should not be ignored. It cannot be integrated into the summary cost effectiveness values or the multi-criteria number, but it is relevant to the selection of the project and, more crucially, to the design of the project. Once a GHG-related project has been identified, a preliminary screening should generate important qualitative information. This should then be used to modify the design of the project so that the key negative impacts are mitigated wherever justified. The revised project will still have some impacts but these will have been passed as "acceptable". This preliminary screening of projects will avoid serious environmental damages, as well as serious political blunders where projects that seem technically acceptable have such negative impacts on key stakeholders that they are bound to fail on political grounds.

### 1.2.15 Conclusions on selection criteria

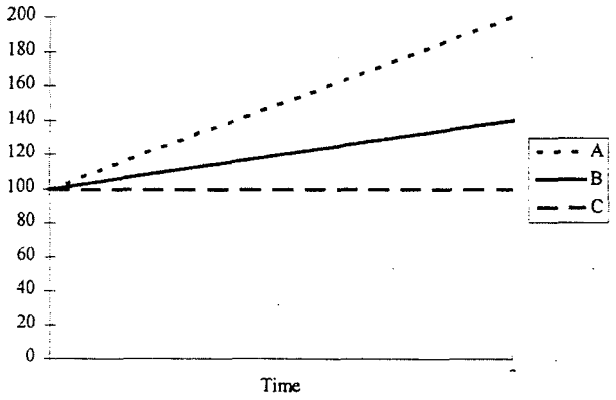
Ultimately the decisions on which projects to undertake is a political one. The screening rules discussed above are a guide to those decisions. As has been noted these rules will not provide unique guidance on which policies or projects to choose. But they will provide a range of indicators on financial costs ($C_{fin}$), full economic costs ($C_{full}$) and on the other quantitative and qualitative impacts that are inputs to the decision-making process.

### 1.2.16 Reporting of quantitative project impacts in GHG emission reduction marginal cost curves

One way of presenting the mitigation scenario results is to use "GHG emission reduction marginal cost curves" (UNEP 1994). This approach has been used for presenting mitigation cost assessments in the UNEP phase two project for $CO_2$ reductions in the energy sector. These marginal cost curves can, in some cases, be created using just information about emissions reductions and project outlays on individual projects. In other cases, the marginal cost curves should be constructed on the basis of integrated sectoral assessments.

The GHG emissions reduction marginal cost curve expresses the relationship between the minimum cost to society of reducing an additional ton of GHG emissions and the corresponding level of emissions reductions. GHG

**Figure 4 GHG emission scenario cases.**




emission reductions are defined as reductions in relation to the baseline.

The emission reduction targets can either be defined in relation to a base year (as for example 1990 emissions) or in relation to future baseline scenario emissions. Figure 4 shows these alternative definitions of emission reduction targets. Line A illustrates future baseline emissions, line C corresponds to the base year emission level while line B represents a reduction scenario. Emission reductions in relation to the baseline scenario involve the comparison of line A and B, while reduction in relation to base year involves the comparison of line B and C.

An example of the cost curve estimated in the UNEP format is shown in Figure 5.This cost curve is a comparison of marginal reduction cost for $CO_2$ reduction in the longterm assessed for the countries participating in the UNEP study ($CO_2$ reduction is measured in % in relation to future baseline emissions; costs are measured as levelised cost).

In the simplest case, where the marginal cost curves for GHG emissions reductions can be constructed on a project by project basis, from the bottom-up, the marginal reduction costs (US$ per tonne of $CO_2$) are calculated as the difference between the cost of following the baseline scenario and the mitigation scenario. A parallel cost assessment for these two scenarios can been done as follows:

$$MRC_t = \frac{CR'_t - CR_t}{R_t - R_t}$$

where:

$MRC_t$ = Marginal reduction costs in the year $t$.

$R_t$ = Total $CO_2$ emissions in the year $t$.

$R'_t$ = Total $CO_2$ emissions in the year $t$ after the implementation of a given project.

$CR_t$ = Total cost of a given scenario corresponding to the emission level $R(t)$.

$CR'_t$ = Total cost of a scenario corresponding to the emission level $R'(t)$.

The stepwise calculation of reduction cost can be reported in cost curves like the ones depicted in Figure 5. The total reduction costs can then be calculated as the area below the marginal abatement cost curve.

### 1.2.17 Implementation costs

In the costs and benefits assessed in traditional CBA, bottom-up studies, and sector studies it is more or less assumed that, despite the costs of the resources involved, like production factors, technologies, final products, overhead costs, etc., no specific activities are necessary to promote policy implementation. It is implicitly assumed that the market itself establishes incentives for the agents to implement the project. Implementation costs are given this only represented by the costs of planning activities, administration, information, training, monitoring and the like. These costs can be termed *administration costs.*

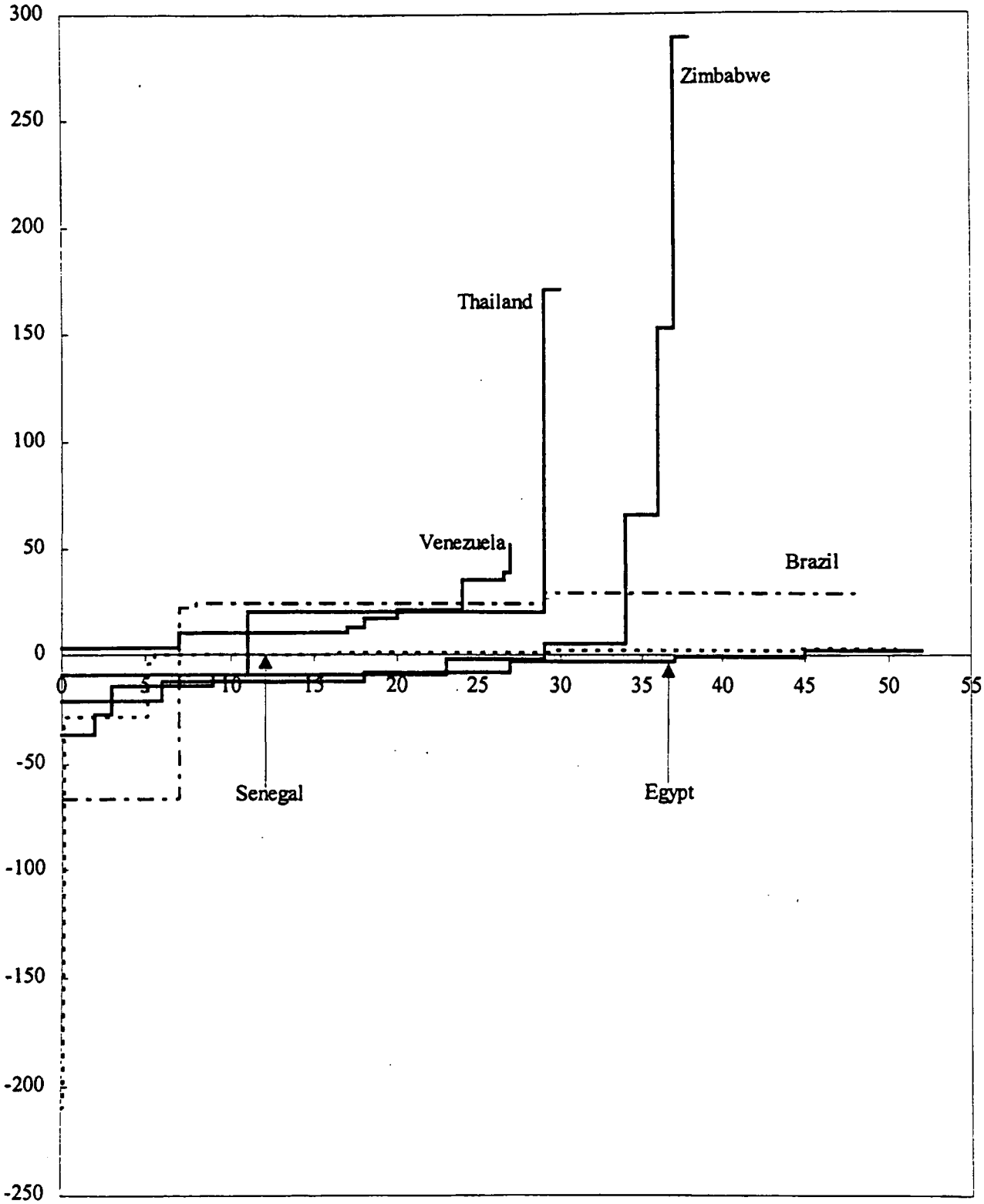
Successful implementation of large scale environmental projects or strategies such as climate change mitigation strategies will, however, typically involve costs that exceed a administration and training costs. The existence of market imperfections, imperfect information, institutional failure, externalities, ill-defined and/or not well enforced property rights etc., indicate that implementation will not be pursued as a friction less exchange process and transaction costs can be significant.

Implementation can then be supported by specific measures to remove and reduce barriers in order to realise the desired outcome of a given project or strategy. These a additional measures can be termed *barrier removal measures* and the analogous costs barrier removal costs. Barrier removal measures are as defined here related to the *short and medium term reduction of transaction costs.* Their effects are not limited to the immediate project or strategy. Thus barrier removal is under what might be labelled *transaction cost policies.* According to the above, implementation costs are defined as follows:

*Implementation costs* cover all costs associated with the expected requirements to realise a project or a sectoral strategy. These costs consist of two separate elements: administration costs and barrier removal costs.

*Administration costs* are the costs of activities that are directly related and limited to short term implementation of the project or sectoral strategy. They include costs of planning, training, administration, monitoring, etc.

Figure 5 Marginal cost curves of GHG emission reduction (UNEP country studies).



*Barrier removal costs* are the costs of activities aimed at correcting market failures directly or at reducing the transaction costs in the public and/or the private sector. These activities should support exchange processes related to project implementation in cases where the implementation meets significant barriers. Examples of barrier removal costs are, costs of improving the institutional capacity, reducing risk and uncertainty; enhancing market transactions, enforcing regulatory policies etc.

Implementation costs according to these definitions include all costs of policy realisation that are additional to other resource requirements such as technology, capital, land, and labour costs and should be assessed at project or sector level.

Barrier removal costs should be measured as social costs. They are incurred to reduce a the social costs in the longer run by making regulation and policy instruments work. This indicates that barrier removal must be seen in a dynamic perspective, i.e. the costs of barrier removal are time and context specific. Correspondingly, although barrier removal activities are aimed at the implementation of a specific project or sectoral strategy, their effects reach beyond that specific project or sectoral strategy.

As part of the Asian Development Bank's (ADB) Regional Study on Global Environmental Issues, 1994, mitigation options for Sri Lanka within the context of the country's development and strategic needs were identified and assessed. Three broad areas in which Sri Lanka has reasonable potential to reduce GHG emissions were shortlisted and possible project areas identified.

- Energy sector
- Transportation sector
- Forestry Sector

Energy sector

The main options available for mitigating emissions in the energy sector are the following:

1. Increase the productivity and efficiency of energy use through sound pricing policies based on the principle of recovering the economic cost of supply as measured by the long-run marginal cost.
2. Introduce programmes to improve the productivity of energy in sub-sectors of industry which are major users of energy by special measures to cut down waste and improve efficiency.
3. Training in conservation and energy audits should be encouraged for both public and private sectors. Although sound pricing policies will provide the most important incentive for improvement in the efficiency of energy use in the private sectors, these supportive measures in all sectors of economy will support a strategy aimed at energy conservation.
4. Special attention should be paid to the small-scale industry and household sectors, introducing programmes to promote conservation and technology advancements which cut down on energy use.
5. A comprehensive strategy for identifying, developing, and using alternative energy sources, particularly the renewable sources, should be formulated and implemented.

Transportation sector

In Sri Lanka, as in other developing countries, the use of private vehicles is rapidly rising resulting in frequent heavy traffic congestion in cities leading to a higher fuel consumption and increased vehicular emissions into the atmosphere. Thus, the mitigation strategies that need to be implemented immediately are as follows:

1. Implementation of legislation on strict emission standards.
2. Identifying and strengthening of the institutional framework for pollution control.
3. Periodic vehicle registration based on maintenance standards and tests.
4. Use of unleaded petrol.
5. Provision of mass transportation systems: the rail and bus transportation systems in Sri Lanka need to be planned according to the increasing commuter needs and improved with sophisticated equipment that pollute less and also consume less energy.
6. Introduction of canal and sea transport systems for cargo transport.
7. City planning should be based on a long-term perspective to achieve an optimum balance between free flow of traffic, citizen access, and minimum use of private vehicles.
8. Promote greater use of telecommunications.

Each of these mitigation strategies requires a well-co-ordinated set of initiatives and measures in order to achieve the desired results.

Forestry sector

A significant measure for mitigating GHG emissions is the enhancement of GHG sinks in the form

of forests and green belts in and around urban areas. The role of forestry as a source of GHG sinks and economic growth should be fully exploited by

(a) maintaining existing forest lands,

(b) improving forest management practices,

(c) expanding forest lands and biomass,

(d) substitution of bio energy fuels for fossil fuels, and

(e) increasing extent and efficiency of the use of forest products.

In the forestry sector, policy strategies for enhancement of sinks fall into two broad categories: conservation and reforestation. Conservation of forests generates twin benefits. Firstly, preserving the existing sources of carbon sequestration when trees are allowed to grow and absorb GHGs and secondly, preventing the release of accumulated carbon stored in the trees after years of growth through deforestation. Reforestation increases the forest cover and the capacity for carbon sequestration.

## 1.2.18 Case examples

This section will present a number of case examples based on climate change mitigation studies for Thailand in order to illustrate important aspects of cost assessments. The examples will for a number of climate change mitigation case examples consider:

- direct costs and benefits.
- indirect costs and benefits.
- ranking of the projects according to financial costs and social costs
- implementation cost assessment.
- development of projects to be supplied for AIJ or CDM financing.

## 1.2.19 Direct costs and benefits

The ALGAS project has included country studies for 12 Asian nations including capacity building, establishment of GHG inventory, baseline GHG emission projections, identifying least cost options and developing national portfolio of technical assistance and investment projects. From this study, a number of project types that would be appropriate in the Sri Lanka context can be identified and are listed in Box 3 below.

Identify main direct costs and benefits that are critical for the cost effectiveness of the projects.

A number of mitigation options have been or are being instigated in Sri Lanka. These include:

The AIJ SELCO-Sri Lanka Rural solar Electrification project.

The GEF Energy Services Delivery Project. The objectives of this project are: a) to encourage the use of environmentally sound renewable energy technologies by the private sector, non-governmental organisations, and co-operatives for grid and off-grid energy services, b) to reduce long-term demand for electricity through demand side management, and c) to strengthen the public and private institutional capacity to deliver energy services through renewable energy and DSM. In addition, the ESD Project has the global environmental objective of mitigating carbon emissions in Sri Lanka.

The GEF Renewable Energy and Capacity Building Project. This project focuses on: a) enhancing the information base about renewable resource potential; b) broadening the familiarity with and confidence in renewable energy technologies, and c) training personnel in the techniques and practices relevant to the expanded use of renewable energy and enhanced energy efficiency.

The South Sri Lanka wind-farm, intended by the Ceylon Electricity Board to generate 200 MW of wind power.

## 1.2.20 Indirect costs and benefits

In addition to direct costs and benefits climate change mitigation projects as outlined in the cost concept section can have a number of indirect costs and benefits including:

- employment.
- income gains and losses of different groups.
- associated environmental changes.
- macroeconomic impacts.
- sustainability.

Markandya has assessed the indirect costs and benefits of a Demand Side Management Project in Thailand. The project was prepared by GEF and the World Bank. A five year Demand Side Management (DSM) programme was proposed with two major objectives (a) to build suffcient institutional capability in the Thai electric power sector and the energy related private sector to deliver cost-effective energy services throughout the economy and (b) to pursue policies and actions that would lead to the development, manufacture and adoption of energy efficient equipment and processes within the economy. The DSM project has four main elements:

a) provide user and manufacturer incentives and consumer education to influence practices and

**Box 3: Possible Mitigation options for Sri Lanka**

| ProjectTitle | Key Objective of the Project | Rationale for project priority | Non-Quantifiable Benefits |
|---|---|---|---|
| *Energy Sector* | | | |
| Fuel Switching for Public Buses in Colombo | promote use of natural gas in transportation buses | Environmental and climate change benefit | less pollution |
| Bioelectricity Using Biomass from Parawood | create value added to parawood waste, abundant biomass | SPP, value added | reduce air pollution |
| Training for Commercial Buildings Energy Auditing | revise policy for energy auditing and conduct training program to train energy auditor | lack of qualified personnal | |
| Efficient Boilers in Small and Medium Enterprises | target selected number of SMEs in energy auditing and upgrading boiler efficiency | SMEs are not competition | |
| Efficient Stoves to Reduce Demand for Fuel Wood | transfer environmentally-friendly technology | reduce illegal deforestation | less pollution and reduce deforestation |
| Sri Lankan :Market Development for Solar Cell Industry | promote wider adoption of solar cell technology option | Renewable energy | less pollution |
| Efficient Batteries | initiate a joint venture for the production of efficient batteries with emphasis on technology transfer, research and development | Technology transfer, potential for export | assist wider adoption of solar energy |
| Integrated Electricity Production Systems for Sustainable Development on Small Island | provide an environmentally-friendly source of electricity production | Environmental and greenhouse benefit | less pollution |
| *Forestry Sector* | | | |
| Fuel Wood for Self Sufficient | enhance self-sufficient renewable energy for villagers | reduce pressure on natural forest | indirect benefits from reforestation |
| Potential Strategies for Private Sector Involvement in Forest Sector Carbon Sequestration Program | promote role of the private sector in reforestation | private sector participation | indirect benefits from reforestation |
| Urban Forestry to Reduce Energy Demand for City Cooling and to Enhance Carbon Sinks | promote carbon sinks and reducing power energy in Colombo and vicinity | save energy, green city | less pollution and indirect benefits from trees |
| Chemical Treatments to Reduce Methane in Rice Production | reduce methane emission from rice production | major emitter of methane | reduce methane |

Table 6: Financial Analysis of DSM in Thailand

Figures in Millions of Dollars

| YEAR | Foreign Capital | Domestic Capital | Total Capital | Total Labour | Total Cost | Tonnes of $CO_2$ Reduced |
|---|---|---|---|---|---|---|
| 1 | 3.05 | 16.03 | 19.08 | 18.72 | 37.80 | 0 |
| 2 | 3.05 | 16.03 | 19.08 | 18.72 | 37.80 | 0 |
| 3 | 3.05 | 16.03 | 19.08 | 18.72 | 37.80 | 0 |
| 4 | 3.05 | 16.03 | 19.08 | 18.72 | 37.80 | 0 |
| 5 | 3.05 | 16.03 | 19.08 | 18.72 | 37.80 | 0 |
| 6 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 7 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 8 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 9 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 10 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 11 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 12 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 13 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 14 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 16 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 18 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 19 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| 20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1160 |
| TOTALS | 15.26 | 80.14 | 95.40 | 93.60 | 189.00 | 17400.00 |

| | |
|---|---|
| Total Cost at 3% $000 | 173.11 |
| Total Reduction at 2% (000tonnes) | 13.50 |
| FICOSTEF $/TONNE | 12.82 |
| Total Cost at 10% $000 | 143.29 |
| Total Reduction at 1% (000tonnes) | 15.30 |
| FICOSTEF $/TONNE | 9.36 |
| Total Cost at 5% $000 | 163.65 |
| Total Reduction at 3% (000tonnes) | 11.95 |
| FICOSTEF $/TONNE | 13.70 |

Table 7: Full Socio-economic Analysis of DSM in Thailand — 3

Figures in Millions of Dollars unless Otherwise Stated

| YEAR | Foreign Capital | Domestic Capital | Total Capital | Total Labour | Total Cost | $SO_2$ Saved Tonnes | $SO_2$ Saved | $NO_x$ Saved Tonnes | $NO_x$ Saved | TSP Saved Tonnes | TSP Saved | Total Emissions Saved | NET COST | Tonnes of $CO_2$ Reduced |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 4.58 | 24.04 | 28.62 | 18.72 | 47.34 | | | | | | | | 47.34 | 0 |
| 2 | 4.58 | 24.04 | 28.62 | 18.72 | 47.34 | | | | | | | | 47.34 | 0 |
| 3 | 4.58 | 24.04 | 28.62 | 18.72 | 47.34 | | | | | | | | 47.34 | 0 |
| 4 | 4.58 | 24.04 | 28.62 | 18.72 | 47.34 | | | | | | | | 47.34 | 0 |
| 5 | 4.58 | 24.04 | 28.62 | 18.72 | 47.34 | | | | | | | | -22.84 | 0 |
| 6 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 7 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 8 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 9 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 10 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 11 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 12 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 13 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 14 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 16 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 18 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 19 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -22.84 | 1160 |
| 20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 1552.58 | 9.38 | 1172.99 | 13.12 | 25.69 | 0.34 | 22.84 | -105.95 | 1160 |
| TOTALS | 22.90 | 120.20 | 143.10 | 93.60 | 236.70 | 23288.64 | 140.70 | 17594.91 | 196.83 | 385.29 | 5.12 | 342.65 | -259.24 | 17400.00 |

| | |
|---|---|
| Total Cost at 3% $000 | -124.98 |
| Total Reduction at 2% (000tonnes) | 13.50 |
| FICOSTEF $/TONNE | -9.26 |
| Total Cost at 10% $000 | 15.84 |
| Total Reduction at 1% (000tonnes) | 15.30 |
| FICOSTEF $/TONNE | 1.02 |
| Total Cost at 5% $000 | -67.13 |
| Total Reduction at 3% (000tonnes) | 11.95 |
| FICOSTEF $/TONNE | -5.62 |

attitudes towards energy-efficient technologies,

b) to develop energy efficiency standards, testing capabilities and monitoring procedures;

c) develop and promulgate appliance codes so as to enforce minimum standards

d) to continue to pursue technological improvements appropriate to Thai conditions.

The DSM programme would be managed by the Electricity Generating Authority of Thailand (EGAT). Actions would be taken in the commercial, residential and industrial sectors with a projected annual electricity saving of 1,427GWh after five years. This is equivalent to a generation saving of 436 million litres of oil, where oil is assumed to be the marginal fuel. The total financing required would be US$189mn over the five year period. Of this, around 84% would constitute local capital, the balance would be foreign. The project offers significant GHG reduction opportunities through the need to defer the building of new generation capacity.

The financial analysis of the project is presented in Table 1 below. It assumes that the benefits of the project will last for 15 years, in other words that the energy savings from this programme will continue for 15 years after the project has been completed. This is an assumption that needs to be tested. The report gives little guidance as to how long the benefits will last, and yet the cost effectiveness of the project is crucially dependent on that. Assuming a life of 15 years the present value of the costs is between $143 and $173 million and the net value of carbon emissions reduced is between 12 and 13.5 thousand tonnes The cost per tonne saved is $9.4 and $13.7 depending on what discount rates are applied. By comparison with other methods of reducing carbon this is in the middle of the ranges. [8]

The economic cost analysis makes the following additional assumptions:

a) Capital (both domestic and foreign) has a scarcity premium of 1.5

b) There are benefits from the reduction in fossil fuel emissions. The amounts of gases emitted per MWh of generation from diesel (which is the fuel that is assumed to be saved are: 798 grams of $SO_2$, 938 grams of NOx and 25 grams of Total Suspended Particulates (TSP). Corresponding annual generation saved is given as 1472 GWh a year after 5 years. The annual value ofthe savings are assumed to be: $3093-8990/tonne for $SO_2$, $5203-17171/tonne for$NO_x$ and $4472-22080/tonne for particulates[9].

The implications of making these adjustments is very substantial. The net costs of the project become negative, so that the fossil fuel benefits alone would justify the project. The resulting values of the social costs, FUCOSTEF are, however, more sensitive to the discount rates than are the corresponding values of financial costs, FICOSTEF. Clearly, therefore a broader analysis would give much higher priority to a DSM project such as this one.

The issues that have not been included in the above are: (a) the question of hidden costs as for example relating to implementation barriers (b) the sustainability; contribution of this project and (c ) uncertainties arising from these estimates. With regard to the hidden costs more work would need to be done to establish what such costs were. This would reduce the cost effectiveness of the project but is unlikely to change the conclusion that on economic grounds the project is more viable than non financial grounds. The sustainability of the project is increased as the dependence on non-sustainable energy is reduced by 1472 GWh or about 2.4 percent of total energy consumption now and about 1 percent of total electricity consumption at the end of the project period. Finally on the question of uncertainty, the values taken in the above analysis for fossil fuel benefits have been relatively conservative. Even more; conservative values could be taken. But this would not change the conclusion that the project has negative cost for most plausible discount rates. Hence it can be assumed that the uncertainty will not affect the judgement that the DSM project is a more desirable one than the financial analysis would suggest.

## 1.2.21 Implementation costs

The section on implementation costs defined implementation costs to include two separate elements namely administration costs and barrier removal costs. The first cost category relates to project planning and training costs, while the latter relate to market failures and transaction costs. In particular this latter cost component is critical in the implementation of end use energy efficiency like DSM programmes, where a large number of individual agents in the business and private sector are going to undertake the specific implementation activities.

The UNEP methodological guidelines includes a list of case examples of barrier removal policies and these are outlined in Box 4 below.

8 The financial analysis should also look at the financial viability of the project for the implementing agencies . This is not reported here as the information is not available. With GEF funding for most of the project, however, this viability should not be difficult to establish.

9 See Markandya 1998 for data sources

**Box 4: Examples of Barrier Removal Policies**

| | $CO_2$ reduction project Specific policy example |
|---|---|
| **Market barriers** | |
| Market creation, possibly With public sector involvement in the transition period | Temporary support to specific demonstration projects |
| Privatisation for example through the establishment of well-defined property rights and enforcement. | Housing markets |
| Regulate competition by introducing more market actors. | Informabon campaigns, soft loans. |
| Environmental taxes | Carbon taxes |
| Support efficiency in savings and investment decisions by deepening financial markets. | Support financing mechanisms (e.g. GEF) |
| Launch technical standards to be met in a given time frame. | Efficiency standards for electricity appliances |
| Price liberalisation, support international competition | Exchange rate devaluation |
| In-flexibility and constraints of established technical systems | |
| Timing of infrastructure investments and transmission | Long term planning of power production |
| Subsidy to capital turnover projects | Specific capital grants |
| Subsidised credit to support research, development and learning processes | Demonstration and research programmes |
| Co~ordination and integration of speciffc climate change mitigation efforts in general investment policies | Information, capital subsidies |
| Institutional barriers | |
| Establish monitoring and enforcement systems | Reporting systems |
| Establish and enforce property right | |
| Institutional set-up for the reduction of risks and/or risk pooling (notably capital market) | Offset market |
| Establishment of specific organisations to reduce uncertainty and transmit information. | |
| Establish international mechanism for technology transfer | "Clean technology mechanism" |
| Human capacity barriers | |
| Training and education activities. | |
| Improvement of decision making processes | |
| NGO involvement in specific areas | |

### 1.2.22 Participation in international financing mechanisms

The Kyoto Protocol has established a framework for the development of a set of international financing mechanisms that are relevant to consider from the perspectives of capital requirements for cost-effective climate change mitigation projects in Sri Lanka. The mechanism of particular relevance for projects in Sri Lanka is the Clean Development Mechanism. This mechanism has not yet been outlined and specified in any detail, but a general characteristic of the CDM include the following elements:

- the CDM will work through a system of certified emission reductions that are supplied from non-Annex I countries to Annex I countries. Independent bodies will be established to conduct verification and certification of emission reductions.
- a price of carbon offsets will be established at international CDM markets. The price will probably be in the range of $10 to $30 per tonne of carbon.
- the costs of verification, certification and other transaction activities will be critical; parameters in the carbon market development.
- national sustainability indicators will be applied to the projects.
- the CDM will probably include the establishment of a fund for adaptation projects.
- CDM financing should be additional to other financing.

*Discuss:*

- potential good Sri Lankan project candidates for CDM financing.
- which projects could potentially be implemented with low implementation costs.
- what will be main national sustainability indicators to assign to CDM projects implemented in Sri Lanka. ;

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# Chapter 13

# THE ECONOMICS OF SUSTAINABLE DEVELOPMENT

*by*

*Pamela Mason*

## 1 INTRODUCTION

The problem of economic and environmental sustainability can be summarised as the problem of maintaining over time the capacity of the economic and environmental systems, and of society more generally, to maintain human wellbeing. Therefore the ethical motives behind sustainability are derived from a concern for the future, and the fear that current paths of production and consumption are putting at risk the wellbeing of future generations.

There has been some argument within the sustainability literature as to whether or not sustainability is a valid problem. For instance, some writers such as Beckerman (1974) and Barnett and Morse (1963) maintain that as environmental resources become more scarce, the market will provide the correct signals for the development of alternatives. Therefore there is no problem of absolute resource scarcity. Other writers, such as Daly (1991), disagree, maintaining that the laws of thermodynamics imply absolute limits to the natural resources that may be consumed over time, and the scale to which the economy can grow. Others still, such as Solow (1991) argue that, while there is no reason to believe that the market will reflect future values sufficiently highly to ensure resource sustainability, sustainability should be thought of as ensuring that the opportunity to create wellbeing does not diminish over time, rather than as an injunction to preserve any particular asset.

The problem of sustainability can be described as consisting of two distinct aspects. The first is a purely economic problem and derives from the fact that current wellbeing, and in particular current consumption, can be increased at the expense of future wellbeing by consuming part of the current stock of invested capital or stocks of natural resources, which have come to be known as natural capital. This can be analysed explicitly for the case of invested or manmade capital, which yields an explicit rate of return. However, the problem can also be analysed for other types of capital as described below. This aspect of the sustainability problem is a failure to limit consumption to the amount that the current capital stock can support. That is, impatience on the part of current generations may mean that current consumption and wellbeing is higher than the sustainable level.

This problem is discussed in section 2 through the role of discounting on the path people choose for consumption over time. This shows that if at any time the utility discount rate is higher than the rate of interest, then according to the neoclassical "maximum discounted utility" criterion, it is optimal for some of the capital stock to be consumed at the expense of declining consumption in the future. We also show that if production is dependent on a non-renewable resource as well as on manmade capital, it becomes more likely that consumption will decline over time.

In section 3 we discuss how the economic models of the efficient use of renewable and non-renewable resources can be constrained to provide a solution of constant consumption. This results in the Hartwick rule, the original and best-known rule for economic sustainability.

The second aspect of sustainability, and the one that is arguably most important for sustainable development, is the problem of environmental sustainability. This involves the possibility that, even if the current generation does desire to consume no more than is sustainable, a failure to account for the services (essential or otherwise) that natural capital provides, could result in effective capital consumption. In section 4 we consider the services that the natural environment provides, ranging from fundamental life-support services to the provision of raw materials for production and the provision of amenity value. The fact that many of these services are pre-requisites for human existence means that they imply constraints to the depletion of natural resources and ecosystems. Section 5 considers the contribution of social capital to sustainability.

In section 6 we discuss the literature on Strong versus Weak sustainability. This is essentially a debate as to the extent to which a sustainability policy must maintain specific kinds of capital stock, especially stocks of natural resources and ecosystems, rather than depleting them and investing in manmade capital as an alternative. The weak sustainability paradigm assumes that different types of capital can substitute for one another, while the

strong sustainability paradigm holds that each type of capital must be maintained in its own right, at least above certain minimum levels. This section also provides a discussion of how recent work on calculating countries "Genuine Savings" is being used as an indicator of sustainability. This is related to the adjustment of national income to account for the environmental and social factors that contribute to wellbeing.

Section 7 considers the extent to which the problem of sustainable development at the local level is different from global sustainability, section discusses the role of economic and environmental policy at the local, national and international level in achieving sustainable development, and section 9 concludes.

## 2 ECONOMIC SUSTAINABILITY AND IMPATIENCE: THE ROLE OF THE DISCOUNT RATE.

This section illustrates the relationship between sustainability and the rate at which people discount their future utility. It is generally thought that in valuing paths of utility over time, people attach progressively less value to utility as it moves further into the future. This phenomenon is incorporated into the standard models of economic growth, e.g. Cass (1966), Koopmans (1965) and Malinvaud (1965). These neo-classical models identify the optimal growth path by maximising an intertemporal welfare function (IWF), which consists of the present discounted value of the utility path. The discount rate measures the rate at which utility becomes less important as it moves further away in time. The social planner derives the optimal solution by maximising the IWF subject to a budget constraint of the form:

Investment = Output - Consumption.

The budget constraint means that the cost of postponing consumption by investing in capital is offset by the return on capital, which permits higher consumption in the future. This type of model is called a PV-maximising model and reflects the fact that efficient markets aim to maximise the present value of economic assets. This model yields the solution that, given the initial level of the capital stock, the economy must invest up to the point at which the marginal product of capital, or the interest rate, is equal to the discount rate. Therefore if the initial capital stock is relatively low, investment will be positive and consumption will rise towards the equilibrium level at which $F_K = \delta$

This solution is illustrated in figure 1. In the figure, the initial capital stock is $K^0$ and the initial consumption level is $C^0$. Here, the cost of delaying consumption is more than offset by the productivity of the investment, so it is worthwhile to invest. Consumption rises and investment remains positive until the equilibrium capital stock, K*, is reached.

If, on the other hand, the initial capital stock is high so that the interest rate is lower than the rate of discount, then investment will be negative and consumption will fall towards the equilibrium level.

**Figure 1: The optimal path of consumption with a constant utility discount rate.**

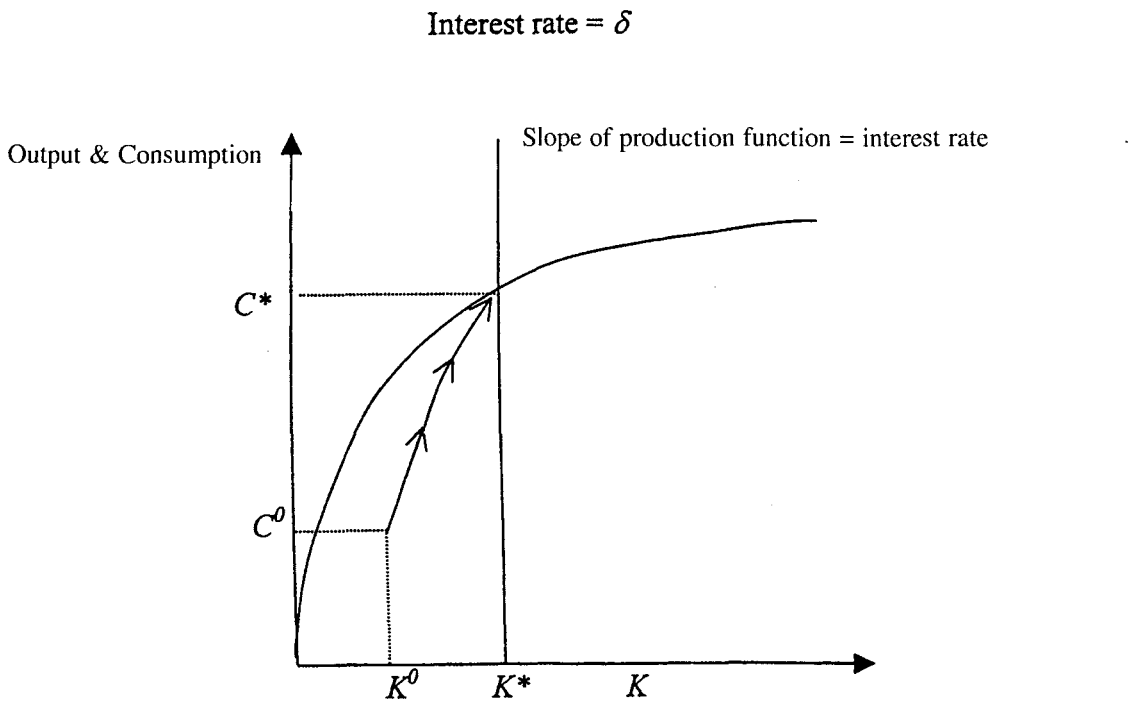
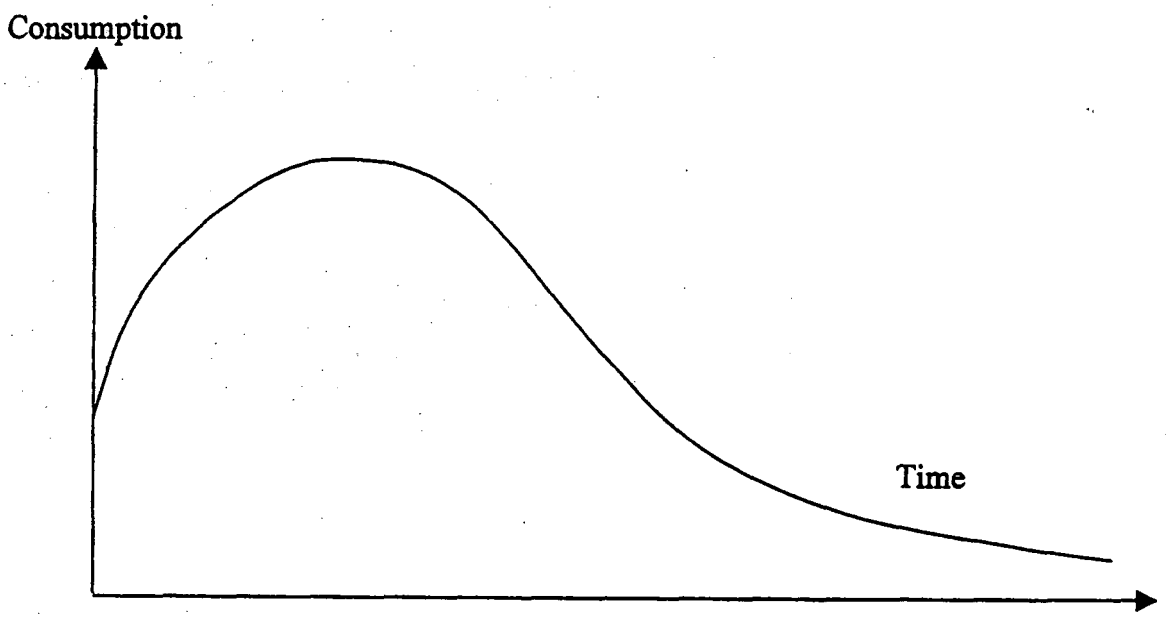


**Figure 2: The consumption path in Dasgupta and Heal's model with the Cobb-Douglas production function.**




This model illustrates the relationship between discounting and intertemporal wellbeing, since it shows that the optimal equilibrium level of consumption depends on the utility discount rate. Moreover, it illustrates the relationship between discounting and sustainability, since if the initial capital stock is greater than the equilibrium level then the optimal solution is to consume some of the initial capital stock. In this case consumption falls over time and so is unsustainable.

In this model the only capital stock is manmade capital. This tends in practice to be accumulated over time from low levels, so in a situation like this the PV-maximising criterion would tend to lead to a sustainable consumption path. In particular, as technical progress increases the productivity of capital, this increases the incentive to invest and increase the capital stock. However, if the capital stock were inherited by the economy rather than being built up by it, then it is possible that the initial stock could be higher than the equilibrium level, and in this case the optimal path would be unsustainable.

It is clear that environmental capital stocks, such as stocks of natural resources, tend to be inherited by economies rather than being built up. The neoclassical model described above was developed solely for the case of manmade capital. However, it can be used to show that in the case of environmental capital if the initial stock level is high so that its marginal product is less than the discount rate, the optimal solution will be to consume the stock unsustainably. This illustrates that the discount rate can affect the sustainability of natural resource use.

The situation becomes more complicated if there is a manmade capital stock and a natural capital stock. This is the situation considered by Dasgupta and Heal (1974) in their seminal paper on the efficient use of non-renewable resources according to the PV-maximising criterion. They set up the optimisation problem to maximise the PV of the utility path with a constant discount rate. They demonstrated that with a low initial manmade capital stock and a high initial resource stock, the initial interest rate is higher than the discount rate. Under these circumstances it is efficient to invest in manmade capital so that consumption rises over time. However, as the stock of capital rises and the flow of resource from the non-renewable stock decreases, the interest rate falls. At a certain point the interest rate falls below the discount rate. From this point onwards, consumption falls, as illustrated in figure 2.

This model illustrates that with a non-renewable resource stock and a manmade capital stock it is possible for the optimal path to decline towards zero over time. This is a direct result of the fact that a positive (and constant) utility discount rate means that the cost of low levels of utility far into the future is outweighed by the benefit of higher levels of utility in the near future

## 3 SUSTAINABILITY AND NATURAL RESOURCE USE.

Section 2 has illustrated how both manmade and natural resources can be exploited over time so as to maximise the net present value of utility. In this section we move on to discuss how such a solution can be constrained by a criterion of sustainability. This means planning the use of natural resources and investment in manmade capital so that consumption does not decrease over time.

Sustainable extraction of a non-renewable resource is, for practical purposes, impossible. However, sustainable resource use can be defined as maintaining a capacity to provide the benefit in question, for example by investing in a substitute for the non-renewable resource or, if the resource is exported, in an alternative revenue-generating asset. This corresponds to Solow's (1991) definition of sustainability as leaving to the future "the option or the capacity to be as well off as we are."

## 3.1 Sustainability and non-renewable resource use

The best known rule for sustainability in the presence of non-renewable resource extraction is the Hartwick rule (Hartwick 1977). This sustainability rule builds on the contribution of Solow (1974), who was the first to prove the existence of a theoretical constant consumption path when a non-renewable resource is depleted. The context of Solow's model was a closed economy in which a non-renewable natural resource is combined in production with manmade capital.

Hartwick (1977) used the same context to derive his original rule for investment for sustainable consumption, identifying the investment in manmade capital along the maximin path. Hartwick showed for this context that along the maximin path, investment in manmade capital is equal to the rents from the depletion of the non-renewable resource. Hence, Hartwick's rule has become known as the "Invest resource rents for sustainability" rule. The context of the problem, which will be referred to as a Hartwick-type economy, is depicted in Figure 3.

The Hartwick rule requires constant consumption in the presence of nonrenewable resource depletion. It ignores the possibility that the use of the resource has negative externalities, and that well-being might be determined by factors other than consumption. Moreover, it was derived using the assumption that manmade capital can substitute indefinitely for the non-renewable resource. This can be made more realistic by assuming the existence of a backstop technology, so that it is no longer necessary for the non-renewable resource stock last for infinite time. Under these assumptions sustainability still resource rents are invested while the resource stock is depleted.

## 3.2 Sustainability and renewable resource use.

A discussion of sustainability is more intuitive in the context of renewable resources. Clearly, for a given stock of a renewable resource there is a level of exploitation, corresponding to the growth of the stock, that is sustainable and does not involve depletion of the stock. Thus, the fundamental difference between renewable and non-renewable resources is that the potential exists for the resource stock itself to be sustainably managed. However, it is also true that if the objective of the owner of the resource is to maximise the monetary value of the stock, it may be that the optimal strategy is to deplete the resource stock.

**Figure 3: Maximin consumption and the Hartwick rule.**

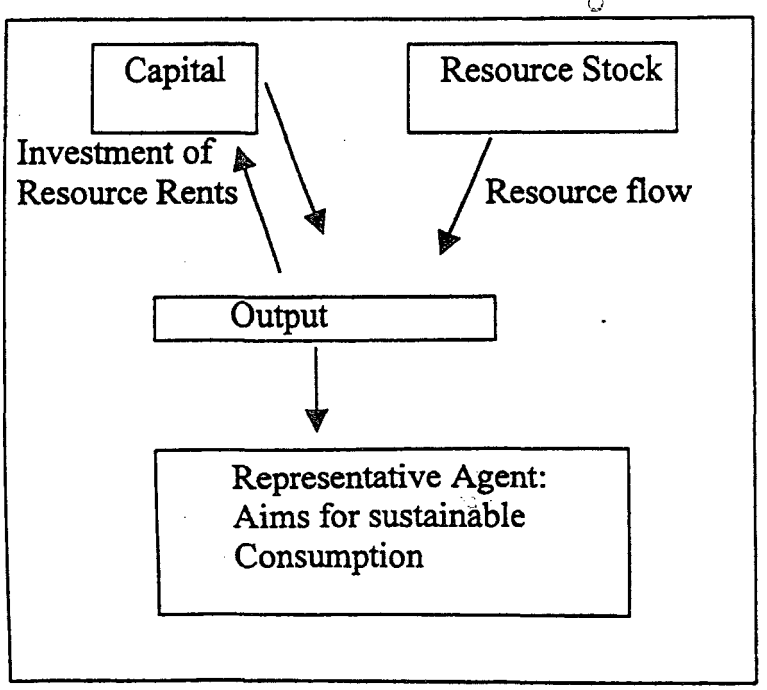


**Figure 4: Efficient resource depletion.**

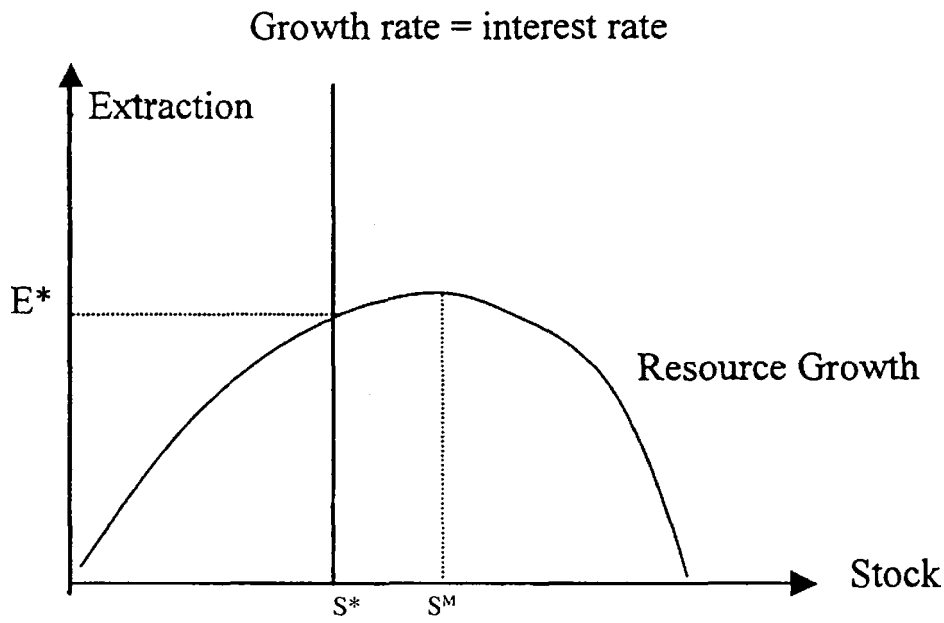



Figure 4 shows how, for a renewable resource stock, the efficient sustainable stock size is determined. The curved line represents the growth of the stock, given the stock size. As the stock increases, its growth increases, reaches a maximum and then decreases to zero once the stock , reaches the "carrying capacity" of its environment. The slope of the curve represents the marginal productivity of the stock. This is an important , factor as it represents the return on holding the stock, rather than depleting it and investing the proceeds in an alternative asset. The interest rate represents the return on an alternative asset, and therefore the opportunity cost of holding the resource stock. At all stock levels above S*, the return on the marginal unit of resource stock is less than the opportunity cost, and so it is efficient to deplete the stock up until this point.

Hartwick (1978) extended his sustainable non-renewable resource model to include renewable resources. He showed that for sustainable consumption, it is necessary for the rents from the depletion of the resource stock (which is extraction minus resource growth) to be invested in manmade capital. For maximum sustainable consumption to be achieved, resource extraction must follow the efficient path described above so that the stock is depleted until the growth rate is equal to the interest rate. If there is no positive stock level at which the rate of growth is equal to the interest rate then the resource stock will be efficiently depleted. Therefore sustainable consumption may well involve unsustainable resource extraction

This discussion of sustainability has considered only sustainability of consumption, and the contribution of natural resources to production. Sustainability of consumption is one definition of economic sustainability. However, economic sustainability requires that the more fundamental services provided by the environment be sustained at or above some minimum levels over time. This can be described as environmental sustainability. The services that must be maintained if environmental sustainability is to be achieved are described in the next section.

## 4 ENVIRONMENTAL SUSTAINABILITY AND THE SERVICES OF THE NATURAL ENVIRONMENT.

The most conspicuous services that the natural environment provides are food and inputs to production, including energy, metals and timber. Energy and metals are non-renewable resources, while timber is a renewable resource. We have seen in the last two sections how these resources can be used sustainably in terms of maximising the level of sustainable consumption that their owners can afford. In addition, the environment provides utility directly via, for instance aesthetic and recreational values.

The natural environment also provides more fundamental services, without which human life on earth would not be possible. These are known as Global Life-Support services, since they provide the basic necessities to allow human life such as food and shelter, and the maintenance of suitable climatic and atmospheric conditions. There is often a trade-off between using natural ecosystems to provide inputs to production, even if this use is sustainable, and preserving them in their

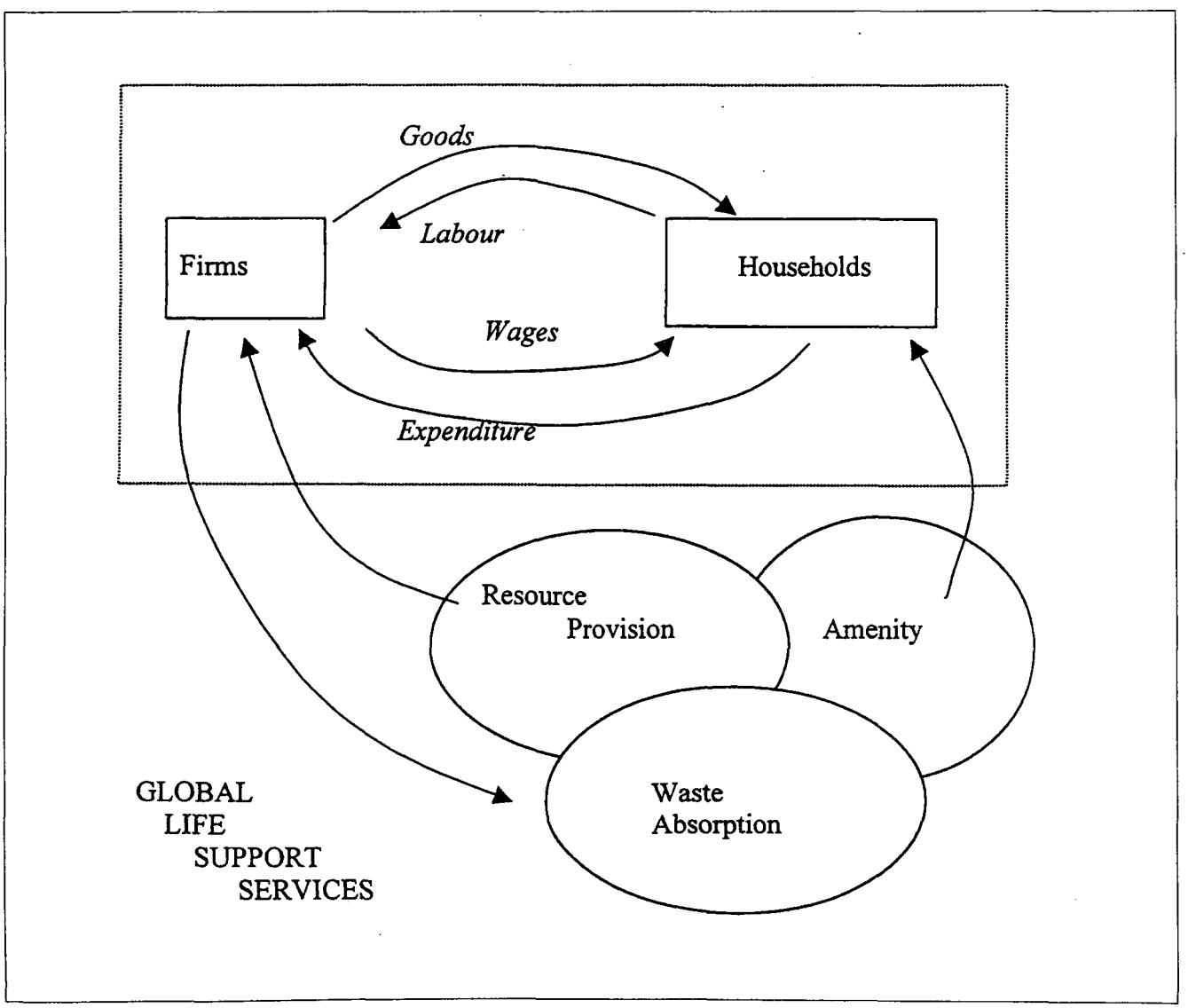
natural condition to maintain life support services. The opportunity cost of preservation is the value that could be obtained from harvesting ecosystems, and from converting the land to an alternative use.

Ecosystems, whether natural or managed, are required to capture the sun's energy and to produce food and raw materials. They regulate the hydrological cycle, which is a direct service to agricultural production. They create and maintain fertile soils, and they break down both natural and manmade waste into nutrients, maintaining soil productivity.

Some of these ecosystem services are summarised in figure 5 below. Another life support service provided by natural ecosystems is climate regulation, both on a local and a global scale. As noted, at a local level ecosystems are necessary for regulation of the hydrological cycle. On a global scale, they are necessary to maintain the composition of the world's atmosphere and therefore its system of climate regulation. The mitigation of global warming is the best known example of this service. A major issue for global environmental sustainability is how far ecosystems can be depleted, and pollutants allowed to accumulate in the atmosphere, without threatening global life-support services. One of the problems in analysing this issue is that scientific knowledge of ecosystem structure is far from complete. This results to a certain extent from the immense complexity and interdependence of species and systems. Therefore, it is very difficult to predict the effect that removing or reducing a population of one species will have on the rest of the species in a community, or on the viability of the ecosystem.

Figure 5 highlights three important facts. Firstly, the environment provides a wide variety of services that improve human welfare, both directly and indirectly. Secondly however, the use of the environment for some services, e.g. waste absorption, reduces its capacity to provide other services, e.g. amenity or resource inputs. Thirdly, the services of the environment can be thought

**Figure 5: The ecological services of the natural environment.**



of as representing "natural capital". Stocks of natural resources, including the capacity to perform global life-support services, can be used unsustainably over time, reducing their capacity to deliver the sarne services in the future. The implication of the second factor is that economic growth should be limited so that the positive contributions of economic growth to welfare are not outweighed by its negative environmental effects. This is the point that Daly (1991) makes in complaining that while in microeconomics, the optimal scale of an economic activity is clearly defined, with further growth uneconomic past the point at which the costs of growth outweigh the benefits, there is no corresponding recognition of an optimum scale of the macroeconomy. The question for environmental macroeconomics is, to what extent should economic growth be allowed to destroy environmental assets?

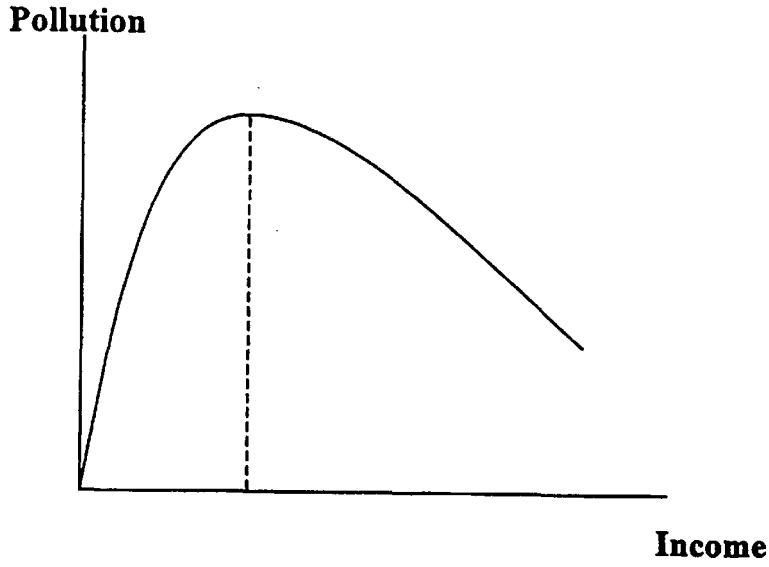
It has been suggested that this is not an issue, since people become richer with economic growth and demand a cleaner environment. This hypothesis is encapsulated in the Environmental Kuznets Curve debate. The Environmental Kuznets curve represents an empirically observed phenomenon, namely the fact that some environmental problems have become less severe as income levels rise. This is shown in figure 6 below.

The Kuznets Curve phenomenon has been used to argue that economic growth is in fact good for the environment, and that the argument that there are limits to the contribution that economic growth can make to human wellbeing is false. However, it is not clear that this is the case. This is mainly because the effect is not observed for all types of pollutant. For energy use and waste, for instance, the relationship between emissions and income is linear and increasing, rather than following the pattern depicted in figure 6. The available empirical evidence is therefore insufficient to draw any general conclusions regarding the existence of an environmental Kuznets curve effect.

A separate issue is the third fact mentioned above, namely that the essential services provided by natural capital stocks may be run down unsustainably over time. This implies that, if we accept that the market cannot reflect all of the values of the services provided by the natural environment, environmental sustainability places a constraint on economic growth. The current section has underlined that natural ecosystems perform fundamental life-support functions without which human life would not be possible. However, because these services are not traded in markets, they are not reflected in the value of conserving natural resource stocks. Therefore, it is possible that when a natural resource stock is depleted efficiently in terms of maximising its net present value, crucial life-support services could be lost.

An example of this conflict could be in the use of forest resources. Many of the world's forests are being depleted, since this is the economically optimal strategy for their owners. These forests, however, also perform global ecosystem services. Firstly, they absorb carbon dioxide and therefore reduce the effects of global warming. Secondly, they contain much of the world's biodiversity. Therefore, global environmental sustainability, which is a pre-requisite for economic sustainability, might require that limits be placed on the depletion of natural resource stocks, even if it appears

**Figure 6: An example of an Environmental Kuznets Curve**



that this means sacrificing some of the monetary value of these stocks.

## 5 SOCIAL SUSTAINABILITY

The discussion so far has focussed on two dimensions of sustainability the economic and the environmental. In the last few years, the discussion of sustainability has extended to the social dimension, in recognition of the fact that economic and environmental factors fail to explain substantial differences in the level of economic development achieved by the countries with similar endowments of human, natural and physical capital and in the outcomes of the similar sets of policies (e.g. structural adjustment) applied to the countries with similar characteristics. This suggests that the three types of capital determine only partially the process of economic growth, overlooking the way in which the economic actors interact and organise themselves to generate growth and development. The missing link is seen as social capital (World Bank, 1997).

There are many competing definitions of social capital but most of them incorporate the following features: relations of trust, reciprocity and exchanges between individuals which facilitates co-operation, common rules, norms, and sanctions mutually agreed or handed-down within societies, networks and groups including access to wider institutions. Recently the debate over social capital has focused on the following issues:

- The role of different forms of social capital in achieving sustainable development;
- The distinction between forms of social capital;
- How social capital can be assessed and whether it can be measured;
- Whether social capital can be created through externally catalysed processes;
- The role of social capital in ensuring good governance and strong civil society

(DFID, 1999).

There is little disagreement about the relevance of social capital for sustainable development in general. However, there is no consensus on which aspects of interaction and organisation it defines, nor about the validity of the term capital to describe this[1]. Least progress has been made in measuring social capital and in determining empirically its contribution to economic growth and development (World Bank, 1997).

The variety of definitions makes it very diffcult to construct a set of indicators to measure social capital. These definitions are multidimensional, incorporating different levels and units of analysis. Moreover, any attempt to measure the properties of inherently ambiguous concepts such as "community", "network" and "organisation" is problematic. In addition, very few long-standing surveys have been designed to measure "social capital", allowing to compile indexes from a range of approximate items, such as measures of trust in government, voting trends, memberships in civic organisations, hours spent volunteering.

In spite of these problems a number of researchers take the view that measuring social capital is not impossible, and several studies have identified useful proxies for it, using different types and combinations of qualitative, comparative and quantitative research methodologies.

In an example of a quantitative study, Narayan and Pritchett (1997) construct a measure of social capital in rural Tanzania, using data from the Tanzania Social Capital and Poverty Survey (SCPS). This large-scale survey asked individuals about the extent and characteristics of their associational activity, and their trust in various institutions and individuals. They match this measure of social capital with data on household income in the same villages (both from the SCPS and from an earlier household survey, the Human Resources Development Survey). They find that village-level social capital raises household incomes.

An example of a qualitative study is Heller (1996), who examines the case of the south Indian state of Kerala, where literacy rates, longevity, and infant mortality rates have long been the most favourable on the sub- continent. Tracing the history of state-society relations in Kerala, Heller shows how the state has played a crucial role in bringing about these results, by creating the conditions that enabled subordinate social groups to organise in their collective interest. However, the state in Kerala has also been hostile to foreign investment and the maintenance of infrastructure, which has made it difficult for a healthy and well-educated population to transfer its human capital into greater economic prosperity.

While there is growing empirical evidence on the benefits of social capital, there are few data on the cost side. It is very diffcult to determine the cost creating an association or a certain level of trust. However, investment decision for social capital requires a comparison of cost s and benefits. To do so, definition of social capital has to be better operationalised.

To sum up, the conceptual aspects of social sustainability focus on social capital. Measuring this, however, remains difficult and largely *ad hoc.*

1 The relationship between social capital and development is not always oriented towards material gain. Benefits from the social capital include enhanced spiritual well-being, a sense of identity and belonging, social status, honour and prestige (DFID, 1999).

## 6 WEAK AND STRONG SUSTAINABILITY: THE MAINTENANCE OF DIFFERENT TYPES OF CAPITAL.

So far we have discussed the optimal maintenance of manmade capital, and the extension of this model for a non-renewable natural resource. We have seen also how this model can be constrained to allow for a sustainable solution, and that this can be extended to renewable resources. Section 4 has discussed the role that the environment plays, over and above the provision of inputs to production, in providing the fundamental life support services necessary for human life, and how calculating maximum sustainable consumption must account for maintaining these services. Section 5 discussed the role of social capital and the social norms of behaviour, such as a respect for other people and their property, which are built up in the process of civilisation, and which may also deteriorate over time.

Natural capital, as we have discussed, performs many different functions. Examples of natural capital stocks are aquifers, which are run down as they provide supplies of water, hydrological systems and fertile soil. Natural capital also consists of stocks of non-renewable resources such as oil and gas, and stocks of renewable resources such as forests and fisheries. The latter produce raw materials, but are also direct sources of utility and many provide life support services. Finally, as discussed in the previous section, the global ecosystem constitutes a capital stock whose capacity to provide life-support services can be depleted

The fact that there are different types of capital stock that contribute to wellbeing has led to a distinction between weak sustainability and strong sustainability, as discussed by Rennings and Wiggering (1997). Weak sustainability can be defined as the maintenance of the value of the aggregated stock of capital. This implies two things. Firstly it implies that different capital stocks can be expressed in common terms, namely in terms of monetary value. Secondly, it implies that different types of capital stock can substitute for each other in a sustainable solution. A definition of weak sustainability might be:

$$\dot{K} + \dot{H} + \dot{SC} + \dot{N} \geq 0$$

where $K$ is manmade capital, $H$ is human capital, $SC$ is social capital and N is natural capital. As long as these capital stocks can be expressed in money terms, then under a policy of weak sustainable development, depletion of the stock of natural capital may be compensated for by investment of the same or greater value in, for instance, manmade capital.

Strong sustainability, on the other hand, requires that each type of capital stock be maintained in its own right, at least above some minimum level. Therefore a strong sustainability constraint can be expressed as follows:

$$\dot{K} \geq 0, \quad \dot{H} \geq 0,$$

$$\dot{SC} \geq 0, \quad \dot{N} \geq 0,$$

In the case of natural capital the minimum level of different types of natural capital stock might be determined by Safe Minimum Standards and the Precautionary Principle. Under a policy of strong sustainability, depletion of the world's forests might be limited so that the remaining stock is sufficient to maintain biodiversity at safe levels, to sustain timber supplies at adequate levels, and to absorb carbon dioxide emissions.

### 6.1 Daly's Rules for Strong Sustainability.

Herman Daly's (1990) criteria for sustainability provide an example of a suggested programme for strong sustainable development. Daly's criteria are as follows:

1. Renewable resources must be harvested at or below the growth rate for some predetermined level of resource stock. ,
2. As non-renewable resources are depleted, renewable substitutes must be developed so as to maintain the flow of services over time.
3. Pollution emissions should be limited to the assimilative capacity of the environment.

In another paper Daly (1995) answers Beckerman's criticisms of the concept of strong sustainable development, saying that it does not mean, as Beckerman maintains, that every species must be preserved and that no non-renewable resource can ever be extracted. Rather it involves acknowledging, in contrast to the weak sustainability paradigm, that natural resources are basically complements to manmade capital in production and that decreased availability of natural resources can be compensated only to a limited extent by increased manmade capital. Therefore Daly does agree with Beckerman that the concept of weak sustainability is illogical, since it assumes that manmade capital can substitute for natural capital, an assumption which Daly says is not borne out empirically.

### 6.2 Rule for strong sustainability: the Shadow Project constraint.

Another rule for strong sustainability, suggested

by Pearce et al (1990), is the shadow project constraint. This is based on the idea that strong sustainability can be interpreted to mean that the stock of natural capital, including ecosystems, must not decrease over time. This is quite a stringent criterion since it implies that no new land may be developed. This could mean that a development project with large economic benefits might have to be foregone because it involves some environmental damage. The shadow project constraint presents a way to avoid this problem. The standard cost-benefit criterion, including environmental, costs and benefits is:

$$\sum_{t=0}^{T} B_t\theta_t - \sum_{t=0}^{T} C_t\theta_t - \sum_{t=0}^{T} E_t\theta_t > 0$$

Where T is the time period over which the costs and benefits of a project are analysed, θ is the discount factor in each period, B is monetary benefits of the project, C is its monetary costs, and E is its environmental costs. Therefore the standard cost-benefit criterion is that the net present value of the benefits of a project must outweigh the net present value of its costs, including the environmental costs. The shadow-project constraint allows for any ecosystem destruction to be compensated by a shadow project which increases environmental quality elsewhere. Given this, as well as the standard cost-benefit criterion, the project must satisfy:

$$\sum_{t=0}^{T}\sum_{i=1}^{n} E_{it}\theta_t \leq \sum_{t=0}^{T}\sum_{j=1}^{m} a_{jt}\theta_t$$

where there are sites at which environmental damage is caused, and a denotes the shadow projects that create environmental benefits. Therefore this condition states that the net present value of environmental damage over the relevant time period must be negative. This is a weak version of the shadow project constraint. A strong version would be that the environmental damage in each time period must be negative.

## 6.3 Rule for strong sustainability: Safe Minimum Standards

Another rule which is associated with land use and non-declining natural capital stock is the Safe Minimum Standards (SMS) approach. This was developed by Ciriacy-Wantrup (1952) and Bishop (1978).

**Table 1: Genuine Savings Data for Several Countries**

| | $\frac{S}{Y}$ | $-\frac{\delta M}{Y}$ | $-\frac{\delta N}{Y}$ | $Z$ |
|---|---|---|---|---|
| Sustainable economies | | | | |
| Brazil | 20 | 7 | 10 | +3 |
| Costa Rica | 26 | 3 | 8 | +15 |
| Czechoslovakia | 30 | 10 | 7 | +13 |
| Finland | 28 | 15 | 2 | +11 |
| Germany | 26 | 12 | 4 | +10 |
| Hungary | 26 | 10 | 5 | +11 |
| Japan | 33 | 14 | 2 | +17 |
| Netherlands | 25 | 10 | 1 | +14 |
| Poland | 30 | 11 | 3 | +10 |
| USA | 18 | 12 | 3 | +3 |
| Zimbabwe | 24 | 10 | 5 | +9 |
| Marginally Sustainable | | | | |
| Mexico | 24 | 12 | 12 | 0 |
| Philippines | 15 | 11 | 4 | 0 |
| United Kingdom | 18 | 12 | 6 | 0 |
| Unsustainable | | | | |
| Burkina Faso | 2 | 1 | 10 | -9 |
| Indonesia | 20 | 5 | 17 | -2 |
| Madagascar | 8 | 1 | 16 | -9 |
| Nigeria | 15 | 3 | 17 | -5 |
| Papua New Guinea | 15 | 9 | 7 | -1 |

It stems from a concern that the type of calculation carried out under cost benefit-analysis cannot be used to plan for sustainability, because the valuation of damage to ecosystems cannot reflect sustainability principles. In the absence of a reliable calculation, it is suggested that ecosystem damage be limited so that the remaining stocks are above safe minimum levels, usually calculated as the minimum levels required for the ecosystem to remain viable.

The SMS rule, therefore is "prevent reductions in the natural capital stock below the safe minimum standard identified for each component of this stock unless the social opportunity costs of doing so are 'unacceptably' large". (Hanley et al 1997). This implies, for example, that pollution emissions and biodiversity loss should be kept below identified safe levels. The indicator of sustainability implied by this criterion is then whether or not the SMS is breached for any class of resource.

## 6.4 Indicator of weak sustainability: Genuine Savings

An example of an indicator of weak sustainability is provided by Atkinson and Pearce's (1993) measure of "Genuine Savings". This indicator takes a country's savings, and deducts from it the value of the depreciation on its manmade capital, and the value of the depreciation on its natural capital. This allows a judgement to be made as to a country's sustainability, since if the genuine savings measure is positive then the country is weakly sustainable. The genuine savings measure is the following:

$$Z = \frac{S}{Y} - \frac{\delta M}{Y} - \frac{\delta N}{Y}$$

where $Z$ is genuine savings, $Y$ is income, $S$ is savings, $S$ is the depreciation on $M$, the country's stock of manmade capital and on $N$, the country's stock of natural resources. The following table provides a summary of the empirical study undertaken by Atkinson and Pearce.

These estimates have been criticised for the same reason that Daly criticised the concept of weak sustainable development. This is essentially that it is illogical to suppose that maintaining the overall level of capital stock, with increases in manmade capital substituting for decreases in natural capital, can constitute a policy of sustainable development Moreover, countries such as Japan, which deplete very few natural resources can count most of their investment, net of manmade capital depreciation, as genuine savings. However, their production is dependent on natural resource imports, which may well derive from unsustainable extraction. Therefore, the high sustainability rating for such countries may be misleading. Atkinson and Pearce's position is that this measure does not necessarily mean that a country is sustainable, but rather that if a country does not pass even such a weak test of sustainability as this, this is a strong sign of unsustainability.

Genuine savings can be thought of as one aspect of the adjustments to a country's Net National Income (its output plus its net investment) that should be made to account for the services provided by the natural environment described in section 4, and any depletion of the capital stock that these services represent. The next section describes how genuine savings are used in the adjustment of national income.

## 6.5 Genuine Savings and Environmentally Adjusted National Income

Traditionally, National Income (Net National Product) has included only activities that involve market transactions. This leaves out many activities and changes, which affect human wellbeing. These include non-marketed goods and services such as housework and looking after old or sick relatives in the home. Leisure, security of employment and of the future, and social relationships are all factors, which influence well-being but are not reflected in marketed output and may conflict with increased production. Moreover, the environmental damage caused when, for example, a forest stock is depleted, is not included. On the other hand, "defensive" expenditures on reducing the environmental damage caused by economic activities, for example cleaning up oil spills, are included as income. For all of these reasons, it has been proposed that NNP is a poor measure of economic welfare.

There have been many suggestions for the correction of national income. Daly (1989) has suggested correcting Net National Product for defensive expenditures and the depletion of natural capital. Daly defines defensive expenditures as "those regrettable expenditures necessary to protect ourselves from the unwanted side effects of our aggregate production and consumption." The depletion of natural capital (both non-renewable as renewable) should be treated analogously to the depreciation of man-made capital. He argues that the level of arbitrariness in depreciation estimates are not necessarily larger than those made concerning the depreciation of man-made capital.

In a seminal paper, Maler (1991) considered the subject of green accounting, discussing corrections on Net National Product (NNP). His four main conclusions are that:

1) The value of environmental services to households and the direct use of natural resources by households should be included in NNP, valued at households' marginal valuation. If the value of environmental services is thus included in NNP, defensive expenditures must not be deducted to avoid double-counting;
2) The value of environmental services and natural resources to firms is already accounted for in NNP through the value of the output of firms;
3) The value of input goods to increase environmental quality should be deducted from NNP since they are intermediate goods.
4) The value of the change in environmental and natural resource stocks should be included in NNP, valued at its (discounted) future value to households and production.

The last of these conclusions is reflected in the calculation of Genuine Savings. Genuine Savings represent the net investment that is added to output, corrected for the depletion of natural resource stocks.

## 7 LOCAL VERSUS GLOBAL SUSTAINABILITY.

In this section we consider that the scale (e.g. local, national or global) on which a policy of sustainability is implemented may influence the policies that are used. Firstly the scale determines the problems that can feasibly be tackled, as opposed to those that must be treated as exogenous. For instance, while maintenance of the system of global climate regulation is a pre-requisite for both global and local sustainability, it is not possible for a local economy to include this requirement in its own sustainability policy.

The scale on which a sustainability policy is implemented also determines the extent to which it must maintain specific assets rather than broad indicators, such as monetary wealth. For instance global sustainability must account explicitly for environmental sustainability as defined above. Local sustainability is dependent on global environmental sustainability as a pre-condition, but does not necessarily depend on the local environment being sustained in its own right. For a small open economy sustainable consumption can be achieved by depleting resource stocks and investing in an alternative asset, thus maintain wealth and therefore consumption generating capacity.

The distinction between local sustainability and global sustainability can be related to the difference between "strong sustainability" and "weak sustainability" as discussed above. We have seen from the discussion above that weak sustainability and strong sustainability are often thought of as competing paradigms (see e.g. Goodland (1995)). However, it might be more useful to recognise that both of these measures can be valid in different contexts. For global sustainability, it is essential to maintain stocks of, e.g. biodiversity, forests, etc, above a "safe minimum standard" level, as well as maintaining material wellbeing over time.

The local economy, while dependent on strong global sustainability, may seek weak local sustainability by maintaining the means to secure material goods, so that sustainability of the local environment is not necessary. For example, a community on a small, forested island could deplete its forest stocks, export the timber and invest the proceeds in a foreign, interest bearing, asset. If consumption is limited to the interest on this asset and the interest rate does not decline then this is an economically sustainable strategy. However, it is clearly not environmentally sustainable in terms of the island ecosystem. Moreover, it depends on the global economy being environmentally sustainable. Thus, a small open economy can pursue weak sustainability while depending on the global economy to achieve strong sustainability by maintaining minimum stocks of environmental assets.

## 8 FINANCING FOR SUSTAINABLE DEVELOPMENT: THE ROLE OF ECONOMIC POLICY

This paper has been concerned with describing the characteristics of paths of economic development that could be described as sustainable, with respect to the use and depletion of natural resources, emissions of, polluting substances, and indicators of improvements in wellbeing over time. It has also described the ways in which the market may not produce sustainable development, due to the fact that the financial incentives to deplete stocks of natural resources are greater than the incentives to conserve them for future use, or to invest in the provision of alternatives. This tendency is often exacerbated by government policies that encourage the unsustainable use of natural resources. Examples of this are subsidies to forestry exploitation that encourage deforestation and subsidies on energy use by industry and the use of chemicals in agriculture. This, section describes briefly the roles of developing country governments, the private sector and the international community in ensuring that the finance is available to implement the policies required for sustainable development.

Finance for sustainable development is usually thought of as being additional funding that would not be provided in the normal course of development. However, there is considerable overlap between finance for

sustainable development and other environmental issues, such as, biodiversity conservation.

Developing country governments need resources to implement the policies associated with sustainable economic, environmental and social development. Sustainable economic and social development policies include the reduction of poverty, the provision of education and clean water and sanitation, the reduction of population pressures and the investment over time in productive capital stocks. Ensuring that development is environmentally sustainable involves limiting, deforestation and the loss of biodiversity, combating soil erosion and air and water pollution and ensuring that resource depletion is sustainable.

**Sources of finance for sustainable development:**

The following sources of finance are available to developing country governments

*1: Reduction of funding needs by eliminating distortions*

The financial requirements of the policies outlined above are considerable. However, they can be reduced at the outset by ensuring that costly subsidies that encourage environmental unsustainability are removed. It is estimated (Panayotou 1995) that the world spends over $ 1 trillion on subsidies of environmentally unsustainable activities These include the subsidies mentioned above, as well as capital subsidies, subsidies to fossil fuel use, water use and subsidies to land clearance and construction. Such subsidies are not only expensive, but they are distortionary, meaning that development is less sustainable than it would be otherwise. Similarly, the net funding needs for financing for sustainable development are reduced to the extent that many projects and policies, such as environmental taxes, generate funds that can be used in other areas of sustainable development financing. However, interim finance is also required to set up the institutions and incentive systems associated with these policies.

*2. Increased use of currently available mechanisms*

*(a) Public policies to increase revenues*

Governments in all countries, not only developing countries, can reduce distortions in resource use and increase sources of finance for economic and social development by including the use of environmental resources in the tax system. Increasing taxes on environmentally damaging activities, such as air and water pollution, creates funds with which conventional taxes, such as taxes on labour, can be reduced, and which can be spent on other environmental and social projects. The setting of environmental standards can also lead to funds being raised by the imposition of charges for violations.

A second source of potential savings for the government budget lies in the introduction of user fees for publicly provided services, such as the provision of water for irrigation. Where the services to be charged for are environmental, as in the case of water and energy provision, and the use of roads, the charges will result not only in lower pressure on the state budget but will also increase the efficiency with which environmental resources are used. This in turn will mean that fewer environmental resources are used, bringing the economy closer to environmental, sustainability.

A third source of funding for sustainable development concerns the implications for economic sustainability of natural resource depletion. We saw in sections 3 and 6 that weak economic sustainability requires that if stocks of natural resources are depleted, investments of similar value must be made in alternative capital stocks so that the total value of the economy's capital stocks does not decline. Currently, however, the failure of governments to charge the private sector for the value of depleted stocks of minerals, timber, oil, gas and fish means that this value is lost to society, and the required investment for sustainability is not forthcoming. Implementing user charges would both increase revenues for environmental protection and reduce excessive depletion of resource stocks.

*(b) Policies to increase private financing of sustainable development*

The first point to note regarding the private financing of sustainable development regards property rights over land and natural resources. Where property rights are insecure, this leads to a lack of incentives to invest in soil conservation and other forms of land improvement. The risk that land, or access to natural resources such as forests or fisheries, could be lost due to insecure property rights provides rather an incentive to maximise the short-term returns by using the resource unsustainably. Thus, policies to formalise property rights for individuals and groups currently using natural resources can provide incentives for the sustainable management of renewable resources.

A second policy that can stimulate environmentally friendly behaviour by the private sector is a deposit-refund systems, whereby incentives are provided for re-usable items used in both production and consumption, such as containers, to be returned for recycling. In the case of environmentally damaging items, the deposit system could provide incentives to return the items for safe disposal.

More explicit policies to capture private finance for environmental protection include **Environmental Bonds**. These are bonds that are posted by private companies at the outset of a project that involves resource extraction, such as forestry or mineral extraction. These bonds are, in theory, equal to the cost of restoring the maximum possible environmental damage. Thus, the company has an incentive to minimise environmental damage, and to repair any damage done. In the event that damage is left, the government can use the bond to finance a clean-up operation. A similar scheme is the use of **Industrial Environmental Funds**. These are charges to industry that correspond to the value of expected waste created. The government can then use these funds to pay to have the damage cleaned up. A third scheme, by means of which the private benefits of public spending can be appropriated, is known as Betterment Charges. Where environmental improvements, such as the clean-up of a city-centre lake or park, result in private benefits such as increased amenity and increased value of properties, these benefits can be captured by charging local property owners for the clean-up operation.

*3. Additional External funding*

External finance is required due to the fact that the requirements of sustainable development often create a gap between domestically available capital and capital requirements due to the long-term nature of the investments and the vew high implicit discount rates that are often present in developing countries where poverty is a pressing problem. External finance is also required where investments require imports that cannot be covered using a country's foreign exchange reserves. This finance must exceed currently available Official Development Assistance (ODA), not least because the amounts paid in ODA are exceeded by payments by developing countries to service their debts. Financing schemes related to international debt, such as debt-for-nature swaps, have been suggested but are not thought to be a major potential source of funding for sustainable development.

External finance is also warranted because environmentally sustainable development by developing countries has global economic and environmental benefits, and indeed may be essential for global environmentally sustainable development. This may require that actions are taken, such at the limitation of pollutants and the conservation of biodiversity, that are not necessarily in the direct interests of the country in question. The country should therefore be compensated for the incremental costs (i.e. the costs over and above the benefits to the country) of these actions.

The following mechanisms have been suggested by means of which international finance for sustainable development can be harnessed. Firstly, International Environmental Taxes could be introduced, involving the taxing of international travel, international trade, emissions of $CO_2$ and international capital transfers (the Tobin tax). However, there are considerable political difficulties with such taxes since their administration would require some form of international government, a concept that many countries are unwilling to consider. A second form of international finance is the use of Carbon offsets. This is a form of joint implementation in meeting $CO_2$ reduction targets, and the Kyoto protocol has resulted in its being incorporated into the convention on climate change. It involves a developed country financing the conservation of forestry in a developing country in order to offset the $CO_2$ emissions of the utility. To qualify as a "certifiable emissions reduction" the reduction in net emissions must be greater than it would have been without the project.

Also related to $CO_2$ reductions are internationally tradable $CO_2$ emissions permits. This scheme is designed to minimise the costs of reducing emissions of greenhouse gasses by a given amount. However, if the permits are allocated on the basis of population, it can also be a source of finance for developing countries, whose $CO_2$ emissions are proportionally lower and which are often allocated more rights than they need to use.

Agenda 21 was launched at the 1992 Earth Summit in Rio, and contained specific provisions for the financing of sustainable development, emphasising the role of ODA, and the overriding importance of raising finance within the developing world by reducing damaging subsidies and increasing the use of revenue-raising economic instruments. Since then, progress on financing for sustainable development has been mixed. While the debt burden of middle-income countries has been reduced, it remains high in low-income countries, constituting a barrier to sustainable development, and the impact of debt-for-nature swaps has been limited. Flows of private finance from developed to developing countries have increased although it is not clear how much this finance contributes to environmental and social sustainability. Moreover, international funds including the Global Environmental Fund (GEF) have played a significant role m providing financing for environmental projects with global benefits, such as biodiversity conservation. Environmentally damaging subsidies have been reduced considerably since Rio, and some progress has been made in the introduction of economic instruments, both as means of enforcing environmental standards and as sources of revenue for sustainable development. However, progress in general has been limited, and considerable work remains to be done.

## 9 CONCLUSION

This unit has attempted to give an overview of the problem of sustainability, and some of the ways in which sustainability is analysed in the literature. It has also discussed the extent to which the problem of sustainability can be incorporated into national income, the standard measure of economic success, in order to indicate whether or not an economy is sustainable.

We have seen that the economic, environmental and social factors that, contribute towards human wellbeing can be thought of as being capital stocks. These capital stocks can be used sustainably, so that the implicit or explicit interest on the stocks can be consumed while leaving the overall stocks intact. Alternatively, they can be used unsustainably, which means that they are allowed to deteriorate, or to be consumed over time. There are two reasons for which the use of these capital stocks might not be sustainable. The first is where people prefer to increase their own current consumption at the expense of future consumption. This tendency can be expressed as a high rate of discount of future utility. The second is where , the services provided by these capital stocks are not fully accounted for in decision-making, so that decisions are taken which result in the deterioration of essential capital stocks. We have discussed examples of this phenomenon, which is particularly likely to apply to environmental capital and social capital.

The two main frameworks within which economic sustainability has been analysed can be described as weak sustainability and strong sustainability. The difference between these two paradigms is that weak sustainability, assumes that different forms of capital can be substituted for one another along a sustainable path. Strong sustainability on the other hand assumes that substitution between, e.g. manmade capital and natural capital is very limited. We have discussed some of the rules and indicators that have been developed in the literature for both weak and strong sustainability. We have also related the two paradigms to the distinction between applying a sustainability policy at a local level, and applying one at a global level. We noted that if the depletion of e.g. natural capital stocks can be given a monetary value, then this value can be used to estimate, "Genuine Savings", which in turn can be used in the estimation of a country's Environmentally Adjusted National Income. Finally, we, considered the mechanisms available by means of which developing countries can raise and obtain the finance necessary to achieve sustainable development.

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# Chapter 14

# ENVIRONMENTAL ACCOUNTING

*by*

*Anil Markandya and Pamela Mason*

## 1 INTRODUCTION

The integration of environmental and economic decisions, which is the goal of sustainable development, requires information at the macro level, which includes both economic and environmental variables. This is the goal of integrated economic-environmental accounts, or 'environmental accounting' as it is commonly known. The development of such accounts is a major development in many countries. The need for consistent data, which allow international comparisons has prompted the Statistical Office of the United Nations to prepare a set of guidelines on the preparation of such accounts. These are known as the **System of Environmental Economic Accounting (SEEA).**

### 1.1 The Debate on Which System of Environmental Accounts to Use

There are a large variety of approaches in the design of statistical systems describing the interrelationships between the natural environment and the economy (United Nations, Economic Commission for Europe, 1991) Two extreme positions can be identified. On the one hand, there is the statistical description that focuses on the natural environment Environmental-economic linkages are described with regard to impacts on the environment. Much of the statistical framework is concentrated on the spatial description of the natural environment, involving the use, for instance, of maps of particular regions (ecosystems or eco-zones). The information is normally presented in physical units. At the other extreme, some statistical frameworks focus on the economy and take environmental-economic linkages into account only in so far as they are connected with actual economic transactions (for example, environmental protection expenditures and actual damage costs). These data systems are more closely related to the conventional national accounts, as they present monetary data on actual transactions in market values. In Figure 1 these two concepts - physical data collection and monetary accounting - are indicated in boxes 1 and 6. Approaches that are located between these two extremes could be classified with regard to the extent to which they incorporate monetary values.

Systems that mainly use physical units could extend the description of the natural environment to include information on the physical flows between the environment and the economy (use of natural resources, flow of residual products). The existing systems of natural resource accounting and environment statistics comprise such data (Figure 1, boxes 1 and 2). This description in physical terms could be further extended to include information on transformation processes within the economy. Material/energy balances comprise a physical description of the use of natural resources, their transformation by production and consumption activities and the flow of residuals back to the natural environment (Figure 1, boxes 2 and 3). Natural resource accounting and material/energy balances overlap, especially with regard to flows between the economy and the environment (Figure 1, box 2).

The description of economic activities in monetary terms has been extended in the case of the SEEA to the valuation of the use of natural environment. Different methods are discussed below. The comprehensive measurement of costs and benefits of economic activities and their environmental impacts is the purpose of such calculations (Figure 1, boxes 5 and 6) (see, for example, Bartelmus, Stahmer and van Tongeren, 1991). Such valuation not only facilitates the incorporation of environmental concerns into economic analysis but also creates a common scale of measurement that allows the compilation of economic-environmental aggregates on a highly condensed level

The SEEA thus covers in principle both national accounts describing economic activities and environmental accounts including all monetary and physical flows that describe the interrelationship between the environment and the economy (Figure 1, boxes 1, 2, 3, 5 and 6). This ideal concept cannot be fully realised at present, since comprehensive data systems for describing the natural environment and its interaction with the economy are still missing. Some ambitious approaches have been advanced in several countries, but no overall description of the natural environment has been realised so far.

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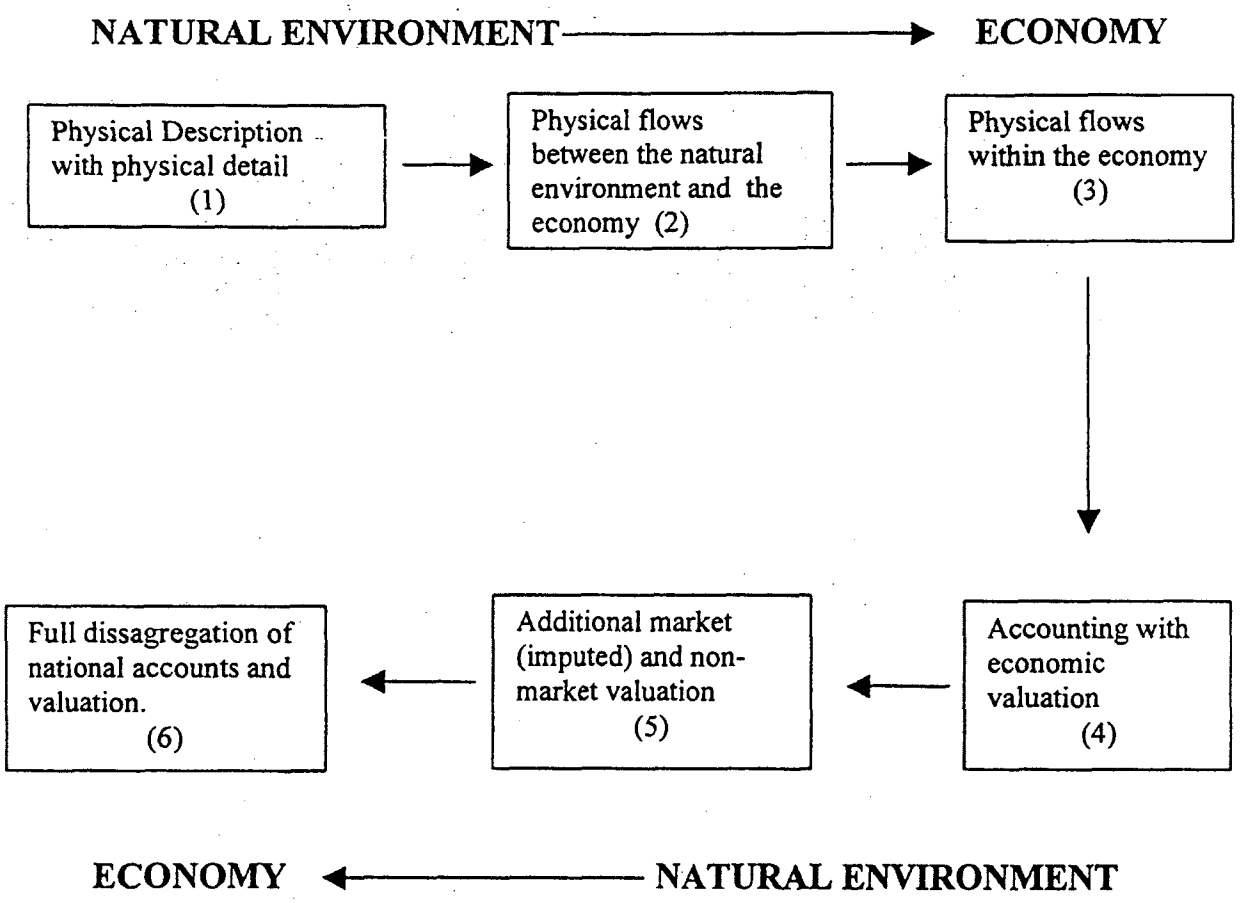
**Figure 1: Different Systems of National Economic-Environmental Accounts**




Distinction between Accounting Systems:

| | |
|---|---|
| 1: | Environmental statistics in the narrow sense |
| 1+2: | Natural resource accounts and environment in a broader sense |
| 2+3: | Material and energy balances |
| 5+6: | Extended accounting systems |
| 1+2+3+5+6 | Satellite systems of integrated environmental and economic accounting (e.g. UN SEEA) |

Source: United Nations (1993)

## 2 APPROACHES TO MONETARY ACCOUNTING

### 2.1 A brief introduction to Monetary Green National Accounting

Gross National Product (GNP), as well as being used as a tool of government policy, is also used as a proxy for welfare. It is for this reason that growth of GNP is seen by governments as a development goal. It has also been used as an indicator of success by international institutions such as the IMF. Governments are thus under pressure to increase GNP, often at the expense of other factors that contribute to wellbeing, and at the expense of environmental degradation, which may compromise the economy's ability to sustain production in the future.

The shortcomings of GNP, as measured by the standard system of national accounting (SNA), as an estimate of wellbeing include the fact that unpaid work is not included, so that GNP does not reflect the full value of output. Leisure is not accounted for, and because more leisure is generally considered to make people better off, a measure that does not account for the time people give up in order to produce a certain level of output does not reflect the net benefit of that output.

The reasons why GNP may not reflect welfare accurately, and the arguments as to whether or not the SNA should be adjusted to account for them, are well-known and thus will be discussed only briefly in this section. These consist of factors that are not included in measured GNP but that contribute to wellbeing, and that may conflict with increased economic output. There are also social and environmental by-products of economic activity that detract from welfare, which, it is argued, should be deducted from GNP. It has also been suggested that there are factors that are included in GNP that do not contribute to welfare and thus should be deducted. These consist principally of expenditures to reduce the unwanted side-effects of economic activity.

### 2.2 The external effects of production.

This section discusses some of the reasons for which GNP may not be a good measure of welfare. GNP measures economic activity in terms of the amount of final demand satisfied by economic output. The use of this figure as a measure of wellbeing rests on the assumption that wellbeing is proportionate to consumption of produced goods. However, economic activity has many negative external effects. These can often be described as environmental externalities, a major example being the pollution (to air, water and soil) caused by production. Some of the damage caused by this pollution is reflected in reduced output, as in the case of reduced agricultural output due to polluted soils and water supplies. This damage is thus reflected in the national accounts, since output figures are lower than they would otherwise have been.

However, environmental damage also directly affects human welfare and this loss is not reflected in the national accounts. Examples of this loss of welfare are the discomfort of breathing polluted air, as well as the increased risk of damage health that this entails. Pollution can also reduce the recreational and aesthetic benefits that people derive from the environment. Thus, for any level of output, welfare can generally be assumed to be higher, the better is environmental quality. However, standard national income does not reflect this and thus by this measure a given level of output implies the same level of welfare however much pollution it generates.

The second measurement problem associated with pollution stems from the fact that some resources are devoted to mitigating its consequences. This defensive expenditure does not contribute to welfare, but because it is final expenditure, it is counted as part of GNP. It has been suggested that these defensive expenditures should be deducted from national income in order to estimate more accurately the net welfare created by economic activity.

Further shortcomings of national income as a measure of welfare include social issues. For instance, no account is taken of the fact that the leisure that is given up in order to increase working hours and thus production has a welfare value. This means that an increase in measured output may not lead to a proportionate increase in wellbeing even in the absence of external costs of production. Likewise, job security may conflict with increased output, but almost certainly contributes to human wellbeing. Thus, an increase in output that is achieved at the expense of increased insecurity will not necessarily lead to a proportionate increase in wellbeing. However, these issues are outside the scope of this discussion.

### 2.3 Depletion of natural capital stocks

Net National Product (NNP) is intended to reflect the longer term implications of economic activity each year, because it includes the value of investments in and depreciation of productive stocks. This means that while the problems associated with the interpretation of GNP apply also to the interpretation of NNP, there is the additional problem that under the SNA, while estimates are made of the depreciation of manmade capital stocks, depreciation of natural capital stocks is not accounted

for. This means that if a country depletes e.g. its forest stocks, its income is inflated by the proceeds, but there is no corresponding deduction to reflect the decrease in value of its asset.

## 3 NET NATIONAL PRODUCT AS A MEASURE OF INCOME AND WELFARE

We have discussed the use of Gross National Product (GNP) as an indicator of the wellbeing enjoyed by an economy's citizens. Net National Product, on the other hand, has been used as an indicator of the economy's income. This section describes the theoretical basis for using NNP as a measure of an economy's income, based on the work of Weitzman (1976). This disagreement is based on the fact that NNP measured as consumption plus the value of net investments is not, even in theory, a measure of an economy's sustainable income.

The theoretical framework described in this section is the framework within which the GARP methodology has been developed. It is therefore useful to provide an explanation of its derivation, and also of the theoretical problems with the framework that have resulted in the proposal of alternative methodologies.

### 3.1 The Theoretical Basis of NNP as a measure of income and welfare.

The neoclassical literature on national income accounting calculates income in the sense of the consumption level that, if maintained at a constant level, would yield the same present value (PV) of consumption as the PV-maximising path. Weitzman (1976) proved, for an economy with a constant consumption discount rate and with no exogenous technical, progress or price changes, that national income defined as consumption plus the value of net investment is equal to the interest rate times the maximised present value of the consumption path. Since the present value of the consumption (or utility) path over time is usually referred to as the intertemporal welfare of an economy, we call this type of income "welfare-based income." That is:

$$Y^W = C + P\dot{K} = r\int_{t=0}^{\infty} C^*(s)e^{-rs}\,ds$$

Where $Y^W$ is welfare-based income, $C$ is consumption, $C^*$ is the PV- optimal path of consumption, $P$ is a vector of prices at which the vector of capital stocks $K$ is valued and $r$ is the consumption discount rate. $PK$ is thus "net investment", which means the sum of the value of changes, both positive and negative, in the vector of capital stocks. Welfare-based income is theoretically attractive since, as the equation above shows, it represents the "return" on the economy's "wealth", where wealth describes the PV of the future consumption path and the rate of return is the consumption discount rate. This provides a welfare basis for the use of national income accounts, since income is proved to be the "stationary equivalent of future consumption" (Weitzman 1976).

We should note at this stage that the context of Weitzman's model is that of the closed economy. Therefore there are no exogenous effects, since all prices etc. are determined within the model. For an open economy, exogenous price changes affect the value of the capital stocks and thus consumption possibilities over time. This effect is not accounted for by standard national income in the Weitzman tradition. This problem has been discussed by several authors, as explained below.

### 3.2 The distinction between NNP and sustainable income

One of the reasons for the lack of clarity in the interpretation of national income, is that the word "income" in the income accounting literature has at least two distinct meanings. The first meaning corresponds to welfarebased income, the theoretical foundations for which have been discussed above. This measure of income corresponds to the assumption of a path of economic growth that maximises the net present value of consumption using a constant discount rate. This path will not necessarily be sustainable, in that under certain circumstances the PV-maximising path may involve decreasing utility at some point in the future[1].

The second main definition of income is sustainable income. This is maximum that can be consumed without reducing the capital stock, and thus reducing future consumption possibilities. This is also known as Hicksian income, since it is based on Hicks's (1946) definition of income as "the maximum amount ...(an economy or individual) ... can spend this week, and still expect to be able to spend the same amount in each ensuing week." (Hicks 1946). However, the term Hicksian Income has itself become ambiguous, since Asheim now uses it to mean welfare-based income, as distinct from sustainable income. To avoid confusion, therefore we will use the term 'sustainable income'.

1 Dasgupta and Heal (1974) illustrate a case in which the consumption path that maximises the net present value of utility is eventually decreasing. This case involves the use of a non-renewable resource with manmade capital in production, the decreasing flow of the resource being compensated by an increasing stock of manmade capital. Beyond the point at which the interest rate on to manmade capital falls below the constant utility rate of discount, the efficient consumption and utility path falls.

Pezzey (1994) and Asheim (1994) showed that, in general, welfare-based income and sustainable income are not equal. If the PV-maximising path is unique and not constant, then welfare-based income in the tradition of Weitzman is not attainable as sustainable income. Rather, welfare-based income identifies an "upper bound" on sustainable income, and is *attainable* only if an unsustainable consumption path can be switched to a sustainable path without changing the supporting prices, that is, if prices are exogenous.

This condition would hold in the case of a single capital stock yielding a constant exogenous rate of interest. It would not hold for the closed economy of Weitzman's analysis. This point contradicts Solow's (1986) synthesis of Weitzman's (1976) proof and Hartwick's (1977) proof that resource rents must be invested along a path of sustainable consumption. Solow argued that if, at any point along a PV-maximising path, the value of net investments is set to zero (so that e.g. any depletion of natural capital stocks is compensated by a similar increase in the value of manmade capital stocks) then consumption will be constant. This was an attempt to show that consumption under the Hartwick rule corresponds to welfare-based income, and gave the misleading impression that income calculated as consumption plus the value of net investments (on the PV-optimal path) is equal to sustainable income as well as welfare-based income. Solow did not acknowledge that his result holds only where all prices are exogenous, for example, for the small open economy facing given international prices.

The simplest example with which we can distinguish between Sustainable and Welfare-based income corresponds to an individual or society that owns a stock of capital, that yields interest at a constant and exogenous rate. The sustainable income for this economy, corresponding to the definition given above, is clearly the constant interest on the stock. The welfare-based income, on the other hand, depends on the economy's utility rate of discount, and thus on the optimal path of consumption. If the economy's utility rate of discount is higher than the rate of interest, then the present discounted value of utility is maximised by a path of consumption and utility that declines over time. This means that the economy's PV-optimal strategy is to consume its capital stock, and its NNP, as measured by the current value Hamiltonian, is at all times higher than the current level of sustainable income[2].

This distinction has caused some difficulty in interpretation of national income, in that income is often thought of as being the amount that one can consume without diminishing the productive capacity of the capital stock[3], and thus as being the amount that can be consumed sustainably. For instance, Ahmad, El Serafy and Lutz (1989) note that "income is sustainable by definition: if it cannot be sustained, then it is wrongly estimated". This statement is an interpretation of income in the Hicksian tradition, as is Heal's (1998) definition of income as sustainable consumption. The fact that welfare-based NNP does not correspond to sustainable income, even in theory, is one of the objections that has been raised to adjusting the national accounts for environmental factors. We will return to this problem below.

There are two alternative approaches to calculating environmentally adjusted national income accounts. These two approaches are firstly the welfare based income

2 This can be proved as follows. The setup of the PV-maximising version of this problem is:

$$\underset{C}{Max}\int_{t=0}^{\infty} U(C)e^{-\delta t}\,dt \qquad s.t.\ \dot{K} = rK - C$$ assuming $U_C>O$, $U_{CC}<O$. The current value Hamiltonian is:

$H = U(C) + \psi_K[rK - C]$, where $\psi_K$ is the current value costate variable on the capital stock. The first order condition is

$\frac{\partial H}{\partial C} = U_C - \psi_K = O$ and the adjoint equation is:

$\dot{\psi}_K = \psi_K[\delta - r]$. Differendating the first order condition with respect to time and substituting the adjoint equation shows that the PV-optimal path of consumption, $C^*$, is such that $\frac{\dot{C}^*}{C^*} = \frac{1}{\eta}(r-\delta)$ where $\eta$ is the elasticity of the marginal utility of consumption. Thus, if $\delta$ is greater than r, the PV-optimal path of consumption falls over time which, for effciency, means that the economy is consuming capital At each point in time, sustainable utility income is equal to $U(C^S)$, where $C^S$ Is sustainable income. NNP is equal to $U(C^*) - \psi_K[C - rK]$. Note that for efficiency $C^* > C^S$ Thus the additional consumption in the PV-optimal case adds to utility over the sustainable case, but since $C^*>rK$, a deduction must be made to NNP for capital consumption. Using the first order condition $U^C = \psi_K$, the utility value of the last unit of extra consumption is equal to the utility value of each unit of capital consumption. However, using the assumption of diminishing marginal utility, the utility value of each previous unit of extra consumption is greater than the utility value of the capital consumption deducted. Thus, NNP is greater than sustainable consumption in each period.

3 Note that if interest rates are constant, sustainable income is equivalent to the interest on the capital stock. If, for example, interest rates fall over time, sustainable income is clearly less than the interest on the capital stock.

GARP approach, and secondly the sustainability standards-based GREENSTAMP approach. The GARP approach is in the tradition of the national accounting framework, and involves calculating the economic costs of the environmental side-effects of economic activity in order to estimate more accurately the net welfare provided by economic activity, and the effects of current economic activity on the economy's future prospects. Section 2.1 provides a brief introduction to the concerns regarding the shortcomings of standard Gross National Product (GNP) and Net National Product (NNP) that have led to the development of green national accounting.

The theory underlying the GARP methodology and in general the use of NNP as a measure of an economy's welfare, and the correction of standard NNP for environmental effects, is due to Weitzman (1976). Section 3 explains how Weitzman proved that NNP defined as consumption plus the value of net investments (including the value of depleted stocks of natural resources, is a measure of the economy's income. It also points out, however, that there is a distinction between income in the Weitzman tradition and sustainable income, in that income calculated in the Weitzman tradition cannot necessarily be sustained over time. Section 4 describes how this framework has been used in the development of the methodology of green national accounting. It introduces also some further theoretical problems with the framework, in particular the fact that it has not so far been developed to account for the effect of capital gains on the long-term prospects of open economies.

Section 4 finishes by describing some of the suggestions that have been made in the literature regarding the items that should be deducted from the national accounts to correct for environmental effects. Section 5 gives a short description of the work that has been done under the GARP project, and goes on to describe the GREENSTAMP project, which has arisen due to the concerns described in this paper that welfare-based income does not relate sufficiently to issues of sustainability

# 4 CORRECTING STANDARD NATIONAL ACCOUNTS FOR ENVIRONMENTAL AND OTHER EXTERNALITIES ;

The shortcomings of the standard national accounts as a measure of wellbeing have long been discussed. In the last thirty years, however, public concern has grown regarding the extent to which environmental damage may offset the positive effects of economic growth on wellbeing, and the extent to which economic growth may be unsustainable due to the depletion of productive environmental resources. The first concerted call for standard national accounts to be corrected to account for the issues highlighted in section 2.2 came in 1989 with the publication of "Environmental Accounting for Sustainable Development", edited by Ahmad, El Serafy and Lutz. This publication reflected the concern among researchers that standard measures of national income provide neither an accurate reflection of welfare, nor an accurate indicator of the long-term prospects of the economy. This section surveys the theoretical work that has been done in the fields of national accounting and economic valuation to correct for these problems.

## 4.1 Green NNP in the Weitzman - Hartwick tradition

Weitzman's analysis as reviewed in section 3.1 showed that the Hamiltonian of an economy's optimisation process can be used to measure its welfare income, since the current-value Hamiltonian represents current utility plus the utility value of net investment, in both manmade and natural stocks of capital. Several authors have used this result in models of resource depletion and pollution to calculate an environmentally corrected net national product known as Green NNP.

One of the early theoretical explorations of correcting the national accounts for environmental and other non-market goods was Maler (1991), who provided a theoretical rationale for deducting certain items of expenditure from national accounts. This was done by constructing a model of an economy in which utility is dependent on a produced good as well as on environmental quality and leisure. Output is a function of invested capital, labour, environmental quality and the harvest from a natural stock. He noted that, since an accurate measure of welfare involves adding to conventional NNP the welfare provided by environmental goods, current defensive expenditures should not be deducted, since this would be double counting. However, the goods used to enhance the stock of environmental resources do not contribute to current welfare, and their value should be deducted. Likewise, the value of changes in the stock of environmental resources should be deducted in the same way as depreciation on manmade capital is deducted.

In a similar vein, Hartwick (1990) suggests netting out the value of decreases in the stocks of renewable and non-renewable resources. Both of these contributions involve using Weitzman's (1976) result that the Hamiltonian of an economy's optimal trajectory represents the utility level that, if kept constant, would constitute a utility path with the same net present value as the optimal path. Thus, using the assumption of a

constant discount rate, the Hamiltonian represents the "return" on the economy's capital stock and thus in this sense can be thought of as the economy's income, or NNP. Hartwick transformed the Hamiltonian into a monetary measure of NNP by dividing through by the marginal utility of consumption.

## 4.2 Theoretical problems with Green NNP as an indicator of welfare

Svensson (1986) and Asheim (1995) noted that if consumption changes over time, the marginal utility of consumption is different in different periods, so the linearisation essentially involves discounting at a nonconstant rate. Weitzman's result depends on a constant consumption discount rate, and therefore this expression for NNP does not qualify as welfare-equivalent income in the tradition of Weitzman. Hartwick (1991) performed a similar exercise, but expresses income in utility terms as current utility plus the current changes in capital stocks valued at their shadow prices. The Hamiltonian is not linearised, and therefore this: represents utility income in the Weitzman tradition. However, the problem remains that this measure of income is not in monetary units, and therefore does not represent a method for adjusting monetary national accounts to represent "true" welfare-based income.

Another significant problem with Green NNP calculated in this way is that it does not account for capital gains or losses resulting from exogenously changing prices, or from exogenous technical progress. This problem derives from the fact that Weitzman's (1976) model, on which Green NNP in the Weitzman-Hartwick tradition is based, is a closed economy model with no technical progress, so that there are no exogenous effects and thus no capital gains.

Usher (1994) pointed out that "the Hamiltonian measure of income is the sum of current consumption and discounted future consumption generated by the activity in the economy during the current year." A more complete definition of income would include the effects of economic activity exogenous to the economy. Asheim (1996) and Sefton and Weale (1996) pointed out that calculating the Green NNP of an open economy whose sole activity is to extract and export a non-renewable resource yields a paradoxical figure of zero, implying that the economy earns no income from the resource stock since the entire revenue must be invested.

Sefton and Weale resolve this counter-intuitive situation by developing a measure of income that accounts for the future paths of resource prices and rates of interest. The income measure derived by Sefton and Weale consists of Green NNP, with the deduction of resource rents applying to resources used in domestic production rather than resources extracted domestically, and corrected for the effect of future of resource price and interest rate changes on the value of the economy's assets. The latter terms are known as capital gains or losses. This means a deduction for an increasing resource price path if the economy is a net resource importer and vice versa. Likewise, there is a deduction for an increasing interest rate path if the economy is a net creditor, and vice-versa. This resolves the paradox of the nonrenewable resource-extracting economy since "the rising resource price allows the exporter to extract an income from the resource stock available for export. For the resource user it implies ... a downward adjustment to its national income. ..".

However, while capital gains should in theory be included in measures of national income, in practice this is very difficult since it involves estimates of future price paths. Aslaksen et al (1990) is the only work of this type reported so far. They calculate capital gains using offficial projections of prices and extraction paths[4].

A third additional problem with the interpretation of NNP is the one pointed out above, namely that even if all of the relevant factors could be measured and included, there is no guarantee that the result would be a measure of sustainable consumption or welfare. Pezzey's (1994) and Asheim's (1994) results imply that if an economy is not on a sustainable path and switching to a sustainable path would alter the prices at which assets are valued then the income level that is attainable as sustainable consumption is lower than the income level that would be measured by NNP in the Weitzman tradition.

## 4.3 Alternative theoretical suggestions for adjustments to the national accounts.

Section 4.1 has dealt with the most mainstream contributions to the theory behind adjusting the national accounts for environmental externalities and the depletion of natural resources. There has been considerable disagreement as to the items that should be deducted from national accounts, whether they should be deducted from GNP or from NNP, and even in some cases whether they should be added or deducted from the accounts. This section provides a summary of the debate, and also a summary of our understanding of the source of the disagreements and of a logical way in which the various positions could be reconciled.

The major environmental factors for which corrections to the national accounts have been

4 Further work in this area is being made possible by the EC-funded SAUNER project, which provides long-term paths of non-renewable resource prices and extraction paths compatible with sustainability of resource use.

recommended are; environmental services both to production and directly to human welfare, environmental damages, defensive expenditures and depletion of productive environmental resources.

### 4.3.1 Environmental Services

It has been suggested (e.g. by Peskin (1989)) that the value of environmental services should be included in GNP. Hamilton (1996) notes that environmental services to production are already reflected in GNP since their contribution constitutes part of the value of production. However, as noted above, the direct contribution of environmental services to human welfare is not reflected in produced output. Thus, the suggestion that the value of these services should be included in GNP reflects the desire that GNP should be a measure of total welfare generated in the current period, including the welfare generated by environmental assets. The difficulty with this of course is in the measurement of the value of the contribution of the environment to human welfare. In particular, since human life would be impossible without the services of the environment, their monetary value could be thought of as being infinite. This implies that the most appropriate aim for GNP is to measure the net contribution to welfare of economic activity during a given period.

### 4.3.2 Environmental Damages

Peskin suggested that environmental damages should also be deducted from GNP, in order that net welfare derived from the economy and the environment should be accurately measured. Harrison (1989) argues in the ' same volume that since environmental damages constitute consumption of natural capital, and that the purpose of GNP is to measure production including the consumption of capital, that environmental damages should in fact be added to GNP, although they should be deducted once more in the calculation of NNP.

### 4.3.3 Defensive Expenditures

Once of the most often-quoted anomalies of standard GNP is that not only are the negative side-effects of economic production not accounted for in the measurement of wellbeing, but the expenditure to reduce these sideeffects is included and thus appears to inflate the measure of wellbeing Daly (1989), for instance, suggests that defensive expenditures should be deducted from NNP since they constitute not a net contribution to welfare but the avoidance of a decrease in welfare. However, Harrison (1989) and Vanoli (1995) argue against this. Harrison argues that such a deduction contravenes standard accounting practice, and that accounting cannot discriminate between items that clearly qualify final expenditure in not being subject to further sale. Vanoli points out that it is moreover extremely difficult to identify those expenditures that are truly defensive. Do we, for example, count the police and health services as defensive expenditures?

### 4.3.4 Resource Depletion

The value of overall productive capital is reduced when stocks of productive natural capital, such as forests and fish stocks are depleted. The fact that standard national accounts do not reflect this is the major complaint of Repetto et al. (1989). The consensus on this issue is that the decrease in value of the asset is measured by the "resource rents" on the decrease in the resource stock, which is equal to the level of extraction in the case of non-renewable resources. This amount should be deducted in the calculation of NNP as representing part of the depreciation of the overall capital stock. As noted in section 4.1, this adjustment is made in calculating green NNP in the Weitzman-Hartwick tradition.

In a practical application of this adjustment, Repetto et al. showed for the case of Indonesia that accounting for the loss of forestry, oil and soils meant that while GDP grew by 7.1% between 1972 and 1984, their estimate of environmentally adjusted NDP grew by only 4%.

El Serafy (1989) proposed a method for calculating the amount that should be deducted from NNP to account for the depletion of nonrenewable resources. This consisted of the annual amount that, if placed in an interest-bearing account, would result at the end of the life of the asset in the accumulation of a fund of the same value as the original value of the resource. It is often thought (see Hartwick and Hageman (1993) and Hamilton (1995)) that El Serafy's "user cost" is equivalent to Hotelling rents when the latter are calculated in the same context used by El Serafy, i.e. a constant extraction path. In fact El Serafy's user cost can be shown to be less than Hotelling rents, by the amount of the interest on the accumulating fund.

The deduction of the value of depleted natural resources from NNP in calculating Green NNP is integral to the point raised by Asheim (1994) and Pezzey (1994), as discussed in section 3.2. We noted that welfarebased NNP in the Weitzman tradition is equivalent to sustainable income only if an economy's path can be changed from an unsustainable to a sustainable one without changing the prices at which economic factors are valued. If this is the case, i.e. if prices are exogenous, then it can be

shown that an economy can, ceteris paribus, move from an unsustainable to a sustainable path by investing the resource rents from depleted natural resources. The amount that remains as consumption is sustainable, and this is the amount calculated as NNP.

However, if prices are endogenous, then an economy on an unsustainable path, in the early stages of development, extracts natural resources more quickly than it would along a sustainable path. Natural resource prices and thus calculated resource rents are therefore lower than they would be on the sustainable path. Attempting to move to a sustainable path by investing resource rents thus calculated would fail because the price of the resource does not reflect its full sustainability value. Investment would be lower than that required for sustainability, and the implied consumption level higher than sustainable income and consumption. This reflects the point made earlier regarding the distinction between welfare-based and sustainable income.

## 5 ADJUSTMENTS TO THE NATIONAL ACCOUNTS IN PRACTICE: THE GARP AND GREENSTAMP APPROACHES.

This section discusses the development and application of two major methodologies whose aim is to assist in the measurement of environmentally adjusted national income, namely the GARP and GREENSENSE methodologies. The GARP project is a major example of an empirical study that gathers the type of information that could be used to estimate a Green NNP, or could be provided as satellite accounts to be interpreted alongside standard national accounts. The project has been conducted in two phases, GARP I and GARP II, both funded by the EC DGXII. The aims of the first phase were to use an advanced methodology developed under the EXTERNE project to provide estimates of the monetary value of environmental damages for the EU countries Germany, Italy, the Netherlands and the UK. The aims of the second phase were to extend the method in terms of the range of pollutants covered, the attribution of the damages to different pollutant sources, and to assess the levels of defensive expenditure on various sources of damage.

The aim of the GARP welfare-based approach is to develop a practically measurable estimate of the net welfare that an economy generates. This is based on the Weitzman foundation as described above. We have noted that the problems with this approach are the following:

- The welfare-based measure is not, even in theory, a measure of sustainable income, because it is derived from the assumption of a PVmaximising intertemporal equilibrium. It represents the amount of welfare that, if held constant, would equal the NPV-maximising path. This is not equal to sustainable income unless the NPV-maximising path happens to be constant.
- Unless the economy is actually at the efficient level of environmental damage implied by the derived environmental values, the prices at which environmental damages are valued will not be the correct prices, and the measure of welfare will be incorrect, even if the aim is to measure welfare-based income rather than sustainable income'
- Some environmental assets provide services essential to the sustainability of hurnan wellbeing, but that are not traded in markets and thus do not have market prices. The depletion of these assets often does not impact directly on current welfare and is not therefore amenable to economic valuation. The sustainability implications o this depletion are not reflected in a welfare-based valuation approach.

The theoretical basis for a second approach to calculating Sustainable National Income (SNI) was developed by Hueting (1989) in response to these difficulties, and in particular to the fact that welfare-based green national income is not a measure of sustainable income. Hueting's suggested methodology is built on the idea of strong sustainability, in contrast to the welfare-based approach, and takes environmental sustainability as a starting point. The basic premise of the SNI concept is that national income can only be sustainable if essential environmental services are themselves sustainable. Thus, sustainable income is defined as the maximum level of net economic output that is compatible with respecting scientifically identified standards of environmental sustainability. This, given publicly declared support for a policy of sustainable development, can be taken to be the "social demand" for environmental goods.

The Hueting method estimates an adjusted GDP, compatible with undertaking the abatement necessary to reach sustainability standards, using existing market prices. This partial equilibrium approach however does not account for the general equilibrium effects of the non-marginal changes in economic activity that would be associated with implementing environmental sustainability standards. Therefore, while the GREENSTAMP team were in sympathy with the aims of Hueting in measuring sustainable national income, they proposed a third methodology that would account for the total price and quantity effects of imposing sustainability standards.

The GREENSTAMP methodology, like the Hueting methodology, is based on a requirement for strong, environmental, sustainability. This approach rejects the monetisation of the benefits of environmental goods in order to apply cost benefit analysis to identify the most efficient allocation of resources at a macroeconomic level. Rather, according to this approach, the appropriate way in which to express society's "demand" for environmental goods is by respecting environmental standards compatible with preserving the ecological base required for sustainability. The GREENSTAMP methodology avoids the problems associated with the Hueting methodology by estimating sustainable national income by empirically calibrating a multi-sector equilibrium model of a national economy and using this to estimate the economic output that is consistent with respecting environmental sustainability standards. However, the major disadvantage with this approach is, arguably, that the complexity of a real national economy is so great that the inaccuracies associated with attempting to model an economy would be so great as to result in inaccuracies at least as great as those associated with the other two methods.

## 6 CONCLUSION

This Chapter has reviewed the theoretical literature on green accounting. The first divide is between those who argue that monetary accounts are desirable and those who take the position that physical information is more valuable. If we go down the monetary path, there are many detailed decisions that have to be taken. These can be summarised as follows:

o Valuing environmental damage (2 approaches)

We have identified two approaches — the welfare based approach and the avoidance cost approach. Ideally both should be looked at, where a clear sustainability criterion can be established, the avoidance cost approach is appropriate. In particular this indicates the costs of achieving sustainability. But this is not always possible, in which case the welfare approach is the better one to use. In any event the welfare approach gives the current costs of economic activity a nd is useful in its own right.

o Defensive expenditures

Information on defensive expenditures is always useful to the policy maker. The question of whether they should be deducted from national accounts remains unresolved. We should also note the formidable problems in identifying pure defensive expenditures. Our advice is to collect data on the obvious and clear cases of such expenditures and present them in a supplementary set of accounts.

o Non-renewable resources

Non-renewable resource stocks are an important component of the capital stocks of many economies. It is thus important to account for the effect of depleting these stocks on the economy's future consumption possibilities. Adjustments to national accounts should therefore include the values of depleted non-renewable resource stocks. Significant progress has been made in the area of valuing natural resource depletion, the World Bank is currently carrying out a comprehensive measurement of the effects of resource depletion on national income.

o Renewable resources

Renewable resources can be depleted in much the same way as can non-renewable resources, if the rate of depletion is greater than their natural rate of growth. Account should be taken of the effect of this depletion by deducting its value from national accounts. Current attempts of the World Bank in this direction are interesting, but remain crude.

In summary the exercise of collecting and documenting environmental items in im a national accounts framework is of greater value to the policy maker than the actual environmentally adjusted GDP measures themselves.

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# Chapter 15

# THE REGULATION AND MANAGEMENT OF WASTE

*by*

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## 1. INTRODUCTION

This module is primarily concerned with outlining the data requirements necessary for

- An appraisal of the private and social costs and benefits of alternative systems of waste stream management.
- An evaluation of the use of economic (and other) instruments designed to optimize waste stream management.

Following this introductory section which covers various preliminary issues and definitions, the next two sections undertake these tasks, in each case building on an appropriate model. The tasks are closely related, since the design of instruments for waste stream management must take account of the fact that

- *Either* existing prices do not fully reflect the private costs and benefits to agents involved in waste stream management.
- *Or* their activities generate additional social costs and benefits which are not reflected in existing prices.

Most work on waste stream management has been carried out in the US and the EU. Though the general analytical principles can be carried over to developing country contexts, obviously the relevant quantitative indicators do not in many cases. Additionally, there may be institutional considerations, which need to be taken into account, particularly to do with

- The organization of productive activities within households and with the structure of the labour market.
- The efficacy of institutions for implementing, managing and monitoring instruments designed to manage the waste stream

The first issue, and others of importance in developing economies, are highlighted in a concluding section.

One of the major difficulties in appraising waste stream management and associated instrument use is to draw boundaries appropriately. Assuming that the overall objective is to minimize the net private and social costs of the waste stream, we need to define which activities are to be considered as part of the waste stream process in the analysis. This is a standard problem in any lifecycle assessment (LCA). Thus it is possible to consider all of the following or some subset of the successive phases.

- The economy's choice of technologies and product-mix in terms of their potential for waste generation.
- The design and production stage of waste generating products.
- Households' and firms' waste-generating decisions after product use.
- The choice between final disposal (through landfill or incineration ) and reuse or recycling, and between various modes of reuse or recycling (though it should be recognised that incineration and reuse/recycling involve - possibly substantial - residual landfill requirements).

Most analyses concentrate on the final two phases, and some incorporate the second. Additionally, we need to consider what displaced activities are to be taken account of, particularly if

- There is energy recovery from incineration and landfill
- Reuse and recycling reduce extraction rates of natural resources.

Though cost minimization is the generally accepted goal, this may impose unrealistic informational requirements. A "waste hierarchy" is widely accepted among policy-makers, prioritising waste reduction at source over reuse or recycling and these in turn over final disposal, largely on the basis of assumed externalities. It is then possible to evaluate the net cost effects of marginal shifts of the waste stream up the hierarchy and the cost effectiveness of the instruments used to achieve this. This sort of analysis is very common.

Likewise, the design of waste-management instruments may have to be considered in terms of their strict efficiency characteristics as well as their

- Fiscal implications
- Income-distribution impact

- Administrative feasibility
- Political acceptability
- Conformity with international agreements and conventions

In the context of developing economies, some of these issues may be strongly related to the points highlighted above (labour market organization and institutional efficacy).

The international dimension is also important for developing economies, since national differences in legislation or in enforcement can encourage cross-border trade in waste. This may not necessarily be inefficient, but certainly requires careful evaluation. There is a possible trade-off between efficient scale in waste changing structure of the waste stream, with higher proportions of non- biodegradable, and actual or potentially hazardous waste. Together these impose severe strains on the limited infrastructure and managerial capacity available for handling waste. It is against this background, that the appraisal and management issues discussed in the next two sections need to be considered

(It should be noted that although there are international conventions on hazardous waste the definition differs with national legislation. Generally, the definition excludes radioactive materials.)

## 2 A FRAMEWORK FOR APPRAISING WASTE STREAM COSTS AND BENEFITS

In this section we evaluate a given volume and composition of waste and consider three options.

- Landfill
- Incineration
- Recycling.

Other options could include reuse and (for organic waste) composting. The assumption of given arisings is an important restriction: not only may this be an object of policy choice, but also the instruments used to manage the choice between landfill, incineration and recycling could themselves influence the volume and composition of arisings.

In terms of costs, there are some significant subdivisions in these options.

a) For given environmental management standards for landfill and incineration, we need to consider the extent of energy recovery.

b) For recycling, we need to consider
   - The choice between bring systems (where waste generators are responsible for delivering waste to collection points) and collect systems (where waste collectors collect from individual waste generators) - in fact there is a spectrum between these
   - For collect systems, whether waste for recycling is co-collected with other waste.
   - The materials to be collected.
   - The extent of separation carried out by waste generators before collection.

We consider the private and external costs and benefits for each option.

Some waste streams, including those widely defined as hazardous, are difficult to dispose of through conventional routes, and often require producer or other specialist involvement in handling; comments are made where appropriate. These include some wastes which increase disproportionately with economic development, such as some automobile- related waste - oil, tyres, lead acid batteries - and some electrical equipment, such as discharge tubes.

### 2.1 Landfill

There are two main components of costs and benefits.

- Collection and transport to the landfill site
- Operation, decommissioning, and post-closure management of the landfill site.

Thus $C_{ls} = C_{lp} + C_{le}$

And $C_{ls} = C_{lt} + C_{lo}$

(where C is cost per tonne of waste, l is landfill, s, p and e social, private and external costs respectively, t and o are transport and operational costs respectively). These are current, undiscounted costs.

#### 2.1.1 Private costs and benefits of landfill

We make some general comments here, which apply, where appropriate, to all the options.

First note that private costs refer to social costs net of externalities, not to private sector costs per se. Throughout, recorded costs may require adjustment to reflect true market input costs of capital, land, and labour. In particular, there is frequently public sector involvement in waste disposal and public sector accounting costs may well diverge from a market costs. Especially in developing economies, public sector operations may have access to land and capital on more favourable terms than the market offers, and may pay labour at above private market rates. Private sector costs may themselves be distorted by market imperfections. Costs must also be adjusted for taxes.

All waste management options are highly sensitive to collection and transport costs (external as well as private). There are large and consistent differences between rural and urban areas: in developed a economies the rural cost premium (including externalities) is of the order of 25%. Given the larger rural populations and poorer transport infrastucture in developing economies, this is a significant issue, somewhat obscured by the problems of dealing with waste generated by fast-growing urban populations.

For landfill sites, private costs of operation must be adjusted to allow for any energy produced (from recovered methane).

Thus,

$$C_{lp} = C_{lpt} + C_{lpo} - R_{lf}$$

(where R is revenue per tonne of waste and f ("fuel") is energy ie recovery).

We assume that recovered energy sells at the same price as energy from whatever source is displaced, so the net private saving (S) from displacing an alternative source of energy is

$$S_{lpf} = C_{lpd} - R_{lf}$$

(where d is the displaced source). This is obviously sensitive to our assumption about the displaced source, which we would normally assume to be the marginal energy producer. (Note that this has to be normalized on a tonne of waste.)

$$\text{So}, \; C_{lp} = C_{lpt} + C_{lpo} - R_{lf} + S_{lpf} = C_{lpt} + C_{lpo} - C_{lpd}$$

(and without energy recovery the final term is zero)

### 2.1.2 Externalities from landfill

Externalities associated with collection and transport of waste are those associated with any transport mode: accidents, air pollution, noise and vibration, and amenity loss. In addition, there may be particular externalities associated with the nature of the loads being transported if these are not securely contained.

Similar externalities apply to the operation (including energy recovery) of landfill sites, though the detailed pollutants etc. may, of course, differ. Some of these externalities may persist post-closure. In addition, there may be significant soil, sub-strata and water pollution, and these, especially, may persist post-closure, since these are stock pollutants with uneven spatial mixing, they are difficult to analyse (and, especially, to esign management instruments for), and sensitive to the location of sites and, in aggregate, to the spatial distribution of sites. These are widely regarded as highly significant, but little information is available which enables systematic investigation. Furthermore, for many established operations, and a fortiori for closed sites, there may be inadequate information about site sealing and wastes filled, and thus even dosage levels of ground and water pollutants over time are difficult to establish. They may well contain "difficult" and even hazardous wastes. So the preliminary stage of a complex analysis is fraught with difficulties. What is again highly likely is that the results are extremely site-sensitive and not easily generalizable.

For some externalities (and also for some private costs) there may be significant trade-offs between transport costs and operational/post closure costs, in, for example, the selection of sites away from more densely populated areas.

If there is energy recovery, we additionally need to consider the externalities associated with the displaced energy source.

Consider a simple two-period case, with transport and operational costs falling in the first period and all post-closure costs in the second period. (this is obviously readily generalized.) We assume that there are no private costs post-closure: if there are private costs (or if there are multiperiod transport and operational costs) then the usual considerations apply to the selection of the appropriate discount rate.

$$\text{Thus} \quad C_{le} = C_{let} + C_{leo} + C_{les}/(1+i) - C_{led}$$

(where s are post-closure costs and i is the one-period interest rate). Later we generally assume that costs have been appropriately discounted where necessary.

Thus (net) social costs *per* tonne of landfilled waste are

$$C_{ls} = (C_{lpt} + C_{let}) + (C_{lpo} + C_{leo} + C_{les}/(1+i)) - (C_{lpd} + C_{led}) \quad [1]$$

(with the final bracketed term going to zero without energy recovery).

## 2.2 Incineration

The incineration cost can be derived in a similar way.

Thus (net) social costs per tonne of incinerated waste are:

$$C_{bs} = (C_{bpt} + C_{bet}) + (C_{bpo} + C_{beo}) - (C_{bpd} + C_{bed}) + hC_{ls} \quad [2]$$

(where b ("burn") is incineration and h is the fraction of incinerated ; waste requiring eventual landfill disposal: in practice the composition of costs from landfill of incineration residuals may systematically differ from that of landfill waste as a whole and then the separate

cost components of landfill would need to be differently weighted).

We would generally assume that transport/ collection costs are similar for landfill and incineration ($C_{lpt} = C_{bpt}$ and $C_{let} = C_{bet}$, at least per tonne/kilometre, if not per tonne), that energy recovery savings are greater ($C_{bpd} + C_{bed} > C_{lpd} + C_{led}$), and that air pollution is the most significant operational externality. Therefore the most significant differential net environmental impact arises from the difference in air pollution from recycling and from the displaced energy source. Thus the choice of the latter is highly significant, since energy sources differ substantially in their pollution effects. However, air pollution costs are among the best understood externalities (which may partly account for the generally poor evaluation of incineration plants as compared with other management options, which may have equally significant but less well understood externalities). We assume that all post-closure costs are transferred to landfill costs.

## 2.3 Recycling

### 2.3.1 Private costs of recycling

The major adjustment that has to be made to transport collection costs is to net out the revenue from recycled materials and savings from reduced extraction of virgin materials, allowing for any differences in energy costs between processing virgin materials and recycled materials.

Thus $C_{rp} = C_{rpt} + C_{rpo} - R_r + (R_v - C_{vp}) - S_{rpf}$

(where r is recycling, v is virgin material; costs of recycling and virgin material extraction are net of any difference in energy costs, which is entered separately as the final term).

It is generally assumed that the revenue from a tonne of recycled material is the same as from virgin material. This may not be the case: recycled materials (particularly in the case of paper) may be of inferior quality, on the other hand, they may attract a premium from g environmentally-conscious consumers. A reasonable approximation is s therefore likely to be the cost of extraction of virgin materials. However, it is possible that, for forecasting purposes we should allow for a growing price premium over time with the growth of environmental awareness. For well-known reasons, private markets may not fully reflect the implications of current extraction rates for future resource availability. Thus the extraction cost may have to be adjusted to incorporate dynamic 'Hotelling' rents: for the most commonly recycled materials, this is unlikely to be significant over foreseeable horizons.

Positive energy replacement savings are assumed: that is, processing virgin materials is relatively energy intensive compared to recycled materials.

Transport and collection costs are likely to be significantly higher per tonne/kilometre) than for landfill and incineration. Given the importance of separating different types of waste for recycling, higher collection costs may be offset by lower operating costs. To some extent the a distinction between collection and operation is artificial and the more important distinction is to do with how the costs of separation are spread across different agents (including waste generators) along the waste stream. Likewise, there may be significant economies of scale in recycling plants, and thus there may be complex trade-offs between transportlcollection costs for different systems and operating costs. (There may be a significant internationl dimension, with cross-border trade assisting the operation of efficiently scaled plants.) There may also be a time dimension with trade-offs between collection intervals and costs of waste storage by different agents: inventory models would be applicable in modelling this. On the other hand extending recycling beyond paper, glass and some metals significantly increases operating costs.

On the revenue side, increasing plant capacity may increase revenues in the short-run, but more importantly may encourage the development of more efficient markets, reducing the volatility observed in the spot prices for many recyclables which is a serious complication in the management of many operations.

More generally we should note that given the existence of scale effects in landfill and incineration as well as in recycling, the choice of optimal mixes for waste disposal options is likely to involve more complex modelling of cost envelopes than has so far been attempted in the literature. Most empirical work proceeds on the assumption of constant unit costs, with possibly exogenous shifts over time predicated on rates of technical innovation in waste disposal and recycling. At best, this is likely to be a local approximation only.

### 2.3.2 Externalities from recycling

Direct externalities from transport/collection and operation have to be adjusted for and also externalities from energy replacement and changed extraction rates of virgin materials. The latter are likely to be highly significant. For example, there are substantial negative externalities in metal mining, but some effects are less clear cut - the implications of paper recycling are not obvious for tree cover, for example

Thus $C_{re} = C_{ret} + C_{reo} - C_{ve} - S_{ref}$

Some "difficult" and hazardous wastes are appropriately recycled, though the costs may be relatively high, since the external costs of other disposal routes are very high and, perhaps more importantly, highly uncertain in the longer run (with, for example, ground and water contamination from landfill. However, these often require specialized handling (and therefore regulation or incentives).

Turning briefly from efficiency to distributional considerations, many developing economies are heavily reliant on extractive industries, and thus increased recycling rates may have serious consequences, both directly and through fiscal impacts. Thus the cost of implementing compensation packages (as opposed to compensation costs themselves) or developing alternative fiscal resources may be a significant issue. However, these would be very difficult to quantify and probably would be better considered as a side constraint on feasible policy choices.

Thus the (net) social costs of per tonne of waste recycled are

$$C_{rs} = (C_{rpt}+C_{ret}) + (C_{rpo}+ C_{reo}) - (C_{vp}+ C_{ve}) - (S_{rpd}+ S_{red}) \quad [3]$$

(and we assume $S_{rpd} + S_{red} > 0$).

## 2.4 An Overall Framework

We now note that some residual from recycling goes to landfill and some to incineration, and also that only a fraction of waste arisings is recycled; remember also that some incinerated waste is ultimately landfilled.

Thus *per* tonne of waste *arisings*

$$C_s = rC_{rs} + (g.r + h.b + l).C_{ls} + (k.r + b).C_{bs}^*$$

(where $r$, $l$, and $b$ are the proportions of a tonne of waste arisings going to recycling, landfill and incineration respectively, with the proportions for the latter two net of recycling residuals, g and k are the proportions of recycled waste ultimately going to landfill and incineration respectively, incineration costs $C_{bs}^*$ exclude disposal of landfill residuals).

By substituting for $C_{rs}$, $C_{ls}$, and $C_{bs}$ from the appropriate expressions [1-3] above, we can derive an accounting expression for the (net) social costs of managing a tonne of waste arisings in terms of direct private costs and externalities, and the displaced energy and material requirements of the waste management options. An indication can then be given of appropriate directions of adjustments in managing the waste stream. If there are clear unit cost differences among the management options, we can reasonably regard these as reliable indicators for local adjustments, but for larger adjustments the earlier comments on cost functions should be strongly borne in mind.

Finally, to enhance the relevance to developing economies we should note that, for reasons to be discussed in the concluding section, there is a serious problem of uncollected waste.

Thus $C_s = u.C_{us} + r.C_{rs} + (g.r + h.b + l).C_{ls} + (k.r + b).C_{bs}^*$

(where $u$ is uncollected waste, and $u$ the fraction of each tonne of waste which is uncollected,

$u + r + (g.r + h.b + l) + (k.r + b) = 1$).

This is all well and good, but serious broadly-based estimates for many of the components of this expression are non-existent, even for the EU and US. And many of the estimates for other components are not much more than guesstimates. As is often the case, there is thus a real risk of bias from over-reliance on the more secure estimates. We return to this below.

### 2.4. 1 EU Estimates

The European Commission (1996) carried out a comprehensive survey of social costs. The private rankings for 1993 averaged across the EU were (ECU/ tonne in brackets) (1 ECU = 1 Euro = $1.13):

- o Incineration (156.8)
- o Collect recycle (109.8)
- o Landfill (95.3)
- o Bring recycle (80.8)

Over time forecast rankings remained the same, except that collect recycle moved to second place by 2001 on the assumption of shifts of both technology and consumer preferences towards recyclables, and this also resulted in a sharp fall in the cost of bring recyclables relative to landfill and incineration. This ranking appears insensitive to reasonable changes in the underlying parameters, and, taking into account the ranges for environmental externalities given below, led to the conclusion that the preferred management option was recycling of appropriate materials and landfill of the remainder.

We now turn to the evaluation of environmental externalities in this study, as an example of the techniques used.

## 2.5 Evaluating Externalities: An EU Example

For the reasons indicated above, the EC study restricted itself to air pollution and to road accident costs from transport of waste. Dioxins were commented on,

since they are widely regarded as significant in some waste management options, but not quantified into the analysis, since the effects of various exposures remain contentious. Ground and water pollution (other than that arising via air pollution)and amenity loss were not considered, on the basis that results were not generalizable and that such estimates as there were derived in unsatisfactory ways. There is no reason to suppose that this will not be true for developing economies, probably a *fortiori*. The main implication of this is almost certainly to significantly undervalue the environmental costs of landfill, particularly ; if this is taken across the average of operational sites rather than newly constructed sites, which are built and operated to progressively higher standards and in less damaging locations (though this of course increases private costs and probably externalities from transport). However this . should not alter the environmental cost/benefit ranking, though it could conceivably affect the overall ranking of landfill and incineration.

For air pollution four major greenhouse gases and three major acidifying pollutants in their impact on the following receptors: human health, forests, crops, fresh waters, and buildings and materials. Standard damage cost estimates (from Fankhauser for greenhouse gases and External for acidifying pollutants) were used, with benefit transfer across the EU. (These techniques are discussed in detail in Module 1). In practice, this meant that the damage cost estimates used employed a limited range of valuation techniques. In particular, there was no use of hedonic pricing, though this would seem particularly useful in relation to t the amenity effects of landfill sites and generally to the more localised impacts of waste transfer, incineration and recycling plants. Contingent valuation was used only for a limited range of air-pollution related health impacts. For the most part, therefore damage estimates are based on dose-response with valuation by yield loss valued at market prices (crops, timber, fish), repair costs, or costs of avoiding behaviour (health, global warming). Thus even for air pollution impacts, only a comparatively limited range of costs were considered as well as a . limited range of valuation techniques within what is one of the largest scale studies of the environmental impacts of waste management available. This reflects the limited range of research which is likely to have transferable characteristics.

The waste management options (together with composting) considered in this section were evaluated. Landfill was considered with no gas recovery and with energy recovery. For both landfill and incineration, displaced energy was assumed to come either from the average EU fuel mix or from a marginal source (assumed to be coal-fired), for reasons indicated, this makes a significant difference, especially for incineration.

The system boundaries for the LCA were

- Air emission and accident costs for all stages of waste transport from waste generators to final disposal, including transport of recycled and incinerated residues to landfill: an average journey length needs to be assumed (and doubled to allow for empty return journeys) which is sensitive to area at least - this journey length could also be presumably used to proxy avoidance costs of amenity loss from site location as in a travel cost method (though this technique has not been used)
- Air emissions from processing of waste and from any associated energy production.
- Avoided air emissions from saved virgin materials and displaced energy production

In detail, the following pathways for environmental impacts were distinguished.

1. Costs associated with collection and transport of recyclables.
2. Costs associated with energy use in MSW management.
3. Costs and benefit associated with the MSW management process.
4. Benefits associated with displaced energy production.
5. Costs associated with production of plastic refuse containers.
6. Costs associated with road accidents in transporting MSW.

Thus the number of pathways considered was again fairly limited, even with the restriction to air pollutants (and road accidents).

Based on 1993 patterns of MSW management, the study concluded that for urban MSW:

- The net environmental costs of landfill were from 3 to 20 across member states.
- The net environmental costs of incineration if the displaced energy was the EU average were from 11 to 23, but there were net environmental benefits of 11 to 27 if the marginal energy source was displaced (landfill does not show a similar sensitivity since little energy is produced)
- Net environmental benefits of recycling were from 111 to 282, excluding Greece and Portugal, which had much lower net benefits of 44 and 17. respectively, for bring systems.

(Figures are ECU/tonne.) These figures do not

appear especially sensitive to forecast changes in parameters through to 2001. The orders of magnitude should be compared with the net social cost figures quoted above.

Greece and Portugal are the poorest member states and so their recycling results may be of particular relevance to developing economies. The low benefits are because

- Virgin materials displaced are not particularly energy intensive most recycling is of paper, with little metal.
- There is a high risk of transport accidents.

We would suppose that these features would be characteristic of many developing economies. However, there may be much informal sector recycling of some materials as well as "official" recycling to offset against this. Informal sector recycling is, though, usually assumed to have high accident and disease risks in the collection and processing of waste.

Generally recycling benefits are highly sensitive to the material recycled, highest for metals and glass, lower for paper and textiles, small for most plastics, and negative for some items (such as plastic film). There may, of course be technical innovation which will alter this picture, especially for plastics.

We conclude that here is a need for detailed study of ground and water pollution, especially for landfill, and for amenity loss, and to establish the scope for benefit transfer of the results. There may be scope for transferring results from other activities on which more research is available, particularly for mining which shares some characteristics with landfill (long-lasting unevenly dispersed ground and water stock pollutants and high levels of local amenity loss). Especially for amenity loss, the use of a wider range of valuation techniques, notably hedonic pricing and travel cost needs to be explored.

*Suggested Case Study 1: Examine the scope for using a variety of valuation techniques, including hedonic pricing and travel cost methods in deciding the location of a new landfill site in Sri Lanka.*

## 3 ECONOMIC INSTRUMENTS FOR WASTE MANAGEMENT

### 3.1 General Issues in the Design and Selection of Economic Instruments

Traditionally, there has been extensive use of regulation to control environmental externalities from waste management and from displaced activities. For reasons of efficiency and implementation (which may be particularly severe in some developing economies due to limited skilled personnel, corruption etc.), there has been increasing discussion and use of economic instruments to improve the management of the waste stream.

Public sector economic instruments are designed to provide fiscal incentives to the various agents involved in the waste stream to modify their behaviour to reflect the true net social costs of their actions, that is to correct distortions in private costs and to allow for environmental externalities. As suggested above, private cost distortions may be severe in developing economies with poorly evolved market institutions. This causes particular difficulties in instrument design.

There are two main categories of instrument

- Those that can modify existing market prices through the introduction of taxes, subsidies etc. on those activities.
- Those that can establish prices for waste-related activities through establishing some market process, such as auctioning permits.

In practice, the distinction between regulation and economic incentives is not clear cut

- Enforcement of regulation requires support from economic incentives such as fines, and if theses are set appropriately they should approximated the true economic incentive.
- The implementation of economic instruments requires regulatory classification and monitoring.

We should also note that

- Some taxes and charges which are primarily intended to be revenue raising may have incentive effects, though these may be weak or perverse: in particular, municipal waste charges are widely used, but these are usually at a fixed rate per household and do not distinguish mixed and sort waste and so these is no incentive to reduce waste generation or to recycle (and if there are costs to sorting and storing waste, there is a disincentive to recycling)
- The private sector has an incentive to provide markets for some types of waste: this is common for industrial waste, but may also apply to some significant parts of the MSW stream especially deposit-refund systems for beverage containers; these tend to decline with economic development as the incentives become progressively less significant, relative to incomes and costs of storage and return, and as retailing patterns change, thus there is a choice between attempting to maintain private

sector schemes and replacing them with public sector schemes - whatever, allowing schemes to lapse then imposes costs of reintroduction.

There is a wide range of possible economic instruments for waste management. Internationally, those implemented or under active consideration include the following

- Raw material charges to discourage use of virgin materials, such as levies on quarried aggregates.
- Enterprise taxation to encourage recycled input use, such as differentiated corporate tax rates to reflect proportions of recycled input or accelerated depreciation of investment in recycling plants.
- Product charges to discourage high waste content, such as packaging levies.
- Sales taxes differentiated to reflect recycled content.
- Waste collection charges at point of collection, possibly differentiated to encourage recycling, such as pay-by-the-bag charges for unsorted waste.
- Waste disposal charges at point of disposal, perhaps differentiated by disposal route or type of waste, such as landfill taxes.
- Charges on disposal-related externalities, such as emissions charges on incinerators
- Deposit-refund systems, these may be particularly appropriate for some types of"difficult" and hazardous waste, to encourage return for recycling or appropriate processing prior to disposal.
- Recycling subsidies to operating or capital budgets.
- Subsidies or tax-breaks on waste management research and development.

These are more or less complex in their implementation requirements and in the sophistication required of waste-management agents in understanding the incentives they are presented with, and thus require particularly careful evaluation in developing economies

Turning explicitly to environmental externalities, the simple principle is to internalise the effects by taxing the activity with net external costs or subsidising the activity with net external benefits or some mix of the two. It is important not to double count. Suppose, for example, that landfill disposal is to be reduced by a recycling programme. The same socially efficient outcome can be secured either by taxing landfill by an amount equal to the net external benefit of switching from landfill to recycling or by subsidising recycling by the same amount. A combined tax and subsidy can be used as long as the difference in the price of landfill and the price of recycling is adjusted by an amount equal to the difference in external costs of the two methods.

In practice, the budgetary implications of taxes and subsidies clearly differ, and especially for developing economies which are severely fiscally constrained this is an important consideration. A common solution is to combine taxes and subsidies, with taxation of negative externalities used to at lest partially finance subsidies on positive externalities, isolating, at least partly, the introduction of wastemanagement instruments from the wider budget. This can also enhance their administrative and political acceptability.

It is important that there is a proper separation of fiscal responsibilities aligned with agencies responsible for managing different stages of the waste stream, so that, say, charges levied by disposers are not credited in the same account as that debited by collectors. This is a common problem with over-arching municipal authorities.

## 3.2 A Model of Optimal Instrument Choice

(This section is based on Menell (1990)). Suppose a product can be supplied in two types of packaging, only one of which can be recycled. The product is identical and marginal production costs differ only because of the packaging p(i) where i =1,2 for non-recyclable and recyclable packaging, respectively, there are no relevant uncompensated externalities in production.

The utility of consumption reflects preferences between different types of packaging: assume all consumers get the same utility (u) from the product when supplied in non-recyclable packaging and the utility from the product in recyclable packaging is

$$u+r$$

where r can be negative if the non-recyclable packaging is preferred.

Consumers have different costs of sorting and storing recyclable packaging, which consists of a constant (d), which may be zero, and a variable based on household characteristics of c.

$$d+c$$

We assume that

- Unsorted recyclables are not recycled.
- r and c are independently and uniformly distributed between given high and low values.

Marginal social disposal costs are s(i) where i

=1,2,3 for non-recyclable packaging, for recyclable packaging not recycled, and recyclable packaging recycled.

Thus on social cost grounds a consumer should buy a product in recyclable packaging if

$$u + r - p(2) - \min (c + s(3), s(2)) > u - p(1) - s(1)$$

or

$$r > (p(2) - p(10)) + \{\min(c + s(3), s(2)) - s(1)\} \quad [4]$$

A consumer who sorts saves society

$$s(2) - s(3)$$

and should do this if

$$d + c < s(2) - s(3)$$

From this we can derive optimal instruments. Charges unrelated to waste disposal by households, as are common, provide no incentive. Consumers only sort if they derive satisfaction from doing so satisfies

$$c < -d$$

and there is no incentive to consider social disposal costs: in [4]

$$s(i) = 0, \text{ all } i$$

We now consider collection charges in some detail, and product charges more briefly, to analyse the setting of instruments to encourage recycling.

Now consider collection charges. A perfect charge would charge consumers the full social cost of disposal, and then consumers would make both purchasing decisions and sorting decisions in conformity with [4]. However the implementation costs of this would be very high. A simpler system is to charge for volume of unsorted waste. Assume that the two types of packaging have the same volume that consumers cannot modify differentially. Then they will buy recyclable packaging if

$$r > (p(2) - p(1)) + \{\min (c, t.v(2) - t.(v(1)\}$$

and they will sort if

$$c > t.v(2)$$

Where t is the charge on volume and v(i) is the volDe of waste (t = 1,2, non-recyclable and recyclable respectively). Thus t has to be set to approximate as closely as possible the decision made on the basis of [4].

Mandatory separation, as often advocated is inferior to this. Consumer will only purchase recyclables if

$$r > (p(2) - p(1)) + c$$

and thus there is less incentive to purchase in the first place, and this will also reduce incentives to innovate in recyclables.

Product charges cannot directly influence disposal choices but do influence purchasing choices. The optimal charge reflects the difference between the social cost of disposing of the non-recyclable and the recyclable weighted by the probabilities of the recyclable being recycled or not. So the optimal charge is,

$$t(1) - t(2) = s(1) - \{h.s(3) - (1-h).s(2)\}$$

where h is the probability of recycling.

And recyclables will then be purchased if

$$r > (p(2) - p(1)) + (t(20 - t(1)) + \min (d + c, 0)$$

Product charges would have longer run effects in encouraging both producer and consumer learning about recyclables.

Other economic instruments can be analysed within this framework using similar procedures. Essentially, using the types of information discussed in Section 2 to establish social costs, the framework indicates 15 the other parameters necessary to set each instrument most efficiently 171 and then provides a ranking of these instruments. It is then possible to compare these with implementation costs (broadly defined to include the type of criteria outlined in section 1 )

## 3.3 An Example of Economic Instruments for Waste Management: The UK Landfill Tax and Recycling Credits.

Time-series and cross-section evidence suggests that landfill availability and prices are one of the main drivers of recycling rates. We have also suggested above that the externalities form landfill have probably not been fully accounted for, even in quite sophisticated studies. Many OECD economies have therefore introduced landfill charges or are considering them. They are becoming a major economic instrument for waste management. The UK introduced a charge on disposals to landfill in 1996, initially at rates of 3 ECU for inactive wastes and 11 ECU for other wastes, since uprated. Most wastes are charged at the higher rate.

These initial rates were low in relation to the then (conservatively) estimated environmental damage from landfill in the UK, primarily to allow for other more desirable waste management options to develop sufficient capacity. In particular, unless appropriate charges were levied elsewhere, a likely effect was a switch to incineration or to export of waste. (in fact, the latter has occurred.) Both may be environmentally undesirable. This is of relevance to developing economies, which are likely

to be even more seriously capacity-constrained in waste-handling: in particular the already serious problem of uncollected waste could be exacerbated and there could be regional dumping.

Probably the most serious implementation problems are those connected with illegal tipping, which is reported to have increased sharply. Given the relatively high standards of monitoring and enforcement in the UK, this would obviously also be a major concern for developing economies.

There is also a trade-off between the need to distinguish the damage from different types of waste and sites in the charge scales, and the increased opportunities that this creates for wrongly classifying or directing waste to minimize charges unless there is still more intensive monitoring. (For this reason, the UK rejected an initial proposal for an *ad valorem* charge on disposal costs.) This again is likely to be a serious problem for developing economies, particularly since they have operational sites constructed and operated at highly variable standards and thus imposing much more variable environmental loads than is common in the UK. Work on waste management in the Slovak Republic, which has a complicated charge scale that attempts to distinguish the standards of sites this to be a problem (European Commission 1997).

Finally, a problem that has caused serious diffficulties in implementing he UK tax is the need for good standard sites to use inactive materials for site construction and management. If these are taxed, there is an incentive to purchase virgin materials for this purpose, which is finally, a problem that has caused serious diffficulties in implementing the UK tax is the need for good standard sites to use inactive materials for site construction and management. If these are taxed, there is an incentive to purchase virgin materials for this purpose, which is environmentally perverse, but if they are not, there is an incentive to overstate the requirements. Since developing economies have an urgent need to improve standards of site construction and maintenance, this is again likely to be a serious issue for them in implementing landfill charges.

As suggested above, one of the major issues in managing landfill is postclosure contamination. There is thus a need to develop instruments that attempt to deal with this, such as refundable performance bonds, which have been discussed in the mining industry for comparable reasons. It is possible to consider a system where landfill licences are sold at prices approximating forecast life-cycle environmental costs. These could be tradeable (offering well-known efficiency gains in providing incentives to achieve environmental standards at minimum cost) and deposited to provide a performance bond.

There are a number of routes by which potential revenues from the landfill charge can be routed to provide support for more environmentally desirable management. These include

- Part of a landfill site can be tax-free if used to sort, recycle or store waste for recycling within a year; only the residual is then charged.
- Part of the tax liability can be claimed as a tax credit for approved donations to environmental bodies. These can include land reclamation, provision of environmental amenities within the site vicinity, and environmental research and education. There should be no financial benefit to the site operator.
- Most importantly, recycling credits were introduced prior to the landfill charge and enable recyclers to claim credits equal to half the cost of the most expensive avoided disposal option. The landfill charge obviously enhances this, increasing the chance that is the most expensive option and if it already is, increasing the value of the credit. Bearing in mind the comments made above about fiscal containment, it would be sensible to consider the introduction of a landfill charge and recycling subsidies simultaneously.

Unless landfill charges are passed through to waste generators in such a way as to change the incentives they face in generating or disposal choices (as analysed above), the effects are limited. The main impact is on the budgets of waste collectors: in fact, in developing economies with tight fiscal constraints on collectors, this effect could be substantial (though not entirely predictable, depending on the relative costs of the various disposal routes used), but, again, the problem of uncollected waste could be exacerbated. (It is important to refer to our earlier comments on separation of budgets of collectors and disposers.)

*Suggested Case Study 2: Design and analyse an appropriate landfill charge plus recycling subsidy for waste management in Sri Lanka.*

## 4 SOME PROBLEMS OFWASTE MANAGEMENT IN DEVELOPING ECONOMIES

Throughout we have suggested points of particular relevance to developing economies, particularly when drawing on work carried out for other economies. Accessible studies for developing economies are fairly

limited in number and variable in quality, the most substantial probably being for the waste paper sector in India (for example, van Beukering, Schoon, and Mani 1996). In this section we draw on this to outline an approach to waste management in developing economies which highlights some key features.

In spite of the generally high proportions of municipal budgets spent on waste disposal, fiscal constraints mean that high proportions (generally around one half) of urban waste are uncollected. The social costs of this must be substantial, in terms, especially of disease, but also through air pollution from burning of refuse, etc. as well as general loss of amenity.

Informal recycling is extensive, accounting for 1-2% of the workforce in developing cities. It relieves both municipal budgets and waste disposal capacity, as well as providing employment and incomes for low-skill workforces. However, there are probably significant externalities involved, especially from accidents and disease in the work force.

For the moment, ignore externalities. Denote net marginal revenue from recycling (assumed all to be informal (or, at least private sector) by Rr which declines with the percentage of waste recycled and eventually becomes negative. Disposal is assumed to always have positive net marginal cost which rises with percentage of waste disposed of. When

$$-Rr = C_{lp}$$

there is no uncollected waste

But informal recyclers chose

$$R_r = 0$$

Which is at a lower level of recycling (h). If (1-h) is disposed of by the municipality, its marginal cost is

$$C_{lp}^{*} > C_{lp}^{\wedge}.$$

and there is no uncollected waste.

If municipal budget (or capacity) constraints mean that waste is disposed of at a level corresponding to

$$C_{lp} < C_{lp}^{*}$$

then there is uncollected waste. One solution to this is then to search for efficiency gains in the municipal sector. Thus far, though, we have not explicitly attributed any cost to uncollected waste.

So suppose now that there are marginal externalities, always negative in disposal, but less than from uncollected waste, and that the externalities in the informal sector may be negative or positive, depending on the balance between those discussed in section 2 and those mentioned just above.

$$C_{ue} > C_{ie} > C_{re}$$

The social optimum now requires

$$-R_r + C_{re} = C_{lp} + C_{le} = C_{ue}$$

Ignore $C_{re}$ for the moment, then this implies higher levels of recycling (i.e. a higher negative value of marginal revenue in recycling) than would occur in the no-uncollected waste example above, but there would be some uncollected waste unless $C_{ue}$ is so high that it does not figure in the solution. If $C_{re}$ is negative, this implies higher levels of disposal and uncollected waste than would otherwise be the case, and vice versa.

More information is needed on the externalities associated with uncollected waste and with informal recycling activities. However, the foregoing provides a basic framework for analysing these issues.

One standard policy response to inadequate recycling rates is to shift recycling on to a more formal basis, partly because of the negative externalities involved, and partly to increase recycling rates. Two important issues are that,

- Reuse rates, for example of newspaper for packaging are high at low income levels, these decline with rising incomes, and informal sector recyclers are not able to recycle as opposed to supply for reuse.
- What we have referred to as informal sector really decomposes into the truly informal sector of waste-pickers and the more formal activities of traders. More formal "Western" type recycling is predicated on high levels of voluntary activity in sorting waste by households. It is not obvious such voluntarism can be relied on in developing economies, rather than the incentive offered by even the very low incomes of waste-pickers The activities of local traders may be important in smoothing the operation of markets, whereas the larger scale recycling operations are vulnerable to market instability, especially if they operate internationally.

Some of this has important income-distributional implications. More formal systems are less labour intensive, and thus with overcrowding in the informal sector from their adoption, the income of the poorest households involved in the waste industry declines in the short run. In the longer run, if recycling replaces still less labour-intensive disposal such as landfill, then the situation reverses. Thus there are complex intratemporal and intertemporal distributional trade-offs, to be weighed against possible effficiency gains, and these should be investigated for cities other than Bombay.

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