SUSTAINABLE DEVELOPMENT: ECONOMICS AND MEASUREMENT

Giles David Atkinson

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Department of Economics
University College London
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To Susana and to Nicholas

Abstract

This thesis is an investigation of the issues that arise in the measurement of sustainable development. It uses as its over-arching framework the theory of extended or green national accounting but this is augmented by other methods and techniques.

In Chapter 1, we introduce the concept of sustainable development and discuss proposals regarding how sustainability is to be achieved and measured; in particular, the genuine savings rate (savings net of asset consumption) and its caveats.

In Chapter 2, we extend the analysis of genuine savings to the depletion of forests in the developing world. We examine both in theory and in practice the accounting problem for the conversion of forest land to slash-and-burn agriculture in the Peruvian Amazon.

In Chapter 3, we consider the effect of international trade on sustainable development. By modelling these trade flows using Input-Output analysis we identify countries that can be characterised as net producers or net consumers of (global) resources.

In Chapter 4, our starting point is the proposition that if corporate sustainability has relevance to the wider sustainability debate, it is via corporate full cost accounting. We illustrate one candidate indicator that can provide the desirable signal that as the firm's external impact diminishes the more 'sustainable' it is indicated to be.

In Chapter 5, we examine the so-called "resource curse hypothesis". We investigate this question using cross-country regressions and offer statistical support for the view that countries where growth has lagged are those where there is a *combination* of natural resource abundance and deficient public policy.

In Chapter 6 we turn to the notion of strong sustainability. In particular, we evaluate candidate indicators that have been proposed to reflect this concern via a specific focus on the conservation of natural assets.

Finally, in Chapter 7 we explore the meaning of equity in an intragenerational context. Using a contingent ranking methodology, trade-offs between competing notions of environmental equity are investigated. The results indicate that these trade-offs do exist and, furthermore, are significant.

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Declaration

1. No part of this thesis has been presented to the University for any degree.

2. Chapter 2 was undertaken as joint work with Kirk Hamilton. At least 50% corresponds to my own work.

3. Chapter 7 was undertaken as joint work with Fernando Machado and Susana Mourato. At least 50% corresponds to my own work.

Signed ______________________ **Professor David Pearce (Supervisor)**

Chapter 1

Introduction

The call for countries to pursue policies aimed at achieving 'sustainable development' was established, both in the Brundtland Report (WCED, 1987) and at the Earth Summit in 1992. Sustainable development has since been adopted as a goal of economic and social development by international agencies, individual nations, local governments and even corporations and has further generated a huge literature. The transformation of these pledges into policy is a formidable challenge faced by decision-makers. It also raises fundamental questions regarding the design of indicators needed to construct suitable policy responses and, furthermore, to monitor progress towards stated goals. In this latter regard, this thesis is primarily an investigation of many of the issues surrounding the *measurement* of sustainable development.

The over-arching framework in this respect is the theory of extended or green national accounting but this is augmented, where appropriate, by other methods and techniques. Hence, the approach that we take is primarily an economic one. Arguably, economists have provided the most consistent and coherent 'story' about sustainable development: what it is, what the conditions are for achieving it, what might have to be sacrificed to obtain it and how it can be measured. Nevertheless, it is important to recognise that interest in sustainable development represents a 'broad church', although the debate continues as to how to accommodate these diverse contributions within a more pluralistic or multidisciplinary approach.

Our pursuit, in this thesis, of further understanding of a range of issues concerning the measurement of sustainable development leads down a number of paths. In each chapter we endeavour to underpin this empirical work with existing conceptual frameworks such as the theory of green national accounting. While this leads to a focus on empirical estimates of extended national accounting aggregate indicators, we also find a role for input-output analysis and econometric analysis in addressing the sustainability measurement problem. The various chapters of the thesis represent distinct research questions and, therefore, can be read separately. Nevertheless, they also form a coherent and consistent whole.

In the remainder of this introductory chapter we review the major elements of the theory and measurement of sustainable development as well as outlining some caveats to the approach that we take. In doing so, we provide a rationale for the main issues discussed in subsequent chapters. These issues are: accounting for (tropical) deforestation, constructing indicators of international trade in resources, corporate sustainability, the relationship between resource abundance and economic performance, strong sustainability as well as an investigation of public preferences for competing principles of environmental equity.

1.1 Definition of Sustainability

The concept of sustainable development itself has been defined in a number of ways since the early shaping of the problem well over a decade ago. One of earliest contributions to this debate, the Brundtland Report, defined it as "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs" (WCED, 1987, p43). Economists have tended to reinterpret this as a requirement to follow a development path where human welfare or wellbeing per capita does not decline over time (see, for example, Pezzey, 1989). A somewhat different focus is proposed by Barry (1999) who argues that sustainability should be seen in terms of not allowing development *opportunities*, available to future generations to decline. It is worth noting that others have made it clear that the emphasis on future generations was only part of the story. For example, the Brundtland definition of sustainable development embodied concern with the poor now (WCED, 1987). These questions pertain to the distribution of welfare or opportunities *within* each generation: i.e. intragenerational equity.

Discussion of the meaning of sustainable development raises interesting questions about what we understand by inter- (and intra-) generational equity. From a practical perspective however, it is whether the broad *conditions* for sustainable development vary with the definition that is a concern (Pearce, 1998). In this respect, these conditions are arguably broadly similar whether couched in terms of opportunities or welfare consequences in that each implies a similar focus on how a portfolio of assets is to be managed over time.[1] This is also relevant to intragenerational concerns if this analysis is extended to questions about access to (and distribution of) productive capacity. In this thesis, however, we define sustainability as non-declining welfare (per capita). In other words, a development path is unsustainable if (at least) the level of current welfare (per capita) cannot be reproduced for each successive generation. Hence, we take future generations as our main focus. Nevertheless, in chapter 7 we do explore in greater detail the meaning of environmental equity within a generation.

1.2 Theory and Measurement of Sustainable Development

By examining theories of sustainable development, competing claims regarding the sacrifices required to make good our obligations to the future can be scrutinised. Although these theories are typically derived from very different world views, common to all is an emphasis on environmental resources as a component of wealth – i.e. natural assets (or capital). This insight necessitates not only an extension of the asset boundary, in accounting terms, but also an emphasis on how these assets are managed.

Broadly speaking, there are two distinct characterisations of the basic conditions required to achieve sustainability. For weak sustainability, the appropriate reference point is one where, on balance, assets are neither created nor destroyed. In contrast, strong sustainability stresses 'limits' to the deterioration of either all or (certain) *critical* natural assets. The rationale for this focus is both the complexity of ecosystems and the absence of substitutes for functions of natural assets such as ecological system maintenance (Norton and Toman, 1997). Farmer and Randall (1998) propose a two-tier approach to the sustainability problem capable, in principle, of reconciling these two theories. This is based upon the concept of a safe minimum standard (SMS) whereby policy-makers follow standard cost-benefit rules unless there is a compelling reason not to; e.g. to conserve a critical natural asset. However, this conservation rule can itself be overridden if its costs are "intolerable". While this

[1] Some contributions do appear to require a change of emphasis, most notably those definitions that focus upon procedural concerns such as the requirement for greater public participation in decision-making (Toman, 1998).

approach is appealing, in practice, little is known in general as to when this policy switch should be made. We do not intend to resolve this debate in this thesis and instead we focus primarily (but not exclusively) on the measurement of weak sustainability in providing a first approximation to understanding the sustainable development problem. However, we comment on the measurement of strong sustainability where appropriate.

For weak sustainability any one form of asset can be run down provided 'proceeds' are reinvested in other forms of assets. Put another way, it is the 'overall' portfolio of wealth that is bequeathed to the future that matters. Hartwick (1977) showed that achieving sustainability[2] required as a 'rule of thumb' that some of the proceeds (specifically, the total rents) from the exploitation of non-renewable resources be reinvested in produced assets. 'Hartwick's rule' states that sustainable development is feasible even if there is an initial reliance on non-renewable resources.[3] This result has been extended to consider the role of renewable resources and changes in environmental liabilities (Hartwick, 1978; Mäler, 1991; Hamilton, 1995d). Finally, Solow (1986) has shown that Hartwick's rule amounts to holding the aggregate stock of capital intact and treating consumption as the interest on that stock. Hence, it is by following an *extended* Hartwick rule that the basic requirement for (weak) sustainability, that the change in the (real) value of assets should not be negative in aggregate, is fulfilled.[4] Put another way, there is no specific focus or special place for the environment and as such there are no particular things that we owe to future generations. It is by passing on some generalised productive potential, broadly construed, that will ensure that welfare is sustained now and into the future (Solow, 1993).

One of the salient links between the discussion of the conditions to achieve sustainable development and the debate regarding how to actually measure sustainability is the theory of green national accounting. The fundamental idea is that the asset boundary in the national accounts should be extended to include natural resources and environmental liabilities in the form of pollution stocks. Hence, depletion of resources and degradation of the environment represents asset

[2] This is defined in Hartwick (1977) as constant consumption.
[3] Dasgupta and Heal (1979) proved that if the elasticity of substitution (between natural and produced assets) is less than 1 then the Hartwick rule is not feasible.

consumption, i.e. depreciation of natural assets, a concept ideally suited to treatment within the national accounts. This insight has been useful as a basis for efforts to construct green alternatives to existing accounting aggregates such as Gross National Product (GNP). Indeed, much of the impetus for defining new green national accounts aggregates can be traced to Hicks' (1946) concept of income: true income is that income in excess of asset consumption. There are now numerous empirical studies that attempt to construct green national accounts in the presence of non-renewable resources (e.g. fossil fuel and mineral resources), living resources (e.g. forests and fisheries) and environmental liabilities (e.g. air pollution) (see, for a review, Hamilton and Lutz, 1996; Hanley, 2000; Atkinson, 2000). In this way, it is hoped that this new information might be provided alerting policy-makers to whether an economy is on or off a sustainable development path.

Pearce and Atkinson (1993) introduced one such measure of sustainable development based on a *net savings* criterion. This indicator can be interpreted as an adjusted national savings measure that accounts for the depletion of natural resources and environmental damage. Hamilton (1994, 1996a) developed welfare measures to account for living and non-living resources and varieties of pollutants in a similar extended national accounting framework – the corresponding savings measure, equivalent to that of Pearce and Atkinson, was termed 'genuine' saving. This is simply the net saving rate in this extended accounting framework where the term 'genuine' is used to provide a distinction from standard national accounting terminology. Genuine savings represent the amount of saving over and above the value of asset consumption – i.e. the sense in which such savings are 'genuine' (Atkinson *et al.* 1997). For a definition of sustainable development as a development path along which welfare does not decline, this definition will be satisfied as long as net savings are not negative (Pearce *et al.* 1996).

The basic framework that underpins the analysis of income and saving in the presence of (natural) asset consumption is the theory of extended or green national accounting. Reference to this framework adapted from Hamilton *et al.* (1997), in the context of the present discussion, allows a synoptic outline of a result with regard to savings and

[4] This assumes constant technology and population.

sustainability. In common with Weitzman (1976), Hartwick (1990) and Mäler (1991), national income is defined along the optimal path of a growth model for a simple (closed) economy. We assume a simple economy in which a composite good can be consumed, invested or used to abate pollution. Welfare U for the representative individual in this economy is a function of consumption and some number of stocks of living and non-living natural resources and pollutants, with the stocks of natural resources (e.g. forests) generally adding to welfare and the stocks of pollutants decreasing it. The measure of wealth W for this society is just the present value of welfare on the optimal path over an infinite time horizon. Wealth at time t is defined as,

$$W_t = \int_t^{\infty} U\big(C(s), M(s)\big)e^{-r(s-t)}ds \tag{1.1}$$

In this expression U is utility or welfare, C is consumption and M is a vector of stocks of natural resources and pollutants that may add to or subtract from utility. The pure rate of time preference r is assumed to be constant.

National income or Net National Product (NNP) is defined as the maximum amount of produced output that may be consumed while leaving wealth instantaneously constant (Pemberton and Ulph, 1998). Hence, the framework is "extended Hicksian," where income is defined as the maximum amount that could be consumed while still leaving oneself *as well off* at the end of the accounting period as at the beginning. Being 'as well off' in the current context implies maintaining the level of the present value of welfare along the optimal path. National income can be written,

$$NNP = C + G = C + \dot{K} + \gamma \cdot \dot{M}\,. \tag{1.2}$$

where γ is the vector of shadow prices for the environmental stocks that support the path which maximises wealth: i.e. scarcity rents in the case of natural resources, marginal damages in the case of pollutants.[5] The interpretation of NNP in expression

[5] Strictly speaking, this is the expression for genuine saving for an autonomous problem. This approach generalises to the non-autonomous case (for exogenous technological change, for instance) in a straightforward manner.

(1.2) is relatively straightforward; it is equivalent to consumption plus the net change in the economy's assets. The latter corresponds to genuine saving in this model.

$$G = NNP - C = \dot{K} + \gamma \cdot \dot{M}. \tag{1.3}$$

Hence, in expression (1.3) genuine saving G is the difference between NNP and consumption and is also equivalent to the sum of the net investment in produced assets and the changes in the various stocks of natural resources and pollutants all valued at their respective shadow prices.[6] As long as the rate of time preference is constant it follows that

$$U_C G = \dot{W} \tag{1.4}$$

In other words, the change in total wealth is equal to the genuine saving rate valued at U_C, which is the marginal utility of consumption – the shadow price of capital in this model. The implication of the result in expression (1.4) is that negative genuine saving results in a decrease in wealth. This leads in turn to declining welfare at some point in the future and according to our definition is not sustainable. It is worth noting that this result holds in both optimal and non-optimal growth models (Hamilton and Clemens, 1999; Dasgutpa and Mäler, 2000).

This provides a theoretical rationale for interest in genuine saving rates. Just as importantly, data now exists to measure genuine saving across countries on a consistent basis. In this respect, estimated rates of genuine saving for a broad range of countries and regions are now routinely published by the World Bank (e.g. World Bank, 1999). The most interesting finding is that many countries appear to exhibit negative genuine savings over the periods for which data are now available. More generally, within the green national accounting literature, the focus on empirical applications has spurred on theorists such as Aronsson *et al.* (1997) to clarify a number of outstanding theoretical points. In turn, many of these conceptual

[6] This framework can be extended to consider, in addition, investment in human capital (see, for example, Atkinson *et al.* 1997).

developments have spurred improved empirical work. Nevertheless, a number of other contributions do constitute important caveats relevant to the savings analysis.

Firstly, it should be noted that genuine saving measures the value of the net change in an economy's wealth. To the extent that population is not constant (e.g. there is population growth) a more satisfactory indicator of sustainability is the change in (total) wealth *per capita.* This is self-evidently important for many developing countries with rapid population growth. However, Hamilton (2000b) finds that it is even important for economies such as the United States. Secondly, the genuine savings rate is a one-sided indicator of sustainability. That is, while it can be shown that (a point measure of) *negative* genuine savings will lead to non-sustainability, observing a point measure of *positive* genuine saving does not indicate that an economy is on a sustainable path (Asheim, 1994; Pezzey, 1994). Further evidence must be sought before a judgement can be made in this respect.[7] However, the policy prescription arising from measuring a negative genuine savings rate is clear: continuing dissaving is not sustainable and must be rectified (Atkinson *et al.* 1997). Thirdly, there are clear difficulties encountered in trying to construct an indicator that provides information about development prospects into the (far-off) future. For example, a crucial determinant of the ability to sustain development is technological change and this is neglected in most simple models.

In this respect, Weitzman and Löfgren (1997) have argued that any proposal to construct an estimate of green NNP (or genuine saving) needs, in addition, to include an annuity term reflecting the dollar value today of future technological change. Moreover, it is argued that for the US, the value of this *technological premium* could be as much as of 40% of GNP. If so, then in practical terms, it is arguable that improvements in technology will ultimately sustain development. However, this optimism rests crucially on a number of accounting issues. What must be ascertained is how much of this premium is currently reflected in the national accounts (e.g. via investment in research and development) and much how is not 'bought' and 'sold' in markets in this way (Pemberton and Ulph, 1998). If economic resources are used to

[7] **These contributions show that for an economy where there is a single exhaustible resource and no pollution, the optimal path is not sustainable, while the measure of genuine saving 'early' in the extraction program for the resource is positive.**

create technological improvements then much of the value of these resources is already captured by existing accounting aggregates. Put another way, if technological change is an endogenous process then, for example, estimates of genuine saving may not need a further technology premium to be imputed.[8]

These caveats illustrate that implementing any sustainability measure remains a complex task – indeed, it would be something of a surprise if this was not so. Nevertheless, it seems reasonable to conclude that the concept of genuine savings and, more generally, green national accounting has given a new focus to the question of how much net wealth is being created and, critically, how domestic savings levels compare with the consumption of a country's assets. Moreover, these concepts and tools also provide potentially useful signals for governments committed to achieve sustainable development. However, green national accounting can be usefully complemented with other approaches relevant to the sustainability problem. For example, while national accounting is ideally suited to the design of national indicators, the search for 'comparable' indicators at different levels of aggregation may necessitate a different approach or framework. Moreover, sustainability indicators are not a substitute for policies to achieve sustainability and it is important to understand the factors that might mitigate against the introduction of these policies.

1.3 Chapter Summaries

In the following chapters to this thesis, we consider a number of empirical issues surrounding the concept of sustainable development or sustainability. Firstly, we consider how asset consumption can be defined in the presence of (tropical) deforestation (Chapter 2). While this investigation is undertaken within an extended *national* accounting setting, two subsequent chapters examine the accounting issues that arise with regard to the measurement of linkages with respect to international

[8] Nevertheless, even if technological improvement is wholly endogenous, existing accounting practice is likely to be inadequate in capturing its full value for two reasons. Firstly, the standard national accounting treatment of research and development (R&D) expenditure is to classify it as intermediate or final consumption (United Nations, 1993) whereas these expenditures are more appropriately treated as an investment. Secondly, where knowledge is a public good, the value of the resources devoted to R&D is likely to be less than is socially optimal. The implication is that the full value of (endogenous) technological change will not be reflected in standard national accounting practice.

trade in resources (Chapter 3) and the external costs or damages associated with the generation of corporate income (Chapter 4).

These chapters therefore extend the (genuine) savings analysis (or related concepts) in a number of ways. Chapter 5 explores this further by examining a policy-oriented question relevant to the sustainability debate. This chapter uses cross-country regressions to ask why resource abundant countries appear to have experienced relatively disappointing economic performance.

While Chapters 2 to 5 take weak sustainability as a starting point, Chapter 6 examines the alternative notion of strong sustainability and specifically the candidate indicators that have been proposed to reflect this concern via a more specific focus on the conservation of natural assets.

Finally, in Chapter 7 we move beyond green accounting to explore the meaning of equity in an intragenerational context. Specifically, by using a contingent ranking methodology, we examine trade-offs between competing notions of environmental equity.

Chapter 2: Sustainability, Green National Accounting and Deforestation

This chapter extends the savings analysis to a domain that seems both particularly topical and important, the depletion of forests in the developing world. To the extent that forests have been considered within extended accounting frameworks, the change in asset value has been conceived primarily as a net change in timber resources. Clearly, for many countries, how the rents arising from the depletion of timber resources are managed is an important determinant of sustainability. However, this emphasis seems to be unduly narrow if forest clearance results in a wider range of asset values, such as (non-market) local and global conservation benefits, that are lost in perpetuity.

In chapter 2, we capture this range of (net) costs within an extended national accounting framework. One crucial component of this accounting problem is a stock change that we characterise as the total value of 'excess deforestation'. This is defined as the amount of permanently cleared land on which the return to agriculture is less than the sum of the values of sustainable timber harvest and carbon sequestration, local and global willingness to pay for conservation. This framework is applied in a case study of Peru, in order to examine this accounting problem in the context of switching of forest to slash-and-burn agriculture in the Peruvian Amazon. Our results show that defining asset consumption (arising from deforestation) broadly, new insight can be gained regarding the sustainability of the development process.

Chapter 3: International Trade and the "Ecological Balance of Payments"

The standard extended or green national accounting problem assumes (implicitly or explicitly) a closed economy. However, a criticism made by Martinez-Alier (1995) is that (weakly) unsustainable countries apparently tend to be located in the developing world, whereas the resources that these countries extract are often traded with developed countries. The specific question raised by Martinez-Alier is whether a full "ecological balance of payments" analysis would show that the US and Japan, which exhibited positive genuine savings in the analysis of Pearce and Atkinson (1993) were actually unsustainable when global resource flows are taken into account.

On one hand, it is unclear why the savings rate of a resource importing country should be reduced to reflect the depreciation of an asset that it does not own. On the other hand, strong demand for the natural resources of an exporting country could plausibly lead such a country down an unsustainable path (although only if its own policies are deficient). If it is reasonable to speculate that importers are concerned about the unsustainable behaviour of resource exporters then modelling these interdependencies is of interest. In chapter 3, we model these flows using Input-Output analysis and trace these resource trade linkages. Furthermore, we identify those countries in the global trading economy that can be characterised as net producers or net consumers of (global) resources.

Chapter 4: Measuring Corporate Sustainability

Although sustainable development has been interpreted primarily as a national (or global) goal, there is increasing discussion of 'sectoral sustainability' and 'corporate sustainability'. While it can be argued whether or not these concepts have any literal interpretation, chapter 4 proposes that if corporate sustainability has relevance to the wider sustainability debate, it is via corporate full cost accounting. For example, to the extent that external costs or damages are directly associated with the generation of a corporate entity's income, the environmental or full cost account of that entity should reflect the magnitude of the damage caused.

This reasoning allows the development of indicators that are comparable to measures of net or 'genuine' saving estimated at the level of the national economy. An intuitive interpretation is that a business is, at least notionally, unsustainable to the extent that adjusted profit or corporate genuine saving is less than zero. Judged over time, this indicator provides the desirable signal that as the firm's external impact diminishes the more 'sustainable' it is indicated to be. This, we argue, provides one reasonable link between an understanding of 'corporate sustainability' in terms of the corporate entity itself and in the contribution of the entity to sustainability in a wider sense.

Chapter 5: Savings, Growth and the Resource Curse Hypothesis

The theory of sustainability suggests that resource abundant countries should invest (some portion of) the proceeds of depletion in other forms of wealth if a sustainable path is to be attained. However, studies such as Gelb (1988) and Auty and Mikesell (1998) have shown how difficult prudent resource and public investment policies can be to achieve. Other studies have posited an apparently negative relationship between resource abundance and economic performance, contrary to intuition. This is the so-called "resource curse hypothesis".

We investigate this question using cross-country regressions and confirm, using new data on resource abundance, a negative (and significant) relationship between natural resources and economic growth. More interesting is the reason for the curse; that is, what is the explanation for the inability of resource abundant countries to manage large resource revenues? We offer statistical support for one perspective on the resource curse, which is consistent with the parallel discussion of sustainability. This is that countries where growth has lagged are those where the *combination* of natural resource, macroeconomic and public expenditure policies has led to a low rate of genuine saving.

Chapter 6: Strong Sustainability and Indicators

In this chapter, we examine the sustainable development problem from the perspective of, for example, ecologists and ecological economists who have taken a rather different approach: strong sustainability. This stresses 'discontinuity' and 'non-smoothness' in ecological systems and hence in the (economic) damages to which ecological impairment gives rise (Pearce *et al.* 1996). In turn, this school of thought places a greater emphasis on the conservation of natural assets within the broader goal of prudently managing a portfolio of assets over time.

Very few advocates of strong sustainability argue that all natural assets must be conserved. More usually, it is argued that there are *critical* natural assets, crucial for human welfare, that have no substitutes and therefore cannot be traded off for other forms of wealth (Pearce *et al.* 1989). A distinct concept is 'resilience'– the ability of an economic, ecological and social system to 'bounce back' from shocks (such as outbreaks of disease in agricultural systems) and to persist despite continuous stresses (Common and Perrings, 1992). It is clear that many notions of strong sustainability are suggestive of relatively sophisticated indicators that are demanding with respect to data needs. However, few practical indicators of strong sustainability exist and the applications that we illustrate in this chapter are primarily restricted to simple indicators of resource scarcity in the context of agricultural production, fuelwood and water supply.

Chapter 7: Balancing Competing Principles of Environmental Equity

Diverse and competing principles of equity have been proposed as relevant to the design of policies to move economies towards sustainability and environmental improvement. For example, the numerous definitions of sustainable development that can be found in the literature, in part, reflect competing notions of intergenerational equity. Furthermore, many environmental policy proposals are based on competing concepts of equity across the current generation. Indeed, it is often forgotten that the original definition of sustainable development in the Brundtland Report (WCED, 1987) encompassed not only intergenerational equity but also intragenerational equity. That is, many would argue that a seriously unequal society is 'liveable' but not necessarily 'sustainable' in the normative sense.

One example of the interpretation of equity in context of the current generation is the burden-sharing problem for environmental improvement programmes; that is, sharing of the costs of an environmental programme among different individuals or groups (in the present). While, it is typically argued that the polluter-pays-principle should guide, in some way, burden distribution, other principles such as ability to pay have also been proposed to be relevant to this policy problem. However, the discussion of environmental equity rarely considers how these competing (and possibly conflicting) principles can be used in conjunction with one another (rather than the exclusive application of a single principle).

In chapter 7, a survey-based approach using a contingent ranking methodology is used to propose a means of reconciling this apparent conflict. This moves beyond seemingly intractable arguments regarding the pre-eminence of specific principles of environmental equity. In doing so, we examine three specific principles: responsibility for pollution (or damage caused), benefit received and income (or ability to pay). The econometric analysis of survey responses casts light on the magnitude of trade-offs between different principles of environmental equity. The results indicate that these trade-offs do exist and, furthermore, are significant. Assessing trade-offs in this way could be an important input into the design of mixed criteria rules for sharing burdens. The wider relevance of these findings to this thesis is twofold. Firstly, to the extent that *intra*generational equity concerns can be considered part of the ambit of sustainable development then this chapter makes a direct contribution. Secondly, the methodology that we outline could be used to address analogous questions regarding some of the diverse concepts of *inter*generational equity that have been proposed.

Chapter 2

Sustainability, Green National Accounting and Deforestation

2.1 Introduction

Emerging interest in the measurement of sustainability has led to an increasing emphasis on green national accounting as a tool for monitoring progress towards or away from this goal. For example, according to one well-known proposition, it is the sign of an extended net saving rate (i.e. savings net of asset consumption) that can be used to evaluate prospects for sustainability. To the extent that forests have been considered within this framework, the change in asset value that occurs when forests are cleared has been conceived primarily as the net change in timber resources. Clearly, for many countries, how rents (arising from the depletion of timber resources) are managed is an important determinant of sustainability. However, this emphasis seems to be unduly narrow in that when forests are cleared a wide range of asset values, such as (non-market) local and global conservation benefits, are often lost in perpetuity.

In this chapter, we capture this range of (net) costs within an extended national accounting framework. One crucial component of this accounting problem is a stock change that we characterise as the total value of 'excess deforestation'. This is defined as the amount of permanently cleared land on which the return to agriculture is less than the sum of the values of sustainable timber harvest and carbon sequestration and local and global willingness to pay for conservation. In other words, what this describes is whether or not the switch in land-use from forest to agriculture is wealth-increasing. In order to illustrate the practical implications of these insights, we estimate the value of 'excess deforestation' and other relevant parameters such as the value of net carbon accumulation arising from deforestation in Peru. These estimates are used to calculate an extended net or 'genuine' saving rate in order to indicate prospects for sustainability in the presence of deforestation.

2.2 Review of the Literature

As a result of the United Nations Conference on Environment and Development in 1992, most governments world-wide have adopted sustainability as a national goal. While discussion continues regarding what it is that policy-makers have committed themselves to, Pezzey (1989) offers a widely accepted definition, that a development path is sustainable if welfare per capita does not decline along the path. Achieving sustainability, in turn, has been equated with an assortment of propositions regarding how an economy should manage its portfolio of assets over time. For example, Pearce *et al.* (1989) define *strong* sustainability to mean non-declining values of both produced and natural assets, while *weak* sustainability is characterised as a non-declining value of total assets (so that substitutions of natural and produced assets are possible).[1]

Pearce and Atkinson (1993) provided one of the earliest suggestions for an indicator of (weak) sustainability: an adjusted national savings measure that accounts for the depletion of natural resources and the environment. Hamilton (1994, 1996a) developed welfare measures to account for living and non-living resources and varieties of pollutants in a similar extended national accounting framework – the corresponding savings measure, equivalent to that of Pearce and Atkinson, was termed 'genuine' saving. This is the net saving rate in an extended accounting framework where the term 'genuine' saving is used to provide a distinction from standard national accounting terminology. It can be shown that for simple optimal and non-optimal growth models, negative rates of genuine saving result in a decline in future welfare (Hamilton and Clemens, 1999; Dasgutpa and Mäler, 2000).[2] Estimated rates of genuine saving for a broad range of developing regions are now published in World Bank (1999).

[1] A more specific requirement is that strong sustainability demands that stocks of certain 'critical' natural assets – assets without obvious substitutes whose loss would entail catastrophic declines in welfare – are maintained above some minimal level (Pearce *et al.* 1996).

[2] It should be noted that the genuine saving rate measures the value of the net change in an economy's assets. To the extent that population in this economy is not constant a more satisfactory indicator of sustainability is the change in (total) wealth per capita (Hamilton, 2000).

The primary goal of this chapter is to extend the savings analysis to a domain that seems both particularly topical and important, the depletion of forests in the developing world. For example, a significant challenge in making good commitments to sustainability lies in the design of policies that seek to move toward an optimal balance between deforestation, defined as the permanent clearing of forested land, and conservation of the world's remaining tropical rain forests. This presents policy-makers with a formidable task in understanding the driving forces behind decisions to clear tropical forest in the developing world (Swanson, 1994). While we address this issue indirectly, in this chapter, we focus primarily upon the evaluation of the net welfare costs of deforestation. Furthermore, we demonstrate how these net costs can be evaluated within an extended national accounting framework. In doing so, we show how deforestation can be analysed formally within the broader sustainability debate.

Estimates of the net welfare loss, arising when forests are cleared, are an important component in gauging the magnitude of the policy response required to address deforestation. Any assessment of the impacts of deforestation needs to encompass the diverse range of market and non-market effects on welfare. This includes the loss of local conservation benefits (Smith *et al.* 1997; Shyamsundar and Kramer, 1996), positive production externalities on agricultural productivity (e.g. soil conservation) provided by standing forest (World Bank, 1992) and global conservation benefits (Kramer and Mercer, 1997). Although the measurement issues involved are frequently complex, existing studies have demonstrated that the magnitudes of these forgone benefits are non-trivial and, moreover, are often empirically comparable with activities that they are replaced by such as agricultural production on cleared land (Pearce *et al.* 1999; Nordhaus and Kokkelenberg, 1999).

Green national accounting can inform this debate in a number of ways, primarily by providing a consistent and coherent framework for analysing detailed and diverse data describing the net welfare cost of clearing forested land. Moreover, given the focus of these accounts on the better measurement of income and wealth, they are ideally suited to evaluating whether the switch of land-use from forest to agriculture is actually wealth increasing (or 'sustainable'). Central to this is an expansion of the asset boundary to account explicitly for changes in land-use, i.e. where land is an asset that has a distinct value depending on the use to which it is put (Hartwick, 1992, 1993b). Accounting for forest resources within this framework, in the presence of deforestation, can be viewed as an exercise in calculating the net welfare cost of moving from forest to an alternative land-use, for example agriculture. Furthermore, this net change in land asset value can be interpreted as a stock change to be subtracted from estimates of income and saving (Hartwick, 1993b).

A wide range of studies has examined forestry and the national accounts (see, Vincent and Hartwick, 1997, for a comprehensive survey). Many of these studies, however, have focused exclusively on accounting for the net accumulation of timber that arises when forested land is cleared. The basic model underlying these calculations views the exploitation of primary forest as akin to a "timber mine" where "reserves" can be augmented via natural growth (Nordhaus and Kokkelenberg, 1999). Net accumulation is defined as net harvest (or net growth) valued at the unit rent or stumpage value for a given timber resource (see, for example, Repetto *et al.* 1989; van Tongeren *et al.* 1993; Howell, 1996). A relatively sophisticated treatment of this problem is Vincent (1999) which takes account of the age class of timber on a unit of land as well as the volume of resource harvested, an issue of particular importance when accounting for timber resources in managed secondary forest.

Where forested land provides benefits in addition to timber, our understanding of the loss of asset value on cleared land extends beyond the net accumulation of timber. Indeed, in acknowledgement of this, a number of studies have constructed accounts that encompass this wider notion of land value across a range of developed and developing countries. Thus, forestry accounts exist for non-timber forest products (NTFP) (Bartelmus *et al.* 1993; Hultkrantz, 1992; Hoffrén, 1996; Scarpa *et al.* 1998), environmental services such as watershed services and soil conservation functions (Aguirre, 1996; van Tongeren *et al.* 1993; Hamilton, 1997a; Hassan; 2000; Torres, 2000) and fuelwood (Peskin, 1989; Katila, 1995). Fewer studies have estimated the value of biodiversity, although Hultkrantz (1992) proposes an estimate, for Sweden, based on the opportunity costs of conserving land. More recently, Haripriya (2000) accounted for the pharmaceutical benefits of forests in India based on an estimate of option value. A particularly novel treatment is Vincent *et al.* (1993) for Malaysia, based on species-area relationships drawn from the biogeography literature[3], where new extinctions (of species) are valued according to an estimate of the unit value of species loss. Several studies such as, for example, Hassan (2000) attempt to account for the value of net carbon accumulation or sequestration, with Anielski (1992, 1993) for Canada providing one of the first accounts of this type.

The primary conclusion of the review by Vincent and Hartwick (1997) is that, "empirical efforts to incorporate forest resources into the national accounts must be guided by economic theory more than they have been" (p50). Put another way, a critical starting point of such efforts needs to be an explicit economic model of forestry in the accounts. This is important in order to ensure that the salient features of deforestation are properly accounted for; i.e. that *all* relevant changes in quantities that arise when switching from forested land to some alternative land-use are valued by their correct shadow prices. More recently, several contributions that address the accounting problem for timber resources have emphasised the link between empirical applications and theoretical models of the net accumulation of timber (see, for example, Seroa da Motta and Ferraz, 2000). However, few – if any – applications (such as Torres, 2000) of the wider variety of market and non-market losses arising from deforestation have followed this lead. Most notably, these broader studies have neglected arguably one of the most important questions when dealing with the change in asset values that occurs when land is permanently cleared of forest: that is, whether a change in land-use is actually wealth increasing (or sustainable).

The contribution of the current chapter is to combine forests and the national accounts, non-market valuation and, most importantly, to demonstrate a clear link between the theory and practice of green national accounting in a developing country. Firstly, we propose an extended national accounting model of deforestation arising from slash-and-burn farming. This builds on a theoretical framework from which can be derived the relevant accounting aggregates for the specified forest accounting problem. Secondly, the results of this model underpin our subsequent case study of deforestation in Peru, using field data from that country wherever possible. The

[3] For a discussion, see for example, MacArthur and Wilson (1967).

clearing of tropical rain forest in Peru provides a good example of the competing market and non-market (net) changes in land assets when forests are cleared. In addition, while few existing studies focus *directly* on the implications of forest resource accounting for sustainability in a developing country, our framework and its focus on the genuine saving rate allows us to offer policy relevant comments on the link between sustainability and deforestation in Peru.

2.3 Accounting for Excess Deforestation

In common with Weitzman (1976), Hartwick (1990) and Mäler (1991), national income here is defined along the optimal path of a growth model for a simple economy with tropical forest. The framework to be employed is "extended Hicksian", and as such is an elaboration of the general model described in chapter 1. Specifically, we focus on accounting for the value of forests in national income.

For the study nation we assume that there is a fixed amount of land that can either be used for agriculture (A, measured in hectares) or is covered by forest (L), and that deforestation is the process of conversion of some amount of forest land into agricultural land. The area deforested each year d is initially covered with a forest stock of density S/L (in cubic metres per hectare), where S is the total stock of timber. On forested land the amount of harvest is given by R.

The standing forest grows according to a function $g(S,L,X)$, with g_S following the usual pattern of being positive, then zero, then negative (i.e., for a fixed L the growth curve is an inverted 'U' which defines a maximum sustainable yield and a long run equilibrium growth rate of zero), while $g_L > 0$. X is the global stock of carbon dioxide (CO_2). This CO_2 fertilises forest growth, so that $g_X > 0$. CO_2 dissipates naturally as described by the function $n(\eta X)$, where η is the share of the study country in the total global stock of CO_2.

Slash and burn is the assumed forest clearance mechanism in the study country. It is assumed that each cubic metre of timber yields α tonnes of CO_2 when burned, and symmetrically, that the growth of one cubic metre of timber absorbs α tonnes of CO_2

(in other words, the only source of carbon in trees is assumed to be atmospheric). The accounting identity for the stock of CO_2, given deforestation, natural growth and dissipation is therefore,

$$\dot{X} = \alpha \frac{S}{L} d - \alpha g - n(\eta X), \tag{2.1}$$

Deforestation is assumed to cost an amount *f(d)* of an aggregate good that can be consumed, invested or spent on deforestation. Standing forest is assumed to provide production externalities. The national accounting identity is therefore given by,

$$GNP = F(K, A, R, L) = C + \dot{K} + f(d). \tag{2.2}$$

It is assumed that residents of the nation value consumption and standing forest, while the stock of CO_2 causes harm. It is also assumed that the rest of the world derives benefits from the existence of the forested area in the study nation. Therefore, the utility function for the forestry model is $U(C, L, X) + U^w(L)$, with $U_X < 0$.

The optimal growth model for this economy is specified as,

$$\max_{C,R,d} W = \int_0^\infty U(C, X) e^{-rt} dt \quad \text{subject to:}$$
$$\dot{K} = F - C - f$$
$$\dot{A} = d$$
$$\dot{L} = -d$$
$$\dot{S} = -R - \frac{S}{L} d + g$$
$$\dot{X} = \alpha \frac{S}{L} d - \alpha g - n$$

For shadow prices γ_i the current value Hamiltonian for this problem is,

$$H = U + \gamma_1 (F - C - f) + (\gamma_2 - \gamma_3) d + \gamma_4 \left(-R - \frac{S}{L} d + g \right) + \gamma_5 \left(\alpha \frac{S}{L} d - \alpha g - n \right). \tag{2.3}$$

From the static first-order conditions we can define the marginal damages from carbon dioxide to be $b \equiv -\gamma_5 / U_C$. The expression for Hicksian income then is derived as:

$$NNP = C + \dot{K} + \left(f' + (\alpha b + F_R)\frac{S}{L} \right) d - F_R\left(R + \frac{S}{L} d - g \right) - b\left(\alpha \frac{S}{L} d - \alpha g - n \right). \qquad (2.4)$$

The last term in expression (2.4) is the value of damages from the net accumulation of CO_2 in the atmosphere. Slash-and-burn therefore adds to the CO_2 stock, while the growth of timber on the remaining forested land reduces it. The preceding term is the value of net reduction in the stock of timber. This has two components: net harvest of timber on forested land ($R-g$); and, timber on deforested land ($S/L \times d$). Before that is the term representing the difference in the shadow prices of agricultural land and forested land. Here, marginal clearance costs (f'), damages from carbon dioxide emissions (αb) and the rental value of the timber that was burned are all part of the difference in prices between these two different uses of land. In a model of optimal deforestation we would expect this latter term – i.e. the difference in the shadow prices of agricultural land and forested land – to be positive (at least before the steady state level of deforestation had been reached). In the real world, a variety of policy distortions and market imperfections can easily lead to *excess* deforestation, where land values do not rise as result of deforestation.

To characterise excess deforestation it helps to derive the efficiency condition for harvesting the marginal hectare in the steady state. At this point the marginal returns to agriculture must just equal the marginal returns to standing forest. Deriving the dynamic efficiency conditions for the shadow prices of agricultural and forested land in the steady state yields,

$$\frac{U_L}{U_C} + \frac{U_L^w}{U_C} + F_L + F_R g_L + \alpha b g_L + F_K\left(f' + (\alpha b + F_R)\frac{S}{L} \right) = F_A . \qquad (2.5)$$

These terms are relatively simple to interpret. The first two are, respectively, the (marginal) willingness to pay (WTP) of national residents and foreigners for a unit of standing forest. F_L is the production externality provided by a unit of forest. $F_R g_L$ is the rental value of the natural growth of forest on a unit of land – this is the sustainable harvest or off-take. $\alpha b g_L$ is the value of the carbon sequestered during natural growth on a unit of land. The next term is interest that would be earned if the sum of the clearance cost, carbon sequestration benefits and timber rental value for the marginal unit of deforested land were put in a bank. Finally, the right hand side is the marginal product of the unit of land under agriculture: i.e. agricultural returns per hectare.

If for a given hectare of land the left-hand side of expression (2.5) is greater than the right, then there is excess deforestation. If it is assumed that there are d^* such hectares and that the land use change is permanent, then the value of excess deforestation is given as,

$$\left(\frac{U_L}{U_C}+\frac{U_L^w}{U_C}+F_L+F_R g_L+\alpha b g_L+F_K\left(f'+(\alpha b+F_R)\frac{S}{L}\right)-F_A\right)\cdot\frac{d^*}{F_K}. \tag{2.6}$$

Note that the terms in F_K cancel, and so when expression (2.6) is subtracted from expression (2.4) to arrive at Hicksian income (expression (2.6) represents the change in asset value for deforested land), the final expression becomes,

$$\begin{aligned} NNP &= C+\dot{K}-F_R\left(R+\frac{S}{L}d-g\right)-b\left(\alpha\frac{S}{L}d-\alpha g-n\right) \\ &\quad -\left(p_L+p_W+F_L+F_R g_L+\alpha b g_L-F_A\right)\cdot\frac{d^*}{F_K} \end{aligned} \tag{2.7}$$

where, $p_L = U_L/U_C$ and $p_W = U_W/U_C$. Genuine savings, G, in this model is defined as $NNP - C$ or,

$$\begin{aligned} G &= \dot{K}-F_R\left(R+\frac{S}{L}d-g\right)-b\left(\alpha\frac{S}{L}d-\alpha g-n\right) \\ &\quad -\left(p_L+p_W+F_L+F_R g_L+\alpha b g_L-F_A\right)\cdot\frac{d^*}{F_K} \end{aligned} \tag{2.8}$$

In other words, genuine saving G is simply net accumulation of assets (in the study country) where assets are defined to include forest resources (providing a wide range of market and non-market benefits). This analysis is a long route to a result that environmental economics would lead one to expect: that there is excess deforestation if the total value of forested land exceeds the return of the same land under agriculture. Nonetheless, what this analysis does is to place this in a Hicksian income accounting framework and, thereby, provide a powerful link to the notion of sustainability.

2.4 Empirical Analysis

2.4.1 Study Country

Our study country is Peru where the rate of deforestation over the period 1980-95 was approximately 0.3% per annum compared to an average for Latin America of some 0.6% over the same period (World Resources Institute, WRI, 1998). In terms of the extent of forest cleared, Peru's deforestation rate is estimated to be approximately 210 000 hectares per year (ibid.). For the purposes of applying our accounting framework, we draw on data from a number of sources and, moreover, wherever possible we restrict attention to relevant Peruvian data (rather than 'similar' data transferred from elsewhere). In particular, in accounting for agricultural and forestry aspects of our model we draw on data from the field as far as is feasible. Field data is obtained from three regions of Peru (illustrated in Figure A2.1.1 in Appendix 2.1). The first study site is Pueblo Libre where the process of deforestation begins with the extraction of commercially valuable timber followed by occupation of land by agricultural colonists typically in plots of 20 hectares (ha). This area is thought to be representative of conditions elsewhere in the country and therefore, with caveats, is a suitable site for extrapolating agricultural data. Our second and third study sites are the rural area near the city of Pucallpa in the Ucayali Region of the Peruvian Amazon and the national forest of Von Humboldt.

2.4.2 Data

Discount Rates

The total value of excess deforestation is a present value such that the value of the net change in forested land asset is the discounted sum of all future net losses attributable to switching land-use on a unit of land. For example, when a hectare of forest is cleared these net conservation benefits – i.e. components of the last term on the right hand side of expression (2.7) are lost in perpetuity. Hence, annual estimates of losses must be converted to present values using some discount rate. Similarly, agricultural returns will be enjoyed into the future and so too this stream of benefits must be discounted.

In principle, in our model, F_K is equivalent to the mean future value of the social discount rate or the social rate of time preference (*SRTP*). The *SRTP* is the fundamental discount rate in growth theory and can be defined as the maximum amount of extra consumption made possible by foregoing a unit of consumption now. In a fixed-technology neoclassical world, this social discount rate would decline to the pure rate of time preference in the long run steady state. However, as a practical matter, setting F_K equal to the current social discount rate might be a working assumption. The *SRTP* can be expressed in the following formula (e.g. Lind and Schuller, 1999):

$$SRTP = r + \theta \frac{\dot{C}}{C}.$$

Here r is the pure rate of time preference (or rate of impatience, the rate at which future utility is discounted), θ is the elasticity of the marginal utility of consumption, and $\dot{C}/C$ is the percentage rate of growth in per capita consumption. The social discount rate is thus the sum of the rate of impatience and the rate of decline in the marginal utility of consumption associated with an extra unit of consumption.

In practice, the appropriate value that this discount rate should take is far from obvious. For example, it has been proposed that projects with global conservation benefits be discounted at a global social discount rate perhaps in the range of 0.5% to 3.0%. The basis for this is described by Lind and Schuller (1999) who argue that plausible values of $\dot{C}/C$ lie between 0.5% to 1.5% and θ of between 1 and 2. The assumed value of r is arguably the most controversial aspect of this calculation. Many contributions to this debate argue that $r = 0$ is the only (ethically) defensible assumption (ibid.) Alternatively, a survey by Pearce and Ulph (1995) concludes that the literature supports values of $0\% \leq r \leq 2\%$, which in turn would support a wider range possible values of SRTP (of 0.5% to 5.0%).

It could also be argued that local conservation benefits should be discounted with a social discount rate specific to Peru. The growth rate of (real) *GNP* per capita in Peru over the period 1965 to 1997 was, on average, 1%. This average belies much variability over this period and in particular following significant economic restructuring the growth rate over period 1990 to 1997 was 4% (World Bank, 1999). Assuming that again r lies between 0 and 2 and that θ lies between 1 and 2 then this suggests a social discount rate in Peru in the range of 1% to 10%. This is a wider range than that suggested for a 'global' social discount rate.

A further complicating factor is that to the extent that flows such as agricultural returns and (sustainable) timber harvest have an explicit financial component (which can be invested) it can be argued that the relevant discount rate is the rate of return on capital or a private discount rate. Pearce *et al.* (1999) note that little is known about private discount rates in developing countries but observe that what evidence exists suggests a powerful explanation for unsustainable agricultural practices. For example, Cuesta *et al.* (1994) found that real personal discount rates of farmers in Costa Rica were in the range of 15-22%. Poulos and Whittington (1999) find that longer-term private discount rates are, in the range of, 11-28%.[4]

[4] Poulos and Whittington also find that short-term personal discount rates are 45-206%. The countries studied were Ethiopia, Mozambique, Uganda, Indonesia, Bulgaria and Ukraine.

It is clear that, in practice, there is significant variation in the magnitude of possible discount rates depending on whether we believe a private or social discount rate to be more appropriate or indeed depending on assumed values of relevant parameters. This reflects the uncertainty surrounding this issue. Unfortunately, from a practical perspective, this complicates the problem of valuing excess deforestation. As noted by Feenstra *et al.* (1999) even small changes in the discount rate can lead to large variations in parameter values. Pearce (1986) outlines the rationale for using different discount rates in conjunction to evaluate different flows in a *given* project. Alternatively, following Seroa da Motta and Ferraz (2000), another pragmatic solution to this problem is to discount each flow using a single value of the discount rate but to vary this rate. In what follows we assume rates equal to 3%, 6%, 10% and 20%, which correspond broadly to the range of possible values described previously. This should provide an indication of the sensitivity of those parameters, which require discounting to variations in the discount rate.

Timber Depletion

Trees on forest land are a living resource and the value of current resource rents is equivalent to the net growth of the resource valued at its rental rate. This rental rate is also known as stumpage value and can be most simply estimated by the net price technique whereby stumpage value is calculated as the market price of timber minus the costs of harvest and processing (e.g. transportation and milling etc.) (see, for a more sophisticated treatment, Vincent, 1999). From a theoretical viewpoint, resource rents should be measured as (the border) price minus *marginal* cost of harvest (including a normal return to capital, *ROC*). In practice, marginal production costs are almost never available and practitioners (as evidenced in the green national accounting literature) fall back on using average harvesting costs. Note that it is likely that this will tend to overstate calculated resource rents if marginal costs are increasing.

With respect to logging in the Peruvian Amazon, Nalvarte (1999) provides estimates of costs associated with six different harvest and processing activities (in the Pucallpa and Von Humboldt areas respectively). These are: matting (locating and marking trees to be felled); felling and blunting (separating the trunk from the tree crown); log making (sawing); haulage; loading; and transportation. An average of the total clearance costs of timber cited by Nalvarte indicates that costs are in the region of $17/m^3, with transportation and loading accounting for just over 60% of this total and haulage for a further 33%. Furthermore, we assume a (normal) rate of return on capital of 15% (i.e. 15% of the market price received for timber, consistent with World Bank, 1997).

Table 2.1 Unit Timber Resource Rents, 1995

	Dollar per cubic metre ($/m^3)
Unit (Export) Price	52.86
Costs	25.03
Matting	*0.13*
Felling/ blunting	*0.78*
Haulage	*5.68*
Log making	*0.17*
Loading	*0.63*
Transport	*9.71*
+ 15% ROC	*7.93*
Unit Rent	27.83

Source: Nalvarte (1999); World Bank (1997)

World Bank (1997) calculates the implicit export price of timber for Peru (from data on the value and volume of roundwood exports), which are then averaged regionally to reduce some of the noise in the data. In 1995, this (implicit average) current export price was $52.86/m^3 in Peru. In practice, timber harvested on a representative unit of forested land in Peru consists of timber of high commercial value (e.g. caoba or mahogany, cedar) and hardwoods and softwoods of medium and low value (Nalvarte, 1999). However, it seems reasonable to infer that this export price satisfactorily reflects the unit value of timber harvested on an average tract of forested land in Peru. Table 2.1 estimates the unit rental rate (i.e. $/m^3) in Peru from these data. Average harvest costs are approximately 47% of price (including a normal rate of return on capital) and unit rental rent is $27.83. Nalvarte (1999) further reports that a representative hectare in the von Humboldt region of the Peruvian Amazon contains about 113 m^3 of commercially valuable timber. This gives an estimate of the value of timber depletion on a unit of land of $3145.

Agricultural Productivity

Converting land to agriculture under slash-and-burn is not the same as receiving a return in perpetuity (e.g. assuming that current year values remain unchanged indefinitely) and so our model is only indicative of this aspect of the land conversion process. The returns to a representative hectare of agricultural land must rather account for the mix of crops grown on that hectare over the productive life of the soil (while taking account of fallow periods). Hence, agricultural returns (π) are defined as follows,

$$\pi = F_{A0} + \frac{F_{A1}}{(1+F_K)} + \ldots + \frac{F_{AT}}{(1+F_K)^T}.$$

That is, the present value of the return to agricultural production on land cleared in a period 0, comprises the discounted (finite) stream of income that occurs from that year until the land's soil fertility is exhausted at *T*. However, land may continue to earn a return after this point if it is converted to pasture, e.g. grazed by cattle, rather than reverting to (secondary) forest. The extent to which deforested land is later converted to pasture varies across Peru. For example, within our study area of Pueblo Libre, pasture is not widespread. Clearly, however, it would be desirable to take some account of the value of pasture in our calculations.

Within the study area for agriculture (Pueblo Libre), data were obtained relating to physical yield per hectare per crop type on cleared land, over the likely productive life of the land. (Table A2.2.1 in Appendix 2.2 provides a detailed summary of the agricultural data used.) Agricultural production, on a given hectare, takes place over approximately 6 years. In terms of the crop types that are grown by farmers the data indicate that the first crop planted, after forest is felled and burned, is rice. This is followed by a period of fallow (while another part of the farm is cleared and planted) and then the planting of corn (maize), plantain and yuca (cassava). These physical outputs are valued at their respective market price (in current prices).

Returns to a unit of land, π, are net of labour and any other productive inputs used in farming. Few, if any, capital inputs, aside from simple tools or fertiliser are used in slash-and-burn production and hence any additional inputs consist primarily of seed. These inputs are valued at the prevailing market prices. Labour time is empirically the most significant input to be accounted for. Data were therefore obtained on labour requirements (i.e. hours/ha/year) associated with the production of crops. Labour time is valued at the hourly agricultural wage. Farmers also participate in other productive activities associated with farming of a representative hectare of land. In particular, fuelwood is collected on fallow land (and from surrounding forest). The value of fuelwood is taken to be the opportunity cost of time spent collecting this resource: i.e. hours-spent collecting wood multiplied by the hourly agricultural wage.

We assume that agricultural returns are constant across land units. Although available data would not permit a more sophisticated treatment of this problem, it is important to note that we would expect marginal returns at the agricultural frontier to be declining. That is, in initial periods the most productive land is brought into use followed by land of more peripheral agricultural value. With respect to our accounting model, if F_A is decreasing in units of land converted then it is becomes increasingly likely that, other things being equal, an additional unit of land converted can be described as excess deforestation. A similar issue is that there will also be differences in productivity across units of land at a given point in time in different parts of the country. Such differences can be expected due to variations in soil fertility and farming practices and given these diverse characteristics of farmland across Peru, some variation from field data estimates is to be expected. There are two main types of land in Peru: Selva Baja – lowland forest which characterises our study area; and Selva Alta – highland forest which characterises some of the land area beyond the study site. In addition, there are two main production systems within the study area itself and beyond; 'colonist production systems' (that is, slash and burn farming as we have described it) and 'indigenous production systems' (practised by the indigenous inhabitants of the forest).

Table 2.2 Agricultural Crop Production in High and Low Forest

Location	Rice (t/ha)	Corn (t/ha)	Yuca (t/ha)	Plantain (bunch/ha)
Pueblo Libre	***1.3***	***1.0***	***10.0***	***600***
Low forest: salt marsh (Ucayali)	2.0	3.0	15.0	600
Low forest (Ucayali)	1.2	1.5	12.0	400
High forest (Ucayali)	1.5	1.5	12.0	625
Low forest (Puerto Inca - Huánuco)	1.3	1.4	8.5	...
High forest (Chanchamayo - Junín)	...	1.2	11.1	...
High forest (Satipo - Junín)	1.4	1.2	12.1	...
High forest (Oxapampa - Pasco)	...	1.3	10.0	...
Low forest (Loreto departament)	2.3	1.5	10.4	...

Source: Nalvarte (1999)

An indication of these differences in productivity, by altitude and by area (region), is provided in Table 2.2. These comparative data suggest that our estimates of yields for corn are relatively low (compared to other areas), are marginally low for rice and yuca and slightly higher, on average, for plantain. One means of reflecting this apparent geographical variation in Table 2.2 would be to test the sensitivity of our estimate of agricultural returns to the assumption that agricultural returns are 50% higher and lower than our data suggest.

We characterise the most common pattern of slash-and-burn agricultural production along the following lines. Firstly, there is production of rice for two years followed by a fallow period of one year (on which fuelwood is collected) after which corn, yuca and plantain is grown on the land. In the first year of production additional costs are incurred in clearing and making good the land. Other costs are those associated with planting, maintenance and harvest. Rice, corn and yuca all have a one-year growing cycle while plantain has a three-year growing cycle. Depending on the discount rate used to evaluate future net returns, Table 2.3 indicates that this pattern of crop production yields a return in the range of about \$490 to \$666. Obviously, the assumption of ±50% increases this range even further as indicated in parenthesis in Table 2.3.

Table 2.3 Agricultural Returns on a Representative Unit of Land

Crop Production	*0.03*	*0.06*	*0.10*	*0.20*
Rice + Corn + Plantain + Yuca	\$666	\$629	\$584	\$490
	(\$333–\$999)	(\$315–\$944)	(\$292–\$876)	(\$245–\$735)
Rice + Corn + Plantain + Yuca + Pasture	\$842	\$717	\$624	\$499
	(\$490–\$1469)	(\$394–\$1179)	(\$328–\$984)	(\$253–\$758)

Source: authors' own estimates from data in Nalvarte (1999)

The second calculation in Table 2.3 imputes an estimate for the value of pasture. Unfortunately, there are appear to be no available data either from our study site or elsewhere in Peru (to our knowledge) that estimate the value of pasture. However, Schneidner (1995) reports estimates from Brazil to the effect that the returns from pasture in the Brazilian Amazon are in the range of \$8/ha to \$24/ha per annum (in 1992). Hartwick (1993b) estimates returns from pasture in Costa Rica (in the same year) to be \$9/ha per annum, at the lower end of this range. In the absence of a satisfactory alternative, we use these data here adjusted to 1995 prices and so adopt a suggested range of \$9/ha to \$26/ha with a central estimate toward the lower end of this range of \$10/ha. Table 2.3 indicates that the value of pasture only appears to make a significant difference to the present value of a unit of land under agriculture only for relatively low discount rates. One reason for this is that these returns relatively late in the land's "life" under agricultural production. (It should be noted that if land reverts back to (secondary) forest then our estimate of the net returns of agriculture should not include the value of pasture.)

The results in Table 2.3 can be benchmarked against other data from Peru. For example, Smith *et al.* (1997) report results from experimental evidence, which indicates that slash and burn agriculture (i.e. growing of rice, yuca and plantain) yields returns in the region of \$380-\$720. This broadly seems to accord with the data in Table 2.3. Smith *et al.* (1997) also asked slash-and-burn farmers in Peru, using a stated preference survey (described in more detail in the next section), for their willingness to accept compensation to set aside a hectare of farmland for conservation. This elicited somewhat higher valuations of (implied) returns per hectare; for example, for a 20% discount rate implied (one-off) compensation is in the region of \$960, slightly above our estimates for the same discount rate.

Sustainable Timber Harvest

For standing forest there is a sustainable off-take of timber that can be harvested while keeping the forest stock constant. Sustainable harvest is equivalent to the volume of natural growth of timber on forested land, g_L, and is valued at the stumpage value or unit rent, F_R. The volume of this yearly sustainable harvest per hectare can be equated to the mean annual increment (MAI) of the forest. When forested land is permanently cleared this MAI is lost in perpetuity. Estimating the value of this harvest requires data on the mean annual increment (i.e. natural growth) per hectare of forest. The magnitude of this average increment will depend on a number of biological (e.g. tree species) and geographical factors (e.g. climate, altitude). Note that this differs from the economic concept of growth based on maximum sustainable yield. That is, the stock level of forest that maximises the volume of timber that can be sustainably harvested each year. A caveat of using data on natural growth is that, in general, it is reasonable to expect that the magnitude of the economic sustainable yield is likely to exceed MAI in primary or old-growth forest.

Various estimates of natural growth or MAI have been discussed in the literature. The common feature of all such estimates is that they are to some degree speculative in nature. Furthermore, there exist no studies specific to natural forest growth in the Peruvian Amazon and, hence, the data that we use for this purpose must be sought from further afield. Solóranza *et al.* (1991) review a range of forest studies citing estimates in the range of 4.8 m^3/ha and 20 m^3/ha. A more recent review from Nalvarte (1999) finds a similar range of estimates for MAI of timber in humid tropical regions of 5 m^3/ha and 19 m^3/ha, with a central estimate for Latin American forests of 12

m^3/ha (in the middle of this range). We use a conservative estimate of 5 m^3/ha, which would appear to be reasonable given the volume of timber on a hectare of land in Peru, as discussed above (i.e. this corresponds to slightly less than 4.5% of the timber stock on a unit of land). This gives an estimate of the *annual* value of sustainable harvest of about \$139 (i.e. 5 m^3/ha × \$27.8). Of course, if land is permanently cleared then this annual harvest is lost in perpetuity and it is therefore the present value of this sustainable harvest that is of interest. This is simply calculated as the annual value divided by the discount rate. Hence, given our assumed magnitudes of the discount rate, the value of the sustainable harvest lies in the range of \$695/ha to \$4633/ha (i.e. \$139/0.2 to \$139/0.03). Recall that the timber value of clearing a unit of land was estimated to be \$3145. In this respect, Pearce *et al.* (1999) cite evidence that, in general, the unit value of sustainable timber harvest is less than the unit value of non-sustainable harvest. In the current context, this finding would appear to be consistent with the use of a higher discount rate.

Local Willingness to Pay for Conservation

Local people are assumed in our model to have preferences for standing forest, as well as consumption enjoyed by clearing these forests. Local willingness to pay to conserve a hectare of forested land is denoted by p_w in our model. The underlying benefits upon which local people place value are arguably diverse and include air purification, maintenance of soil and water quality in addition to the provision of food (e.g. game meat), shade and shelter (see, for example, van Kooten *et al.* 1999). Smith *et al.* (1997) value the local benefits to farmers in the Peruvian Amazon of moving from land-use based on slash-and-burn agriculture to (increased) forest conservation. The study elicits farmers' (implicit) willingness to pay (WTP) for local conservation benefits using the contingent valuation method.[5] Firstly, farmers were asked how much they would be willing to accept in compensation in order to conserve a unit of forested land rather than clear this land for slash-and-burn farming. Secondly, forest services were in effect "sold back" as, in a follow-up question, farmers were asked

[5] For a recent review and discussion of this methodology see Bateman and Willis (1998).

how much they would be willing to reduce their stated compensation given that increased forest conservation provides them with environmental benefits.

Smith *et al.* (1997) estimate that the implied mean local WTP per annum (specifically, farmers' WTP) for conservation is in the range of \$55–\$72 with a central estimate of \$67. These annual benefits (which we assume to also embody production externalities, F_L) are converted to present values using our range of discount rates.

Table 2.4 Local Willingness to Pay for Conservation

Discount Rate	Local WTP		
	Central	Lower-bound	Upper-bound
0.2	335	275	360
0.1	670	550	720
0.06	1117	917	1200
0.03	2233	1833	2400

Source: adapted from Smith *et al.* (1997)

The results are shown in Table 2.4. Not surprisingly, these results are highly sensitive to choice of discount rate. If we use a discount rate of 20% then local WTP is in the range of \$275–\$360 with a central estimate of \$335. However, if the discount rate is 6% then WTP lies between \$917 and \$1200 and the central estimate is \$1117. Notwithstanding this sensitivity to choice of discount rate, reviews of studies of local benefits of forest conservation by, for example, Pearce and Moran (1994) and Lampietti and Dixon (1995) (in Latin America and beyond) suggest some plausibility for the estimates of local WTP in Smith *et al.* (1997). However, the recent review by Pearce (1999) also makes it clear that this literature appears to offer a wide range of possible values, owing to differences in benefit coverage, study methodology and quality of these studies.

Global Willingness to Pay for Conservation

Global willingness to pay, p_w, has been characterised in the literature in a number of ways. The most commonly cited is non-use value. Hence, it has been argued that the global population could place a value on the existence of standing forest, and its resident species, regardless of current or future use.[6] Although there are a number of studies that value the non-use benefits of conserving specific species, there are few examples that value the global benefits of forests. Indeed, the only suitable example is Kramer and Mercer (1997) (henceforth, KM) who estimate the mean willingness to pay of US households to conserve 5% of the world's remaining tropical rain forests. This is based on expert consensus that to maintain the integrity of the global rain forest ecosystem would require protection of 10% of remaining forest, half of which is already currently under some form of protection.

Respondents in KM's survey were asked for their WTP based on the perceived benefits of rain forest conservation, which in turn are primarily based on non-use value (including biodiversity). KM used two distinct CVM elicitation mechanisms – referendum and payment card – yielding mean WTP of \$21 and \$31 per household respectively. Furthermore, the confidence interval around these values implies a range of WTP of between \$8 and \$40. It is important to note that this is a 'once-and-for-all' payment. Hence, respondents are assumed to discount the future benefits that they would receive from the conservation programme at some unknown (average) rate.

We use KM's range of WTP taking \$26 per household as a central estimate. These estimates are adjusted, using the dollar GDP deflator, to convert to 1995 prices, such that for present purposes the range we use is \$9-\$43 with a central estimate of \$28. Using these estimates as the basis for global willingness to pay, within our model, raises the following problems. Firstly, there is no corresponding estimate for the WTP of households outside of the US. Nevertheless, it is reasonable to speculate that households in other (presumably, high-income) countries have preferences for standing forests elsewhere. One approach is to transfer US WTP, adjusted for income differences, across OECD[7] countries. Furthermore, we need to translate this adjusted WTP into an estimate of global WTP per unit (i.e. ha.) of land. By dividing the

[6] A more complex benefit of conserving habitat is an (indirect) use-value whereby the global population enjoys the (ecological) functions provided by biodiversity residing in standing forest (Perrings *et al.* 1995). However, there arguably exist no data, in this respect, that are suitable for the purpose of greening national accounts. An additional complication is that it is sometimes claimed that these functions are really local, rather than global, benefits.

product of WTP per household and total number of households by the number of hectares conserved under the proposed scenario, an estimate of this unit value is calculated. This is necessarily crude because not only does it rely on only one study of global preferences for forest conservation but also assumes that these preferences do not differ across high-income countries and that global WTP is constant across each unit of forest conserved. The latter is a particularly strong assumption. However, we retain it in order to provide a placeholder for global conservation benefits within our framework.

On this basis, our calculation of global WTP (or WTP across OECD countries[8]) is as follows. Firstly we weight KM's (adjusted) estimates of WTP (upper, lower and central estimates) by an income weight: i.e. the ratio of country *i*'s GDP per capita in Purchasing Power Parity (PPP) to US GDP PPP per capita (in 1995). These values are then aggregated across households in OECD countries: for simplicity, we assume that the US ratio of population to households prevails across the OECD, which gives an estimate of approximately 300 million households. This generates an aggregate conservation "fund" across the OECD of \$2.2–\$10.4 billion with a central estimate of about \$6.7 billion. This is nearly 2.5 times a corresponding fund based on US households alone, indicating the importance of trying to include preferences for conservation of those resident in other (high-income) countries. The final step is to derive a per hectare valuation that we can use as an estimate of global WTP for conserving a unit of forested land in Peru. Assuming that KM's programme conserves an additional 46 million ha of forest[9] then global WTP is in the range of \$47/ha–\$225/ha with a central estimate of \$147/ha. (If the magnitude of WTP was based on US households alone this range would be \$18-\$85 per ha.)

Net Carbon Accumulation

The accumulation of a unit of CO_2 in the (global) atmosphere is akin to an addition to a (global) liability. Specifically, this change is equivalent to the present value of the

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[7] Organisation for Economic Co-operation and Development.
[8] This includes the European Union countries, Switzerland, Norway, Canada, Australia, New Zealand, Japan and South Korea.
[9] This constitutes 5% of the world's tropical forest in the year in which KM carried out their study.

future damage arising from net CO_2 accumulation when a hectare of forest is cleared. Net CO_2 accumulation enters our model in a number of ways (as indicated by the final term on the right hand side of expression (2.7)). Firstly, slash-and-burn adds to the global CO_2 stock. Secondly, the growth of timber on remaining forested land reduces it. Lastly, Peru's "share" of the global carbon stock gradually dissipates. In addition, CO_2 also enters our expression for excess deforestation given that carbon is sequestered by natural growth on a unit of land, if that land remains forested. Thus, if this land is permanently then the sequestration benefit associated with this sustainable off-take is lost in perpetuity.

Boscolo *et al.* (1997) propose a methodology that can be adapted to estimate the net CO_2 or equivalent carbon emitted when slash-and-burn farmers clear forested land. According to this approach *NCA* is the difference in carbon (C) stock over time for slash-and-burn (land use, *j*) and forest conservation (land use, *i*) and can be written,

$$NCA = \sum_{t=0}^{T} \frac{(C_{j,t+1} - C_{j,t})}{(1+d)^t} - \sum_{t=0}^{T} \frac{(C_{i,t+1} - C_{i,t})}{(1+d)^t}.$$

This carbon is discounted using some discount rate (*d*). The rationale for this is that a unit of carbon emitted (or sequestered) in the future is less costly than a unit emitted (or sequestered) in the present. That is, the latter will contribute to the global stock of CO_2 (and its impacts) sooner rather than later in time. Net carbon, in our model, will not be solely determined by the *initial* clearing and burning of trees to make way for farming. For example, agricultural activities sequester (and emit) carbon through crop production and fallow. An additional complication is that while our model assumes that land is cleared in perpetuity, in practice, an estimate of *NCA* will be sensitive to assumptions regarding land-use after soil fertility is depleted and the farmer has abandoned the land. In particular, the quantity of carbon that is stored in pasture will be very different to the quantity of carbon that is stored if the land reverts to secondary forest.

Our estimate of *NCA* requires data on net changes in carbon under both agriculture and standing forest. Smith *et al.* (1997) review a range of Peruvian and other Latin

American data and we draw on these findings here. For example, based on an estimate of the carbon content in natural forest in the Pucallpa region in Peru, it is assumed that forest, on average, stores some 180tC/ha. in above ground biomass (Ricse *et al.* 1996) with Schroeder and Winjum (1995) estimating root biomass to be 20% of that above ground. Crops and fallows are estimated to accumulate biomass at the annual rate of 12tC/ha above and below ground (Szott *et al.* 1994; Woomer *et al.* 1996). (While it is also the case that soil stores carbon, Smith *et al.* (1997) note that it is reasonable to assume that there is no net difference in soil carbon under forest and agricultural uses on a unit of land.) Turning to the question of what happens to a unit of land after farming on it ceases; we base our estimates on either of two outcomes. Firstly, if land is used as pasture it accumulates carbon at the same rate as cropland in the first year under this use and subsequently, there is no further *net* accumulation of carbon. Secondly, if land is completely abandoned it reverts to secondary forest. This forest gradually accumulates carbon (at a rate of 12tC/ha/year) until the climax biomass (216tC) is reached. (For an illustration of this process see Figure A2.3.1 in Appendix 2.3.)

The quantity of *NCA* on a unit of land is valued using an estimate of the shadow price of carbon. Efforts to estimate the dollar value of damage per tonne of carbon emitted have typically calculated optimal carbon taxes or the costs associated with a doubling of carbon dioxide concentrations in the atmosphere from pre-industrial levels ("$2 \times CO_2$") (IPCC, 1996). Strictly speaking, within our model, it is the former that applies to our accounting problem. In practice, however, it would appear that this choice of methodology has had only a marginal bearing on findings in the literature.[10] A commonly cited estimate is Fankhauser (1994) (using the $2 \times CO_2$ approach) who estimates climate change damage to be \$20/tC (rising to \$28/tC by 2030).[11] This dollar value per tC emitted is equivalent to the discounted sum of future damage arising as a result of additions to atmospheric carbon CO_2.[12] We adopt Fankhauser's

[10] David Maddison (pers. comm., CSERGE, UCL).

[11] This increase is due, in Fankhauser's model, to projected increases in global population and wealth (and therefore the cost of damage) over time and the impact of future emissions on atmospheric concentration.

[12] This is a different issue to discounting of carbon released in the expression for *NCA* (adapted from Boscolo *et al.* 1996). That is, it is the necessity of comparing units of carbon released at *different* points

estimate as carbon's shadow price, b, in our model. This amounts to valuing carbon released in 1995 at \$20/tC and that carbon in future years is valued by assuming that this unit damage grows at a constant rate of 1.1% per annum. We also adopt Fankhauser's estimates of a lower bound of \$6/tC and an upper bound of \$45/tC in order to characterise the sensitivity of our estimates to uncertainty about the magnitude of climate change damage.

Table 2.5 describes estimates of the dollar value of *NCA* per hectare of land. This discounts carbon at a 3% rate, consistent with estimates in the literature of the social discount rate appropriate for discounting climate change impacts (see, for example, Lind and Schuller, 1999). These results describe the value of *NCA* depending on whether a unit of land becomes pasture or secondary forest when farming ceases. In the former, there is a net loss of carbon as a result of the land-use switch as, over time, a greater quantity of CO_2 is released than is subsequently sequestered under agricultural uses. In the latter, there is – over the long-term – no net loss of carbon arising from the change in land-use, as all CO_2 released is later sequestered in equal quantities (i.e. $\sum NCA_t = 0$).

Table 2.5 Value of Net Carbon Accumulation per Unit of Land

	Pasture	Secondary Forest
Low CO_2 Damage – \$6/tC	\$1188	\$289
Central CO_2 Damage – \$20/tC	**\$3907**	**\$1071**
High CO_2 Damage – \$45/tC	\$8665	\$2259

As the results in Table 2.5 show, it makes a significant difference whether we assume that after slash-and-burn farming ceases, the land gradually reverts back to (secondary) forest or is used for pasture.[13] For example, in the case of the latter the

in time that provides the rationale for discounting *NCA* (rather than comparing the stream of future damages, arising from a unit of carbon released at a *given* point in time).

[13] Of course, these results are also sensitive to the choice of discount rate. That is, changing the discount rate used to weight *NCA* across time changes the "relative price" of carbon emitted in a given period of time such that for discount rates greater than 3% *NCA* in later periods is assigned an ever-declining weight.

value of *NCA*/tC/ha is according to our central estimate reckoned to be just over \$3900. In contrast, \$*NCA*/tC/ha is about \$1100 if abandoned land is instead allowed to revert to (secondary) forest. The reason for this difference is clearly that secondary forest sequesters and stores relatively large quantities of carbon after land is abandoned. Indeed, there is in fact no net loss in carbon under this scenario although *NCA* in Table 2.5 has a non-zero value. This is because the effect of discounting in the expression for *NCA* is to give greater weight to the relatively large amount of carbon is released when the land is being prepared (i.e. trees cleared and burned) for farming than in later years when secondary forest begins to accumulate.

Finally, there are two additional elements to the CO_2 aspect of our model. The first is related to the calculation of the dissipation of Peru's "share" of the global carbon stock. For simplicity, we assume that this is zero on the basis that Peru's share of historical global CO_2 emissions can be thought of as being trivially small and that dissipation of the global atmospheric CO_2 is relatively slow (Hamilton and Clemens, 1999). The second calculation is the loss of the sustainable off-take of timber on a unit of land. Moreover, it is the carbon that is sequestered by this natural growth that is of particular interest here. The value of this lost carbon is likely to be significant if this forest is lost in perpetuity.

Table 2.6 Carbon Value of Foregone Sustainable Harvest (\$/tC/ha)

	Discount rate	Pasture	Secondary Forest
Low CO_2 Damage – \$6/tC	3%	\$718	\$11
Central CO_2 Damage – \$20/tC	**3%**	**\$2392**	**\$36**
High CO_2 Damage – \$45/tC	3%	\$5383	\$81
Low CO_2 Damage – \$6/tC	*6%*	*\$278*	*\$9*
Central CO_2 Damage – \$20/tC	*6%*	*\$928*	*\$28*
High CO_2 Damage – \$45/tC	*6%*	*\$2087*	*\$64*

The carbon value of the (foregone) sustainable harvest is indicated in Table 2.6. The data underlying these calculations are: (a) regarding quantities, that our earlier estimate of the sustainable harvest of $5m^3$/ha translates into a carbon benefit of approximately 2.3/tC/ha (above and below ground); and (b) regarding prices, that carbon is valued as in the calculation of *NCA* described above. If land is permanently cleared (i.e. slash-and-burn followed by pasture) then, using a 3% discount rate, this value of lost carbon is estimated to be \$2392 (in the range of \$718-\$5383). While these results are clearly sensitive to choice of discount rate, it is also interesting to note that if land reverts to (secondary) forest (following the cessation of farming) then the carbon value of this lost sustainable harvest is *significantly* lower. For example, the central estimate (using a 3% discount rate) in this scenario is \$36/ha while the value under pasture is two orders of magnitude greater.

2.5 Results and Discussion

Our foregoing model and discussion makes it clear that the deforestation accounting problem raises varied empirical issues regarding (net) changes that arise when a unit of land is deforested. As an illustration, Table 2.7 summarises our previous discussion of the value of excess deforestation assuming that a unit of land is permanently cleared (i.e. slash-and-farming followed by pasture). As regards choice of discount rate, carbon values are discounted at a 3% rate and all other values (sustainable timber harvest, local WTP and agricultural production) are discounted using a 20% rate. On this basis, excess deforestation is estimated to be \$3070/ha.

Two points are worth noting in this respect. Firstly, the range of possible values is large, i.e. somewhere between \$1482-\$7421. This is largely explained by alternative assumptions regarding the magnitude of discount rates and the damage caused by a tonne of carbon emitted. Secondly, the loss (in perpetuity) of the sustainable harvest of timber accounts for a significant proportion of the value of excess deforestation. This loss is made up of both the commercial value of the lost timber and the carbon that would have sequestered by this natural growth. By contrast, the sum of local and global WTP (non-use value) is (marginally) not in itself enough to offset agricultural returns. Put another way, it is the foregone sustainable harvest that appears to 'tip' the balance such that the switch from forest to slash-and-burn agriculture can be characterised as, other things being equal, wealth-decreasing.

Table 2.7 Value of Excess Deforestation in 1995

	$/ha
Local WTP	$335
	($275-$360)
+ Global WTP	$147
	($47-$225)
+ Sustainable timber harvest	$695
+ Carbon value of sustainable harvest	$2392
	($718-$5383)
- Agricultural returns	$499
	($253-$758)
Excess Deforestation	$3070
	($1482-$7421)

In order to estimate genuine savings in Peru in 1995, the data in Table 2.7 and other relevant forest-related data must be aggregated across total land deforested (which we take to be 210 000 ha in 1995). These data are then combined with other relevant economic variables, including gross savings, depreciation of produced capital and depletion of resources (including forest resources). The sources of these data are World Bank (1999) and World Bank (1997). In 1995, the gross savings ratio was 19.9% (the period average of this ratio during 1970-1997 was 20%) while the depreciation of produced capital was 12.2%. World Bank (1997) also estimates that resource rents (as a % of *GNP*) were relatively high during the 1970s and 1980s – e.g. between 4 and 18% (with a period average of 9%) reflecting the depletion of oil resources (and to a lesser extent, various metals such as copper and nickel). However, during the 1990s resource rents declined and in 1995 were 1.5% of *GNP* in Peru.

Table 2.8 estimates genuine saving both in dollar terms and as a percentage of *GNP*. Genuine saving is defined here as the rate of gross saving subtracting depreciation of produced capital and resource depletion, with the latter defined so as to include both non-forest and forest resources. What Table 2.8 shows is that while, on one hand, gross savings rate in 1995 was 19.9%, genuine savings is, on the other hand, only 2.4%. Furthermore, genuine saving lies in the range of -0.5% to 4.0%. In other words, once a relatively wide definition of asset consumption is taken into account, genuine savings in Peru are, in the aggregate, either negative or at best positive but low relative to *GNP*.[14] Furthermore, the net change in forest assets alone that occurs when land is permanently switched from forest to agriculture is 3.7% of *GNP* (in the range of 2.1% to 7.1%). Indeed, a significant proportion of this consists of non-market benefits lost in perpetuity (although the sensitivity of estimates to assumptions regarding, in particular, *NCA* should be noted). This offers at least some insight as regards how green national accounting can be used to re-evaluate development prospects in the presence of deforestation.

Table 2.8 Genuine Savings and Deforestation in Peru, 1995

	$ million	% GNP
Gross saving	11354.8	19.9
- Depreciation of produced capital	6977.9	12.2
- Non-renewable resource depletion	850.3	1.5
- Timber depletion	660.5	1.2
- Net carbon accumulation	832.2	1.4
	(249.5 – 1819.7)	(0.4 – 3.2)
- Excess deforestation	644.7	1.1
	(311.2 – 1558.4)	(0.5 – 2.7)
Genuine saving	1389.2	2.4
	(-512.0 – 2305.4)	(-0.5 – 4.0)

It is worth mentioning that our approach neglects the notion of strong sustainability. This is characterised by Pearce *et al.* (1989) as the idea that there are some amounts of critical natural capital that must be preserved if welfare is to be maintained – there are

[14] It is interesting to note that this basic conclusion does not change if a discount rate of 10% is used to discount "national" values such as local WTP, sustainable timber harvest and agricultural production.

essentially no substitutes for certain natural assets.[15] Many experts and lay-people alike would claim that tropical rain forests are critical stocks of living natural resources that provide life-support functions. One way of capturing this notion of a critical amount of rainforest, within our extended accounting framework is by assuming that,

$$U_L^W \to \infty \text{ as } L \to \bar{L}^+,$$

where $\bar{L}^+$ is the critical area – i.e., as forested area declines to the critical amount, arbitrarily large losses in (global) welfare are associated with deforestation of a marginal hectare.[16] The resulting excess deforestation would show up in our model as a large loss in land value. Thus, if global preferences are taken into account, the optimal program will also be strongly sustainable, because the rapid increase in global willingness to pay will quickly reduce the amount of deforestation. While this approach can handle strong sustainability in principle, in practice it requires good measures of global willingness to pay for conservation and sufficient scientific and economic information (concerning the damages resulting from loss of rainforest) for preferences to reflect the appropriate trade-offs that would underpin this willingness to pay.

Finally, in our deforestation model, local consumption (i.e. forest clearance) decisions reduce welfare in other countries. In other words, the optimal mix of forest and agricultural land is different to that which currently prevails given that farmers, for some reason, cannot capture the value of conservation benefits. In reality, attempts to reduce excess deforestation will have to translate these values into transfers that farmers in Peru can appropriate. Such mechanisms, in essence, have been put into effect by the Global Environment Facility (GEF) and endorsed in international environmental agreements such as the Convention for Biodiversity. In this deforestation case, the property right (to the forest) lies with the forested country (and

[15] Hence, this concept is distinct from those interpretations of strong sustainability as conserving natural assets in some aggregate sense. We explore the concept of strong sustainability in more detail in chapter 6.
[16] This could correspond to a physical process, such as rapid deterioration in forest quality and diversity once a critical threshold has been breached.

moreover, farmers). This implies that the reduction in genuine savings associated with excessive deforestation (from a global perspective) is linked to the extent to which compensation would be owed to the forested country if it reduced its deforestation to globally optimal levels. To put this another way: if there were actual international mechanisms that transferred income to the forested country in order to reduce deforestation, then excess deforestation would represent a tangible (as opposed to hypothetical) loss in land value. Our deforestation accounting problem therefore reinforces the rationale for actual international mechanisms that transferred income to a forested country in order to reduce deforestation. However, it is useful to have the implications for green national accounting and sustainability made clear as well.

2.6 Conclusions

We have investigated how a developing country might green its national accounts for current 'excess' deforestation and have illustrated how our framework can be implemented in practice. While, much of the literature to date has largely focused on the net accumulation of timber, we have presented an extended national accounting model of the costs and benefits of deforestation in which land-use is switched from forest to slash-and-burn agriculture. In particular, in this framework, it is the value of excess deforestation and net carbon accumulation that emerge as additional parameters of potential importance when accounting for the changes in wealth that occur when land is permanently cleared of forest. Using a range of market and non-market data, reflecting these changes, we have constructed an estimate of the genuine savings rate for Peru. This is based on the extended net saving rate derived from our model (i.e. *NNP minus consumption*). Sustainability requires that genuine savings should not be negative in the aggregate.

Our calculations apparently show that the Peruvian genuine savings rate in 1995 was either negative or at best positive but low in proportion to *GNP* in that year (i.e. in the range of -0.5% to 4.0%). This result is found despite a robust (gross) savings performance in Peru. A number of caveats and suggestions for future research are worth discussing. While our theoretical model allows for the change in asset value on land cleared of forest to be specified in relatively precise terms, in practice there is considerable uncertainty regarding the data needed to calculate these terms and the appropriate magnitude of discount rates. This gives rise to potentially wide ranges of values for much of the data as we have seen. As regards outstanding conceptual issues, there remains a need to investigate, in more detail, how the notion of strong sustainability can be accommodated into our analysis in both theory and practice.

Finally, we have made some inferences regarding the implications of green national accounting for national and global initiatives to conserve the world's remaining tropical forests, which are worth further exploration. Our findings reinforce the rationale for a commonly cited appeal for actual international mechanisms that transferred income to a forested country in order to reduce deforestation. The contribution of green national accounting, as we have shown, is to illustrate this in terms of the reduction in genuine savings associated with excessive deforestation. This, in turn, is linked to the extent to which compensation would be owed to the forested country if it reduced its deforestation to globally optimal levels.

Appendix 2.1 Accounting for Deforestation

2.1.1 Key to Symbols Used

A = Land used for agriculture (hectares, ha.)

L = Land covered by forest (ha.)

d = Deforestation (ha.)

S = Total stock of timber (cubic metres, m^3)

R = Timber harvest on forested land (m^3)

X = Global stock of carbon dioxide (tCO_2)

g = Natural growth of standing forest (m^3)

g_L = Unit natural growth of standing forest (m^3)

α = tonnes of CO_2 per m^3 of timber

n = Share of global CO_2 stock of study country

η = Rate of natural dissipation of atmospheric CO_2

GNP = Gross National Product

NNP = Net National Product

G = Genuine Saving

K = Produced capital stock

C = Consumption

F_A = Unit agricultural returns

F_L = Unit production externality provided by standing forest

F_R = Unit timber rents/ stumpage value

F_K = Marginal productivity of (produced) capital

f' = Marginal cost of forest clearance

b = Unit marginal damage of CO_2

p_L^W = Unit global willingness-to-pay for standing forest

p_L^l = Unit local willingness-to-pay for standing forest

r = Rate of time preference

U = Utility/ welfare

H = Current-value Hamiltonian

γ_i = Shadow prices

W = Total wealth

2.1.2 Deforestation and National Accounting Model

For the study nation we assume that there is a fixed amount of land that can either be used for agriculture (*A*, measured in hectares) or is covered by forest (*L*), and that deforestation is the process of conversion of some amount of forest land into agricultural land. The area deforested each year d is initially covered with a forest stock of density S/L (in cubic metres per hectare), where S is the total stock of timber. On forested land the amount of harvest is given by R.

Standing forest grows according to a function $g(S,L,X)$, with g_S following the usual pattern of being positive, then zero, then negative (i.e., for a fixed L the growth curve is an inverted 'U' which defines a maximum sustainable yield and a long run equilibrium growth rate of zero), while $g_L > 0$. X is the global stock of carbon dioxide (CO_2). This CO_2 fertilises forest growth, so that $g_X > 0$. CO_2 dissipates naturally as described by the function $n(\eta X)$, where η is the share of the study country in the total global stock of CO_2.

Slash and burn is the assumed forest clearance mechanism in the study country. It is assumed that each cubic metre of timber yields α tonnes of CO_2 when burned, and symmetrically, that the growth of one cubic metre of timber absorbs α tonnes of CO_2 (in other words, the only source of carbon in trees is assumed to be atmospheric). The accounting identity for the stock of CO_2, given deforestation, natural growth and dissipation is therefore,

$$\dot{X} = \alpha \frac{S}{L} d - \alpha g - n(\eta X), \tag{A.1}$$

Deforestation is assumed to cost an amount $f(d)$ of an aggregate good that can be consumed, invested or spent on deforestation. Standing forest is assumed to provide production externalities. The national accounting identity is therefore given by,

$$GNP = F(K, A, R, L) = C + \dot{K} + f(d). \tag{A.2}$$

It is assumed that residents of the nation value consumption and standing forest, while the stock of CO_2 causes harm. It is also assumed that the rest of the world derives benefits from the existence of the forested area in the study nation. Therefore, the utility function for the forestry model is $U^I(C,L,X)+U^w(L)$, with $U_X<0$ and where U^I and U^W refer to the utility of residents of the study nation and the rest of the world respectively. Utility or welfare, U, to be maximised is defined as $U = U^I(C,L,X)+U^w(L)$.

The optimal growth model for this economy is specified as,

$$\max_{C,R,d} W = \int_0^\infty Ue^{-rt}\,dt \quad \text{subject to:}$$
$$\dot{K} = F - C - f$$
$$\dot{A} = d$$
$$\dot{L} = -d$$
$$\dot{S} = -R - \frac{S}{L}d + g$$
$$\dot{X} = \alpha\frac{S}{L}d - \alpha g - n$$

For shadow prices γ_i the current value Hamiltonian for this problem is,

$$H = U + \gamma_1(F - C - f) + (\gamma_2 - \gamma_3)d + \gamma_4\left(-R - \frac{S}{L}d + g\right) + \gamma_5\left(\alpha\frac{S}{L}d - \alpha g - n\right) \quad \text{(A.3)}$$

The first order conditions for this accounting problem are:

$$\frac{\partial H}{\partial C} = 0 = U_C - \gamma_1 \quad \Rightarrow \gamma_1 = U_C \quad \text{(A.4)}$$

$$\frac{\partial H}{\partial R} = 0 = \gamma_1 F_R - \gamma_4 \quad \Rightarrow \gamma_4 = U_C F_R \quad \text{(A.5)}$$

$$\frac{\partial H}{\partial d}=0=(\gamma_2-\gamma_3)-\gamma_1 f'-\gamma_4\frac{S}{L}-\gamma_5\alpha\frac{S}{L}$$
$$\Rightarrow(\gamma_2-\gamma_3)=U_C f'+U_C F_R\frac{S}{L}+\gamma_5\alpha\frac{S}{L} \tag{A.6}$$

From the first-order conditions we can re-write the current value Hamiltonian in expression (A.3) as,

$$H=U+U_C\dot{K}+\left(U_C f'+(\alpha\gamma_5+U_C F_R)\frac{S}{L}\right)d-U_C F_R\left(R+\frac{S}{L}d-g\right)+\gamma_5\left(\alpha\frac{S}{L}d-\alpha g-n\right)$$

The Hamiltonian in expression (A.3) is measured in terms of utility. However, it would be useful to have an expression that is more readily recognisable as a national accounting aggregate. That is, we need a rationale to measure *NNP* in more familiar terms – i.e. its dollar value. Hartwick (1990) relied on assuming a linear approximation of utility ($U(C)=U_C C$) and then dividing the Hamiltonian by the marginal utility of consumption U_C to derive *NNP*: i.e. $NNP = H/U_C$. While this is a practical proposal that has been used relatively extensively (Hamilton, 1997c) various critiques of this practice have been offered (see, for example, Aronsson *et al.* 1997).

In particular, Pemberton and Ulph (1998) have proposed that the rationale for measuring *NNP* does not depend on the specific utility function used. Rather it derives from applying an extended Hicksian concept of income to the accounting problem. This income concept is "extended" Hicksian because income or *NNP* is defined as that amount of produced output that can be consumed while leaving the present value of *utility* (i.e. wealth) instantaneously constant.

Hamilton (1997c) has generalised this approach to a model that allows for both production of natural resources and pollution emissions. The key points, from that contribution, that allow the Hamiltonian to be re-written as *NNP* are the following. Firstly, the Hamiltonian (e.g. as in expression (A.3)) is simply current welfare plus changes in various assets or 'genuine' (or net) saving: $U + U_C G$. For a constant rate of time preference r, this expression is equivalent to the product of r and (util-dominated) wealth: $U + U_C G = rW$ (for proof, see for example, Dasgupta and Mäler, 2000). In turn, the change in wealth can be written as: $\dot{W} = rW - U$ or $U_C G = \dot{W}$. That is, genuine saving, G, can be interpreted as the change in total (net) wealth. Therefore, $\dot{W} = 0$ if $G = 0$ (as $U_C > 0$). In other words, wealth is constant only if genuine saving is zero. Hicksian income or the maximum amount of produced output that can be consumed (leaving wealth constant) is *consumption plus genuine (or net) saving.* Given that $NNP = C+G$, this provides the rationale for measuring NNP in conventional (i.e. dollar) terms. That is, in the context of current accounting problem , and defining the marginal damages from carbon dioxide to be $b \equiv -\gamma_5 / U_C$, the expression for Hicksian income is:

$$NNP = C + \dot{K} + \left(f' + (\alpha b + F_R)\frac{S}{L} \right) d - F_R \left(R + \frac{S}{L} d - g \right) - b \left(\alpha \frac{S}{L} d - \alpha g - n \right). \quad \text{(A.7)}$$

The last term in expression (A.7) is the value of damages from the net accumulation of CO_2 in the atmosphere. Slash-and-burn therefore adds to the CO_2 stock, while the growth of timber on the remaining forested land and natural dissipation of atmospheric CO_2 reduces it. The preceding term is the value of net reduction in the stock of timber. This has two components: net harvest of timber on forested land ($R-g$); and, timber on deforested land ($S/L \times d$). Before that is the term representing the difference in the shadow prices of agricultural land and forested land. Here, marginal clearance costs (f'), damages from carbon dioxide emissions (αb) and the rental value of the timber that was burned are all part of the difference in prices between these two different uses of land.

It is also worth noting how, for example, global preferences for standing forest relate to the expression for NNP in expression (A.7). There are two points that need discussing in this respect:

Firstly, it might be argued that, for example, the welfare that residents in the rest of the world currently derive from standing forest needs to be accounted for: i.e. as a "consumption-like" item. An estimate of these (non-market) benefits would certainly be of interest if we wished to measure of economic welfare (see, for example, Hamilton, 1996a). However, given the definition of Hicksian income as the maximum amount of *produced* output that can be consumed (while leaving wealth constant), this offers a rationale as to why the value of the current flow of these (non-market) benefits might not be measured in *NNP*.

Secondly, in this model, land is an asset which can be used to for crop production or standing forest. In turn, the price of land under these distinct uses will depend on different factors. In particular, we can infer more about what factors determine these prices if we examine the steady-state conditions in expressions (A.8) and (A.9) below. In expression (A.8), we can see that (steady-state) γ_2, the (steady-state) shadow price of agricultural land, is related to the marginal returns to agricultural land (F_A). In expression (A.9), (steady-state) γ_3, the (steady-state) shadow price of forestland, is related to a range of factors such as the welfare enjoyed by citizens in the rest of the world from a hectare of land under standing forest (U_L^W).

At the margin, if land clearance is costless, we would expect the value of land under these two competing uses to be equal. In other words, for the marginal hectare, we would expect that (in the steady-state) $\gamma_2 = \gamma_3$ and that farmers are indifferent between land clearance and forest conservation (i.e. the marginal benefits of clearance are just equal to the marginal costs). When land clearance is costly there is some additional term reflecting investment in land-use change that we must be taken account of. In terms of our forestry model, it can be recalled from expression (A.7) that this investment term is related not only to marginal clearance costs (f') but to damages from carbon dioxide emissions (αb) and the rental value of the timber that was burned. This term drives a wedge between the value of land used for agricultural production and value of land under standing forest. Hence, if land clearance is costly, and we observe deforestation (i.e. d>0) then we can reasonably assume that agricultural returns less the costs of that investment must at least just equal the returns (and all of the marginal values this is made up of) from keeping the land under standing forest.

Expression (A.7) is an expression for income that we would expect to prevail if deforestation was optimal. However, there is a strong rationale for arguing that the income measure that we should be interested in is one where deforestation is non-optimal. Thus, in the real world we would expect that a variety of policy distortions and market imperfections can easily lead to *excess* deforestation. In the current context, "excess" can be interpreted as deforestation over and above that which would have prevailed on the optimal path. In such circumstances, the income measure in expression (A.7) might not be not a good indicator of the change in the value of the land asset. Put another way, a proper evaluation of the social costs of land clearance would make *NNP* reflect better the change in land asset value.

To characterise "excess deforestation" it helps to derive the efficiency condition for harvesting the marginal hectare in the steady state. At this point the marginal returns to agriculture must just equal the marginal returns to standing forest. This can most straightforwardly be done by examining the following dynamic first order conditions.

First recall that the expression for $\gamma_2 - \gamma_3$ can be written as follows, after substituting in the expression for *b:*

$$(\gamma_2 - \gamma_3) = U_C(f' + (F_R + b\alpha)\frac{S}{L})$$

The dynamic first order conditions for these shadow prices are given by:

$$\dot{\gamma}_2 = r\gamma_2 - \frac{\partial H}{\partial A} = r\gamma_2 - U_C F_A \tag{A.8}$$

$$\dot{\gamma}_3 = r\gamma_3 - \frac{\partial H}{\partial L} = r\gamma_3 - U_L^W - U_L^I - U_C F_L - U_C F_R g_L - U_C b\alpha g_L \tag{A.9}$$

These are the standard first order conditions for the maximisation problem that can be used to be derive the steady-state conditions for the accounting model. Hence, each shadow price change ($\dot{\gamma}_i$) is defined in terms of the product of the rate of time preference, r, and shadow price, γ_i, and the change in the Hamiltonian with respect to the appropriate stock i.

However, in what follows, we are interested in how these conditions can be used to define "excess deforestation" and therefore Hicksian income or *NNP* away from the optimum. Subtracting expression (A.8) from expression (A.9), and substituting the expression for $\gamma_2 - \gamma_3$ gives,

$$\dot{\gamma}_2 - \dot{\gamma}_3 = rU_C(f' + (F_R + b\alpha)\frac{S}{L}) + U_C(F_L + F_R g_L + b\alpha g_L - F_A) + U_L^W + U_L^I \quad \text{(A.10)}$$

We define one more dynamic first order condition for the shadow price of produced capital, $\gamma_1 = U_C$:

$$\dot{\gamma}_1 = r\gamma_1 - \frac{\partial H}{\partial K} \quad \Rightarrow \quad \frac{\dot{U}_C}{U_C} = r - F_K \quad \text{(A.11)}$$

That is, the rate of change of marginal of utility of consumption is equal to the rate of time preference minus the marginal productivity of capital (or the interest rate). Expression (A.11) implies that $r = F_K$ in the steady state where all rates of change fall to 0. Therefore expression (A.10) reduces to the following in the steady state:

$$F_K(f' + (F_R + b\alpha)\frac{S}{L}) + F_L + F_R g_L + b\alpha g_L + \frac{U_L^W}{U_C} + \frac{U_L^I}{U_C} - F_A = 0 \quad \text{(A.12)}$$

This expression for the shadow prices of agricultural and forested land in the steady state can be re-written as follows,

$$\frac{U_L^w}{U_C} + \frac{U_L^I}{U_C} + F_L + F_R g_L + \alpha b g_L + F_K\left(f' + (\alpha b + F_R)\frac{S}{L}\right) = F_A. \quad \text{(A.13)}$$

These terms are relatively simple to interpret. Starting with the left-hand side of the expression, the first two terms are, respectively, the (marginal) willingness to pay (WTP) of foreigners and national residents for a unit of standing forest. F_L is the production externality provided by a unit of forest. $F_R g_L$ is the rental value of the natural growth of forest on a unit of land – this is the sustainable harvest or off-take. $\alpha b g_L$ is the value of the carbon sequestered during natural growth on a unit of land. The next term is interest that would be earned if the sum of the clearance cost, carbon sequestration benefits and timber rental value for the marginal unit of deforested land were put in a bank. Finally, the right hand side is the marginal product of the unit of land under agriculture: i.e. agricultural returns per hectare.

If for a given hectare of land the left-hand side of expression (A.13) is greater than the right, then there is excess deforestation. If it is assumed that there are d^* such hectares and that the land use change is permanent, then the value of excess deforestation is given as,

$$\left(\frac{U_L^I}{U_C}+\frac{U_L^W}{U_C}+F_L+F_R g_L+\alpha b g_L+F_K\left(f'+(\alpha b+F_R)\frac{S}{L}\right)-F_A\right)\cdot\frac{d^*}{F_K}. \tag{A.14}$$

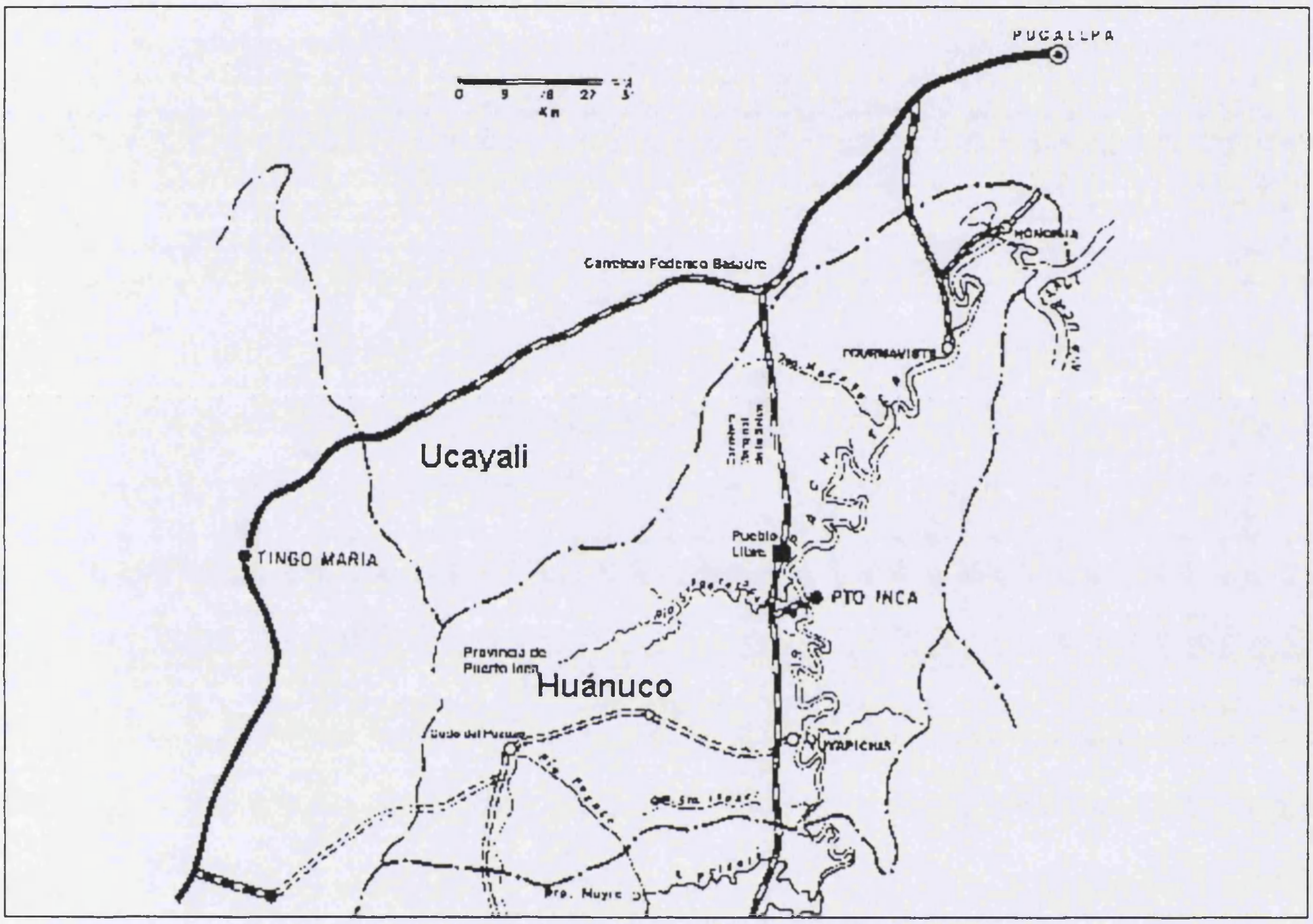
Note that the terms in F_K cancel and so, when expression (A.14) is subtracted from expression (A.7) to arrive at Hicksian income or NNP when there is excess deforestation, the final expression becomes,

$$\begin{aligned} NNP = C + \dot{K} - F_R\left(R+\frac{S}{L}d-g\right)-b\left(\alpha\frac{S}{L}d-\alpha g-n\right) \\ -\left(p_L^I+p_L^W+F_L+F_R g_L+\alpha b g_L-F_A\right)\cdot\frac{d^*}{F_K} \end{aligned} \tag{A.15}$$

where, $p_L^I = U_L^I / U_C$ and $p_L^W = U_L^W / U_C$. Genuine savings, G, in this model is defined as $NNP - C$.

Appendix 2.2

Figure A2.2.2 Map of Study Area



Appendix 2.3

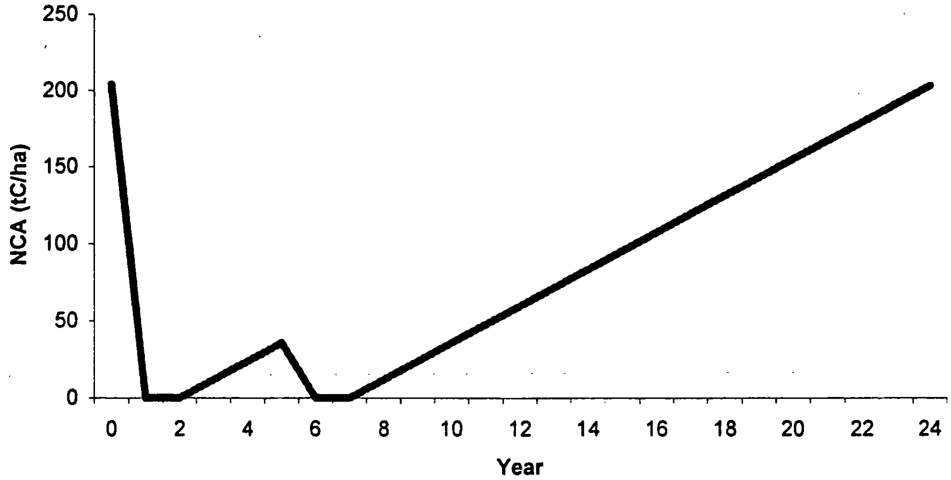
Table A2.3.1 Basic Agricultural Data

	Output	Unit Price	Labour	Wage	Capital	Unit Cost
		New soles	*Days/ha/yr*	*New soles*		
Rice	1333 kg	0.79	55	13.00	Seed 8 kg	1.00
					Seed Service	100.00
Corn	1000 kg	0.35	25	13.00	Seed 12 kg	1.00
					Sack 20 units	0.30
Yuca	10000 kg	0.10	34	13.00		
Plantain	600 bunches	3.00	29	13.00		
Fuelwood	5000kg		25	13.00		

Source: Nalvarte (1999)

Appendix 2.4

Figure A2.4.1
Net Carbon Accumulation: farm reverts to (secondary) forest



Chapter 3

International Trade and the "Ecological Balance of Payments"

3.1 Introduction

The role that international trade plays in measuring sustainable development has come under recent scrutiny, reflecting in part the wider and diverse debate about trade and sustainability. For example, by relaxing domestic natural resource constraints it has been argued that international trade allows any particular country to deplete natural assets abroad by importing its natural resource requirements. In turn, it is proposed that this apparent insight nullifies or alters sustainability criteria that are based on the investment of current resource rents regardless of their final destination. However, at least in the specific case of trade in commercial natural resources, it is arguable that the onus is on resource extracting countries to make provision for the loss of domestic natural assets whether for export or not. In this way, it can be argued that the fact that resources are often traded does not necessitate an alteration in recent proposals for measuring sustainability, particularly those based on savings rules. Nevertheless, it is also the case that international trade creates and reflects interesting interdependencies whereby (primarily) developed countries rely on imports of natural resources from (primarily) developing countries. Indeed, several of the former have expressed concern over their 'responsibility' for the depletion of resources elsewhere – for both selfish and altruistic reasons – and this may be of particular interest where an exporter is believed to be on an unsustainable path.

For these reasons we argue that it remains of considerable interest to examine these flows in a way that is analogous to an "ecological balance of payments" analysis. This, in turn, could form the basis of policies to assist exporters in adopting prudent resource and public investment policies. To facilitate this we use an Input-Output framework that allows us to re-attribute resource depletion from country of extraction to country of final-use. The empirical section of this chapter applies an Input-Output model to data on the global trading economy combined with data on natural resource depletion in 1980, 1985 and 1990. Our results provide a quantitative assessment of the significance of imports of resources – direct and indirect – required by say, Japan, the United States and the European Union. These results can also be disaggregated to permit an examination of trade relations vis-à-vis individual resource exporting countries. It is interesting to note that many of these resource exporters appear to be unsustainable at least on the basis of the criterion that the savings rate net of asset consumption (i.e. genuine savings) should not be negative.

3.2 International Trade and Sustainable Development

A basic requirement for (weak) sustainability is that the change in the (real) value of assets should not be negative in aggregate. In other words, in order to achieve sustainability, a country that is liquidating its natural assets must set aside sufficient economic resources to finance investment in other forms of wealth. For example, a resource-extracting country that follows a Hartwick-type rule to invest the current rents from resource depletion will ensure that, other things being equal, in the aggregate the change in the real value of assets is non-negative (Hartwick, 1977; Solow, 1986). In other chapters, we have described how this has led to a focus on adjusted net savings measures that account for the depletion of natural resources and environmental damage. Pearce and Atkinson (1993) and World Bank (1997) provide examples of this indicator – which Hamilton (1994) termed 'genuine' saving – for a range of developed and developing countries. One criticism of the measurement of genuine savings and its conceptual rationale is that it does not distinguish between those natural resources that are for export and those that are not. There are two distinct critiques that are worth noting in this respect.

The first concern is that international trade introduces novel measurement issues because of the respective signs of terms of trade effects for resource exporting and importing countries. This has been explored in the context of formal economic models of resource depletion and trade. Asheim (1986) examines the operation of the Hartwick rule when prices for exported resources change, Hartwick (1994) has explored the case of two countries that trade a natural resource, with one being a net exporter and the other a net importer and Sefton and Weale (1996) have modelled a similar situation. These contributions are all general equilibrium analyses, which require resource prices to increase according to the Hotelling rule.[1] The effect of an increasing resource price is a terms-of-trade effect or capital gain, which means that the annuity value of future changes in the resource price must also be accounted for when calculating savings requirements. For the resource exporter what this means is that investing some amount less than the full current resource rent (i.e. the Hartwick rule) will produce a constant consumption path (the definition of sustainability in these models).[2,3] Under these circumstances it follows that the effect of the changing terms-of-trade is to boost the Net National Product (NNP) and genuine savings of resource exporters and shrink those of resource importers.

In practice, how much saving the resource exporting country should do is determined by its expectations about future resource prices. The economic theory of exhaustible resources predicts these prices *will* rise and this drives the results of e.g. Sefton and Weale (1996). However, long-term historical trends indicate a decline in the price of many (subsoil) resources including oil (Nordhaus, 1992). In view of this, Vincent *et al.* (1997) conclude that under-saving is likely to be the greater concern for policy-makers in resource exporting countries. Put another way, if it is reasonable to expect that these terms-of-trade will continue to decline in the future (for example, because of technological change) then it is arguable that resource exporters should be investing some amount more (not less) than the full current resource rent in order to ensure a sustainable path. Hence, while trade in these frameworks may well alter the way in which sustainability needs to be measured, the implication of this for say resource exporters is ambiguous. Indeed, Atkinson *et al.* (1997) speculate as to whether, in view

[1] This states that the resource rental rate (equal to the resource prices in these models) must increase at a percentage rate equal to the endogenously determined interest rate.
[2] Pezzey (1998) analyses the specifics of this saving requirement. The optimum strategy for a resource exporter is to deplete its resources and invest (some portion of) the proceeds abroad. However, Pezzey suggests that the benefits of this strategy will be realised only to the extent that all exporters do not follow the same strategy.
[3] Hamilton, Atkinson and Pearce (1998) propose an additional term for the effects of the exogenous paths followed by international interest rates: the annuity value of future changes in international interest rates. That is, the present value (discounted at the domestic interest rate) of future changes in international interest rates times the stock of foreign assets.

of this, a balanced and prudent view is that the standard Hartwick rule still offers a useful rule of thumb.

The second criticism, exemplified by an important point made by Martinez-Alier (1995) is that (weakly) unsustainable countries apparently tend to be located in the developing world. Many developing countries are highly dependent on resource extraction activities and the depletion of these assets often means that high levels of savings need to be generated if aggregate real wealth is not to be run down. Given that these resources are often traded with developed countries, Martinez-Alier's implicit criticism is how does this resource trade affect sustainability and its measurement? Put another way, could international trade lead countries down an unsustainable path, and could indicators such as genuine savings, which do not consider trade to fundamentally alter analysis of the sustainability problem, mask this?

The specific question raised by Martinez-Alier (1995) is whether a full "ecological balance of payments" analysis would show that the US and Japan, which exhibited positive genuine savings in the analysis of Pearce and Atkinson (1993) (and World Bank, 1997), were actually unsustainable when global resource flows are taken into account. A more formal analysis is proposed by Klepper and Stähler (1998) who show that a resource importer, which perhaps is leaving its own resources intact as a result of a strict (unilateral) restriction on the use of domestic uses, could be characterised as 'buying' sustainability at the expense of a resource exporter. By modelling the total value of resource trade between countries, one response to this is that it is the savings of a resource importer that should be debited for use of a resource (Proops *et al.* 1999; Proops and Atkinson, 1998; Bailey and Clarke, 2000). However, it is unclear why the savings rate of a resource importing country should be reduced to reflect the depreciation of an asset (the resource stock of the exporting country) that does not belong to it, so the logic of the question may be faulted. Strong demand for the natural resources of an exporting country could plausibly lead such a country down an unsustainable path, but only if its own policies are deficient – for instance, if resource royalties are not captured, resource tenure is insecure or resource rents are not invested in other assets.

In the end it is the resource and public investment policies of the resource exporters that determines whether or not they are on a sustainable path. Nevertheless, developed countries are to a large extent reliant on foreign resources to support their domestic economies and, moreover, several of these countries – e.g. the Netherlands and Germany – have expressed concern over their *responsibility* for resource depletion in other countries. A variety of motivations underlie this concern such as externalities associated with resource extraction (Bosch and Ensing, 1995). This specific rationale complicates the measurement problem, as the magnitude of resource trade is likely to be a poor proxy for these externalities. However, a distinct concern is over what is done with the proceeds of resource depletion and this could provide the basis for focusing on resource trade itself. For example, studies such as Gelb (1988) and Auty and Mikesell (1998) have shown how difficult prudent resource and public investment policies can be to achieve. In a related vein, Sachs and Warner (1995) and chapter 5 in this thesis demonstrate that resource-dependent economies have enjoyed lower rates of growth than other economies since 1970. Furthermore, World Bank (1997) calculates negative rates of genuine saving for a range of resource-exporting countries.

More broadly, resource-importing (developed) countries could be concerned about unsustainable behaviour on the part of resource-exporting (developing) countries for reasons both of self-interest and altruism.

From a purely self-interested point of view, importers may be concerned about the security and stability of supply of natural resources. If important sectors of their economies are dependent on resource imports, then supply shocks and price shocks are potentially quite damaging. Unsustainable behaviour on the part of resource exporters – overly rapid depletion of sub-soil resources, for instance – creates risks for importers. Tracing the flows of natural resources in international trade and measuring the degree of dependence on individual countries or regions may therefore be in the interest of developed countries.

From a more altruistic viewpoint, governments in developed countries provide considerable amounts of finance on concessional terms to aid the development of poorer countries. Since unsustainable behaviour is in essence the consumption of assets, countries providing development assistance are increasingly concerned about the effectiveness of this assistance when aid recipients are on an unsustainable path – these types of concerns underlie the 'greening' of development finance institutions such as the World Bank.

Identifying resource trade linkages to particular developing countries or regions may be of interest to wealthier nations in targeting their development assistance, particularly 'policy-based' loans or grants with conditionality aimed at policy reform. In other words, a means of informing these concerns would be to provide an analysis of the extent to which economic activity in (primarily) developed economies is dependent on resource imports from (primarily) developing countries. This is analogous to the ecological balance of payments analysis as envisaged by Martinez-Alier (1995). However, it is important to note that while this could highlight the policy failures of resource exporters, we do not ascribe 'responsibility' for this to importers in the sense of debiting (explicitly or implicitly) a country's savings rate for the resources that it imports. That is, the interpretation of responsibility needs to be made with care.

There are a number of concepts and analytical approaches that could be used in this respect to construct this ecological balance of payments analysis.

For example, Rees and Wackernagel (1994) argue that the impacts of the economic activity of an individual country can be viewed in terms of the aggregate land area required to meet the needs of its population relative to the country's carrying capacity or available land. This is the so-called 'ecological footprint': the extent to which a particular country (or region) is reliant on resources from elsewhere to support domestic economic activity. These needs can be expressed in a number of ways such as the land required to satisfy nutritional requirements or by converting fossil energy into land required to grow the equivalent biofuel.[4] If this required area is larger than the area actually available to that country, then in this sense the country has an ecological footprint deficit (see also chapter 6). However, there are a number of problems with this

[4] For example, for a country's carbon dioxide emissions, the ecological footprint is calculated as the land area needed to absorb emissions of carbon dioxide – or alternatively the area required to produce an equivalent amount of energy using renewable resources.

approach. Arguably, the most problematic is that the main role of the 'footprints' notion appears to be the provision of a rhetorical focus for those who believe that sustainable development is threatened by international (resource) trade. For example, Rees and Wackernagel propose that one of the main policy implications of this work is that a country should reduce its ecological footprint primarily by reducing its reliance on traded resources (and increase the degree of self-sufficiency). That a country (or countries) may be 'sustainable' by following this route of greater self-sufficiency is debatable. However, it is extremely unlikely that a sustainable level of welfare will be maximised by this strategy.

An alternative and more appealing approach is suggested by Input/Output (I/O) analysis (see, Miller and Blair, 1985; Førsund, 1985). For example, Pedersen (1993) has used an I/O framework to analyse net exports of transboundary ("acid rain") pollution in Denmark vis-à-vis the rest of the world. Similarly, Young (1996) examines the relative pollution intensity of traded (export-oriented) sectors and non-traded sectors in the Brazilian economy. An analogous framework could be used to construct an ecological balance of payments that quantifies resource trade interdependencies between countries. It should be noted that this is not the only means of carrying out this analysis. A somewhat different approach is adopted by Bailey and Clarke (2000) using a computable general equilibrium framework to model and forecast sustainability prospects in the world economy taking into account resource trade between major trading blocs.

A useful feature of I/O analysis is that not only can direct flows of resources be examined but also those indirect flows. For example, although Japan imports timber resources from Indonesia or Malaysia, a significant portion of these resources could be embodied in produced goods for subsequent export to another country, say the USA. A reasonable definition – in terms of where the resource ultimately ends up satisfying (domestic) final demand – suggests that 'responsibility' for this particular resource depletion be attributed to the USA. In the remainder of this chapter, we develop an analytical approach based on I/O analysis that permits the calculation of the direct and indirect flows of resources in international trade, thereby extending the methods presented in Proops and Atkinson (1998) and Proops *et al.* (1999).

3.3 Framework for Analysing Flows of Resources in International Trade

We wish to attribute resource depletion to the country where it eventually goes to support (domestic) final demand. Input-output analysis captures these interactions in two ways. Firstly, there are *direct* exports of domestically extracted resources such as crude oil. This is typically what is conceived of as 'resource trade'. Secondly, an additional aspect to trade is *indirect* flows whereby resources are embodied in produced goods destined for markets abroad. As an example, country i might directly import resources from country j, which it then uses as an input in the production of traded goods (e.g. manufactures) for subsequent export to country k. In this instance, the input-output framework will attribute the resource depletion not to country i but to country k.

We seek to re-attribute resource depletion (i.e. current resource rents) from the country of extraction to the country where the resource was actually 'consumed'. Our framework for calculating these direct and indirect flows of resources is set out below but first, it is useful to define two indicators that elucidate these issues:

(1) We denote by N the (total) *domestic resource depletion* required to support Gross National Product (GNP). This is simply the value of depletion of domestically extracted resources familiar in green national accounting: i.e. the product of the unit resource rent and quantity of resources extracted or harvested. It is useful to note that N has two components: domestic extracted resources that are consumed domestically and (direct) export of resources abroad.

(2) The second measure N* is the *global resource consumption* required to support domestic final demand (i.e. consumption plus investment). N* excludes the domestic resources used to produce exports, and includes all of the foreign resources consumed in making up some portion of domestic final demand.

Next we define [N-N*]. This is a summary indicator of the ecological balance of payments. A positive dollar value of [N-N*] indicates that a country's use or consumption of global resources to support its own domestic final demand is *less* than the total resources it uses (i.e. depletes) to support its GNP. Put more simply, [N-N*]<0 indicates that a country is a net consumer of global resources. Examples of countries that are likely to have a negative dollar value of [N-N*] are Japan and the United States. Conversely, [N-N*]>0 indicates that a country is a net producer of global resources. Examples of these countries are likely to be resource abundant economies such as Indonesia. More generally, the calculation of [N-N*] across countries permits the quantification of the degree to which the developed world is reliant on the resources of developing countries to support their domestic economies.

In order to quantify these trade interactions we consider a simple two-country global economy. The basic accounting identities for these countries are:

$$\begin{aligned} X_{12} - X_{21} + C_1 + I_1 &= Y_1 \\ X_{21} - X_{12} + C_2 + I_2 &= Y_2 \end{aligned} \tag{3.1}$$

where, Y_i is country i's total output (or GNP); C_i is consumption in country i; I_i is investment in country i and; X_{ij} are the exports from country i to country j.

Relating the imports into each country to the GNP of that country, we define the following import coefficients:

$$q_{ij} \equiv \frac{X_{ij}}{Y_j}$$

So, we can write:

$$X_{ij} = q_{ij} Y_j \tag{3.2}$$

Substituting for X_{ij} in the above accounting identities and re-writing in matrix form we obtain,

$$\begin{pmatrix} -q_{21} & q_{12} \\ q_{21} & -q_{12} \end{pmatrix} \begin{pmatrix} Y_1 \\ Y_2 \end{pmatrix} + \begin{pmatrix} C_1 + I_1 \\ C_2 + I_2 \end{pmatrix} = \begin{pmatrix} Y_1 \\ Y_2 \end{pmatrix} \tag{3.3}$$

This result can be generalised to several countries giving,

$$q_{ij} = \begin{bmatrix} \dfrac{X_{ij}}{Y_j}, \ i \neq j \\ \dfrac{-\sum\limits_k X_{kj}}{Y_j}, \ i = j \end{bmatrix}$$

This generalisation to several countries, permits the calculation of Y in each country based on the domestic final demand (C+I) in all countries. We can re-write the output accounting identity in expression (3.3) in condensed matrix form.

$$\mathbf{Qy} + (\mathbf{c} + \mathbf{i}) = \mathbf{y} \tag{3.4}$$

This can be re-organised to give (where **I** is the unit matrix):

$$(\mathbf{c} + \mathbf{i}) = (\mathbf{I} - \mathbf{Q})\mathbf{y} \tag{3.5}$$

Solving expression (3.5) for **y**, by matrix inversion, gives:

$$\mathbf{y} = (\mathbf{I} - \mathbf{Q})^{-1}(\mathbf{c} + \mathbf{i}) \tag{3.6}$$

Next we define a resource depletion vector - 'n' (i.e. the product of the resource rent, p, and the extraction rate, R, expressed as a proportion of Y, in any country:

$$n_i = \frac{pR}{Y}$$

Then the depletion of total domestic resources required to support GNP, N, given by:

$$\mathbf{N} = \hat{\mathbf{n}}(\mathbf{I} - \mathbf{Q})^{-1}(\mathbf{c} + \mathbf{i}) \tag{3.7}$$

Where the 'hat' notation indicates that the entries in the vector **n** (or **c+i** below) are the diagonal elements of otherwise null matrices. An alternative expression allows the calculation of the total direct and indirect consumption of global resources required to support the domestic final demand of each country:

$$\mathbf{N}^* = \mathbf{n}'(\mathbf{I} - \mathbf{Q})^{-1}(\widehat{\mathbf{c} + \mathbf{i}}) \tag{7.8}$$

The calculation of [N-N*] is our summary indicator of an 'ecological balance of payments', in that it measures *net* consumption of global resources. In the following section, we provide an empirical application of this framework.

3.4 Empirical Application: Global Trading Economy

Our empirical analysis describes an ecological balance of payments for the global trading economy in 1980, 1985 and 1990. This is made up of 95 individual countries (OECD – 23; (Former) Soviet Union and Eastern Europe – 8; Africa – 23; Central and South America – 18; Middle East – 11; Asia – 11; Oceania – 1). The remaining trading blocs are residual regions, namely: the 'Rest of Africa', the 'Rest of Central and South America', the 'Rest of Asia' and the 'Rest of Oceania'. Data on the depletion of commercial natural resources and Gross National Product (GNP or Y) are taken from World Bank (1997). The resource data consist of depletion values for crude oil, timber, zinc, iron ore, phosphate rock, bauxite, copper, tin, lead, nickel, gold and silver. Trade flows data are taken from OECD (1994) and IMF (various). Hence, while our analysis cannot be claimed to be a full ecological balance of payments – in that we only analyse a subset of natural assets – the data covers relatively comprehensively (but not exhaustively) commercial natural assets.

Figure 3.1 illustrates net consumption of global resources [N-N*] by region for the year 1985. What this indicates is whether or not a particular region is either: (i) a net producer of global resources, i.e. [N-N*]>0, or; (ii) is a net consumer of global resources, i.e. [N-N*]<0. For example, as is expected, OECD countries are, on average, net consumers of global resources. It can also be seen that Sub-Saharan Africa (SSA) is a net producer of resources – although this is clearly not as pronounced as for the

Middle East & North Africa (which represent the big oil exporters). The SSA case mainly reflects resource exports of oil (Nigeria, Congo, Cameroon and Zaire[5]), metals (Zaire and Zambia) and timber (Cameroon). Latin America is a net producer of resources reflecting, to some extent, oil extraction in Venezuela, Mexico and Ecuador.

Figure 3.1 Net Consumption of Global Resources [N-N*]: By Region, 1985

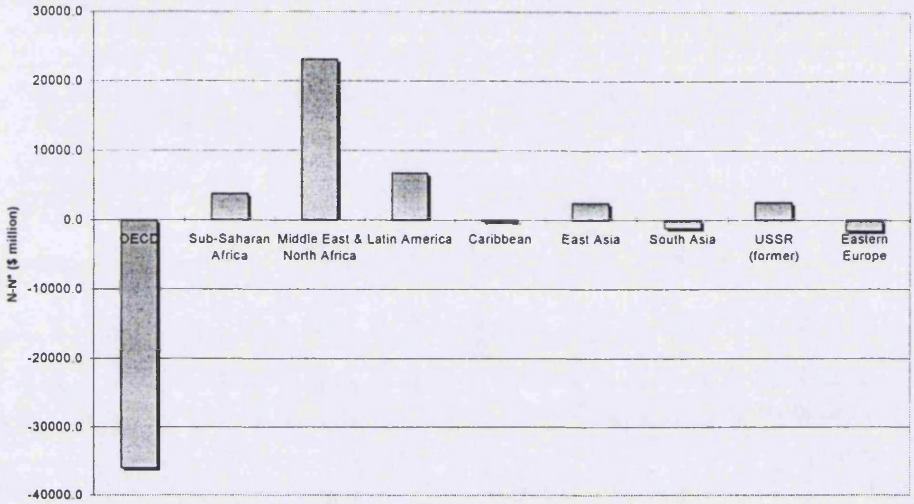
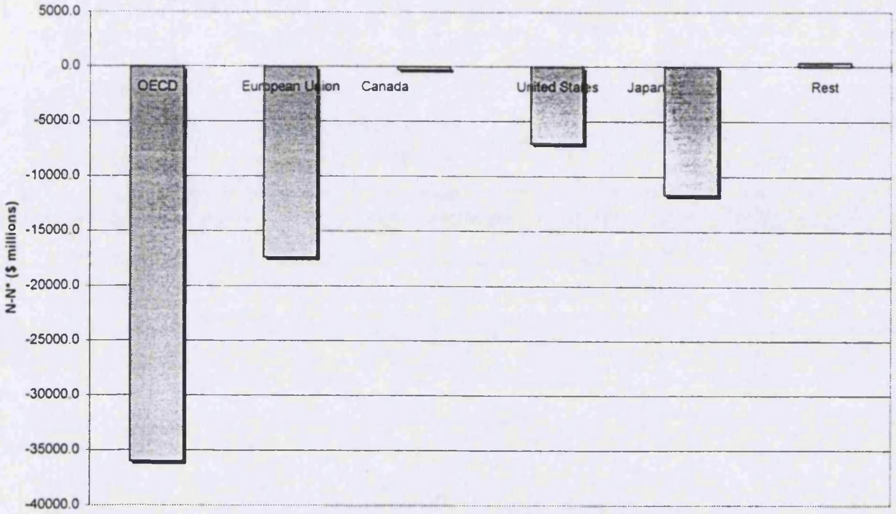


Figure 3.2 disaggregates the OECD region and in doing so reveals that much of its consumption is accounted for by the European Union trading bloc. The United States has relatively large endowments of its own resources but in order to support its large domestic final demand must import resources from abroad and is, as a result, a net consumer of global resources. While a proportion of this consumption is 'direct' so too is a proportion accounted for by resources embodied in the produced goods that the US imports. Japan is the largest net consumer (in dollar terms) of any individual country. However, even though it has few resources of its own, this is offset to an extent because Japan exports produced goods to the rest of the world and, as discussed, these exports will have resources embodied in them: the 'indirect effect'. Canada is a slight surprise here as it is (marginally) a net consumer of global resources, although it is a large exporter of oil (in terms of value of its other resources) and metals. It should be noted, however, that Canada imports produced goods from the rest of the world particularly the US and it is this indirect component of resource trade that is likely to be driving this finding.

[5] Now the Democratic Republic of Congo.

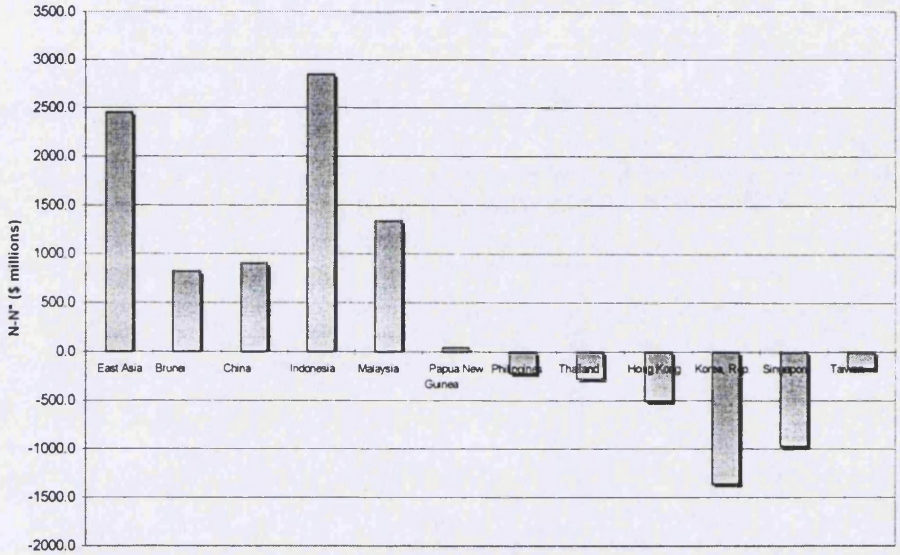
Figure 3.2 Net Consumption of Global Resources [N-N*]: OECD, 1985



The results in Figure 3.1 revealed that East Asia is, as a region, a net producer of global resources. Of course, this conclusion is an aggregate of different experiences across countries in this region. Hence, Figure 3.3 disaggregates [N-N*] across East Asia as follows. Net producers of resources include Indonesia, Malaysia and China, primarily reflecting exports of oil and timber to the rest of the world. In contrast the 'tiger' economies – Taiwan, Singapore and Hong Kong – are net consumers of global resources. Clearly, the growth of domestic final demand in these small open economies has resulted in relatively large resource import requirements. Interestingly, both the Philippines and Thailand are also net consumers of resources from the rest of the world.

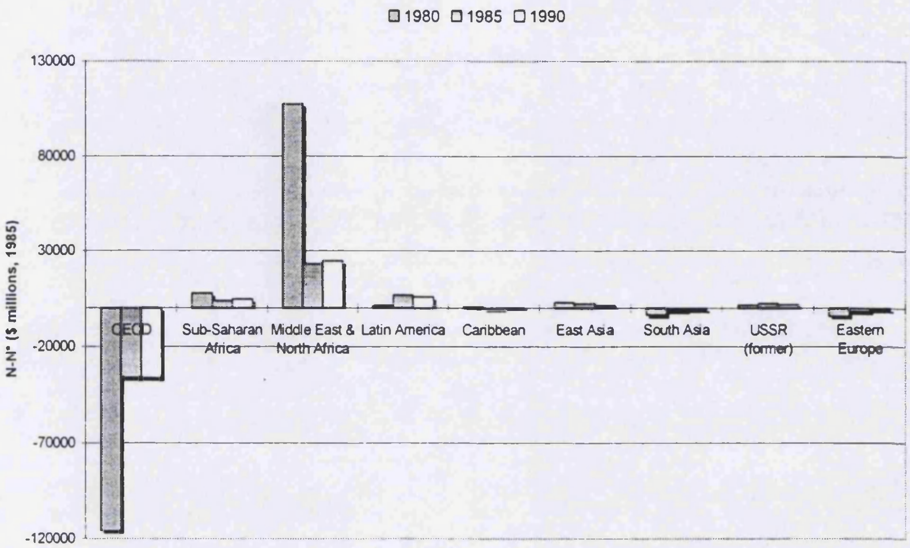
Some important caveats to this empirical analysis need to be borne in mind, not least the relatively high degree of aggregation that we have used in our I/O model. Underlying this is an assumption that the value of resource depletion in a dollar of exports is equivalent to the value of resource depletion in a dollar of GNP. This will probably understate actual resource exports (and thereby imports to other countries). It is likely that a greater proportion of the exports of say, oil producers will be made up of resources. Secondly, it is also likely that traded produced goods (e.g. heavy manufacturing) are more resource intensive than non-traded goods (e.g. some light manufactures and services). Correcting these biases would impose greater data burdens on our Input-Output framework necessitating at the very least identification of sectors such as primary production, services and manufacturing in each country.

Figure 3.3 Net Consumption of Global Resources [N-N*]: East Asia, 1985



It would also be interesting to examine the evolution of these linkages over time. The values of [N-N*] in 1980, 1985 and 1990 (in 1985 prices) are illustrated in Figure 3.4. Clearly, variations in estimates of [N-N*] over time will depend on changes in the price (specifically the relevant rental rates) as well as changes in the quantity of resources traded. The OECD and Middle East & North Africa dominate the overall picture. Figure 3.4 shows that in terms of the dollar value of [N-N*], consumption of global resources by the OECD decreased significantly in 1985 with little change in 1990 (relative to 1985). The mirror image of this is the experience of Middle East and North Africa. Clearly, these results reflect primarily changes in the international price of resources and in particular oil. This general downward trend over the period is also exhibited (albeit less pronounced) in SSA and East Asia. By contrast, a similar trend to that prevailing in the OECD is experienced in South Asia and Eastern Europe.

Figure 3.4 Net Consumption of Global Resources [N-N*]: 1980, 1985, 1990



The clear exception in Figure 3.4 is Latin America where [N-N*] increased (but was slightly less in value in 1990 than in 1985). It may be that – even in the face of declining international resource prices – some of these countries have attempted to increase their exports of resources to the world to earn foreign exchange in order to service external debt. For example, World Bank data on the physical quantity of resource exports indicate that Mexico and Ecuador both increased the quantity of oil exports by 51% and 56% from 1980-1990 respectively.[6] Indeed, Figure 3.4 illustrates graphically that the analysis of linkages over time based only on the total value of resource flows is significantly affected by the (short-term) volatility of prices for certain resources. This suggests that it would also be interesting to extend our empirical model to analyse physical flows of traded resources as well as the total value of these flows.

3.4.1 The Impact on Individual Countries

The above discussion described the extent to which a particular region or country relies on importing resources from the rest of the world to support its domestic final demand. It would also be interesting to identify those individual countries from which, for

[6] Kirk Hamilton (pers. comm.)

example, Japan's reliance on resource imports originates. This can be achieved by analysing in detail the elements of the total resource use matrix (**T**):

$$\mathbf{T} = \hat{\mathbf{n}}(\mathbf{I}-\mathbf{Q})^{-1}(\widehat{\mathbf{c}+\mathbf{i}}) \qquad (3.9)$$

For example, the column sum in **T** denoting Japan describes the dollar value of resources that Japan uses to support its domestic final demand. The relevant individual elements of this column in turn constitute the dollar values of resources (extracted in and) exported from individual countries (in our sample) to Japan. Hence, while it is often asserted that Japan's domestic economy is heavily reliant on the imports of resources from Malaysia and Indonesia, our framework enables us to evaluate this claim in quantitative terms.

Table 3.1 provides a summary of estimates of imports from countries upon whose resources the major trading blocs of Japan, the United States (US) and the European Union (EU) were particularly reliant in 1985. This reliance is described in dollar terms and relative to the GNP of the exporter. Regarding the specific question of Japan's dependence on the resources of Indonesia and Malaysia, our estimates indicate that Japan imported resources valued at $1500m from Indonesia, equivalent to 1.7% of Indonesian GNP. For Malaysia, this magnitude is somewhat lower at $386m although this still corresponds to some 1.2% of Malaysian GNP. Although the degree to which Japan relies on imports from Indonesia and Malaysia is large, the absolute dollar value for Saudi Arabia is larger. Japan is less dependent on Zambia and Mauritania, in dollar terms, but these values appear to be empirically significant relative to GNP in those countries.

With respect to the experience of the US, Table 3.1 indicates that this country imports its resources primarily from South and Central America. In particular, for Mexico and Venezuela the value of resource exports required to support the United States' domestic economy are $2429m and $1448m respectively. For Mexico, this is equivalent to 1.4% of its GNP and similarly, in Ecuador the value of resource depletion that is required to support the US's domestic final demand is some 3.7% of Ecuador's GNP. For the Congo, these proportions are 13.6% and 25.2% respectively. Lastly, Table 3.1 also identifies a number of African countries upon which EU countries rely for resources to support EU final demand. Relative to GNP, the results for Cameroon, Zambia, Egypt and Algeria are all worth noting.

Table 3.1 Resource Dependency by Country, 1985 – A Summary

Country	Resource exports ($m)	% of GNP of exporter	Genuine Savings (% of GNP)
Japan – Imports from:			
Saudi Arabia	1861.7	1.9	-30.7
Malaysia	385.9	1.2	4.8
Indonesia	1500.2	1.7	2.4
Mauritania	8.4	1.2	-20.8
Zambia	46.3	2.0	-31.8
US – Imports from:			
Congo	276.4	13.6	-22.2
Ecuador	408.3	3.7	-16.0
Trinidad and Tobago	208.6	2.5	-23.4
Mexico	2424.8	1.4	-4.2
Venezuela	1447.5	2.2	-23.9
EU – Imports from:			
Cameroon	352.2	4.2	4.0
Zambia	59.7	2.6	-31.8
Egypt, Arab Rep.	636.2	2.1	-2.9
Algeria	1160.1	2.0	-14.1
Nigeria	1314.2	1.6	-27.6

Source: author's own estimates; World Bank (1997)

The final column in Table 3.1 provides an estimate of the period average genuine saving rate (1980 to 1990) in the resource exporting countries that are reported here. The data are derived from World Bank (1997). Genuine savings are defined as (gross) savings net of asset consumption: i.e. the depreciation of produced capital and depletion of non-renewable and living resources.[7,8] It is interesting to note that in the majority of these countries, genuine savings were negative. Indeed, the only exceptions to this are Indonesia, Malaysia and Cameroon. However, the remaining 12 countries in Table 3.1 appear to have been on an unsustainable path in that, on balance, they have been liquidating wealth. Moreover, the average genuine savings rate was more than -10% of GNP in 10 of these countries. For example, all of the countries listed as those upon which the US has significant resource trade links have negative genuine saving, often significantly so as in the case of Ecuador, Trinidad and Tobago, Venezuela and the Congo.

Drawing a link to our earlier discussion, this should not be interpreted as meaning that the US is responsible for unsustainable behaviour in these countries. It is arguable that strong demand for the natural resources of an exporting country is only a proximate cause of unsustainable development, although some commentators have posited a more direct link between external indebtedness and resource exports (see, for a discussion, Pearce *et al.* 1995). More generally, whether a country is on or off an unsustainable path will largely depend on whether its own policies are deficient and the ease of correcting this deficiency. Nevertheless, it can be questioned as to whether resource importers will be entirely unconcerned about sustainability prospects of its trading partners. This could be either because this could create risks for importers or because the long-term welfare of exporters is also a source of concern for importers. A tentative policy application of our framework is that, given this hypothesised concern, this analysis may provide the building blocks needed by importing countries if they wish to target assistance to exporting countries that are on an unsustainable path.

3.5 Conclusions

We began by examining the claim that international trade in resources alters the criteria by which sustainability can be measured. Two distinct aspects of this claim have been

[7] That is, oil, natural gas, coal, nickel, iron, bauxite, copper, zinc, lead, tin, phosphate, gold, silver and forest resources.

[8] It should be noted that the theoretical frameworks underlying the measurement of genuine savings and our I/O analysis are somewhat different. That is, the latter assumes fixed coefficients whereas the former assumes substitution between relevant assets (or inputs).

proposed in the literature. Firstly, while there is a theoretical case for taking account of terms-of-trade effects in assessing the savings requirements that a resource exporter (or importer) needs to make, it is less clear – in practice – whether this implies more or less saving relative to the conventional Hartwick rule. Secondly, there appears to be even less justification for adjusting savings rates to reflect imports of resources. That is, the onus is on the exporting country to make adequate provision for the assets which it owns regardless of whether these are for export or not.

However, it is arguably the case that there are several (perhaps related) reasons why an "ecological balance of payments" analysis may be of interest to, for example, policy-makers in resource importing countries. Thus, we have identified a number of reasons – both selfish and altruistic – why this might be so. Hence, we have attempted to quantify these interdependencies firstly, by assessing the net consumption of global resources (i.e. [N-N*]). Secondly, we can further disaggregate the implied resource imports of each country (the dollar value of global resources used to support a country's final demand) in order to identify of those countries upon which Japan, the USA and European Union are particularly reliant. Our analysis calculates in quantitative form what has been the subject of much speculation in qualitative form: that developed countries are reliant on developing countries for resources and that, furthermore, it can be claimed that many (but not all) of the latter are on an unsustainable path (that is, have negative genuine saving).

The observation of negative genuine savings is one example of the emerging evidence that many notable resource exporters have found the implementation of prudent resource and investment policies difficult to achieve. While resource importers are not responsible for the unsustainable behaviour of exporters, one implication of our analysis is that the latter may wish to assist, in some way, the former back onto a sustainable path. The link to our analysis of resource trade is that plausibly this assistance could come from those countries, which are particularly reliant on these exported resources. Of course, the extent to which this is a realistic proposition depends on the degree to which developed countries care about unsustainability in other countries.

A number of other extensions to our model suggest themselves. Firstly, it would be interesting to separate out direct and indirect flows of resources in international trade. Secondly, changes in the value of resource flows over time are clearly significantly affected by the volatility of the prices of many of the resources that are commonly traded. It therefore would also be useful to examine changes in the volume of resources traded in addition to the value of these resources. Thirdly, regarding the question of the conservation of living resources, it would be worthwhile to distinguish between living resources (e.g. timber resources) and non-living resources (non-renewables) in future work.

Chapter 4

Measuring Corporate Sustainability

4.1 Introduction

As a result of official responses to both the Brundtland Commission (WCED, 1987) and the United Nations Conference on Environment and Development in 1992 (The 'Earth Summit'), most governments have adopted sustainable development as a national goal. An emerging debate is how economic sectors or businesses contribute to this objective. This has resulted in a number of concepts such as 'corporate sustainability' and 'corporate environmental responsibility' as well as a plethora of proposals to monitor progress towards corporate sustainability. There are two broad responses to this measurement problem. The first begins with the proposition that there is little in the notion of the 'sustainable business' or 'corporate sustainability' beyond defining a set of pragmatic guidelines whereby a corporate entity can monitor and improve its environmental performance. The measurement issue here is to find meaningful environmental indicators that capture the flavour of the broader sustainability debate; for example, by conveying environment-economy linkages. The second response is that lessons drawn from the green national or 'macro-' accounting literature allow us to define more formally what it means for a business to be either 'sustainable' or 'unsustainable'. Common to both approaches is an increased emphasis on 'micro-' accounting for external pressures or impacts attributable to a corporate entity.

By focusing on the latter response, we aim in this chapter to contribute to the current debate surrounding the meaning of full cost accounting. That is, accounting for a corporate entity's internal and external costs generated as a result of its economic activity (Canadian Institute of Chartered Accountants, CICA, 1997). Specifically, we focus on the generation of external environmental costs such as pollution and, furthermore, link this to the current debate regarding sustainable development. Indeed, the whole issue of evaluating external costs associated with environmental change within corporate accounts is judged to be underdeveloped in a recent survey by de Koning-Martens and van der Ende (1997). Hence, we outline one way in which full cost accounting can be extended and developed and propose an indicator of corporate sustainability that emerges from this process. As an empirical illustration we provide an example for air pollution and the UK corporate sector, although, in principle, the approach can be extended to other forms of pollution. Sustainable development is undoubtedly a complex notion open to numerous interpretations and many issues surrounding the usefulness and reliability of full cost accounting remain unresolved. However, we argue that, far from being impossibly confusing, it is possible to take practical steps towards measuring corporate sustainability.

4.2 Sustainable Development and Full Cost Accounting

The most celebrated formulation of sustainable development is that given by the World Commission on Environment and Development (the Brundtland Commission) (WCED, 1987). Over a decade on, substantial progress has been made in clarifying the many controversial issues that have emerged since this early shaping of the problem in the Brundtland Report (Pearce and Atkinson, 1998). The concept of sustainable development itself has been defined in a number of ways. The Brundtland Report defined it as "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs" (WCED, 1987, p43). Many have tended to reinterpret this as a requirement to follow a development path where human welfare or wellbeing does not decline over time (see, for example, Pezzey, 1989). It is this definition that we adopt in this chapter, although it is worth noting those interpretations that view the problem as one of not allowing development *opportunities* to decline or a requirement for greater public participation in decision-making (Barry, 1999; Toman, 1998).[1] If sustainable development is accepted as a desirable objective for society, then current decisions need to be made with some degree of concern for impacts on future generations. Much of the formal discussion

[1] Some contributions such as Monsanto (1998) like to stress the numerous definitions of sustainable development that have been proposed. Indeed, there is a danger that sustainable development or

regarding the sustainability of development paths has been conducted, either implicitly or explicitly, with the national or global economy in mind (Mäler, 1991; Atkinson *et al.* 1997). There has been, however, increasing discussion which has looked at sustainability from the perspective of smaller spatial scales (such as cities or regions) (Nijkamp and Perrels, 1994) and economic entities (such as sectors or firms) (Gray, 1990). It is the latter, i.e. corporate sustainability, which we focus upon in the remainder of this chapter.

Controversy arises when we consider how sustainability is to be achieved; i.e. what are the conditions for attaining sustainable development. While there is no single unified theory of sustainable development, all theories share a common theme in recognising that future welfare is determined by what happens to wealth over time. Indicators of sustainability, therefore, tend to emphasise either stocks of wealth or, more specifically, how a portfolio of assets is managed over time. The contribution of the recent debate makes it clear that portfolio management must also take account of natural wealth: for example, changes in environmental liabilities arising as a result of pollution. It is on the relative importance of components of the portfolio of assets that opinions diverge into two broad camps. The first is weak sustainability and states that total wealth should not decrease over time. Put another way, it is the 'overall' portfolio that is bequeathed to the future that matters. On this view, there is nothing special about natural wealth insofar as its liquidation is accompanied by adequate and offsetting compensating investment in other assets.[2]

In contrast, strong sustainability suggests a greater emphasis on the conservation of natural assets within the broader goal of prudently managing a portfolio of assets over time (for a more detailed discussion of this concept see chapter 6). For advocates of strong sustainability, it is the physical protection of absolute levels of natural assets that is a prerequisite for sustainability. The rationale underlying this is that natural assets provide complex ecological functions, crucial to the maintenance of life, and,

sustainability can be defined to mean anything and to justify any behaviour. One means of distinguishing credible statements is with reference to a well-grounded theory of sustainability.

[2] Weak sustainability does not permit overly rapid depletion of non-renewable resources or imply that excessive environmental degradation does not matter. Hence, in a world of 'over-pollution' and 'over-

furthermore, that these critical functions cannot be substituted for other assets (Norton and Toman, 1997). Natural assets that exhibit these characteristics are described by Pearce *et al.* (1989) as *critical* natural assets. It has been argued that allowing the stock size of these critical assets to fall below a threshold level could result in serious, if not catastrophic, consequences for welfare (Pearce *et al.* 1996). However, the problems involved in firstly, identifying critical assets and secondly, determining thresholds are far from trivial and are at the frontier of interdisciplinary research (Hamilton, 1997b). Yet, some progress is being made in developing decision-making rules such as safe minimum standards (SMS). A SMS is defined by Farmer and Randall (1998) as representing the suspension of standard arrangements for managing a given resource in order to prevent irreversible outcomes providing that this policy switch itself does not incur intolerable costs.

The predominant approach within the corporate environmental accounting literature is to take a definition of (usually, strong) sustainability from the broader literature and redefine it in this specific context. For example, Hawken (1993) argues that a firm is behaving sustainably if it does not reduce the capacity of the environment to provide for future generations. Similarly, Bebbington and Gray (1997a) assert that at a minimum the sustainable business is one that leaves the environment no worse off at the end of each accounting period than it was at the beginning. At first blush these definitions are enticing, although somewhat stringent, serving both to focus attention on responsibility for environmental impacts and to suggest that firms should be more responsive to these concerns. In addition, Bebbington and Gray (1997a) also note that the trend by groups such as the Business Council for Sustainable Development towards avowedly pragmatic programmes of action intended to be economically sound while decreasing environmental impacts.

Despite the on-going debate, there has emerged some consensus that a prerequisite to understanding sustainable development is the construction of green accounts and sustainability indicators. At the 'macro-' level, green national accounting has been proposed while at the 'micro-' or corporate level, considerable attention has been

depletion' one way to increase sustainability would be to implement policies that prevent wasteful use of resources or reduce pollution levels (Atkinson *et al.* 1997).

devoted towards corporate environmental accounting and reporting. The latter covers a diverse range of activities with, currently, little standardisation of existing practice (United Nations, 1997). If comparisons of environmental performance are to be made across the corporate sector this will require an overarching framework within which to evaluate these activities. For example, within the domain of green national accounting, the United Nation's Satellite Environmental and Economic Account (UN SEEA) serves this function. This, in turn, is an adjunct to the conventional UN System of National Accounts (SNA). The SEEA framework embodies natural resource accounts, resource and pollutant flow accounts, environmental protection expenditure accounts and the estimation of green accounting aggregates (United Nations, 1993; Atkinson *et al.* 1997). It is important to note that its primary role is to lend some degree of coherence to an otherwise impenetrable mass of data and indicators.

Within the corporate environmental accounting literature similar concerns can be identified (see, for example, Bennett and James, 1997). Particular attention has been given to those activities that explicitly identify and account for environmental expenditures incurred by firms. In other words, expenditures that have actually been incurred reflecting a corporate decision to address an environmental impact. At the 'macro-' or national level, environmental expenditure accounts typically have been used to address questions about the total economic burden of environmental protection and the distribution of this burden between sectors. Within individual corporations, a primary use is for monitoring which production techniques and inputs will minimise internal environmental expenditure in order to attain a given protection goal. The separation of environmental expenditures from general operating costs allows the former to be allocated to processes, products or budgets and, where relevant, to properly appraise capital budgeting decisions (Bennett and James, 1997).

It is argued that recording these environmental expenditures separately allows interested parties to gauge the extent to which firms are sensitive to environmental concerns (United Nations, 1997). Of course, high levels of environmental expenditure do not necessarily correlate with good environmental performance per se. In many respects, moreover, the goal of 'gauging concern' is better served by measuring external costs. In other words, pressures on the environment that, at present, go unabated. An external cost can be defined as a cost imposed by an entity as a by-product of its economic activity on third parties (e.g. households). For example, one such by-product is pollution, which imposes costs on others through adverse effects on health states or environmental quality. These costs are external because the firm takes no account of this outcome when deciding on the amount of pollution that it should emit (see, for example, Tietenberg, 1996). However, a firm that presently faces insufficient incentive to internalise the costs that it imposes on others arguably also has little incentive to monitor and report this activity. Hence, the search for *credible* and *meaningful* frameworks for reporting external costs has been a defining characteristic of much of the corporate environmental accounting literature (see, for a review, Mathews, 1997). There do exist significant and countervailing reasons for interest in these measurement issues. This includes the anticipation of potential future legislation to restrict pollution, 'green consumerism' and the environmental concerns of employees (de Koning-Martens and van der Ende, 1997).

One interesting development is the often-controversial process of quantifying external costs or 'full cost accounting' (see, for example, Gray, 1992; Rubenstein, 1994; CICA, 1997).[3] There is no single definition of full cost accounting. An excellent review is provided in CICA (1997) and from this is distilled a relatively wide definition based upon the integration of an entity's internal costs (including internal environmental costs) and the external costs of its activities. Put another way, in terms of environmental imputations, this includes the value of expenditures currently made in order to abate pollution and the costs of pollution generated that currently goes *unabated*. Hence, in principle, full cost accounting provides a comprehensive framework for evaluating the environmental aspects of corporate economic activity. Furthermore, the definition of full costs could include *all* those costs associated with an entity's economic activity not just those associated with environmental impacts. For example, the 'social' aspect of the measurement problem is an important feature of this where questions regarding quality of the work environment have been central (Mathews, 1997). However, following CICA (1997), we focus in this chapter, upon

[3] Firms may also provide external benefits, as well as external costs. There are two points here. Firstly, it is important to clarify to what extent the benefit is a 'true' externality and not captured by the firm. Secondly, even if a firm does provide an external benefit it is unlikely that it will provide it at socially efficient levels. In general, the empirical analysis of this issue is less than complete.

environmental costs and furthermore those environmental costs that are generated as a by-product of economic activity: i.e. external costs.[4]

It should be noted that full cost accounting is not the only means that has been proposed to take account of corporate-level environmental impacts. For example, an influential alternative is the triple bottom-line approach embodying indicators of economic, social and environmental concerns, presented in heterogeneous units. This is outlined, for example, in the guidelines produced by the Global Reporting Initiative (GRI), an influential group of non-governmental organisations, professional associations and multinational corporations (see, for example, GRI, 1999). Nevertheless, while we acknowledge this interest, full cost accounting remains a useful, if not exclusive, approach to understanding the external impact that the corporate sector imposes on society. Hence, European Commission (1992) proposes that exercises should be undertaken that better incorporate the 'full costs' of economic activity into corporate accounts. Contributions from CICA (1997) and Epstein (1996) also demonstrate the on-going significance of full cost accounting. Furthermore, it is arguable that this approach is more consistent with discussions in both the 'macro' (i.e. national) accounting and the broader sustainable development literature.

Full cost accounting for external costs feasibly covers a range of activities including elementary monitoring of physical indicators (such as environmental 'pressures'), as found currently in some company environmental reports or more sophisticated analysis of the 'full costs' of a firm's activity, where this is possible. For example, a handful of innovative efforts exist that account not only for firm-level environmental pressures but also for external impacts or costs caused by these pressures. Perhaps the most interesting feature of full cost accounting is that where practical costs are estimated in monetary terms (CICA, 1997). How this is to be achieved in practice is less clear and evidence from practical applications appears to indicate that most progress has been made in estimating firm-level external costs originating from routine emissions from point sources arising from, for example, energy generation (Ontario Hydro, 1993).

[4] Nevertheless, it is important to note that environmental impacts are associated with a diverse range of effects including human health, in addition to adverse effects on say, ecosystems.

There are more complicated, and largely unresolved, issues with respect to the treatment of emissions or discharges from non-point or diffuse sources. One much cited example is by BSO/Origin, a Dutch computer software consultants, which recorded a monetised imputation reflecting its external impact in an 'environmental value added statement' (Huizing and Dekker, 1992).[5] Ontario Hydro embraced a mode of accounting familiar from studies of energy externalities (see, for example, Smith, 1996; European Commission, 1995). This study not only modelled the dispersion of the firm's emissions but also quantified subsequent physical impacts and further used economic valuation techniques to translate physical effects into monetary terms. Tuppen (1996) notes that both AT&T and Dow Chemicals are considering a similar approach to the costing of external impacts. In the remainder of this chapter, we examine the rationale for full cost accounting of this type and evaluate its practical 'value-added' for measuring corporate sustainability.

4.3 Accounting for External Costs

From society's point of view the interesting question can be thought of in terms of the contribution of a given entity (e.g. business or sector) to sustainability defined in the wider sense (e.g. nation). From the entity's own perspective, the extent to which its contribution impinges on the sustainability of its own activity will also be of concern. The key to defining 'corporate sustainability' is to reconcile these two outlooks and full cost accounting for external costs provides one means of explicitly making this link. Thus, our concern is how existing accounting information relating to economic performance can be augmented with indicators that provide signals regarding the overall sustainability of a corporate entity and its wider environmental performance. In particular, the rationale for full cost accounting can be illustrated with reference to an extension of the *polluter pays principle* (PPP) to the domain of accounting. If the property right to a clean environment lies with downwind or downstream entities (i.e. 'victims') such as households, then damages caused by externalities are attributable to the polluter. In accounting terms this damage is a notional liability. It follows that any

[5] This re-stated the conventional profit and loss account in value-added form and attributed an explicit monetary value to environmental impacts. These included all upstream and downstream, direct and

external effect that is directly identified with the generation of the polluter's income be properly attributed in a full cost account. Indeed, this generalises to any decrease in welfare suffered by any individual in the global population as a result of emissions or discharges from the polluter.[6]

This says nothing in particular about how we might measure pollution damage (and thereby supports a range of potential accounting activity). However, if damage could be expressed in money values then it is possible to adjust the polluter's recorded income, in a complementary full cost account, by that amount corresponding to damages (specifically, externalities) directly associated with the generation of its income. This sum can be interpreted as the amount that, at least notionally, should be set aside in order to compensate the recipients of the pollution emitted. What then does this allow us to say about the sustainability of the entity? If there existed some mechanism to make a liability actual by enforcing property rights then this would represent a tangible decrease in the entity's income available say for investment in new assets. It is in this sense that, other things being equal, the entity is less sustainable as a result of the damage that it causes. Note that we are limiting in practice our inquiry to weak sustainability as well as restricting our attention to environmental aspects of the problem. In principle, this framework could be extended to consider stronger forms of sustainability or aspects of social sustainability. Whitby and Adger (1996) provide an example of the potential usefulness of full cost accounting as proposed here. They construct a land-use account extended to include imputations describing externalities that farming activities cause not just in the land-use sector itself but in a global context (for example, emissions of greenhouse gases). The imputed items represent the value of damage caused by the polluter and furthermore are directly associated with the generation of farming income.

Insofar as we have defined responsibility for pollution incidence, we have understood this in a direct sense only. Clearly, this will not appeal to everyone's perception of ultimate moral responsibility (or 'blame') for damage caused. Nor does the principle

indirect costs associated with the firm's activity. Impacts were valued at the costs of abating the damage caused.

[6] Chapter 7 in this thesis examines a situation where the PPP 'competes' with other desirable criteria.

that the polluter should pay, which is so prevalent in the rhetoric of environmental policy, necessarily explicitly define who the polluter ultimately is. For example, some firms might argue that responsibility resides with final consumers. In other words, much of the damage caused can be attributed to the end of satisfying final demand in other sectors. It follows that damage could be charged instead to the environmental account of these final consumers. Pearce and Newcombe (1998) describe a complex notion of responsibility based on sharing of the 'blame' between producers and consumers across a given product chain. For example, a furniture manufacturer may wish to demonstrate accountability to its customers by documenting that its timber inputs come from sustainably managed sources even though it is not directly responsible for unsustainable forestry practices. Whether any one definition of responsibility is correct in that it will satisfy all criticisms is doubtful. This could only be achieved if there was an unequivocal 'social contract' to inform this complex question.

4.3.1 Full Cost Accounting and Valuation

If the direction of policy at both the national and the European level continues to develop in line with the 'polluter pays principle' there exists some impetus for full cost accounting as described above. In the case of Ontario Hydro it was explicitly stated that the anticipation of future policy and enhancement of future competitiveness were the primary reasons driving its full cost accounting exercise (US EPA, 1996).[7] Indeed, the UK Landfill Tax was introduced in 1996 to internalise some of the external costs associated with the disposal of solid waste; thereby confronting waste generators with the 'full costs' of disposal (Brisson, 1996). An estimated value of damage caused by disposal provided the policy target for setting the tax rate: i.e. tax per tonne of waste disposed (Riley, 1996). The use of damage valuation in 'target setting' is currently being considered in a proposed UK Aggregates Tax. It is likely that the full or social cost of externalities caused by this sector will be used to calibrate any resulting tax charged on each tonne of aggregate extracted (ENDS

[7] Similarly and for much the same reason, within Europe there has been increasing interest by the corporate sector in exercises that value the social costs of fuel cycles such as the European Union funded ExternE work (European Commission, 1995, 1999).

Report 280, 1998). Hence, there does seem to be a policy context to placing monetary values on environmental impacts in the way suggested above.

By monetary valuation we mean the value in 'dollar' terms of the stream of benefits that society derives from the environment. Measuring changes in this stream could provide the linkage that is sought between human welfare and the environment in discussions of sustainability at the corporate level and beyond. Of course, these benefits are precisely those that typically are not bought and sold in markets. In contrast, conventional accounting practice is founded on the recording of transactions made in the market or arising from observed transactions. The value of pollution damage is just like any other monetary magnitude in that it consists of a price multiplied by a quantity. This price, in principle, is intended to reflect *marginal willingness to pay* defined as the amount of income that an individual will be prepared to give up in return for a small (i.e. a marginal or unit) improvement in say, environmental quality. In accounting terms, any change in a stock is valued by the marginal willingness to pay for a unit increment in that stock. This stock can be thought of as an environmental liability and, hence, pollution is an accumulation of a liability. Furthermore, if damage to stocks, in turn, can be traced back to an 'emission' (e.g. nation or some other entity) then it becomes possible to express this 'price' in terms of the unit value of damage caused by a pollutant: e.g. dollar per tonne of particulate matter. Indeed, the theory of green national accounting tells us that this is, in principle, the ideal way in which to account for pollution (Hamilton, 1996a). Hence, unit (marginal) values damages can be multiplied by total emissions attributable to a sector or corporate entity.[8]

One alternative response would be to impute the level of abatement costs, or maintenance costs, that the firm would incur if it actually reduced emissions to some predetermined level (Gray, 1992; United Nations, 1993). Many feel that environmental quality should be improved such that it is a natural corollary that the cost of pollution is the expenditure required to take us back to some desirable point. A

[8] However, other valuation procedures might also be relevant. Definitions of liability often refer to (some portion of) the area under the (marginal) damage curve (Segerson, 1994). Unless this curve is horizontal, a

further rationale would be that if there was a potential policy intervention, e.g. some form of regulation, that forced the firm to reduce its emissions to this level, this could provide valuable information on the impact on the firm's bottom-line. This is a distinct exercise to the accounting for internal expenditures described earlier as it involves quantification of the costs of pollution not yet abated. As we show below in the empirical part of this chapter, data on the unit damages of *air* pollution emissions, based on willingness to pay concepts, are increasingly available for developed economies, as are data on the levels of air pollution emissions. Data on abatement costs are generally not collected, but models embodying the appropriate technological and economic information to estimate marginal costs are becoming available. Even so, maintenance costs are far from straightforward to estimate. Hence, maintenance cost is not a simple alternative to methods based on willingness to pay. In addition, it is easy to demonstrate that basing an imputation on abatement costs can grossly understate the impact on human welfare where economies are 'over-polluting' (Hamilton, 1996b).[9]

The focus on estimating damage costs also offers a desirable signalling property. Parker (1997) notes that standard accounting techniques are frequently perceived as 'penalising' firms for tackling environmental problems. By incurring abatement expenditures the company, in turn, incurs increased operating costs and, other things being equal, reduced profits. If damages were expressed in comparable monetary units then these expenditures could be demonstrated in the context of a comparable decrease in damage caused. Moreover, a decrease in damage can be also reported over time. Hence, accounting for external costs may provide firms with an incentive to search for economic ways of decreasing these costs and thereby lead to a more socially beneficial use of environmental resources (Ontario Hydro, 1993). CICA (1997) also argue that this information is useful to internal users both in terms of evaluating alternative strategies and practices, monitoring progress towards goals and to external users such as investors and consumers. Valuing pollution also permits direct comparison with other relevant economic magnitudes in the same units and, therefore,

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charge based on marginal willingness to pay times the units of pollutant emitted will overstate the amount that is required to compensate victims.

[9] That is, relative to the social optimum.

conveys information about corporate environmental performance in a way that makes sense both to environmental and financial managers.

It is important to note that the suggestion that the external costs of pollution might be valued in corporate accounts is not new and can be traced back to, for example, Estes (1976). However, Mathews (1997) argues that these initial contributions, by and large, have not had a lasting influence on business or accounting communities. This, in turn, can be attributed to doubts that the costs of pollution can be reliably valued and a perceived reluctance of corporate entities to sign up to any number purporting to measure external costs. Given this complexity many would argue that physical data on pressures is the best that can be hoped for. If so, then improving the informational value of these data is a major challenge for environmental reporting. In the context of the discussion in this chapter, it is important to ask how the more recent debate regarding full cost accounting for external costs differs from its predecessor.

Reticence on the part of corporate decision-makers to divulge environmentally sensitive information is probably the trickier aspect of the problem to overcome. In this respect, corporations have little incentive to reveal environmental data not based on compliance driven overheads (Bennett and James, 1997) or to reveal 'bad news' in general. For example, Gray, Bebbington and Walters (1993) report that only 3% of companies report contingent liabilities (i.e. likely future dollar losses for the firm that can be estimated with reasonable certainty) in their annual company reports. It is arguable that this also implies that corporations will be reluctant to reveal any meaningful environmental data, which could even call into question the advantages of those reporting frameworks that the corporate sector has been more favourably disposed towards (such as the triple bottom-line). Some of the controversy can be explained by identifying more clearly the audience for which the information is intended. Thus, while it may be socially desirable that corporations provide some indication for external users of external costs reluctance on the part of those same corporations is understandable where, for example, these estimates might later be used as evidence of legal liability (CICA, 1997). Yet, at the same time, it could be perfectly rational for a corporation to estimate these costs for internal purposes perhaps either as an input to designing an environmental management strategy based on where it can, most effectively, make a difference. In turn, the need for this information could arise as a result of the anticipation of future public policy or the desire to play a more active role in influencing the design of public policy.

One of the central difficulties often raised by the accounting profession, is that valuing external costs requires putting a price on 'goods' and 'bads' that lie outside of the market and thus cannot be estimated using familiar, tried and trusted, techniques. Demonstrating that these non-market valuation methods are now sufficiently 'robust', therefore, is crucial. Clearly, significant progress has been made in this respect since Estes (1976) (see, for example, Bateman and Willis, 1999). However, some critics also argue that the environment 'should not be valued' or traded off against other desirable objectives (see, for example, Hines, 1991 and for a critique of the moral philosophy underlying this, Pasek and Beckerman, 1996). More usually it is argued that society's preferences for environmental goods cannot be boiled down easily to a single money value expressing willingness to pay (Hueting, 1980). This task is further complicated for routine emissions as environmental 'prices' are only one input into 'downstream' models of dispersion and impact (e.g. on human health or living resources). This is the basis of the approach taken by Ontario Hydro (US EPA, 1996) the end product of which can be expressed as the monetary value of damage caused per tonne of pollution emitted. Considerable progress has been made in estimating non-market environmental values in this way (Smith, 1996). For example, these have been especially researched in the context of the external costs of energy use – primarily air pollution – although little progress has been made for other environmental impacts, especially biodiversity. The external costs of energy have been measured in the USA in several studies (Oak Ridge National Laboratory and Resources for the Future 1994; Rowe *et al.* 1995, Smith, 1996) whilst the ExternE programme in Europe has established similar values (European Commission, 1995, 1999).

Given that there is a resource cost required in order to implement a system to monitor pollution and its impacts, full cost accounting for air pollution may be feasible only if use can be made of these existing results. This is known as benefits transfer (Desvousges *et al.* 1998). However, this process commonly uses single values as if they are applicable across all contexts and locations and so contains substantial (potential) margin for error. This error is, in turn, due to factors such as meteorology and differences in population at risk across locations (Krupnick and Burtraw, 1996).[10] Fortunately, some progress is being made in our understanding of this problem by the extension of existing dispersion models to consider multiple sources at different locations within and across countries (European Commission, 1999). However, it could be some time before estimates emerge that are robust against such criticisms (Pearce, Hamilton and Atkinson, 1999). Hence, it is with these caveats in mind that we illustrate how valuation estimates might be integrated into full cost accounting exercises in the next section.

4.4 Valuation and Full Cost Accounting – Examples

We wish to estimate the cost of air pollution attributable to corporate entities caused by five air pollutants: carbon dioxide (CO_2); methane (CH_4); sulphur dioxide (SO_2); nitrogen oxides (NO_X); and, particulate matter (PM10). Data describing physical emissions by sector are beginning to emerge largely as a result of on-going efforts to compile environmental accounts by national and international statistical offices (see, for example, Office for National Statistics, ONS, 1998). Examples of similar data at the company level can also be found in some environmental reports. Estimates of unit damages per tonne of pollutant emitted are shown in Table 4.1. These are drawn from an evaluation of energy externalities within the United Kingdom (European Commission 1995). The CO_2 and CH_4 estimates are from Fankhauser (1994). The rows of Table 4.1 represent "receiving agents" (broadly conceived). These are the ultimate effects of polluting activities on human health (e.g. morbidity and mortality), and non-health (e.g. forest damage, material and buildings damage). The corresponding unit values indicate the damage done by a tonne of pollutant vis-à-vis its impact on each receiving agent.

Pollutants such as SO_2, NO_X and PM10 are implicated in significant damage to health (Maddison *et al.* 1995). However, the impacts associated with different pollutants are not simply additive. Recent epidemiological evidence appears to suggest that SO_2 and

[10] For global pollutants such as carbon dioxide these location problems do not exist – damage is the same no matter where the unit of pollutant was emitted. Of course, there is significant uncertainty regarding the impacts arising as a result of the accumulation of greenhouse gases in the atmosphere.

NO_X impacts occur via the formation of sulphate and nitrate aerosols in the atmosphere (see, for a discussion, Pearce and Newcombe, 1998). Given that PM is itself a mixture of sulphate, nitrate and other aerosols adding impacts together risks double counting. In view of this, Table 4.1 only attributes damage to health to PM10 and hence damage caused by SO_2 and NO_X is restricted here to those direct effects associated with non-health categories (e.g. damage to buildings). Climate change, arising from emissions of CO_2 and CH_4, is likely to be characterised by a range of health and non-health impacts. Table 4.1 reports a conservative estimate, from Fankhauser (1994), of the unit damages, caused by the release into atmosphere, of CO_2 and CH_4.

Table 4.1 Marginal Damage Per Tonne of Pollutant Emitted (£)

	SO_2	NO_X	PM_{10}	CO_2	CH_4
Health			19 500 - 44 800		
Non-Health	300 - 670	200 – 280			
Total	300 - 670	200 – 280	19 500 - 44 800	12	80

Source: European Commission (1995); Pearce and Newcombe (1998); Fankhauser (1994)

A candidate indicator is 'green' value-added analogous to that constructed by BSO/ Origin (Huizing and Dekker, 1992). This indicator could signal how much value-added a sector has generated after imputations reflecting environmental damage have been 'charged'. Table 4.2 combines published data on physical emissions in ONS (1998) with the (central estimates of) monetary damage in Table 4.1. Green value-added is then estimated for 10 sectors, in 1987 and 1994, by deducting the value of environmental damage (row 3) from value-added (row 2). Table 4.2 indicates that damage varies considerably by sector. Moreover, there is significant variation in damage in proportion to a sector's value-added. Not surprisingly, those sectors that are relatively damage-intensive include electricity, agriculture, air transport and building materials.

Table 4.2 **Environmental Damage and "Green" Value-Added in the UK Corporate Sector (£ million 1990 prices, unless stated)**

	Agriculture		Chemicals		Electricity		Construction		Textiles	
	1987	1994	1987	1994	1987	1994	1987	1994	1987	1994
Value Added (VA)	5600	8810	10320	12870	5600	6830	29400	31330	7320	6380
Env. Damage	865	462	228	173	2742	1955	688	630	42	44
% VA	*15.4*	*5.2*	*2.2*	*1.3*	*49.0*	*28.6*	*2.3*	*2.0*	*0.6*	*0.7*
Green VA	4735	8348	10092	12697	2858	4875	28712	30700	7278	6336

	Food Processing		Air Transport		Building Materials		Post & Telecomm.		Finance	
	1987	1994	1987	1994	1987	1994	1987	1994	1987	1994
Value Added (VA)	14370	15530	2170	3620	4010	3890	12290	16250	103300	119900
Env. Damage	241	241	212	296	341	210	4	62	118	112
(% VA)	*1.7*	*1.6*	*9.8*	*8.2*	*8.5*	*5.4*	*0.0*	*0.4*	*0.1*	*0.1*
Green VA	14129	15289	1958	3324	3669	3680	12286	16188	103182	119788

Source: Office for National Statistics (1998); see also sources listed below Table 4.1.

There is, however, there is a marked decrease in environmental damage in some sectors between 1987 and 1994 particularly where damage in 1987 was relatively high. The result for the electricity sector is particularly striking. It appears to indicate that damage decreased from almost 50% of the sector's value-added in 1987 to less than 30% in 1994. This, in turn, is attributable to decreases in emissions of all major air pollutants (except CH_4). In the agricultural sector, much of the reduction in environmental damage is caused by reductions in PM10 emissions. Emissions of CO_2 and SO_2 from agricultural sources, in contrast, have increased while CH_4 releases have stayed roughly constant.

An interesting extension is a comparable analysis at the company level. At present this may prove difficult due to the absence, in general, of the prerequisite physical data (in addition to valuation problems). However, this would allow us to comment on the sustainability of a company's activities using an indicator that has proved to be useful at the macro-accounting level. This is the genuine saving rate and is an indicator of the rate at which an economy is on balance creating or liquidating its wealth or assets (Atkinson *et al.* 1997, World Bank, 1997). Persistently negative genuine saving is unsustainable in that welfare will have to decrease at some point in the future if (net) wealth is being run down. The analogous indicator at the company level is the following: any individual firm in generating its economic output incurs operating costs and it is the value of resources left, after these costs are deducted, that corresponds to resources, potentially available, for investment and saving. A crude upper bound on this magnitude is the firm's net profit (before tax). If, in a full cost account, damage is 'charged' against net profit then a suitable indicator of the green economic performance is an adjusted profit rate. This, in turn, can be thought of as a 'corporate genuine saving' (CGS) rate. Thus, other things being equal, the greater the value of environmental damage that is charged the less resources there are, at least notionally, to invest in new assets held by the firm or to distribute in the form of dividends to the firm's owners. It is in this sense that the firm is less 'sustainable' as a result of the pollution damage for which it is responsible.[11]

[11] Re-stating this problem in terms of the contribution of a company (or sector) to sustainability in general is more complicated. Clearly, the firm by contributing to the liquidation of assets elsewhere is, other things being equal, in turn contributing negatively to sustainability elsewhere or overall. However,

It is important to note that the CGS rate is distinct from Bebbington and Gray's (1997b) sustainable cost calculation (SCC). Gray (1992) defines the SCC as that cost that must be borne by a corporate entity in order to restore some previous level of environmental quality. There are, of course, similarities in that SCC is, in principle, also a firm-level indicator using monetary values. The difference lies in the interpretation of the estimate of the cost of pollution. Specifically, SCC is based on abatement cost whereas CGS is based on damage cost. In general, estimates of pollution costs, based on each, will diverge (perhaps significantly) *unless* the prevailing level of emissions corresponded to the socially efficient level. This mirrors our discussion of the relative merits of maintenance vis-à-vis damage costs above.

We illustrate an indicator of CGS, using data available in the public domain, for PowerGen plc one of the UK's largest electricity generating companies. In 1990, the company emitted approximately 12% and 26% of the UK's total emissions of CO_2 and SO_2 respectively (PowerGen, 1996; ONS, 1998). Nevertheless, considerable efforts are being made to reduce these emissions within the company and the electricity sector in general. Specifically, we are interested in how this improvement in environmental performance can be described over time. On a conservative assumption of damage caused, i.e. using the lower-bound estimates of unit damages in Table 4.1, Table 4.3 sets out this hypothetical full cost account for this electricity generator over the period 1992 to 1996. Damage caused by all four pollutants (i.e. excluding CH_4) shown has declined over the period. In particular, SO_2 damage has fallen significantly reflecting the company's move from coal-fired electricity generation to gas-fired technologies. Nevertheless, according to the "bottom-line" CGS rate, the sum of the damage caused exceeds profits in the years 1992 through to 1995. In other words, on our definition of sustainability, the company appears to have been behaving unsustainably during this period.

recasting this problem in terms of CGS, complicates this judgement depending on the degree to which a business' retained earnings influence household saving decisions (and thereby the level of aggregate saving) (Kauffman, 1993). If this offsetting process is entire then, other things equal, sustainability in the aggregate, as measured by a country's genuine saving rate, may be unaffected by the firm's saving decision.

Table 4.3 Accounting for Pollution Damage: The Case of Electricity Generation 1992-1996 (£ millions, current prices)

	1992	1993	1994	1995	1996
1. Turnover	3097	3188	2932	2885	2933
- Operating costs	2771	2736	2455	2354	2240
= Operating profit	326	452	477	531	693
+ Other income	33	-27	8	14	-6
2. = Profit on ordinary activities	359	425	485	545	687
CO_2 damage	256	227	229	219	212
+ SO_2 damage	279	248	218	198	162
+ NO_X damage	43	37	35	32	27
+ PM10 damage	491	338	311	264	215
3. = Pollution damage	1068	850	793	713	616
as % of profit	*328*	*188*	*166*	*134*	*89*
4. Corp. Genuine Saving (2.-3.)	-709	-425	-308	-168	71

Source: Published Annual Reports and Environmental Reports of PowerGen (various years).

A number of points are worth making here. Firstly, the policy signals provided by this indicator need to be interpreted with care. For example, there is an apparent implication that companies can compensate for increased pollution with higher profits. This will depend, on standard cost-benefit rules, whether the economy is over- or under-polluting relative to the social optimum. For example, if it is the former then, in terms of a simple trade-off between output and pollution, £1 of (additional) output can only be generated at a social cost of more than £1 (Tietenberg, 1996). Secondly, we use the term 'behaving unsustainably' advisedly in that it refers to a notional sustainability. That is, it is calculated *as if* the pollution magnitudes in Table 4.3 were actually charged. Thirdly, there is ample scope to improve existing estimates as new data – e.g. based on better knowledge of the pollution-human welfare (wellbeing) link – become available and the results we present are of illustrative value only. Nevertheless, the results in Table 4.3 offer a first approximation of the cost of polluting activities and moreover suggests that corporate sustainability, subject to the above caveats, yields a meaningful indicator of corporate environmental performance. Furthermore, this is a headline indicator of environmental responsibility that can be understood by a potentially wide audience. Finally, Table 4.3 indicates that the company's CGS rate became positive in 1996. What this provides is an overall signal that the company has made significant progress in reducing the environmental impacts of its economic activities. In other words, this framework can be used to signal 'good news' to the extent that it exists.

4.5 Conclusions

We have proposed that the notion of 'corporate sustainability' does have meaning beyond the provision of a unifying standard for diverse questions concerning the corporate sector and the environment. This suggests that to the extent that external costs or damages are directly associated with the generation of a corporate entity's income, the environmental or full-cost account of that entity should reflect the magnitude of the damage caused. This can result in the development of indicators that are comparable to measures of net or genuine saving estimated at the level of the national economy. An intuitive interpretation is that a business is, at least notionally, unsustainable to the extent that adjusted profit or corporate genuine saving is less than zero. Judged over time, this indicator provides the desirable signal that as the firm's external impact diminishes the more 'sustainable' it is indicated to be. This, we argue, provides one reasonable link between an understanding of 'corporate sustainability' in terms of the corporate entity itself and in the contribution of the entity to sustainability in a wider sense.

There remain significant problems in accounting for impacts in money terms. For example, although considerable progress has been made with regard to estimating the external costs of energy use, doubts remain over the scope for 'benefits transfer'. Thus, it may be some time before a database of robust and transferable estimates of damages are available. However, we would argue that further investigation of these issues, in the context of full cost accounting, remains worthwhile and can yield useful information regarding our understanding of corporate sustainability. The appropriate treatment of living resources such as forests and biodiversity is more problematic. Many of the ecological functions giving rise to these values are poorly understood and so in practice crucial but largely unresolved measurement issues are encountered. It should be noted, however, that this would be true of any approach to corporate green accounting that attempted to tackle these same concerns. There will remain a conflict, or at the very least a tension, between full cost accounting frameworks and corporate sensibilities. What is required is careful assessment of the 'value-added' of moving from simple pressure indicators to more sophisticated frameworks based on impact assessment. Lastly, corporate sustainability has been defined narrowly here. It would be interesting to explore how downstream and upstream effects can be dealt with using this framework and how this approach can be extended in order to account for diffuse sources of pollution. What we have illustrated here is a starting point or building block whereby a number of distinct but important issues can be addressed in a consistent fashion. Assuming suitable progress can be made in addressing the questions raised above, we have shown how monetary data reflecting the impact of the corporate sector could be a practical green accounting tool.

Chapter 5

Savings, Growth and the Resource Curse Hypothesis

5.1 Introduction

The recent re-emergence of interest in the sources of economic growth has provided a reminder that a range of policy-related variables can have a persistent influence on economic growth rates. Parallel contributions to the theory and measurement of sustainability have focused on the implications of imprudent use of natural resources and inefficient levels of environmental degradation for sustaining economic development. One important link between these two strands of the economic growth literature is the paradoxical but seemingly robust finding of a negative and significant relationship between natural resource abundance and the growth rate of per capita Gross Domestic Product (GDP). This finding has been characterised as the "resource curse hypothesis" and it is not surprising that it has led to significant efforts to identify the reasons for the curse.

Using cross-country growth regressions, we confirm this finding using a direct measure of natural resource abundance: the share of resource rents in GDP for a range of natural resources including fossil fuels, minerals and timber. We then turn to the interesting question of the relationship between growth and saving, where now we employ a measure of 'genuine' saving, adjusted for resource depletion. We show that there is a clear positive relationship between per capita GDP growth and genuine saving across countries for the period 1980 to 1995. It is stressed that this finding amounts to more than support for the proposition that, other things being equal, a higher (gross) savings or investment rate is associated with a higher rate of economic growth. Indeed, there appears to be a surprisingly low correlation between genuine and gross savings and investment rates. In other words, genuine saving provides new information (rather than serving only as a proxy for more familiar indicators of asset accumulation).

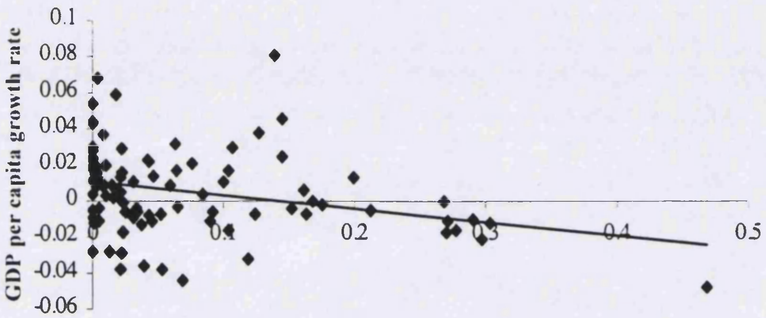
We also show that it is the inability of resource-rich economies to transform this natural good fortune into (genuine) saving that explains the resource curse. Furthermore, our results provide some evidence that the 'vehicle' whereby the curse is delivered is in the form of unsustainable government policy. In other words, one interpretation of these findings is that large resources revenues, on average, have permitted governments to pursue 'wasteful' current expenditure perhaps in order to appease powerful rent-seeking groups. This, in turn, has effectively led to a dissipation of the benefits of resource abundance. These results offer another perspective on the resource curse hypothesis: countries where growth has lagged behind the average are those where the *combination* of natural resource, macroeconomic and public expenditure policies has led to a low rate of (genuine) saving.

5.2 Review of the Literature

The proposition that, other things being equal, resource abundance (i.e. wealth) should increase the level of economic welfare that a country can sustain into the future is well established (see, for a review, Auty and Mikesell, 1998). Put another way, resource-rich countries have a distinct economic advantage over (otherwise identical) resource-poor counterparts. However, a significant literature has drawn attention to the finding that many countries in practice do not appear to have enjoyed this benefit. This pattern of development in the presence of resource abundance has become known as the *resource curse hypothesis*. While it is to be expected that the experiences of different countries will be mixed, this proposition has now been observed in a range of case studies of resource abundant economies (see, for example, Gelb and Associates, 1988; Auty, 1993, 1994).

Davis (1995) disputes this finding, however, by comparing the development performance of a sample of resource-rich and -poor countries. While Davis makes the point that the curse is not necessarily an empirical generality, Figure 5.1 shows that a simple inspection of this relationship over the period 1980 to 1995 points to the opposite conclusion. A straightforward interpretation of these (period average) data is that there does appear to be a (weak) negative correlation between economic growth (the growth rate of GDP per capita in Purchasing Power Parity) and resource abundance along the lines suggested by the resource curse.

Figure 5.1 Growth and Resource Depletion, 1980-1995



Of course, there is a critical caveat that must placed on this simplistic inference. What is of interest is whether or not there is evidence of a resource curse, *other things being equal*. This effectively reconciles the findings of Davis (1995) with the existence of a curse found elsewhere in the literature. Specifically, it is not that resource-rich countries have necessarily been 'outperformed' in an absolute sense. Rather, the empirical question is; is there a negative relationship between economic growth and resource abundance once we have controlled for the influence of other variables on economic growth?

One means of testing this proposition is to use cross-country regressions of the determinants of economic growth. There are now numerous such studies (see, for a comprehensive review, Temple, 1999). The framework used is based on the standard neoclassical model of growth extended to include the influence of economic, policy and demographic variables (Barro, 1997). This approach contrasts with much of the resource curse literature, which is case study based. Clearly, cross-country regressions, based on relatively large samples, allow broad lessons to be drawn out but trade off generality of results for descriptive detail. However, in order to quantify the effect of any given variable on long-term growth this method is arguably an effective – but not exclusive – means of doing so (Temple, 1999). It is perhaps surprising then that few statistical studies of growth determinants have dealt with those questions surrounding natural resource abundance. A notable exception is Sachs and Warner (1995, 1997) which finds statistical evidence for the resource curse hypothesis over the period 1970 to 1990, even controlling for the effect of other economic and policy variables on economic growth (such as openness, quality of public institutions and regional specific influences).

If there is indeed a robust and negative relationship between economic growth and natural resource abundance then there exists an apparent divergence of theory from practice, in that the former makes the (intuitively reasonable) prediction that resources are an economic benefit. Hence, the finding of a resource curse begs the more fundamental question regarding the circumstances under which a curse might develop. McMahon (1997), in an authoritative review of the literature, outlines a plethora of suggested mechanisms which can be characterised as based either on purely economic phenomena or on wider policy failures that cause resource rents to be dissipated (i.e. in effect, wasted).

Economic explanations include "Dutch disease" effects whereby a boom in the resource sector leads, via an overvalued exchange rate, to declining fortunes elsewhere in the traded economy, particularly the agricultural and manufacturing sectors (Corden and Neary, 1982). Furthermore, if the latter are characterised to a greater extent by economies of scale (e.g. based on learning-by-doing) then relative decline could impact negatively on economic activity in the aggregate (Matsuyama, 1992). Sachs and Warner (1995) attribute their resource curse findings largely to Dutch disease effects.

One alternative perspective places the blame for the resource curse on a perceived avoidable failure by policy-makers (e.g. national governments) to convert resource abundance into sustained economic gains. It is worth noting that the basic description of this viewpoint, i.e. that resource revenues have often been dissipated rather than saved for the future, highlights the salient link between the resource curse hypothesis and sustainability, defined as a development path along which welfare is non-declining (Pezzey, 1989). Sustainability is achieved in turn, according to one well-known rule of thumb, by a policy to invest the rents from resource depletion in alternative forms of wealth (which under certain conditions) ensures that, in the aggregate, the change in the real value of assets is non-negative (Hartwick, 1977; Solow, 1986).

Pearce and Atkinson (1993) provided one of the earliest suggestions for a practical indicator of sustainability based on this proposition. This was an adjusted national savings measure that accounts for the depletion of natural resources (and the environment). Hamilton (1994, 1996a) developed welfare measures to account for living and non-living resources and varieties of pollutants in a similar extended national accounting framework – the corresponding savings measure, equivalent to that of Pearce and Atkinson, was termed 'genuine' saving. Estimated rates of genuine saving for a broad range of developing regions are published in World Bank (1997) and Hamilton and Clemens (1999). The results of these studies make it clear that negative genuine savings characterise a number of countries and regions over the period 1970 to 1995. It can be shown that observing negative genuine savings means that not only will welfare decline along the path but, additionally, so will GDP (Hamilton, 1995b). In other words, negative genuine saving is associated with declining GDP.

Figure 5.2 Growth and Genuine Savings, 1980-1995

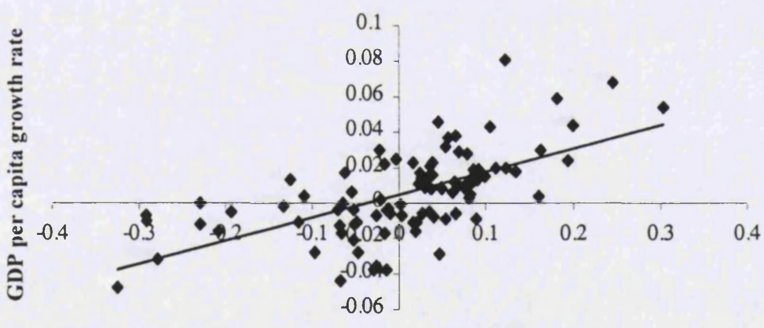


Figure 5.2 illustrates graphically the simple relationship between genuine saving and economic growth across countries and appears to show that countries with low rates of genuine saving have also experienced lower growth of GDP per capita. Although subject to the same caveat as Figure 5.1, it does appear to offer an appealing basis for

further discussion of the resource curse. Put another way, there appears to be a link between economic growth determinants and sustainability *via* the resource curse hypothesis. To the extent that resource rents are consumed rather than invested, this will show up in a lower rate of genuine saving, other things being equal. Not only does this threaten the long-term sustainability of these economies but it also may show up in lower growth.

This does not explain why some resource abundant countries have low genuine savings while others do not. However, this is arguably attributable to the absence of effective institutions to reinvest productively the proceeds of resource depletion. For example, Gelb and Associates (1988) find that resource revenues are often committed by national governments to supporting existing political and economic institutions. Moreover, expenditure by government may be subject to a "ratchet effect" where unrealistic spending commitments, made when resource revenues are relatively buoyant are less easily reversed when revenues fall (McMahon, 1997).

Lane and Tornell (1996) focus in particular on this characteristic of variability of resource windfalls. They propose that what underlies past policy failures is submission by national government to rampant rent-seeking behaviour by powerful interest groups within countries. They provide a theoretical model leading to what is termed a *voracity effect*. This reflects a process where the growth benefits of windfalls are dissipated when national government is weak *and* where there is significant rent-seeking activity amongst competing societal groups. Lane and Tornell do not directly examine resource abundance but rather, terms-of-trade windfalls, which they find, using cross-country regressions of the period 1970-1990, have a negative effect on economic growth. However, they are also able to show that it is the existence of both terms-of-trade windfalls and the voracity effect that accounts for lagging growth. This in turn they interpret as demonstrating that the benefit of any positive windfall is subsequent 'lost' in the form of transfers to rent-seeking groups.

It would appear from this discussion that an appealing explanation of the resource curse hypothesis is that resource abundance allows countries to pursue bad policies longer than otherwise would be desirable or 'sustainable'. Moreover, the policy inference is that it is only sound government policies combined with the prudent allocation of resources revenues to saving that can turn the discovery and exploitation of natural resources into sustained increases in income. This link to the discussion of sustainability is interesting because that literature has to date focused largely upon extending national accounting frameworks (see, for example, Vincent, 2000). However, we show that statistical analysis of cross-country experience also has much to offer our understanding of the sustainability problem. In particular, the contribution of this chapter is to provide further support for the link between the resource curse hypothesis and policy failure based on data from the period 1980-1995 and to cast our findings in the light of the current sustainability debate. Of particular interest is the relationship between, on one hand, the GDP growth rate and genuine saving and, on the other, resource abundance and saving.

5.3 Basic Model

The framework for growth determination is set out in Barro and Sala-I-Martin (1995) and can be written,

$$g_y = f(y, y^*)$$

where, g_y is the growth rate of per capita output, y is the current level of per capita output and y^* is the steady-state level of per capita output.[1] The growth rate is decreasing in y for given y^* and increasing in y^* for given y. The proposition that for two, otherwise identical, countries it is the country with the lowest y that enjoys a higher growth rate is known as *conditional convergence*. Put another way, an economy will enjoy a faster growth rate the further it is from its own steady state value of output (Barro and Sala-I-Martin, 1995).

The level of y^* is dependent on government policies and a potentially wide array of other economic variables such as human capital accumulation (Mankiw, Romer and

[1] While strictly speaking it is output per *worker* that is the variable of interest, in practice data limitations have led to the focus on per capita output (Temple, 1999).

Weil, 1992). A change in any one of these variables will affect not only the steady-state value of per capita output but also the per capita output growth rate. For example, the effect of some beneficial change in government policy, by raising the level of y^*, is a positive influence on the growth rate of per capita output (i.e. the distance between y and y^* is now greater). However, the impact of this benefit is subject to diminishing returns which in the long-run restore the growth rate to that value consistent with the rate of (exogenous) technological progress. However, as Barro (1997) notes, the transition to this long-run rate of growth, in practice, can take a long time (i.e. decades) and so the effects of government policy etc. can persist for long periods.

The existence of this non-trivial lag makes the measurement of the relationship between these variables and the growth rate of per capita output or GDP of considerable policy relevance. What recent cross-country growth studies attempt to do is identify and quantify the individual influences on growth of these variables through econometric analysis. Aron (1997) outlines two broad approaches to this measurement problem. These are, on one hand, sets of structural questions based on extensions of the (human-capital augmented) Solow neoclassical growth model (Mankiw, Romer and Weil, 1992) and, on the other, reduced-form models which are characterised by the omission of variables of initial human capital or investment (in produced capital). A measure of resource abundance, in either approach, is just another relevant characteristic of the economy that affects the value of the steady state (Aron, 1997). The expectation in the context of this model is that (initial) natural resources raise the steady-state level of per capita output. This also implies that resource abundance should raise the growth rate of per capita output. The resource curse hypothesis therefore contradicts this basic result and it is the empirical investigation of this divergence of theory and practice that we focus upon for the remainder of this chapter.

Before we proceed, it should be noted that the analysis of cross-country growth regressions is not without its critics. Temple (1999) outlines several of these main concerns. These include: parameter heterogeneity (i.e. what statistical generalities be made by analysing countries with diverse characteristics?); model uncertainty (where results are weak in that a different model specification changes the significance of findings); endogeneity (is the magnitude or change in a variable a cause or consequence of economic growth?); and, measurement error (of the underlying data for particular variables). Indeed, Temple notes that the consideration of these (and other) difficulties has led to two contrasting responses. The first is to reject or at least regard with scepticism cross-country growth regressions as inherently unreliable and perhaps further to argue that alternative methods such as country case studies offer the most reliable way to address the question of why growth rates differ. Conversely, proponents of cross-country regressions have pursued ever-increasing sophistication in estimating these regressions, most usually in using panel data techniques.

It is important to evaluate both of these responses in the context of the current chapter. On the one hand, the resource curse literature has well illustrated the value of studies of the economic performance of specific resource abundant countries. However, it would seem equally valuable to complement this work with a broader statistical investigation of the resource curse. This would provide a basis to see if various propositions in the case-study literature are capable of statistical generalisation (and just as importantly, vice versa). In this respect, we echo the conclusions of Temple (1999) on the reciprocal importance of these two approaches (in the context of the more general question of why growth rates differ). On the other hand, it has been (often implicitly) argued that the move from relatively simple regression techniques to more sophisticated econometric estimation confers substantial benefits in terms of the reliability of the conclusions. However, the current chapter is an example of the former and therefore it is appropriate to offer some comments in defence of this approach. Principally, it is argued here that given the relatively small attention given to the role of resource abundance in explaining differences in growth rates, then a simple approach offers useful initial insights.[2] Notwithstanding this claim, the clear caveat to our results is that a richer econometric analysis would be desirable in future research, in order to confirm or deny our findings.

[2] As Sachs and Warner (1995) and Lane and Tornell (1996) have shown.

5.4 Empirical Analysis

5.4.1 Data

Our data covers the period 1980 to 1995 and consist of initial period (1980) and period average data.[3] The variables used are described below. In particular, per capita GDP growth and initial GDP per capita are taken from World Bank (1999). Years of education attainment come from Barro and Lee (1993). Data on investment and saving ratios, resource rents and genuine saving are from World Bank (1997). All other data are generated from World Bank (1999) unless otherwise indicated. Countries are excluded from the regressions on the basis of availability of data alone. World Bank (1997) includes data on resource rents and genuine saving for a total of 103 countries, and given the relative novelty of these data, are worth discussing in more detail. Unit rents for non-renewable (or renewable) resources are calculated by subtracting country-specific average costs of extraction (or harvest) from the world price of a given resource (Hamilton and Clemens, 1999).[4] In World Bank (1997) these resources are oil, gas, coal (hard and brown), bauxite, copper, iron, lead, nickel, phosphate, tin, zinc, gold, silver and timber (forest resources). In any given year, these unit rents are multiplied by the quantity of the resource in question that is extracted (or harvested).[5] This gives the total resource rent or value of resource depletion. Genuine savings rates are calculated by subtracting the value of resource depletion from the net saving rate (i.e. gross saving minus the depreciation of produced capital) in a given year.

5.4.2 Results

Table 5.1 reports key summary statistics for the whole sample and according to whether countries are classified as having above and below mean resource rents (respectively, above and 'equal to or below' the period average of 6.9% of GDP). The

[3] The most notable feature of the period 1980 to 1995 is that the prices of resources, in particular oil, were relatively buoyant in the initial year 1980 than were corresponding prices at the end of the period.
[4] Strictly speaking, the resource rents is defined as price minus *marginal* costs. Data based on average costs will overestimate rents if marginal costs are increasing in extraction.
[5] Note that for timber resources it is only that portion of timber harvest that exceeds the natural growth of the forest that is valued.

Table indicates that the GDP growth rate (GDP8095) was higher in countries with below mean rents (i.e. these countries enjoyed, on average, more than 1% higher growth than countries with above mean rents). In addition, countries with few resources also started the period with higher per capita income than their resource-rich counterparts.

Table 5.1 Summary Statistics

	Full Sample	RENT ≤ 0.069	RENT > 0.069
GDP8095	0.006	0.016	.0002
LGDP80	3256	5181	2745
EDU	5.0	6.9	4.3
INV	.216	.230	.211
SAVE	.178	.184	.176
RENT	.069	.001	.095
GS	.007	.092	-.026

With respect to indicators of wealth accumulation, there is little apparent difference in the period average savings ratio (SAVE) according to whether countries are resource-poor or resource-rich while gross investment (INV) are, on average, less than 2% higher in the latter. It is interesting then to note that there does exist a wide disparity (of more than 10%) between period average genuine savings (GS) according to the category in which countries fall. Genuine savings are negative (on average, -2.6% of GDP) in resource-rich countries, and are positive (on average, 9.2% of GDP) in resource-poor countries. These summary data appear to indicate mixed fortunes according to resource abundance, especially with regard to economic growth and genuine saving.

Our basic cross-country regression is reported in regression (1) in Table 5.2. The dependent variable is the period average of the growth rate of GDP (in PPP) per capita (1980 to 1995). Standard errors are reported in parenthesis and are corrected for heteroskedasticity using White's procedure. The independent or explanatory variables are the natural log of GDP (in PPP) per capita in 1980, human capital, measured in terms of years of educational attainment in 1980 (of population over 25) (EDU) and the investment ratio in 1980 (INV80) and period average of the share of resource rents in GDP (RENT). The following dummy variables control for regional factors: Sub-Saharan Africa (SSA); Central America (CAM); Latin America (LAAM); Middle East and North Africa (MENA); and, East Asia (EASIA). Regression (1) provides support for conditional convergence. The results imply a rate of convergence, on average, of about 1% per year (i.e. 0.9%). Educational attainment is positively associated with the growth rate (although is only significant at the 10% level). An increase in mean years of schooling per person, on average, leads to approximately a 0.2% increase in the per capita GDP growth rate. The investment ratio is significant (at the 10% level), indicating that higher investment effort may have a pay-off in the form of a higher growth rate. It should be noted that the magnitudes of the coefficients on initial GDP and education attainment, while significant, are lower than is found elsewhere in the literature. This is most probably caused by the exclusion from the regression of other potentially important variables that explain growth rates (Lane and Tornell, 1996).

Turning now to the question of the resource curse, the coefficient on our variable indicating resource abundance, RENT, is both negative and significant. A 10% increase in the share of resource rents in GDP regression is estimated to decrease the growth rate of per capita GDP by 0.5%. Regression (2) reports results instead using resource abundance in the initial year (RENT80). Again, the coefficient is negative and significant but notably lower than the coefficient on the period average of the same variable. Sachs and Warner's (1995, 1997) proposed indicator of resource abundance, the share of primary product exports in GDP (SXP), can be also be shown to predict the "resource curse" (regression (3)) in that the coefficient on this variable is negative but is only significant at the 10% level. To the extent that resource rents are a more direct measure of depletion, we would expect this variable to "outperform" SXP. Unfortunately, regression (4) shows that while RENT does indeed perform 'better' than SXP, neither is significant when both are included in the same regression. RENT becomes significant (at the 5% level) only if the dummy variable MENA is excluded from the regression.

Table 5.2 Resource Abundance and Growth: Resource Rents vs. Resource Exports (standard errors in parentheses)

	(1)	(2)	(3)	(4)
LGDP80	-.0086	-.0086	-.0073	-.0082
	(.0031)	(.0031)	(.0033)	(.0034)
EDU	.0017	.0019	.0018	.0018
	(.0009)	(.0010)	(.0010)	(.0010)
INV80	.0514	.0476	.0429	.0424
	(.0271)	(.0265)	(.0284)	(.0293)
SXP			-.0254	-.0010
			(.0152)	(.0227)
RENT	-.0502			-.0425
	(.0188)			(.0227)
RENT80		-.0281		
		(0.011)		
SSA	-.0268	-.0262	-.0256	-.0272
	(.0060)	(.0061)	(.0065)	(.0066
CAM	-.0162	-.0156	-.0199	-.0194
	(.0041)	(.0041)	(.0042)	(.0040
LAAM	-.0098	-.0097	-.0103	-.0090
	(.0047)	(.0046)	(0047)	(.0048)
MENA	-.0093	-.0091	-.0113	-.0086
	(.0053)	(.0055)	(.0061)	(.0053)
E. ASIA	.0256	.0266	.0281	.0273
	(.0100)	(.0100)	(.0102)	(.0103)
C	.0676	.0662	.0569	.0659
	(0.195)	(.0200)	(.0218)	(.0226)
R^2	0.59	0.59	0.61	0.62
N	91	91	87	87

Statistical work on the resource curse, to date, has focused on indicators of resource abundance in the aggregate: that is, the value of *total* resource abundance. However, World Bank data on resource rents can be disaggregated into its constituent resources. In principle, this allows us to examine the impact of *specific* resources on growth. For example, it may be the case that only a subset of resources is characterised by the curse, while other resources have had an unequivocal positive impact on growth. If so, we might expect a positive and significant coefficient on at least some individual resources. While we might not expect it to be the case that the revenues of certain resources are more difficult to manage per se, different countries in our sample will be abundant in different resources. To the extent that some of these countries have been able to manage their resources effectively (relative to others), we might expect this variation to be reflected in a 'differential' curse across specific resources.

At present, however, World Bank data on abundance of specific resources arguably has too few observations to generate meaningful results except for oil, gas, and timber (forest resources). Results of regressions for these resources are reported in Table 5.3.[6] For example, regression (2) indicates a strong negative impact of resource abundance in the form of gas reserves. A 10% increase in the share of gas depletion in GDP is associated with a 1.4% decrease in the GDP per capita growth rate. Interestingly, timber (regression 3) is not only associated with a strong resource curse but this finding is also robust against the inclusion of the aggregate variable, RENT (regression 4). Indeed, this effect on growth implies that a 10% increase in the share of timber rents in GDP leads to (approximately) a 3% decrease in the GDP per capita growth rate. For a country such as Kenya where timber resources accounted for a period average of 6.4% of GDP this result predicts that GDP per capita would have been just over one third larger by the end of the study period if not for the effect of this specific resource curse (i.e. $\exp[0.3 \times 0.064]$). This offers further support for the oft-cited conclusion in the forestry literature regarding the failure of governments to capture rents adequately in the forestry sector (van Kooten *et al.* 1999).

[6] World Bank (1998) disaggregates SXP into exports of ore, fuel and food. Regressions including these variables as indicators of resource abundance resulted in insignificant coefficients.

Table 5.3 Disaggregating the Resource Curse (standard errors in parentheses)

	(1)	(2)	(3)	(4)
LGDP80	-.0149	-.0146	-.0109	-.0106
	(.0035)	(.0039)	(.0034)	(.0034)
EDU	.0029	.0030	.0019	.0015
	(.0010)	(.0011)	(.0009)	(.0009)
INV80	.0446	.0968	.0234	.0351
	(.0268)	(.0357)	(.0241)	(.0268)
RENT				-.0334
				(.0191)
OIL	-.0470			
	(.0211)			
GAS		-.1436		
		(.0281)		
FOREST			-.3009	-.3053
			(.1343)	(.1382)
SSA	-.0238	-.0360	-.0301	-.0302
	(.0069)	(.0102)	(.0061)	(.0060)
CAM	-.0191	-.0207	-.0198	-.0198
	(.0044)	(.0038)	(.0040)	(.0036)
LAAM	-.0081	-.0089	-.0146	-.0131
	(.0052)	(.0054)	(.0044)	(.0043)
MENA	-.0084	-.0122	-.0113	-.0100
	(0055)	(.0066)	(.0053)	(.0050)
E. ASIA	.0156	.0245	.0227	.0225
	(.0150)	(.0060)	(.0103)	(.0103)
C	.114	.0100	.0933	.0918
	(.0229)	(.0242)	(.0236)	(.0245)
R^2	0.69	0.82	0.60	0.61
N	52	49	83	83

In summary, our results so far appear to confirm support for the resource curse hypothesis. However, the more interesting issue is how this curse arises, a question that we examine in two stages. Firstly, we examine the link between resource abundance, (genuine) saving and growth. Secondly, we investigate the policy-based reasons for the failure to turn resource abundance into sustained increases in per capita output.

Savings and Growth

Various studies have investigated the contribution of wealth or factor accumulation on growth. The empirical evidence appears, on balance, to be ambiguous in that while some studies have found a significant role for the investment ratio (e.g. Levine and Renelt, 1992) other studies such as Barro (1997) do not find evidence for such a role, once other important variables have been controlled for. Sachs and Warner (1997) include the savings ratio (the share of gross savings in GDP) as an explanatory variable but do not find a significant role in explaining the economic growth experience (in Africa) over the period 1970 to 1990. The interesting question in the context of the current chapter is the extent to which focusing instead on the genuine savings rate provides new information on the accumulation of factors or wealth. Genuine saving measures the extent to which countries are, on balance, liquidating or creating national wealth. Moreover, the genuine savings rate contains information regarding the extent to which the proceeds of resource depletion have been used to finance investment (rather than current consumption), whereas conventional investment and saving ratios measures only gross accumulation.

Table 5.4 Pairwise Relationships Between Savings and Investment Rates

	GS	SAVE	INV
GS	1.0000		
SAVE	0.2489	1.0000	
INV	0.4460	0.6896	1.0000

It could be the case, however, that either gross savings or investment is an adequate proxy for genuine saving. That is, the genuine savings rate is highly correlated with conventional measures of factor or wealth accumulation. Table 5.4 reports correlations between period average savings, investment and genuine saving. While savings and investment are predictably highly correlated, the correlation between genuine savings and savings is slightly less than 25%. There appears to be a stronger correlation between investment and genuine saving but even for the period average data the correlation is less than 45%. Therefore, it would appear on the brief inspection of these data that genuine saving potentially does provide new and additional information (while noting but that genuine saving and investment are fairly highly correlated).

Table 5.5 evaluates whether this additional information is actually valuable in terms of understanding the determinants of economic growth. In other words, have countries with higher rates of genuine saving enjoyed, on average, higher growth rates? Regression (1) in Table 5.5 indicates that there is a positive and significant correlation between genuine saving (in 1980) and the growth rate of GDP per capita. It predicts that a 10% increase in the genuine savings ratio is associated with a 0.3% increase in the growth rate of GDP per capita. This prediction holds even if the investment ratio, in the initial period, is included as an explanatory variable (regression (2)). In this sense, it is plausible to claim that countries with lower genuine savings rates experienced, on average and other things being equal, lower economic growth.

Regressions (3) and (4) in Table 5.5 investigate this further by splitting the sample into two. The first (smaller) sample includes only those countries where the genuine savings rate (in the initial period) is negative (i.e. GS80 < 0). The second (larger) sample includes those countries where genuine savings are greater or equal to zero (i.e. GS80 ≥ 0). In order to evaluate the effect of resource abundance within these sub-samples a dummy variable is created. This is equal to 1 when RENT > 0.029 (its median value) and 0 otherwise. This variable, RENTMED, is negative in regressions (3) and (4) but is only significant in the former (and then only at the 10% level). That is, countries with low (in this case, negative) genuine savings are also those where, on average, a resource curse can be identified. In this sample, countries, which had above median resource abundance, were associated with 2.3% lower growth. That is, one (tentative) interpretation of these results might be that the resource curse arises when resource abundance is accompanied by low genuine saving.

Table 5.5 Genuine Savings and Growth (standard errors in parentheses)

	(1)	(2)	(3)	(4)
			GS80≤0	**GS80>0**
Log(Y80)	-.0089	-.0101	-.0106	-.0095
	(.0031)	(.0031)	(.0063)	(.0037)
EDU	.0019	.0021	.0052	.0013
	(.0010)	(.0010)	(.0033)	(.0011)
INV80		.0321	.0864	.0228
		(.0271)	(.0423)	(.0364)
GS80	.0338	.0305		
	(.0156)	(.0150)		
RENTMED			-.0232	-.0024
			(.0131)	(.0050)
SSA	-.0251	-.0255	-.0301	-.0375
	(.0061)	(.0062)	(.0062)	(.0067)
CAM	-.0159	-.0161	-.0038	-.0143
	(.0041)	(.0041)	(.0062)	(.0047)
LAAM	-.0095	-.0095	-.0291	-.0115
	(.0045)	(.0045)	(.0133)	(.0060)
MENA	-.0094	-.00112	-.0209	-.0143
	(.0052)	(.0057)	(.0128)	(.0077)
E. ASIA	.0273	.0243	.0168	.0250
	(.0098)	(.0094)	(.0078)	(.0119)
C	.0772	.0781	.0869	.0858
	(.0200)	(.0195)	(.0400)	(.0247)
R^2	0.58	0.58	0.53	0.66
N	91	91	32	59

Of equal interest is just why some countries save resource revenues and why others do not. Table 5.6 reports the results of simple regressions of the effect of policy-related and economic variables on the independent variable, the period average genuine savings ratio. Firstly, it appears that the degree of openness of economies has an influence on the level of genuine savings. For example, openness (TRADE) (as proxied by the share of exports plus imports in GDP) is positively (and significantly) associated with genuine saving (regression (1)). Moreover, the share of taxes on international trade in total government expenditure (TAXTRADE) is negatively associated with genuine saving (regression (2)). Although such regressions are overly simplistic, they do accord with the expectation that if a prudent strategy for resource-abundant countries is to invest the proceeds of depletion abroad (perhaps until productive domestic investment opportunities arise) (McMahon, 1997), then outward-oriented policies could encourage part of the required (genuine) savings effort.

Table 5.6 Simple Regressions of Factors Influencing Genuine Saving (standard errors in parentheses)

Dependent variable: GS

		Coefficient	Constant	R^2	N
(1)	TRADE	.0555	-.0283	0.06	91
		(.0238)	(.0183)		
(2)	TAXTRADE	-.1514	.0385	0.04	92
		(.0692)	(.0170)		
(3)	POWER	-.0684	.0446	0.08	66
		(.0243)	(.0131)		
(4)	POP8095	-5.606	.1140	0.27	97
		(.8775)	(.0165)		
(5)	RENT	-.8812	.0671	0.47	97
		(.0882)	(.0099)		
(6)	RENT80	-.51	.0568	0.43	97
		(.0609)	(.0099)		

Similarly, a crude indication of the importance of domestic policy on genuine savings is provided by correlating this variable with the variable POWER constructed by Lane and Tornell (1996). This variable consists of two components. The first, is a measure of weak national government and the second is a measure of the extent of rent seeking. POWER is a dummy variable taking a value of 1 if government, in a country, is weak and rent-seeking behaviour is strong and 0 otherwise. Regression (3) shows that this variable is negatively related to the rate of genuine saving, perhaps indicating that the impact rent-seeking on the liquidation of net wealth within countries. Population growth (POP8095) also enters the regression negatively (regression (4)).

Hence, the results indicate that a 1% increase in the population growth rate is associated with a 5.6% decrease in the genuine savings rate. This appears to lend some tentative support to the posited impact of population growth on sustainability outlined in Hamilton (2000b).

Finally, regressions (5)-(6) in Table 5.6 offer an indication that while resource abundance is associated with an increase in the genuine saving rate this is not proportional. In other words, a 1% increase in the share of resource depletion in GDP is not necessarily associated with a corresponding 1% increase in the genuine savings rate. For example, in regression (1) the period average share of resource depletion (RENT) is regressed on GS. Hamilton (2000a) interprets the coefficient on the variable RENT as the marginal propensity to consume resource rents such that a 1% increase in the share of resource rents in GDP results in (approximately) only a 0.88% increase in consumption. It can be argued, however, that this relationship is rather mechanistic in that by definition the calculation of the genuine savings rate means that an increase in RENT leads to a corresponding fall in GS. In order to 'minimise' this mechanistic influence on GS8095 we also estimate this relationship using initial resource abundance (RENT80) in order to measure this propensity to consume. Table 5.7 indicates that this reduces the coefficient such that a 1% increase in the share of resource rents in GDP results in (approximately) a 0.5% increase in consumption.[7] In words, on average, it appears that only half of resource rents are actually saved rather than consumed. It would be interesting to know what the reasons (proximate or otherwise) for this inability to transform resource rents into saving, especially given the apparent favourable effect of genuine savings on the economic growth rate. It is to this question that we now turn.

Mismanagement of Resource Abundance

It may be that there is no straightforward relation between the resource curse and resource abundance: e.g. the relationship may not simply be proportional to resource

[7] This result is robust against the inclusion of a range of other variables (results not reported here) including initial per capita GDP, human capital and regional dummies. The inclusion of these variables increases the coefficient on RENT80 (which remains significant) to -0.56.

depletion. Resource abundant countries might find it disproportionately difficult to manage those resources. Conversely, countries with relatively few resources may find it easier to convert these resources into a sustained increase in output.

One way of testing this proposition would be to introduce a quadratic term: i.e. RENT squared as in Table 5.7. Regression (1) indicates that the coefficient on $RENT^2$, while negative, is insignificant. If, as in regression (2), the investment variable is omitted then this coefficient again has the expected sign but is only significant at the 10%. Moreover, the variable RENT is no longer significant (and now has a positive sign). On balance, there appears to be at best tentative evidence for the proposition that it is countries with disproportionately greater resource abundance that have faced particular problems managing those resources. More clear evidence would be desirable but even this would still leave important elements of an explanation of why the resource curse occurs.

McMahon (1997) posits a number of mechanisms whereby resource abundance allows governments either to implement undesirable policies or to prolong the duration of existing inefficient (i.e. growth-dampening) policies. Some of these effects are relatively sophisticated involving, for example, the ratcheting upwards of government expenditure whereby public spending programmes implemented when resource revenues are relatively high, which are not reversed if resource prices decline. Lane and Tornell (1996) argue that the larger the resource windfall the greater the incentive of weak government to yield to the demands of powerful rent-seeking interest groups. More generally, while some studies have found a negative role for government size (e.g. Barro, 1997) the review by Temple (1999) suggests that, on balance, the evidence is ambiguous. In the current context, it is the combination of resource abundance and policy that matters in terms of explaining the resource curse. That is, have governments tended to use the revenues of exploiting resource wealth to pursue 'bad' (or, at least, short-term) policies? e.g. policies that raise consumption at the expense of (genuine) saving.

Table 5.7 Testing for a Non-linear Resource Curse (standard errors in parentheses)

	(1)	(2)
LGDP80	-.0074	-.0057
	(.0031)	(.0030)
EDU	.0016	.0013
	(.0010)	(.0010)
INV80	.0462	
	(.0272)	
RENT	-.0009	.0185
	(.0423)	(.0424)
$RENT^2$	-.1617	-.2066
	(.1064)	(.1162)
SSA	-.0264	-.0260
	(.0060)	(.0060)
CAM	-.0161	-.0158
	(.0042)	(.0044)
LAAM	-.0107	-.0115
	(.0045)	(.0045)
MENA	-.0088	-.0071
	(.0051)	(.0047)
EASIA	.0259	.0305
	(.0100)	(.0098)
C	.0772	.0553
	(.0200)	(.0203)
R^2	0.58	0.58
N	91	91

This impact of policy decisions in explaining the resource curse can be gauged by interacting policy variables with a measure of resource abundance. Examining the signs and significance of the resulting interacted variables allows the extent of this policy failure to be evaluated. Interacting our variable RENT with various policy-related parameters would be one means of doing this. However, in order for the results to be meaningful in terms of a straightforward interpretation of the coefficients, it would be better to transform these variables into dummies. Estimating the mean value of the each variable and assigning a value of 1 if the value of that variable for a country lies above the mean and 0 otherwise creates these dummy variables. For example, RENTMEAN = 1 if the RENT > 0.069 and RENTMEAN = 0 if RENT ≤ 0.069, where 0.069 is the mean value of RENT. All government policy variables are also expressed in this way: for example, DGEXP is a dummy variable of the share of government expenditure in GDP (GEXP) which takes a value of 1 if GEXP is greater than its mean value and 0 otherwise.

Regression (1) in Table 5.8 introduces both of these dummy variables, RENTMEAN and DGEXP, as explanatory variables.[8] The latter is intended to be a general indicator of size of national government in that it reflects the proportion of output that government spends to finance public consumption and investment. Neither is significant although both have the predicted negative signs. However, the inclusion of the interaction of DGEXP with RENTMEAN (regression (2)) does have an impact on our interpretation of the resource curse. This interaction variable is both negative and significant. The interpretation of this coefficient is that those countries where RENTMEAN = 1 and DGEXP = 1 have, on average, a 1.9% lower growth rate of GDP per capita. Furthermore, the coefficient on RENTMEAN, which in regression (1) was negative (but insignificant), is now positive and significant (at the 10% level). Those countries that do not use resource rents to finance higher levels of government expenditure enjoyed beneficial effects of resource abundance in terms of, on average, a 1.5% higher GDP per capita growth rate.

[8] It should be noted that this interpretation of the results in Table 5.8 holds if we use RENT as a measure of resource abundance rather than RENTMEAN. However, the conclusions do not seem to be robust against the inclusion of RENTMED (instead of RENTMEAN). This is despite the apparent significance of RENTMED in previous growth regressions in this chapter (see Table 5.5).

Table 5.8 Policy Failure and the Resource Curse

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
LGDP80	-.0091	-.0087	-.0062	-.0063	-.0083	-.0080	-.0043	-.0048
	(.0034)	(.0033)	(.0035)	(.0035)	(.0039)	(.0036)	(.0036)	(.0036)
EDU	.0024	.0024	.0014	.0015	.0015	.0011	.0005	.0007
	(.0011)	(.0010)	(.0011)	(.0011)	(.0012)	(.0011)	(.0013)	(.0013)
INV80	.0359	.0399	.0240	.0221	.0402	.0425	.0090	.0170
	(.0294)	(.0292)	(.0310)	(.0320)	(.0321)	(.0313)	(.0291)	(.0092)
RENTMEAN	-.0011	.0150	-.0013	.0045	-.0031	.0021	-.0033	.0052
	(.0039)	(.0077)	(.0038)	(.0050)	(.0046)	(.0052)	(.0038)	(.0072)
DGEXP	-.0019	-.0185						
	(.0045)	(.0080)						
RENTMEAN		-.0185						
* DGEXP		(.0080)						
DGINV			.0006	.0027				
			(.0050)	(.0058)				
RENTMEAN				-.0094				
* DGINV				(.0069)				
DGCON					-.0005	.0041		
					(.0053)	(.0064)		
RENTMEAN						-.0158		
* DGCON						(.0090)		
DWAGE							-.0014	.0053
							(.0048)	(.0073)
RENTMEAN								-.0170
* DWAGE								(.0092)
SSA	-.0278	-.0261	-.0279	-.0272	-.0295	-.0299	-.0301	-.0323
	(-.0063)	(.0063)	(.0069)	(.0071)	(.0063)	(.0065)	(.0076)	(.0076)
CAM	-.0203	-.0191	-.0163	-.0164	-.0145	-.0139	-.0126	-.0176
	(-.0048)	(.0047)	(.0049)	(.0048)	(.0042)	(0035)	(.0039)	(.0050)
LAAM	-.0110	-.0137	-.0124	-.0134	-.0115	-.0112	-.0139	-.0131
	(.0046)	(.0041)	(.0051)	(.0045)	(.0060)	(.0060)	(.0046)	(.0043)
MENA	-.0138	-.0137	-.0085		-.0130	-.0110	-.0079	-.0080
	(.0072)	(.0070)	(.0064)		(.0078)	(.0069)	(.0064)	(.0063)
E. ASIA	.0269	.0276	.0236	.0248	.0256	.0259	.0249	.0213
	(.0103)	(.0101)	(.0127)	(.0129)	(.0110)	(.0107)	(.0117)	(.0120)
C	.0688	.0625	.0532	-.0066	.0683	.0650	.0490	.0477
	(.0214)	(.0209)	(.0246)	(.0063)	(.0247)	(.0232)	(.0254)	(.0249)
R^2	0.61	0.62	0.51	0.51	0.58	0.60	0.52	0.58
N	84	84	83	83	79	79	76	76

Regression (2) in Table 5.8 appears to offer preliminary support for the proposition that resource abundance has allowed countries, on average, to pursue policies of expenditure that have lowered the economic growth rate. This expenditure has two main components: public consumption and investment. We might speculate that government investment could have a positive or negative relationship with growth. On the one hand, public investment might be interpreted as a productive use of economic resources. On the other hand, investment by government might tend to be unproductive, in that the rate of return on publicly funded projects is lower than the market (or social) rate of the return. Indeed, it may be the case that the existence of resource abundance only serves to stimulate unproductive investment booms. Some of this could be explained by rent seeking as described by Lane and Tornell (1996). McMahon (1997) describes various unproductive investment booms in resource rich countries, especially those occurring in the construction sector, as constituting a possible source of the dissipation of resource revenues. If so, then we might expect that increased public investment (relative to output) might dampen the growth rate of GDP per capita in resource rich countries.

Regression (3) includes a dummy variable of the share of government investment in GDP as an explanatory variable. The coefficient on DGINV is positive but insignificant (regression (3)). The coefficient on the interaction of DGINV and RENTMEAN in regression (4) is negative but is also statistically insignificant. Some evidence is found for a negative impact of government consumption, in the context of resource abundance. Clearly, some expenditure items presently counted as current consumption in national accounts can be thought of contributing to long-term welfare (e.g. current education expenditures). However, we could speculate that governments have also tended to use resource wealth to appease rent-seekers primarily in the form of *current* expenditure items. This means that the coefficient on a variable that interacts government consumption with resource abundance should be negative and significant. Regression (6) finds this to indeed be the case at least at the 10% level. Countries which are resource rich (i.e. RENTMEAN = 1) and where government consumption is relatively high (i.e. DGCON = 1) experienced, on average, 1.6% lower economic growth.

One of the mechanisms identified by McMahon (1997) whereby government consumption is increased in the presence of resource wealth is via inflation of wages and salaries of government employees. Hence, we introduce as a variable, possibly reflecting the success of rent-seeking in the public sector, the share of government wages and salaries in government expenditure. Regression (8) demonstrates that the coefficient on the interaction DWAGE and RENTMEAN is negative and significant although only at the 10% level. The coefficient on RENTMEAN is now positive but remains insignificant. What these results appear to (tentatively) show is that once the effect of resource abundance on government consumption, via increases in the wages and salaries of civil servants, is controlled for those countries which have used resource abundance to finance additional consumption in this way experienced a 1.7% lower growth rate, on average.

The caveat to these specific findings is that few studies using cross-country regressions have identified a robust relationship of any particular policy variable with growth (Temple, 1999). Hence, it could be that variables such as DWAGE are better interpreted as a proxy for policy in general, rather supporting than any specific mechanism. Even were this the case, it does appear reasonable to speculate that, taken as a whole, the results in Table 5.8 appear to provide support for the proposition that the resource curse is policy-induced. That is, it is the dissipation of resource abundance, through government policy – that explains the resource curse.

5.5 Conclusions

Resource abundance should, in principle, confer an economic advantage on resource-owning countries. However, various studies have found that, in practice, these benefits have not been realised. This is the so-called "resource curse hypothesis". In this chapter, we confirm this result by demonstrating that our main indicator of resource abundance, the share of resource rents in GDP, is negatively correlated with the GDP per capita growth rate. These results show that a 10% increase in resource abundance is associated, on average, with a 0.3-0.5% decrease in the economic growth rate. Even if natural resources are not the most significant factor in the growth process, the dollar value of (accumulated) lost per capita output over the period 1980 to 1995 should still be a source of concern. Other specific resources, such as forest resources, are seemingly characterised by an even stronger curse. This makes it imperative that we understand what explains this resource curse.

We have also demonstrated that the genuine savings rate is positively correlated with per capita GDP growth. Indeed, a 10% increase in genuine saving is associated with a 0.3% increase in the economic growth rate: i.e. a magnitude, which is approximately the inverse of (the lower bound of) the resource curse. This offers a perspective on the curse that is appealing given prevailing theories of sustainability. Resource revenues can be invested so as to create a sustained increase in the level of per capita output. Those resource abundant countries that have suffered from a curse appear to those that have low (or negative) genuine savings. We have shown how these predictions from the sustainabilty literature can be analysed statistically, in the context of cross-country growth regressions. In particular, we find some evidence to support the view that the outward manifestation of resource mismanagement might be the use of resource revenues to finance government expenditure. This suggests that avoiding the curse entails a *combination* of natural resource and government policies that have the effect of raising the rate of (genuine) saving.

Appendix 5.1 Countries in Sample

Central America

Barbados
Dominican Rep.
Grenada
Haiti
Jamaica
Trinidad and Tobago
Belize
Costa Rica
El Salvador
Guatemala

Latin America

Mexico
Argentina
Bolivia
Brazil
Chile
Colombia
Ecuador
Paraguay
Peru
Suriname
Uruguay
Venezuela

Middle East and North Africa

Bahrain
Iran
Israel
Jordan
Saudi Arabia
Syrian
Algeria
Egypt
Morocco
Tunisia

East Asia

Hong Kong
Indonesia
Korea, Rep.
Malaysia
Philippines
Singapore
Thailand

Other Asia

China
Myanmar
Papua New Guinea
Bangladesh
India
Nepal
Pakistan
Sri Lanka

OECD

Australia
Austria
Belgium
Canada
Denmark
Finland
France
Germany
Ireland
Italy
Japan
Luxembourg
Netherlands
New Zealand
Norway
Portugal
Spain
Sweden
Switzerland
United Kingdom
United States

Western Europe

Greece
Turkey

Sub-Saharan Africa

Benin
Burkina Faso
Burundi
Cameroon
Central African Republic
Chad
Congo
Cote d'Ivoire
Gabon
Gambia, the
Ghana
Guinea-Bissau
Kenya
Madagascar
Malawi
Mali
Mauritania
Mauritius
Namibia
Niger
Nigeria
Rwanda
Senegal
Sierra Leone
South Africa
Togo
Uganda
Zambia
Zimbabwe

Appendix 5.2 List of Variables

GDP8095	=	rate of growth of GDP per capita (in purchasing power parity) 1980 to 1995.
LGDP80	=	log of GDP per capita (in purchasing power parity) in 1980.
EDU	=	educational attainment (mean years of education) in 1980.
INV80	=	investment ratio in 1980.
INV	=	investment ratio (average 1980 to 1995).
SAVE80	=	savings ratio in 1980.
SAVE	=	savings ratio (average 1980 to 1995).
GS80	=	genuine savings ratio in 1980.
GS	=	genuine savings ratio (average 1980 to 1995).
RENT80	=	share of resource rents in GDP in 1980.
RENT	=	share of resource rents in GDP (average 1980 to 1995).
RENTMEAN	=	dummy variable of RENT (mean, period average).
RENTMED	=	dummy variable of RENT (median, period average).
OIL	=	share of oil rents in GDP (average 1980 to 1995).
GAS	=	share of gas rents in GDP (average 1980 to 1995).
FOREST	=	share of timber rents in GDP (average 1980 to 1995).
SXP	=	share of primary product exports in GDP (average 1980 to 1995).
SSA	=	dummy variable for Sub-Saharan Africa.
CAM	=	dummy variable for Central America.
LAAM	=	dummy variable for Latin America.
EASIA	=	dummy variable for East Asia.
MENA	=	dummy variable for Middle East and North Africa.
GEXP	=	share of government expenditure in GDP (ave. 1980 to 1995).
GINV	=	share of government investment in GDP. (ave.)
GCON	=	share of government consumption in GDP. (ave.)
WAGE	=	share of public wages and salaries in government expenditure. (ave.)
DGEXP	=	dummy variable of government expenditure.

DGINV	=	dummy variable of government investment.
DGCON	=	dummy variable of government consumption.
DWAGE	=	dummy variable of public wages and salaries.
TRADE	=	degree of openness: share of exports plus imports in GDP (average 1980 to 1995).
TAXTRADE	=	share of taxes on international trade in total government expenditure (average 1980 to 1995).
POP8095	=	Population growth rate per annum (average 1980 to 1995).
POWER	=	Dummy variable of relative strength of national government and rent-seeking (Lane and Tornell, 1996).

Chapter 6

Strong Sustainability and Indicators

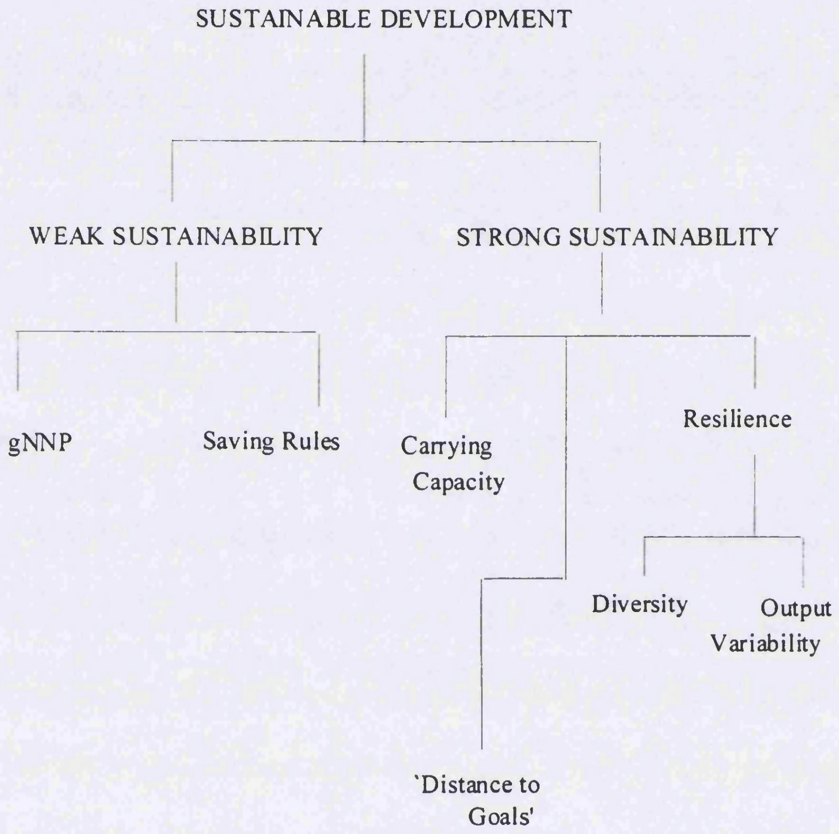
6.1 Introduction

In this chapter we survey the operational meaning attached to the concepts of strong sustainable development. Specifically, we evaluate the contribution of this perspective to the sustainability measurement or indicators debate. While there appears to be no single approach, three elements can be identified; namely, 'distance-to-goal', carrying capacity (including ecological footprints) and 'sustainability as resilience'. As we will discuss, each approach is broadly suggestive of different indicators and we provide illustrations where these data exist. Furthermore, it is arguable at present that there does appear to be an inverse relationship between theoretical appeal and (current) operational significance. Put another way, the approach that appears to offer the most rigorous link to the concept of the strong sustainability (i.e. 'sustainability as resilience') has so far been less suggestive of practical indicators than less appealing alternatives.

It is useful to view these approaches in the context of the wider sustainability debate. Figure 6.1 summarises the prevailing view that the analysis of sustainable development can be split into two (apparently) distinct perspectives on the policy problem facing decision-makers. The key distinction is that 'strong sustainability' (SS) derives more of its foundations from ecological science than does the first, 'weak sustainability' (WS) (but nonetheless, both SS and WS are informed by ecology). This fundamental distinction between SS and WS are: (a) that the former denies to a greater or lesser extent, substitutability between natural assets and other assets – human and produced capital; and (b) SS stresses 'thresholds', 'discontinuity' and 'non-smoothness' in ecological systems and hence in the economic damages to which ecological impairment gives rise.[1]

[1] The conventional environmental economics literature addresses this issue in terms of non-convexities. See Baumol and Oates (1988) and Burrows (1979).

Figure 6.1 A Typology of Sustainability Indicators



In effect, SS takes as its starting point ecological imperatives and this dictates the subsequent form of economic analysis. By contrast, WS begins with standard assumptions in economics and this in turn shapes the form in which ecological and environmental concerns are evaluated. Yet, as we argue below there does exist, in principle, ample scope for reconciling these views of the world most obviously by identifying, for example, *critical* natural assets. Ultimately, much of the choice between SS and WS is arguably to do with issues of fact about the world in which we live. However, since the facts themselves are uncertain, in practice, choosing between SS and WS will also depend, for example, on attitudes to and judgements about behaviour under uncertainty about information.

In terms of indicators of sustainability, different indicators are relevant to the two paradigms, as illustrated in Figure 6.1. The WS paradigm emphasises the substitutability of produced and natural assets and hence focuses on aggregate measures such as: (a) green national income or green Net National Product (gNNP) and, (b) genuine savings. In the main, these indicators are intended to measure environmental performance at the national level, but it is as well to note in passing that the distinction between WS and SS also affects microeconomic indicators; in particular, it affects cost-benefit measures.[2] The elaboration of WS indicators has been the subject of other chapters. In the current chapter, we primarily focus our attention on those indicators suggested by the SS paradigm.

These indicators tend to focus on ecological assets, functions and processes and, in particular, stress 'limits' to the deterioration of natural assets.[3] Indicators in this category include: (a) 'distance to goal' approaches in which, for example, deviations of environmental indicators from sustainability 'targets' are aggregated to derive an overall performance indicator (Hammond *et al.* 1995). The green national accounting counterpart of distance to goals is offered by Hueting, Bosch and de Boer (1992) and is couched in terms of the costs of reaching these goals; (b) measures of carrying capacity: e.g.

[2] See Barbier, Markandya and Pearce (1990); von Amsberg (1992) and chapter 4 in this thesis.
[3] In this respect, the SS paradigm has common elements with the earlier *Limits to Growth* literature; see Meadows *et al.* (1972).

incorporating supply/ demand ratios for resources; (c) measures of resilience, most of which have yet to be developed. One suggestion involves indicators of biological diversity, since resilience is assumed to be a function of diversity (Arrow *et al.* 1995; Common and Perrings, 1992). For agriculture-dependent systems, an indicator of yield variability may also be relevant.

The remainder of this chapter is organised as follows. In section 2, we present an overview of the basic approaches to the analysis of sustainable development as well as examining the existing scope for reconciling weak and strong sustainability both in theory and in practice. We then proceed in section 3 to assess the specific contribution of SS to the indicator debate by evaluating those approaches predicated on 'distance-to-goals', carrying capacity and 'sustainability as resilience' respectively. Finally, we offer some concluding remarks with respect to progress made in measuring strong sustainability.

6.2 Sustainability Models

Sustainable development can be defined as some indicator of human welfare or well-being per capita which does not decline over time (Pearce *et al.* 1989; Pezzey, 1989). At this point there is no significant divergence between WS and SS as both paradigms broadly accept this definition. The approaches are differentiated with regard to the conditions required to satisfy the achievement of sustainable development. Taking the capital or asset base as the means whereby future welfare will be sustained, this implies that some form of conservation or maintenance of capital as a condition for sustainability. This 'constant capital' concept underlies, for example, the contributions of Solow (1986) and Mäler (1991).

The basic debate about sustainability models relates to the *conditions* set for the capital stock or wealth. In WS the operative constraint is the overall stock of capital since all forms of capital are assumed to be substitutable (Solow, 1993). Although natural capital is a form of wealth whose services contribute to welfare it has no 'special role' as such in this scheme. However, this does not permit 'overly' rapid depletion of non-renewable resources or imply that environmental degradation does not matter. For example, in the latter case these actions involve a loss in future welfare unless accompanied by compensating investment, even if pollution emissions are at the optimum (Hamilton and Atkinson, 1996).

In SS, a sustainable development path (in part) is achieved by conserving certain components of the natural capital stock (Pearce *et al.* 1989). Proposals for indicators of SS have been particularly closely associated with ecological economics. This school represents a loosely assembled body of ideas, and in common with other dissenting schools, the most notable unifying theme is a mistrust of neoclassical (environmental) economics; in particular, the perceived inability of mainstream economists to integrate ecological imperatives into their analysis. However, ecological economists are also increasingly offering a positive contribution to the analysis of environmental problems. For example, progress is being made in defining ecological economics as an interdisciplinary framework rather than a wholly new analytical model (Pearce and Barbier, 2000) and in identifying core elements of research such as indicators (Arrow *et al.* 1995).

The SS paradigm requires arguments to the effect that environmental assets are special, i.e. do not have practical substitutes. Within formal models of optimal growth, for exhaustible resources the optimal growth path is not sustainable if resources are essential for production, the pure rate of time preference is positive and the elasticity of substitution between produced assets and natural resources is less than 1 (see Dasgupta and Heal, 1979, ch. 10; Hamilton, 1995a). Hence, the degree of substitutability is therefore critical in determining whether WS is possible with exhaustible resources. For example, if the elasticity of substitution is greater or equal to 1 then 'savings rules', whereby investment in produced assets should offset the value of resource depletion, provide a test of sustainability. More broadly, moving beyond the analysis of exhaustible resources in formal growth models, one essential idea underlying SS is that a given amount of a resource must be preserved intact, in order that it may continue to provide critical services.[4] Pearce *et al.* (1990) present candidate arguments for conservation of natural assets: irreversibility; uncertainty; and, scale of effect.

Firstly, *irreversibility* means that while (within reason) produced capital can be created, destroyed and created again natural assets typically cannot (especially living species). Moreover, while the creation of a produced or new technology may be deemed to be an adequate compensation to future generations for an irreversible transformation of a natural asset, this assumes that we have perfect knowledge of future relative prices (Hamilton, 1995c). If future generations value nature much more highly then such compensations could be either unfeasible or inadequate. (It is also worth noting that some human capital may also be characterised by irreversibility attributes – e.g. indigenous knowledge embodied in social custom.) Secondly, there is extensive *uncertainty* about the workings of ecological systems and hence about the consequences of impairing the functionings of those systems. Lastly, there is the problem of *scale of effect* in that the existence of thresholds and discontinuities may well result in large-scale damage once thresholds are exceeded. It is important to note that while for example, uncertainty has led to proposals for decision-making rules such as the precautionary principle, entailing the conservation of certain resources without full scientific knowledge of the consequences (Perrings, 1991) it is the combination of these three factors which suggest a 'precautionary approach' to the degradation of natural capital (Dasgupta, 1982) and hence motivate a SS rule for natural assets characterised by these attributes.

Some attempts have been made to reconcile WS and SS. Most recently, Farmer and Randall (1998) have re-asserted the concept of a safe minimum standard (SMS) whereby policy-makers follow standard cost-benefit (or WS) rules unless there is a compelling reason not to; e.g. to a conserve critical natural asset. However, this conservation rule can itself be overridden if its costs are "intolerable". Hence, this suggests that policy-makers might adopt a two-tier approach to achieving and measuring sustainable development. Clearly, the key question is how demarcations of the tiers are to be identified in practice

[4] It is important to note that there is no single approach guiding the analysis and measurement of SS. See, for example, Daly (1992) for a discussion of wide range of candidate SS conditions, each of which are suggestive of different indicators.

(Norton and Toman, 1997). Many policy choices faced in the real world are considerably more complex than suggested by the distinction between WS and SS. Hence, there could plausibly exist no straightforward choice between running down the stock of a resource or conserving that stock at some level.[5] In other words, there is likely to be considerable uncertainty regarding which policy regime should apply: e.g., which natural assets are critical and which are not. Moreover, there exists considerable scope for disagreement, among both experts and the public, as to when this policy switch should be made, introducing a further arbitrary element to two-tier approaches to the sustainability problem. However, Woodward and Bishop (1997) demonstrate formally that it is precisely this disagreement that motivates decision rules such as SMS, in the face of this uncertainty, as rational criteria for decision-making.

In principle, the indicators implied by this two-tier approach could be conceived in the following way (Pearce *et al.* 1996). A critical resource does not necessarily need to be kept intact, but if the stock size falls below a certain level then catastrophic consequences will potentially result (in that significant welfare losses would result if this threshold were not observed). Put another way, while some amount of the resource needs to be conserved; there is a residual that can be exploited. One candidate critical resource is tropical rainforest.[6] If preserving some quantity of the rainforest is considered to be critical for the long-term welfare of humanity, the effect of this preservation is to reduce (but not eliminate) the quantity of forest that can be harvested. The key indicators for this living-resource economy operating under this (two-tier) regime will be twofold: are stocks of critical natural assets declining? and are genuine savings rates (adjusted for the *change* in the real value of the non-conserved forest stock) persistently negative? A positive answer to either of these questions would be an indication of unsustainability.

In terms of indicators then, SS – in general – will tend towards environmentally focused measures, supplemented by a weak sustainability measure. This reflects the concern with conserving (certain) natural assets within the broader goal of managing a portfolio of

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[5] For example, one such ambiguity concerns coal as an abundant resource and as a global pollutant (Pearce *et al.* 1994)

[6] Chapter 2 outlines a comparable way of looking at the SS approach in the context of deforestation.

assets sensibly over time. By contrast, (strict) WS will focus primarily on assets generally, with no special highlighted measure for natural capital or assets. However, expressed this way, the *information* required for SS and WS indicators is in some respects similar. For example, savings rules provide essential indicators of sustainability whether SS or WS is the development paradigm. Furthermore, both approaches have to have measures of natural capital to be operative (as well as measures of other capital assets). Nevertheless, each approach also raises distinct measurement issues. Most notably, in the SS approach, non-substitutability is alleged to give rise to aggregation problems (see, Faucheux *et al.* 1994).[7] That is, natural capital has to be measured separately to human and produced and arguably in non-comparable terms (or units). But the 'constant capital' rule can be preserved in this context since SS requires that each proposed index be non-declining.

6.3 Indicators of Strong Sustainability

In Figure 6.1, we identified three proposals for indicators of SS.[8] One unifying feature of these indicators (especially carrying capacity and distance-to-goal approaches) is their definition in terms of exceeding limits or thresholds – to be set by ecological criteria. In moving to actual indicators, we require a more detailed specification of these ecological limits. Typically, these consist of 'sustainability constraints' with respect to environmental resources, e.g. pollution should not exceed the assimilative capacity of the environment; harvest of a renewable resource should not be greater than natural growth (e.g. Daly, 1992).[9] Underlying these constraints is the goal to maintain the stock of *each* resource intact. If the definition of sustainability is non-declining human welfare, then what is argued is that welfare is decreased unless *all* specified ecological constraints are observed. Put another way, it is proposed that all those natural assets being measured have no substitutes. However, while lack of substitutability may be an appropriate

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[7] However, proposals for relatively highly aggregated approaches have also been made (see, for example, Hannon, 1999; Vitousek *et al.* 1986)

[8] It is worth noting that this is not intended as an exhaustive list in that other distinct contributions exist in the SS literature (see, for example, Faber *et al.* 1996).

[9] It is more difficult to think of an appropriate constraint governing the use of non-renewable resources on these terms where growth is by definition zero.

characterisation of some classes of environmental assets, it is asking too much to argue that this is true of all such assets. (Neither need it be the case that limiting factors will remain the same over time, Cohen, 1995.) This illustrates the importance of a theory of strong sustainability with which to underpin our concern about a particular natural asset (or portfolio of natural assets).

Similarly, because SS stresses the importance of some environmental assets – and hence the need to know whether they are increasing or decreasing in size and quality – it is typically argued that physical data can provide the basis for the construction of indicators. While physical indicators are essential inputs into any measure of sustainability, whether in themselves they contain adequate information for appraising whether an economy is or is not on a sustainable path of development is debatable. The essential reason for this is that the indicators usually lack 'weights', i.e. measures of importance. Again, the appeal of (physical) indicators is only superficial unless underpinned by a satisfactory theory of strong sustainable development (i.e. providing the rationale for interest in the change in physical indicators). In other words, indicators of SS arguably should not be judged solely on the basis of practical merits alone (although this is clearly important) but also whether the rationale for indicators is characterised by a substantive theory of strong sustainable development.

6.3.1 Distance to Goals

Distance to goal or performance indicators measure, in effect, the percentage deviation of current environmental conditions from some threshold or goal set by environmental policy. Hence, although not necessarily indicators of SS per se, these data do provide a link to these concerns. The pre-eminent examples of this work are Adriaanse (1993) and Hammond *et al.* (1995). These studies concentrate on the development of environmental policy performance indicators, specifically designed to measure progress towards meeting targets set by environmental policy. Indicators are developed around a number of themes and provide the basis for weighting and combining disparate physical measures into an integrated indicator. Policy-makers then set "sustainability" targets for each theme, typically based on an assumption regarding the assimilative capacity of the environment. The themes that have been developed in the Netherlands are: climate change; acidification; eutrophication; dispersion (of pesticides, toxins such as cadmium and mercury, and radioactive substances); disposal (of solid waste); and, disturbance (from odour and noise) (for a discussion, see Atkinson *et al.* 1997).

A suggested link to SS is provided by the work of Hueting (1980), Hueting *et al.* (1992) and more recently Ekins and Simon (1998). One basis of this approach is a rejection of valuation techniques derived from environmental economics. For example, Hueting (1980) argues that no matter how sophisticated these techniques might become, theoretically correct shadow prices for ecological functions (where no price is observable in a market) cannot be found. In its place, Hueting proposes an indicator of the costs of reaching (specified) sustainability targets. The definition of sustainability used in this context is the sustainable use of each environmental function usually with reference to scientifically set standards. Calculating the resulting 'avoidance cost curve' involves two elements. Firstly, the cost of technical measures to reach the sustainability standard must be established and secondly the cost of shifting economic activities to less environmentally burdensome counterparts is calculated – so-called 'structural measures'. These differ according to the function analysed, be it for non-renewable, living resources or environmental stocks (such as clean air). Ekins and Simon (1998) have recently restated this approach in proposing an analogous indicator: the sustainability gap. The focus is on the distance between the current level of impact on a natural asset and a sustainability standard, whether in physical or monetary terms.

To the extent that sustainability thresholds, in the distance to goal approach, represent reasonable policy targets, indicators that measure progress towards (or deviations from) these targets will be useful to policy-makers. However, two key questions must be asked: firstly, are specified targets justifiable (in that they have in terms of SS) and secondly are indicators of the distance to each target expressed in meaningful terms?

With regard to the former, few contributions (to the distance to goal approach) appear to offer a theoretical justification for indicators proposed. In other words, it is arguable that our conceptual understanding of SS and 'criticality' is not furthered by these efforts. Hueting has consistently argued that the justification lies in the revealed 'preferences' of policy-makers and the public for sustainable development. But whether this general concern for future generations can be used to justify a proposed focus on wide range of specific natural assets and indicators is debatable.[10] The emphasis of contributions such as Ekins and Simon (1998) on the costs of reaching sustainability raises several issues. This approach neglects the benefits of achieving sustainability or, implicitly assumes that benefits exceed costs for all specified natural assets. Although, this criticism could be directed at many contributions to the sustainability debate (weak and strong), this is particularly important in the current context as the costs of reaching SS may be extremely high indeed (Pearce and Barbier, 2000). There is also the issue of aggregation of both physical and monetary data on distance. Adriaanse (1993) aggregates the former by measuring indicators in terms of percentage deviation from some threshold or goal and then by adding up the percentage deviations across environmental themes. Constructing an aggregate indicator involves just one weighting scheme, based on contributions to particular problems, out of many possible schemes (Atkinson *et al.* 1997). The estimated cost of reaching these targets is one alternative. However, the calculation of least cost options (e.g. to mitigate environment impacts) that satisfactorily take into account important determinants of cost such as secondary benefits is a far from trivial task.

6.3.2 Carrying Capacity

The notion of a carrying capacity is drawn from biology. It states that a given area can only support a given population of a particular species and at this upper limit – the carrying capacity – population will have reached its maximum sustainable level. In order to apply this notion of a saturation point to human populations a number of complicating factors arise, not least that we have to consider not only the level of population but also the level and, more importantly, the composition of economic activity. Inevitably, in developing an indicator of carrying capacity, a number of simplifying assumptions are

[10] In addition, it can be argued that Hueting's critique of environmental valuation has not kept pace with

usually made at the risk of a resulting loss of realism such as, for example, how the socio-economic world is characterised. Some ambitious attempts have been made to estimate a global population carrying capacity, with results indicating that sustainable populations are well below existing population levels or wildly in excess of predicted stationary levels (see respectively, Ehrlich, 1992 and Kahn and Simon, 1984).

More usually, carrying capacity indicators tend to be pessimistic regarding judgements about technology; that is, the ability of people to expand the carrying capacity of the earth (Cohen, 1995). For example, Ehrlich (1992) speculates that, at industrialised levels of economic activity per capita, the level of global population that would not exceed carrying capacity thresholds is two billion, about one-third of the *current* population. In the carrying capacity schema each asset is to be maintained intact and presumably the scale of economic activity or population is set by whichever constraint is reached first, as suggested by Rees and Wackenagel (1994). This is known as Liebig's Law and implies that,

$$CC = \min CC_i \tag{6.1}$$

that is, overall carrying capacity (CC) is determined by the lowest carrying capacity of each of the i assets in the system.

A variant of this empirical approach to carrying capacity is to link population levels to relatively simple indicators of resource use. Some commonly cited examples are land use for food production, fuelwood availability and water use as we illustrate below. Various problems are associated with these indicators, such as aggregating the different requirements of a heterogeneous (global) population, the absence of any specified interactions between various constraints, and possibilities for trade where carrying capacity is defined locally (e.g. nationally) (Cohen, 1995). Neither do these indicators necessarily address the measurement of progress in protecting critical natural assets (e.g. biodiversity). One positive comment that may be made in their defence is that they can be

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recent methodological and practical developments in that field (see, for example, Pearce *et al.* 1999).

interpreted as a first stab at developing poverty or scarcity measures that introduce the notion of sustainable use of resources.

Agricultural Output: The Carrying Capacity of Land

The ability of an individual to obtain adequate nutrition can be regarded as a basic need to ensure survival and adequate functioning. Similarly the ability of a nation to provide adequate nutrition for its population might be interpreted in this way. At any point in time population carrying capacity based on food output can be written,

$$CC = \frac{Q_{\text{MAX}}}{C_{\text{MIN}}} = \frac{qL}{C_{\text{MIN}}} \tag{6.2}$$

where Q_{MAX} is the maximum output obtainable. This is made up of available land that is capable of being cultivated (L) and output per unit of land (q). C_{MIN} is some per capita food consumption from which individuals derive their calorific requirements defined in terms of required Q/N (where N is population). In a variant of this simple expression, Rees and Wackenagel (1994) estimate that the world can support roughly a doubling a population from 1992 levels. Such calculations are inevitably crude as they represent highly simplistic descriptions of production potential on cultivatable land. The extent to which policy can be inferred from these indicators is therefore likely to be dubious. For example, even judged on its own merits, any final estimate will be extremely sensitive to the technical assumptions underlying the parameter, Q_{MAX}.

One large empirical study (Higgins *et al.* 1982) on agricultural carrying capacity calculated three different measures each based on different assumptions regarding technology: (i) low inputs (traditional crops, few inputs, no conservation measures); (ii) intermediate inputs (basic inputs, improved crop varieties, simple conservation measures); and (iii) high input (full use of inputs, improved crop varieties, conservation measures and the best mix of crops). However, if even the production possibilities on 'available' land can be characterised satisfactorily, other – equally crucial – analytical considerations need to be brought to bear. For example, it may even be the case that Q_{MAX} is an unreasonably optimistic view of potential agricultural output due to prevailing economic incentives (Srinivasan, 1989). In many countries the agricultural sector's terms-of-trade are distorted due to domestic policies biased towards urban sectors and the protection of agricultural producers in the developed world. Such factors are likely to influence farmers' incentives to produce. Furthermore, in this simple calculation, negative environmental feedbacks are typically ignored (e.g. the expansion of agriculture into ecologically fragile areas, soil erosion, loss of crop species diversity).

The assumption that it is *absolute* shortages of food that are (primarily) responsible for famine, hunger and malnutrition can also be subjected to scrutiny. The apparent co-existence of abundant global food supplies with this on-going suffering has been explained by relative shortages i.e. inequitable distribution of supplies (Sen, 1982; Dreze and Sen, 1987). This inequality in distribution is attributed to 'entitlement failure' – the failure of a right or entitlement to an adequate supply of food, in turn caused by the lack of purchasing power brought about by poverty. The entitlement critique suggests that even a very favourable population carrying capacity can mask relative shortages where the purchasing power of higher income groups outbids that of the lower income groups. In areas where even substantial changes in agricultural yields have been outstripped by population growth these tendencies will only aggravate absolute food shortages.

The Sustainable Use of Water Resources

While globally abundant, only a tiny fraction of the world's water resources are a usable source of freshwater (World Resources Institute, 1992). Yet even this would be sufficient to meet global demand *provided that it was evenly distributed.* An imbalance of supply and demand can lead to problems across regions, countries and indeed within countries. Constraints on water availability are therefore local rather than global.

The general form of a carrying capacity for water resources is basically the same as for agricultural production above,

$$CC_w = \frac{W_{MAX}}{W_0 / POP_0} \qquad (6.3)$$

where W_{MAX} is the maximum sustainable supply, W_0 is total present withdrawal from this flow and POP_0 is the present level of population. (W_0/POP_0) therefore indicates water use per capita: it is present levels of per capita use that we wish to sustain.

Maximum sustainable supply is defined as precipitation minus (surface and groundwater) evaporation and transpiration (Dubourg, 1992). Precipitation is channelled into groundwater systems and rivers via soil and vegetation. This is known as 'run-off'. Calculating the reliable portion of run-off is complex, where as much as two thirds can be dissipated (Clarke, 1991). Reliable run-off corresponds to the maximum sustainable supply of water in any period (defined here as a year). One operational definition of W_{MAX} is annual internal renewable water resources (net of river flows between countries) as described in World Resources Institute (1994). This appears to be close to the concept of maximum sustainable supply although measurement is based on average conditions and will disguise seasonal, interannual and long-term variations (Agnew and Anderson, 1992). It is important to note that this approach only takes into account water *quantity* rather than *quality* issues (or, in turn, linkages between the two).

Falkenmark (1984) has also developed several indicators of water resource scarcity. One such indicator looks at W_{MAX} per capita and within broad bands describes whether particular countries might experience water scarcity problems. This is described in more detail in Table 6.1. This illustrates how many countries in each given region fall within a particular category describing water scarcity (as proxied by country-specific W_{MAX} per capita) For example, within Sub-Saharan Africa, 6 countries are described as facing chronic scarcity on the basis of Falkenmark's criterion that W_{MAX} per capita lies between $1000m^3$ and $500m^3$. In the arid Middle East and North Africa region, 9 countries face severe water constraints, characterised by Falkenmark as a water barrier whereby scarcity of this resource potentially threatens economic development.

Table 6.1 Falkenmark's Water Interval by Region

Indicator	Comment	Sub-Saharan Africa	North Africa & Middle East	Asia	North & Central America	South America	Europe	Former USSR States
$W_{MAX}>10\ 000m^3$ per capita	Little or no problem concerning scarcity	12	-	8	7	11	5	2
$10\ 000m^3>W_{MAX}>1000m^3$	Possible general problems	20	3	11	7	1	16	10
$1670m^3>W_{MAX}>1000m^3$	Possible specific problems: 'water stressed'	4	3	1	1	-	-	-
$1000m^3>W_{MAX}>500m^3$	Chronic scarcity	6	2	-	-	-	-	1
$500m^3>W_{MAX}$	Water barrier: economic development threatened	1	9	1	-	-	-	2

Source: Falkenmark (1984) and World Resources Institute (1995)

It is important to note that W_{MAX} is likely to be unevenly distributed within countries. The potential for redistribution within nations will depend on spatial substitutability. Diverting water flows is not achievable without fairly major infrastructure investment. Hence these requirements are often beyond the budgets of many developing nations and, furthermore, the incremental costs of developing new sources have begun to rise sharply in many countries (Munasinghe, 1992). In view of this, the potential for better management of existing sources is stressed by those who see water scarcity as a problem of poor resource management, i.e. current relative shortage as opposed to absolute scarcity (White, 1984). While there is no one particular level of water use associated with a particular level of overall development, improved water resource management is particularly important if the goal is to raise the level of per capita use. Falkenmark (1984) estimates that in order for Africa to sustain levels of water use of $500m^3$ per capita, an average water use efficiency of 35 percent would need to be achieved. (Average water use per capita in Africa was $153m^3$ in 1990.)

Where water is particularly scarce and flows between countries via large river systems, there is a potential for conflict arising from competition for these flows (e.g. upstream damming) (Alam, 1990). Of course, many countries possess extensive supplies of groundwater or fossil water lying in shallow or deep aquifers thus representing a potentially important source of freshwater especially in arid regions. However, the rate of recharge is usually extremely low and groundwater stocks can be effectively non-renewable. To the extent that this stock can be treated as a 'mine' of water (i.e. little interaction of groundwater with the water cycle), its depletion is analogous to that of a non-renewable resource, depending on the degree of substitution that can be assumed.

Sustainable Fuelwood Production

The burning of biomass fuels such as fuelwood is an essential source of energy in many developing countries. Fuelwood provides 62 percent of energy requirements in Africa and 34 percent in Asia, and in the Sub-Saharan region this figure is sometimes around 90 percent (Alexandratos, 1989; Armitage and Schramm, 1989). Combined with high population densities this dependence threatens the continued existence of fuelwood resources and presents some problems in terms of general levels of forest and woodland loss, in some areas and regions.

Again, we begin with the formula similar to that given in expressions (6.2) and (6.3),

$$CC_{POP,f} = \frac{F_{MAX}}{F_0 / POP_0} \tag{6.4}$$

where F_{MAX} is the maximum sustainable supply of fuelwood. F_0 is current fuelwood demands, which expressed as a ratio to current population gives current per capita fuelwood use. Again, sustainability in any period is indicated when $F_{MAX} \geq F_0$. If $F_{MAX} < F_0$ then population is unsustainable, in the sense that the fuelwood carrying capacity is exceeded. F_{MAX} in effect refers to the natural regeneration rate of biomass, which in turn represents the sustainable yield at any one time. Natural regrowth will differ for reasons such as species type and climate. For example, regeneration rates will be slow where climatic conditions are unfavourable as in high altitude, mountainous regions as well as in arid and semi-arid areas (World Resources Institute, 1986).

Table 6.2 presents estimates based on FAO data on fuelwood demand (production) and fuelwood supply (regeneration). Obviously, if the ratio of demand to supply is greater than unity, then current rates of harvest must be unsustainable. The carrying capacity estimate simply links this information to data on population levels in 1990 and projected levels in 2000. The estimates also take into account the ecological concern that persistent rates of overharvest will *reduce* the sustainable yield and therefore population, over time. Hence, Nigeria's estimated sustainable population is 138 million in 1990. Yet by the end of that decade this was predicted to fall to about 132 million. However, our estimate of carrying capacity predicts that Nigeria's population may well be unsustainable by 2000, at least in terms of a fuelwood constraint. It is worth noting that while Nigeria has substantial oil resources, pressure on fuelwood resources persists owing to the lack of suitable refineries (Armitage and Schramm, 1989).

Table 6.2 Fuelwood Availability and Sustainable Populations in Africa (millions)

Country	Sustainable and (Actual) Population, 1990		Sustainable and (Predicted) Population, 2000	
Algeria	4	(25)	4	(31)
Burkina Faso	4	(9)	3	(12)
Chad	4	(6)	4	(7)
Ethiopia	37	(51)	36	(67)
Gambia	>1	(1)	>1	(1)
Kenya	14	(24)	14	(31)
Madagascar	16	(12)	15	(16)
Malawi	2	(9)	2	(11)
Mali	8	(9)	7	(12)
Morocco	9	(25)	9	(30)
Nigeria	138	(96)	132	(128)
Senegal	6	(7)	6	(10)
Sudan	21	(25)	21	(33)
Tanzania	16	(25)	15	(33)
Tunisia	>1	(8)	>1	(10)
Uganda	2	(16)	2	(22)
Zimbabwe	10	(10)	9	(12)

Source: adapted from Atkinson (1993) and World Bank (1994)

All of the countries described in Table 6.2 are either unsustainable in 1990 or are predicted to be unsustainable by the year 2000. Some of these countries are not surprising, being countries located in arid North Africa with few forest resources. Pressure on forest resources in other countries such as Malawi has been well-documented elsewhere (see, for example, Pearce and Warford, 1993). The size of this threat in terms of energy use will depend upon the availability of substitute fuels, technological measures that increase the productivity of the scarce resource, e.g. more efficient stoves, or sufficient foreign exchange with which to purchase imported refined fuels. However, it is arguable that, for simple (local) economies, an indicator of fuelwood scarcity may be a reasonable indicator of sustainability or rural poverty.

Carrying Capacity: A Summary

Although indicators of carrying capacity less than ideal as sustainability indicators, they do have the virtue of being relatively easy to compute using existing data. Furthermore, it is worth noting that these data may have more specific uses and a number of points are worth summarising. First, computing carrying capacity for African nations suggests that some populations are well in excess of it. As a technical statement of a country's current or immediate future prospects this information may be of some interest. We have not investigated the issue here but it would be interesting to know the extent to which those nations with 'excess' populations are also those nations with low economic growth. In fact, Barro and Sala-I-Martin (1995) do find some tentative evidence that growth in per capita Gross Domestic Product (GDP) was negatively related to population growth and fertility variables for a cross-country regression of mainly developing countries over the period 1965-1985. One fact may then explain the other. However, Barro and Sala-I-Martin also find evidence of a positive link between GDP and fertility for the lowest income countries (and a negative relationship past some income level), suggesting the existence of a Malthusian population 'trap', in this context, which could have important implications for carrying capacities. Focusing on carrying capacity may enable us to single out those resources that are truly constraining. The evidence that exists appears to suggest that it may, for example, be water or fuelwood rather than food or soil.

Ecological Footprints

The concept of ecological footprints (EF) as developed by Rees and Wackernagel (1994) is an extension of efforts to construct indicators of the extent to which carrying capacity is appropriated by human activity. What EF purport to measure are biophysical limits where the common unit used to indicate the human impact on these limits is land area (see also chapter 3). EF can be calculated as follows. Firstly, various components of consumption in a given region are identified and measured such as food and energy. Secondly, for each consumption item, an estimate is made of the land area needed in order to generate the resources involved in that consumption. Thirdly, these land areas are simply added to determine the complete EF of the region. Lastly, the footprint is compared with the *actual* size of the region generating the footprint.

Analysing 51 countries, Wackernagel *et al.* (2000) find that most developed countries have significant ecological deficits of around 3-4 hectares per capita (ha/cap). The only developed economies with ecological surpluses are the Scandinavian countries, Iceland, Canada, Australia and New Zealand. Selected countries with EF deficits are illustrated in Table 6.3. The highest deficit is Singapore with 6.8 ha/cap, not dissimilar to Hong Kong's at 5.1 ha/cap. The United States, Germany, UK and the Netherlands also appear to have significant per capita deficits. Some developing countries are also in deficit including China, India as illustrated in Table 6.3. Globally, Wackernagel *et al.* (2000) calculate that the EF exceeds the world's ecological capacity: average demand is 2.8 ha/cap compared to the notional 2.0 ha/cap available.

Table 6.3 Ecological Footprint Deficit for Selected Countries

Country	EF deficit/cap = ha/cap	Population = million	Total EF deficit = million ha
United States	-3.6	270.0	-972
China	-0.4	1238.6	-495
India	-0.3	979.7	-294
UK	-3.5	59.1	-207
Germany	-3.4	82.1	-279
Denmark	-0.7	5.3	-4
Netherlands	-3.6	15.7	-57
Hong Kong	-5.1	6.8	-35
Singapore	-6.8	3.2	-22

Sources: Wackernagel *et al.* (2000); World Bank 1999.

As a rhetorical device for summarising in an intuitive way the impact of certain activities and regions on the environment, it is arguable that the notion of EF is useful. It is less clear that there are wider novel or useful policy implications to be gleaned. Chapter 3 cautioned against one apparent implication for restricting trade flows. Wackernagel *et al.* (2000) have added additional policy suggestions such as raising the productivity per unit land area, lowering resource use per unit of consumption or reducing consumption itself. It is the latter that is particularly contentious. Firstly, even if this goal was thought to be desirable no single country has an incentive to reduce consumption unless others do the same. Furthermore, it is doubtful whether sufficient co-operation between countries could be sustained to achieve this goal jointly. Secondly, converting 'ecological deficits' into absolute terms by multiplying through by population shows that poor countries with large populations have larger ecological footprints than rich countries with smaller populations as shown in Table 6.3. It would be difficult to argue that any policy designed to reduce consumption in China and India is desirable although this appears to be a logical consequence of the EF approach.

Van Kooten and Bulte (2000) have also discussed a detailed list of criticisms of the EF approach. These include the apparent paradox suggested by, on the one hand, ruling out the possibility of substitution between forms of capital – as advocates of EF explicitly do – and then, on the other hand, implicitly assuming substitution by aggregating different types of land. In this respect, it is the lack of a theory of SS or theoretical framework supporting EF that most probably leads to such crucial issues being dealt with in an inconsistent way. Furthermore, the whole approach appears to rule out the possibility of sustainability in certain regions. For example, while countries such as Canada and Australia appear to do well in terms of their apparent EF, economies that are small and highly urbanised – Singapore and Hong Kong – feasibly cannot meet their ecological demands from their own land either now or in the future. Leaving aside the role of EF as a rhetorical device vis-à-vis public awareness, the usefulness of this indicator appears to have been over-stated by some.

6.3.3 Ecological Resilience

Resilience has been interpreted in a number of ways (Pearce and Barbier, 2000). The most influential vis-à-vis the sustainability debate is that interpretation where resilience determines the persistence of relationships within a system and is a measure of the ability of these systems to absorb changes and still persist (or change) (Holling, 1973). In turn the *degree of resilience* of the system determines whether ecological productivity (e.g. ecosystem functioning) is largely unaffected, decreased either temporarily or permanently or, in the extreme, collapses altogether, as a result of stress or shock (Conway and Barbier, 1990). *Stresses* are small and predictable changes such as increasing erosion and salinity. However, while small these forces can have large cumulative effects. *Shocks* on the other hand are relatively large, temporary and unpredictable such as, for example, a new crop pest or a rare drought. A system that is unable to respond is, in some sense, unsustainable if the stresses and shocks are themselves not capable of control, or, for some reason, are unlikely to be controlled. In turn, capability of response to stress and shocks is usually, but not necessarily, thought to be correlated with diversity of capital, either in the sense of a wide portfolio of natural and produced assets, or a wide portfolio of natural biological assets (Holling, 1973; Conway 1985, 1992; Common and Perrings, 1992).

A measure of the degree of resilience could be interpreted as an indicator of the *degree of sustainability* of the system. However, it less clear what this means for the sustainability of human development. In the green national accounting literature this link is provided by models that connect changes in human welfare to resource depletion and environmental degradation. Hence, green accounting tells us how resource and environmental issues are linked to sustainable human development. The task for the resilience school is to provide a similar link between changes in resilience and sustainability in this way. Although Common and Perrings (1992) have offered one such possible interpretation based on measures of biodiversity, this preliminary work is not yet suggestive of a set of feasible indicators, although standard measures of species diversity and keystone species are clearly candidates, as we discuss below. Pearce and Barbier (2000) note that most attempts to measure resilience have drawn upon the concept of the speed of recovery rather than the interpretation of Holling (1973) (see, for example, Pimm, 1984; 1991).

It is clear that resilience is not something that can be observed directly and so the search for indicators leads in the direction of measuring *inputs* that are thought to contribute to resilience, or the *outputs* that are believed to be affected by changes in resilience. Examples of the former are indicators of biodiversity. If resilience is positively related to biodiversity – as Common and Perrings (1992) and Arrow *et al.* (1995) tentatively suggest – then indices of diversity might be a useful input-based indicator of resilience in ecosystems. The ecology literature offers many such measures (Krebs, 1985) and a useful summary of information about species and habitat diversity is provided by the World Conservation Monitoring Centre (1992).

However, several problems remain. The first of these, as noted above, is that it is not at all obvious how given measures of diversity-resilience map onto sustainability. A related problem is the absence of any clear baseline or 'threshold' to assess the degree of (or deviation from) sustainability using these indicators. The genuine savings approach, for example, has a natural measure of the degree of sustainability, since zero genuine saving defines the borderline between sustainability and non-sustainability. Zero or 'low' diversity would appear to qualify as low sustainability on a diversity measure, but there is no obvious scaling involved. Unless our criterion is to be non-declining diversity, at what value of the diversity index does sustainability become threatened?

Even if non-declining diversity (say, from existing levels) is to be our benchmark, there is a second problem of an empirical nature where available data are often constrained to a single point in time, whereas the relevant measure of sustainability is the change in diversity, not the amount of diversity. That is, the level of diversity is a stock-like quantity whereas it is the change in that stock that is thought to have implications for the degree of resilience (and, in turn, sustainability).

Finally, meaningful interpretations of the data, even if they do exist, are often unclear. For example, the appropriate spatial scale of the index is not simple to determine. Are such indicators meaningful on a national scale or relevant to particular ecosystems regardless of national boundaries? It is likely that national measures of diversity may tell us little about resilience or sustainability in general, although they may be useful in giving a preliminary answer to a different question, namely, where to direct conservation funds most effectively (see, for example, Moran *et al.* 1995).

If we restrict our focus on ecosystems to agricultural systems then a mixture of input- and output-based indicators is suggested. An example of an indicator emerging from *output-based measures*, the loss of resilience in agricultural systems is reflected in variability in crop yields. Upward trends in production may be associated with increasing variability of yields from year to year. In the limit, without any counteracting mechanism, these fluctuations might become so extreme that output collapses. A measure of this variability is changes in the coefficient of variation of crop productivity over time. This is actually a measure of stability (i.e. the ability of a system to return to an equilibrium state after a small, temporary disturbance arising from normal fluctuations and cycles in the surrounding environment). In practice, stability and resilience are likely to be closely and positively linked and hence the distinction may not be critical (Pearce and Prakesh, 1993). For example, Hazell (1984) shows that in India, while the annual growth rate of cereal production was 2.7 percent between 1952/3 and 1977/8, the coefficient of variation around this trend is 4.5 percent per annum before 1965/6 and 6 percent per annum thereafter.

Table 6.4 shows estimates of crop yield variability in India for the period from 1955 to 1989. While other (e.g. climatic) factors need to be controlled for in an analysis, this period also coincides with the structural shift embodied in the green revolution and reveals that, in all but two cases, greater output has been achieved at the expense of greater variation in yields over time.[11] In the case of coarse cereals and groundnuts this change in variation has been highly significant. Conway (1985, 1992) has provided a framework for this kind of analysis – agroecosystem analysis – whereby indicators of a number of desirable attributes of the system (e.g. productivity, resilience and equity) can be assessed. The extent to which farmers can either insure against loss of income in low productivity years or smooth their consumption by saving a portion of high productivity harvests also requires further investigation.

Table 6.4 Crop Yield Variability in India, 1955-1989

Crop	CV pre-Green Revolution	CV post-Green Revolution	Change in CV (%)
Rice	5.1	7.0	**+37**
Wheat	7.2	5.8	-19
Coarse Cereals	3.5	7.3	**+108**
Pulses	8.3	8.5	+2
Foodgrains	3.8	5.4	**+42**
Groundnut	5.5	12.1	**+220**
Cotton	8.8	8.9	+1
Sugarcane	5.2	4.4	-15

Source: Pearce and Prakesh (1993) adapted from Ninan and Chandrashekar (1991)

Indicators of diversity as discussed above are not irrelevant to this analysis. One of the principal explanations of the loss of resilience/sustainability in agricultural systems is changes in biological diversity. While there are a number of different definitions of biodiversity (Pearce and Moran, 1994), here we refer to *genetic diversity* (i.e. the genetic

[11] Note that increased variability of yields is not considered in assessments of technical agricultural possibilities drawing on the carrying capacity concept.

information contained in the genes of plants, animal and micro-organisms). The significance of genetic diversity is often highlighted with reference to global agriculture and food security. The reliance of the majority of the world's human population is on a small number of staple food species (often high-yield varieties), which in turn rely on supply of genes from their wild relatives to supply new characteristics, for example to improve resistance to pests and diseases (Cooper *et al.*, 1992). Anderson and Hazell (1989) cite statistical evidence for variability induced by genetic uniformity of crops.

Of course, other factors are implicated in this process, not least an increased reliance on artificial fertilisers, pesticides and technology. Perrings (1994) suggests that the need to continually substitute produced assets for natural assets in the form of genetic diversity in agriculture can be seen as a cost of the loss of resilience. As such he suggests that the loss of sustainability of a system can be "...measured by the value of increasing quantities of herbicides, pesticides, fertilisers, irrigation and other inputs needed to maintain output at or above current levels...[and]...where output fails, the costs of relocation where soils or water resources have been irreversibly damaged" (Perrings, 1994, p.39). Here, the weak sustainability aspects of substitutability – produced assets for natural assets – are linked to the strong sustainability consideration of ecological functions and the properties of ecosystems. A key question in this context is the extent to which the substitution option is as effective as in the past.

6.4 Discussion and Conclusions

We have identified three distinct sets of indicators associated with strong sustainability. The common feature of this literature is most usually a focus on thresholds – that is, the assimilative capacity of the environment or, for living resources the sustainable harvest, which it is argued must not be exceeded. The question of which of these constraints should be observed on the grounds of criticality, however, typically is not directly addressed, such that society is expected to observe all specified constraints on the *assumption* of criticality.

It is also notable that much of this work is abstract in that few proposed indicators have been based on actual data. The work of Adriaanse (1993) and others in the Netherlands is the exception here. Where data have been generated – e.g. population carrying capacity – these have usually been an overly simplistic description of technical constraints and the level of population deemed to be consistent with these constraints. Empirical applications have tended to be restricted to questions concerning absolute availability of agricultural production, fuelwood and water supply. To the extent that these indicators are intended to inform policy this is not helped by the absence of meaningful socio-economic content to these indicators. However, in providing data regarding resource/ poverty links, these simple indicators may have some value, as long as appropriate caveats are made. In contrast, a somewhat different indicator based on carrying capacity – ecological footprints – seems to create more confusion than it resolves.

The diversity-resilience/sustainability link although less researched from the point of view of indicators, is one promising attempt to add analytical content to the concept of SS. Arguments in favour of conserving diversity, as opposed to the stock of biological resources, have been advanced in terms of ethics, aesthetics and economics (Wilson, 1988; Pearce and Moran, 1994). It is also argued, however, that biodiversity conservation is required for *insurance* purposes, e.g. for sustaining output, and in particular agricultural output, in the face of risks to the genetic base of existing output. The argument here is that the risk of system collapse is higher the less diverse the system. If so, then diversity does have a link to sustainable development as many suspect (Arrow *et al.* 1995).

Output variability or some other indicator related to agricultural production are most relevant to the developing world where agriculture comprises a significant proportion of GDP. In developed countries agricultural output contributes such a relatively small proportion of total output that these indicators are of little strategic importance (although of course these countries may need to import their food requirements from countries affected by large variability). If pressures on agricultural resilience are increased by both demand for food and by modern farming practices, sustainability might be threatened. Meeting these demands has obvious benefits to these countries, in the form of scarce foreign exchange, with the possible downside of loss of sustainability in the form of resilience.

In applying the resilience concept to ecosystems in general, affected 'outputs' will, more often than not, be intangible and non-market. Measures such as the coefficient of variation will not be operational. However, measures of resilience have more appeal than the carrying capacity approach in that indicators of biodiversity are stressed as central to the measurement problem and many would now agree that diversity is a critical natural asset (see Schulze and Mooney, 1994). Yet, as we have seen, practical indicators of biodiversity fall short of what we require to measure sustainability. Problems include the absence of a baseline required to make sustainability judgements and the inadequacy of existing data restricted as it is to single point estimates. In this respect, agricultural diversity while only one aspect of a wider concern with respect to biodiversity, is a promising start. For these reasons, it is arguable that resilience, whilst attractive from a theoretical and SS standpoint, has, at the moment, little to offer for the development of practical indicators that will help in decision-making.

Assuming, however, that the requisite advancement will be made a promising programme for measuring sustainable development could emerge from apparently disparate strands. That is, sustainability most probably requires both an avoidance of persistently negative genuine saving and decline in stocks of critical natural assets or resilience. Progress with respect to the latter part of this measurement challenge will provide a crucial test of whether strong sustainability is more than just rhetoric.

Chapter 7

Balancing Competing Principles of Environmental Equity

7.1 Introduction

Decisions regarding how to share the burdens of environmental policy typically reflect a mixture of efficiency and equity criteria. Put another way, relevant concerns include both how the economic resources needed to achieve a given goal can be used most effectively and how the burdens associated with reaching this goal are distributed. It is the latter problem that concerns us in this chapter. For example, one popular argument as to why different people should be assigned different burdens is that people differ in terms of their 'ability to pay'. A rule for sharing burdens that takes into account differences in ability to pay is argued to be, other things being equal, more equitable. Clearly, a problem arises if there exists more than one relevant principle or argument for assigning this same burden. In the context of paying for a public environmental improvement policy, for example, 'responsibility' for causing the problem could dominate or compete with alternative principles such as ability to pay. Specifically, the problem for decision-makers is to weigh up, either implicitly or explicitly, the relative importance of competing principles.

While we do not claim to fully resolve this debate, we propose a relatively novel approach insofar as it pertains to the discussion of environmental equity. We show that by using a survey methodology, commonly used to elicit respondents' preferences for changes in the provision of environmental goods, we can move beyond seemingly intractable arguments regarding the reconciliation of competing principles of environmental equity. In doing so, we examine, via the econometric analysis of responses, how individuals prioritise and trade-off characteristics or attributes that reflect competing equity principles. Our results suggest that such trade-offs do exist and, furthermore, are significant. In addition, we examine whether the award of a burden is influenced by respondents' selfish preferences, as it is entirely plausible that individuals consider their own attribute levels when ranking different burden sharing criteria. Finally, we discuss the way in which our results can inform theories of distributive justice and sustainable development and illustrate further how to evaluate the relative importance of competing principles.

7.2 Competing Principles of Environmental Equity

The burden-sharing problem refers broadly to the question of 'who should pay' for the costs of environmental protection. Even within the environmental domain, this covers a potentially large range of policy concerns. On one hand, there is the question as to how burdens should be distributed for those policies implemented within a nation. The burden in such instances refers to the cost of the regulation, which can be construed, for example, as the economic cost of the mandated distribution of abatement effort.[1] However, it is important to note that a burden can be interpreted in many different ways. The decision as to where to site a hazardous waste facility is basically one of where to assign a burden (which in this instance is an environmental risk) to some location (Linnerooth-Bayer and Fitzgerald, 1996). Transport policy such as congestion pricing imposes different costs (or burdens) on different groups (Goodwin, 1990). Another distinct class of problem is where burdens of policy have to be shared across national boundaries, the pre-eminent example of which are policies to mitigate climate change (see, for example, IPCC, 1996).

In order to illustrate this problem, we examine the introduction of a hypothetical public programme to reduce the incidence of local air pollution in a major European city (Lisbon). In this example, the burden-sharing problem refers to how the total financial cost, arising from a given allocation of abatement or pollution reduction effort, is distributed. For example, the abatement of polluting substances might be allocated to actors in a way that is estimated to be the least costly option to reach a pre-specified ambient air quality goal. However, in principle, this financial cost could be distributed such that certain actors, where deemed appropriate, are assisted financially with some part of their required abatement effort. What this assumes is that the cost of the policy does not change according to how it is distributed across parties. This emphasis allows us to focus on arguments for distributing burdens based

[1] An alternative interpretation is to ask who should undertake a given abatement effort.

on equity grounds alone. We acknowledge that this is a relatively simple interpretation of the burden-sharing problem, in that real-world phenomena of this kind are undoubtedly far more complex. However, we argue that, by limiting our focus in this way, this will allow us to gain an important and potentially far-reaching insight into the extent to which different principles of environmental equity relevant to the problem might be traded-off against one another by individuals deciding 'who should pay'. However, it is worth noting that the approach that we outline could be extended to consider more complicated scenarios such as, for example, trade-offs between equity and efficiency both now *and* into the future.

Even if we restrict our attention in this way, there remains the crucial question as to what principle, or set of principles, should guide burden distribution? Matters are relatively simple if only one criterion is held to be relevant to the chosen problem. In practice, however, there are likely to be numerous candidate principles each offering an appealing argument as to why burdens should be distributed in a particular way. In addition, disputes may arise concerning the appropriate process to be used to determine distribution: e.g. market allocation vs. politically negotiated agreement (Linnerooth-Bayer and Fitzgerald, 1996). In general, we would expect different rules, in turn based on competing principles, to result in very different distribution of burdens and the empirical evidence appears to bear this out (Burtraw and Toman, 1992; Rose *et al.* 1998).

Arguments 'for' and 'against' principles relevant to our burden-sharing problem have ranged over both philosophical and pragmatic terrain. For example, Kvendorkk (1994) and Rose *et al.* (1998) discuss how various general theories of distributive justice can inform questions of environmental equity. In contrast, Elster's (1992) concept of local justice suggests that the suitability of a particular principle (or principles) will vary across different burden sharing problems and that evidence for this can be found by observing how institutions distribute burdens in practice.[2] Similarly, Young (1994) proposes that burden-sharing criteria emerge as practical rules of thumb in response to specific policy questions. For the purposes of this chapter, we need to ask two questions. Firstly, on the basis of philosophical and

[2] Elster's own examples range from the allocation of household work to military service in wartime.

pragmatic criteria, what principles or common themes can be identified as relevant to determining how to distribute environment-related burdens? Secondly, if it is credible that a single principle is insufficient for this task how are trade-offs between competing principles to be measured?

Regarding the determination of relevant principles, it can be argued that much of the environmental debate has been associated with the argument that it is the polluter who should pay for a programme of environmental improvement (e.g. OECD, 1975). In other words, burdens should be distributed according to responsibility for, or contribution to, an environmental problem. Indeed, a significant body of environmental legislation, at the national and international level, has been influenced by the polluter pays principle (PPP) (Tobey and Smets, 1996). This is consistent with what Bromley (1997) describes as the trend towards assigning property rights to 'victims' of pollution. However, Bromley also introduces the caveat that, in practice, a varying degree of protection is offered given other societal concerns and priorities. Two additional 'concerns and priorities' spring to mind. The first is the distribution of income and wealth, in that society might deem it relevant to distinguish between 'rich' and 'poor' polluters. The second is an alternative distribution of property rights that harnesses, in some degree, the incentive that exists for beneficiaries to secure environmental improvements (i.e. net benefits).

Individuals (or groups) will differ not only in terms of responsibility but also in terms of their ability to pay for burdens awarded to them. If concerns regarding the distribution of income (or wealth) are also thought to be relevant then the allocation of burdens between parties should presumably reflect differences in ability in pay. Indeed, within the context of environmental policy, considerable support for this proposition can be found. Hence, Karadeloglou *et al.* (1995) note that a general principle within the European Union is that a higher burden should be borne by relatively high-income member states. Within nations, the principle of distributional neutrality, or at least something approximating it, has led to discussion of mechanisms that combine the attainment of environmental goals with assistance to those low-income or vulnerable individuals hardest hit by a policy change (Pearson, 1995).

The question of whether alternative means of distributing property rights are relevant is also interesting but more contentious. What this means is assigning, at least part, of the burden to those individuals or parties who benefit as a result of an increase in environmental quality. Put another way, this is a 'beneficiary pays principle' (BPP). Examples of the BPP can be found in the theory and practice of international environmental agreements[3], although less clear support can be found for this proposition at the national level and below. Indeed, Fermann (1997) balks at the prospect of burdens being assigned to 'victims' where a recognisable polluter exists. Nevertheless, while not without its difficulties, the principle that actors might be expected to pay in proportion to the benefits they enjoy is an established proposition in deciding how provision of various public goods is to be financed (Young, 1994).

It is important to note that we are not assuming that, for example, ability to pay substitutes for responsibility as a burden sharing principle. Rather, it is argued that some mixed system, which combines these principles, may be desirable. If so, we might think of the burden-sharing problem as a trade-off where competing principles are balanced in some way. Trade-offs of this type characterise much of Elster's local justice problem but appear to have been less thoroughly explored in the context of environment-related burdens.

If an analogous trade-off characterises the environmental burden-sharing problem then the challenge is to establish a methodology capable of evaluating this. To this end, the aim of this chapter is to use our discussion of three principles for distributing burdens to inform the nature of equity trade-offs for our local air pollution reduction programme. That is, to explore the trade-offs involved in deciding how to allocate burdens to those individuals who are 'responsible' for the problem, 'benefit' from the policy change implemented or are 'able to pay' for the programme. While other principles are undoubtedly relevant to this burden-sharing problem, we feel that these three arguments capture the broad flavour of much of the debate.

[3] This is often a response to the strategic issues that emerge when sovereign countries negotiate together rather than a reflection of equity concerns (but see, Eyckmans, 1997).

There is presumably much to be learned from detailed case studies of the resolution of burden-sharing disputes in past environmental agreements and policies (Albin, 1994). However, an alternative and novel means of exploring these trade-offs – the use of survey instruments – has been proposed by Young (1994). For example, survey instruments within the field of environmental economics, such as contingent ranking, designed to elicit preferences for environmental goods provide a promising vehicle for evaluating trade-offs (Foster and Mourato, 2000; see, for a review, Hanley and Mourato, 1999). Furthermore, Ryan *et al.* (1998) examines how multiple criteria can be evaluated, in the context of health-care, using a similar survey methodology. Cropper and Subramanian (1995) combine these two policy contexts by examining respondents' relative preferences for different types of environmental health and public health programmes. While none of these studies explicitly take account of the preferences of respondents towards equity, Linnerooth-Bayer and Fitzgerald (1996), using a somewhat different survey approach, examined individual preferences for different equitable criteria for selecting hazardous waste sites in Austria. This study finds evidence for conflict between competing and desirable criteria characterising the problem but does not evaluate the existence of trade-offs between conflicting positions.

By using a survey methodology, based on contingent ranking, we hope to provide valuable information by focusing on such trade-offs in a consistent fashion. This allows us to elicit responses from a sample of individuals, randomly drawn from the population at large, which reflect society's preferences for environmental equity or more specifically, attributes reflecting different burden-sharing principles. This information is of interest for two reasons. Firstly, these data may provide a direct test of the proposition that individuals (or society in general) are willing to trade-off competing principles of environmental equity. Secondly, this could yield a set of weights reflecting the relative importance of each principle. This could be especially useful if what is desired is a burden sharing rule that is akin to a point system, where burdens are assigned on the basis of an actor's 'score'.

An interesting parallel to this discussion is concern with equity and burden sharing in the climate change literature (Linnerooth-Bayer, 1999). For example, IPCC (1996) has reviewed a plethora of proposed rules in this context where indicators reflecting notions of equity include per capita income levels and (past and present) emissions of greenhouse gases. The most interesting aspect of that debate, for this chapter, is that several of these rules attempt to combine principles (in order to allocate greenhouse emissions reductions). For example, Wirth and Laslauf (1990) combine indicators based on emissions per capita and emissions per unit of Gross Domestic Product (GDP) by assuming that each carries equal weight. To the extent that perceptions of equity within nations can have a significant bearing on how equity is perceived in an international setting (IPCC, 1996) our analysis may offer some insight into this discussion. Clearly, however, applying the methodology we propose to this wider realm would create complicated but intriguing sampling issues.

7.3 Methodological Framework

7.3.1 Contingent Ranking Method

The methodology adopted for our burden-sharing experiment is the contingent ranking (CR) method. CR is a survey technique used in marketing, transport and environmental research to model preferences for bundles of characteristics of goods and to isolate the value of individual product characteristics typically supplied in combination with one another (Beggs, Cardell and Hausman, 1981; Smith and Desvousges, 1986; Foster and Mourato, 2000). The conceptual framework typically used to provide the rationale for CR is random utility theory (McFadden, 1973). In a CR experiment respondents are required to rank a set of alternative options. For example, these options might correspond to various environmental programmes. Each option is characterised by a number of attributes, which are offered at different levels across options. Respondents are then asked to rank the options according to their preferences or simply to choose their most preferred programme. One advantage of CR is its unique ability to deal with situations where programmes are multi-dimensional and where trade-offs are of particular interest. Its main limitation is the cognitive difficulty associated with complex choices between bundles with many attributes and levels.

In order to investigate individual preferences for environmental equity, we consider a particular policy scenario. This relates to a city (Lisbon) where urban air pollution levels, which are mainly caused by automobile emissions, can be significantly reduced if a particular fixed-cost public environmental programme is put in place. The programme has a positive net social benefit (that is, its aggregate benefits, in the form of pollution damages avoided, are greater than its cost). Our concern is how to distribute its budgetary cost amongst the city's inhabitants. We assume that whatever rule is used for distributing the programme's financial burden, there will be no allocative efficiency effects such that the choice of burden-sharing rule can be made on equity grounds only. Hence, participants in the ranking experiment were asked to rank, according to their preferences, different ways of distributing the burden of the environmental programme among different specified groups of the population. These population groups differed with respect to: responsibility for pollution (or damage caused); benefit received from the programme; and, ability to pay.

According to the random utility model, respondents will select the burden-sharing allocation that maximises their personal utility/satisfaction. Since all the determinants of individual choice cannot be observed, the utility function (U) for each respondent i can be decomposed into two parts: a deterministic element, which is a linear index of the attributes (X) of the j different alternative burden-allocation options in the choice set; and a stochastic element (e) which represents unobservable influences on individual choice. This specification is shown in expression (7.1).

$$U_{ij} = bX_{ij} + e_{ij} \tag{7.1}$$

where X represents the relevant attributes of the various groups of the population that might be sharing the burden of the environmental programme.

Thus, the probability that any particular respondent prefers burden-sharing option g in the choice set to any alternative option h, can be expressed as the probability that the utility associated with option g exceeds that associated with all other options, as stated in expression (7.2).

$$P[(U_{ig} > U_{ih}) \forall h \neq g] = P[(bX_{ig} - bX_{ih}) > (e_{ih} - e_{ig})] \tag{7.2}$$

Under the assumption of an independently and identically distributed random error (e_{ij}) with a Weibull distribution, Beggs, Cardell and Hausman (1981) developed a rank-order logit model capable of using all the information contained in a survey where alternatives are fully ranked by respondents. The probability of any particular ranking of burden-sharing alternatives being made by individual i can be expressed as:

$$P_i(U_{i1} > U_{i2} > \ldots > U_{iJ}) = \prod_{j=1}^{J} \left[\frac{\exp(bX_{ij})}{\sum_{k=j}^{J} \exp(bX_{ik})} \right] \tag{7.3}$$

The parameters of the utility function can be estimated by maximising the log-likelihood function given in expression (7.4).

$$\log L = \sum_{i=1}^{N} \log \prod_{j=1}^{J} \left[\frac{\exp(bX_{ij})}{\sum_{k=j}^{J} \exp(bX_{ik})} \right] \tag{7.4}$$

Note that the contingent ranking model does not allow respondents to be indifferent between options presented in the ranking sets, i.e. no ties are allowed between options. This may introduce bias in respondents' stated choices which is more likely to occur when ranking least preferred alternatives as respondents may find it difficult to identify which option they dislike the least. This problem could lead to a less statistically significant model overall as respondents who are truly indifferent between two options are forced to randomly choose between them.

7.3.2 Experimental Design

We conducted two split-sample contingent ranking experiments in the city of Lisbon (Portugal) during April 1998. Overall, 516 people were interviewed, with 257 people receiving the first experimental treatment and 259 receiving the second. The samples were randomly drawn and representative of the Lisbon population. The interviews were conducted in person by a professional survey company with previous experience in stated preference surveys. The questionnaires included sections on attitudes towards the environment, preferences towards various health issues, burden-sharing ranking questions and socio-economic characteristics. The following opening information on the ranking section was offered:

B-1 As you may well know, air pollution could be the cause of a number of diseases. Now imagine a programme to reduce air pollution in Greater Lisbon that would cost around 2 thousand million Portuguese escudos. The local authorities in Lisbon would then have to decide on what basis different groups of people should bear the burden of paying for this programme.

Experiment 1

In the first experiment, respondents were asked to rank 6 hypothetical groups of individuals in terms of which group they thought should bear the burden of paying for the environmental programme specified. These population groups differed according to three relevant characteristics or attributes: responsibility for the pre-programme level of air pollution; the benefits received from the programme; and, ability to pay for the programme. The ranking card used is reproduced in Figure 7.1.

Figure 7.1 Sample Ranking Card for Experiment 1

	Group					
	A	B	C	D	E	F
Health State	Benefits	Benefits			Benefits	
Income Level	Low	High	Low	High	Low	High
Responsibility	Pollutes		Pollutes			Pollutes
RANK						

Each population attribute was assigned two levels:

Responsibility for pollution: (a) participating in activities that contribute to the original pollution problem in the city, such as driving a car that gives off relatively dirty exhaust fumes (coded as 'pollutes'); (b) not participating significantly in activities that contribute to the pollution (coded as a blank space)

Health benefits received: (a) experiencing improvements in health as a result of the pollution reduction programme (coded as 'benefits'); (b) not noticeably benefiting from improved health in this way (coded as a blank space)

Level of income: (a) earning below the average income of 100 thousand Portuguese Escudos (PTE) per month (coded as 'low'); (b) earning four times average income or more (coded as 'high')

For 3 attributes with 2 levels each this design gave a total of 8 combinations. Two of these combinations could be eliminated as they corresponded to combinations that were either dominant or dominated by all those remaining. Respondents therefore were presented with only 6 choices to rank, as depicted in Figure 7.1.[4]

The environmental programme specified that cost was the same no matter how burdens were allocated and respondents were further informed that group size and the age and number of dependants of each group member were constant across all individuals. That is, population groups could only be distinguished on the basis of responsibility, benefits and income. Respondents were then asked to think carefully about which groups of people should bear the costs of the specified programme and to subsequently rank the groups from 1 to 6, where '1' indicates who should pay first and '6' who should pay last.

The empirical counterpart of model (1) estimated in this split-sample experiment can therefore be specified as:

$$U_{ij} = a_1 P_{ij} + a_2 B_{ij} + a_3 Y_{ij} \tag{7.5}$$

where U_{ij} represents the utility that individual i draws from choosing group j, while the attributes are responsibility for pollution in group j (P_{ij}), benefits received from the programme by group j (B_{ij}) and income level of group j (Y_{ij}). The utility function

[4] In general, 4-6 options are seen as the most that respondents can sensibly be asked to rank (Smith and Desvousges, 1986).

(7.5) can be regarded as an individual preference function for variations in the welfare of the various groups of society. The parameters a can be interpreted as weights attached to the various characteristics of society. In other words, they reflect the 'burden sharing priority' attached to various population attributes. The attributes are all dummy variables taking a value of 1 if there is a positive responsibility for pollution, benefit from the programme or high income, and zero otherwise. This model will be henceforth referred to as the 'dummy variable model'.

Experiment 2

In the second experiment, respondents were presented with two ranking questions. The first asked respondents to rank hypothetical population groups on the basis of responsibility and benefits alone. Card A in Figure 7.2 reproduces the ranking card used. Hence, for the purposes of this question, income was held constant across individuals and the focus is solely upon the property rights issue. The second question asked respondents to rank individuals on the basis of responsibility and income alone (benefits being held constant across individuals). This question allowed us to examine competing principles based respectively on property rights and concern regarding the distribution of income. The card used in this question is reproduced in Card B in Figure 7.2.

This simplified design, whereby population groups are represented by only two attributes, has the advantage of allowing more than 2 levels for each attribute, while still preserving the number of options respondents are asked to consider (6 choices). In effect, each attribute was now set at three levels each:

<u>Responsibility for pollution</u>: 3 levels of annual contribution to pollution, expressed in monetary terms, were considered: low (0 PTE), medium (5 thousand PTE) and high (15 thousand PTE). This reflects the fact that people are not equally responsible for pollution. For example, some people do not possess a car and therefore are hardly responsible for air pollution; while others have highly polluting cars or drive many miles per day.

Health benefits received: 3 levels of annual benefit from the programme, expressed in monetary terms, were considered: low (0 PTE), medium (7.5 thousand PTE) and high (20 thousand PTE). This reflects the fact that not everyone will benefit in the same way from the pollution control programme. For example, people suffering from asthma will benefit more than others.

Level of income: 3 levels of monthly income were considered: low (100 thousand PTE), medium (250 thousand PTE) and high (375 thousand PTE).

Figure 7.2 Sample Ranking Cards for Experiment 2

Card A	Group					
	A	B	C	D	E	F
Benefit/year	7.5	0	20	20	7.5	0
Pollutes /year	15	15	0	5	5	5
RANK						

Card B	Group					
	A	B	C	D	E	F
Pollutes/year	0	5	15	15	0	5
Income/month	250	375	100	250	375	250
RANK						

The fact that three attributes are used instead of two, as in the previous experimental treatment, allows the investigation of the presence of non-linearities in individual preferences for burden-sharing rules. Note also that, in this second experiment, the attribute levels are specified in monetary units (i.e. PTE).[5]

[5] Specifying attribute levels in this way is only one means whereby non-linearities can be examined. In the case of the polluter, the monetary units indicate the total value of pollution damage for which the polluter is said to be responsible. For beneficiaries, the units correspond to the total value of benefits received; that is, willingness to pay for a marginal improvement in health multiplied by physical benefits received. Of course, in practice there are problems in valuing environmental benefits in this way. Nevertheless, respondents did not find this to be a factor affecting their ability to make consistent choices on the basis of the information provided.

As before, respondents were asked to rank these various population groups in terms of their burden sharing priority (that is, they are asked 'who should pay first'). The empirical models estimated in this second split-sample experiment can be specified respectively as:

$$U_{ij} = w_1 P_{ij} + w_2 B_{ij} \tag{7.6}$$

$$U_{ij} = z_1 P_{ij} + z_2 Y_{ij} \tag{7.7}$$

where U_{ij} represents the utility that individual i draws from choosing group j, P_{ij} is the responsibility for pollution of group j, B_{ij} is the benefits received from the programme by group j and Y_{ij} is the income level of group j. The utility function and parameters have the same interpretations as before. The attributes are now continuous variables as described before. Therefore, this model will be referred to as the 'continuous variable model'.

7.4 Results

Table 7.1 presents some sample statistics and shows that both sub-samples used in the experiments have very similar characteristics.

Table 7.1 Descriptive Statistics

Variable	Sub-sample I (Dummy Model)	Sub-sample II (Continuous Model)
Sample size	257	259
Males (no.)	114	101
Age (average)	45	46
Income (average monthly, '000 PTE)	179	189
Income non-response	108	133
Petrol (average monthly, '000 PTE)	18	19
Health status (from 1 very good to 5 very bad)	3.08	3.06

7.4.1 Experiment 1

Table 7.2 illustrates the results for the dummy variable model. The first column contains the estimates of the basic model represented by expression (7.5). Clearly, responsibility, i.e. 'pollutes', is the most important attribute, an outcome that is consistent with the predominance of the 'polluter-pays-principle' in environmental policy discussions. However, the results also indicate marked evidence of a trade-off between responsibility and other attributes in that the coefficients on 'benefits' and 'income' are highly significant. Indeed, the sum of the coefficients on the 'benefits' and 'income' variables (i.e. 0.76 + 1.73) is greater than the coefficient on the responsibility variable (i.e. 2.09). This would appear to suggest that an individual who both benefits from the programme and has high income should pay more, other things being equal, than someone who is responsible for the environmental problem but has low income and does not benefit in any way from the policy change, at least in the context of this experiment.

Table 7.2 Dummy Variable Model Results (t-ratios in parenthesis)

Variable	Basic Model	Model Without Non-Response	Individual Interactions Model
Benefits	0.7583	0.70286	0.24257
	(11.263)	(7.539)	(0.565)
Income	1.7268	1.79034	2.01896
	(26.725)	(20.118)	(13.541)
Pollutes	2.0919	2.07676	2.20495
	(32.824)	(23.661)	(16.855)
Monthly Petrol*Pollutes	—	—	-0.000008
			(-1.266)
Health Status*Benefits	—	—	0.15854
			(1.108)
Monthly Earnings*Income	—	—	-0.000001
			(-1.819)
N	198	105	105
Log-Likelihood	-1055.091	-556.5621	-555.0633

Notes:
1. In these models the choice attributes are dummy variables: 'Benefits' – 1 if benefits from the programme; 0 otherwise; 'Income' – 1 if high; 0 if low; 'Pollutes' – 1 if is responsible for pollution; 0 otherwise.
2. The individual specific variables are coded as following: ' Monthly petrol' – individual monthly expenditure in petrol; 'Health status' – 1 to 5, with 1 if very good and 5 if very bad; 'Monthly earnings' – monthly individual earnings.

An interesting question is the extent to which the preferences of each respondent for burden sharing allocations are independent of his or her particular position in society. For example, does the knowledge of a given respondent that he or she has relatively high income bias his or her response away from options that assign burdens to high income individuals? The answer depends on whether selfish motives influence responses or whether the choice between alternative allocations is made *as if* respondents were under Rawls' 'veil of ignorance' (Rawls, 1972). We specified variables to capture this aspect of responses using survey data on respondents' monthly petrol expenditure ($Petrol_i$), health status ($Health_i$) and monthly earnings ($Earnings_i$). The impact of socio-economic factors can be gauged by interacting socio-economic variables with the attributes of the alternatives and examining the signs and significance of the resulting interacted variables.[6] The empirical model with interactions estimated can be specified as:

$$U_{ij} = a_1 P_{ij} + a_2 B_{ij} + a_3 Y_{ij} + a_4 Petrol_i{*}P_{ij} + a_5 Health_i{*}B_{ij} + a_6 Earnings_i{*}Y_{ij} \qquad (7.8)$$

The third column of Table 7.2 shows that allowing for individual interactions does not appear to add much to our discussion. The signs on the coefficients on *Petrol*P* and *Earnings*Y* are negative as we would expect. However, the former is insignificant and the latter only significant at the 10% level. The coefficient on *Health*B* is positive but insignificant. Owing to the considerable number of non-responses, particularly with regards to respondents' monthly earnings, we re-estimated the basic model on the sub-sample of respondents who had answered all questions. This provided a check on whether the comparison between the basic model and the model

[6] Owing to the statistical specification of the logit model it is not possible to incorporate socio-economic regressors directly into the utility index given by expression (7.5). The random utility framework is based on differencing the values of attributes across different alternatives and any attribute that does not vary across alternatives would simply drop out of the model as a result of this process. Interacting socio-economic characteristics circumvents the problem because the interacted

with interactions might be biased due to the fact that each model was estimated on a different sample. The second column of results in Table 7.2 reveals that removing non-responses from the basic specification does not alter the conclusions drawn from the basic model.

Table 7.3 investigates the trade-off between attributes further by analysing how many respondents behaved in a way possibly consistent with lexicographic preferences. Lexicographic ranking is defined as a tendency for respondents to rank questions solely with reference to one of the attributes, ignoring all other differences between options. A more complete form of lexicographic ranking may arise when respondents also refer to a second attribute to discriminate between those options that 'tie' in terms of their attribute of prime interest. Such behaviour may either reflect the use of a simplifying heuristic to aid the ranking process or may be a manifestation of underlying preferences that are truly lexicographic. This is important because it is entirely possible that, for example, our findings compatible with say, the PPP whereby respondents always rank according to responsibility relying on other attributes only in the event of ties.

Table 7.3 Respondents with Possible Lexicographic Rankings

Ranking	% Respondents (n=198)
P, Y, B	19%
P, B, Y	2%
Y, P, B	11%
Y, B, P	2%
B, Y, P	0
B, P, Y	1%
Total	35%

Note: *P* – Pollutes; *Y* – Income; *B* – Benefits. For example, '*P,Y,B*' means that respondents first ranked burden-sharing allocations according to responsibility for pollution; then, in case of ties, ranked according to income; and finally decided upon any remaining ties according to the benefits from the programme.

variable *both* contains individual-specific socio-economic information *and* varies across alternatives in the choice set.

Analysis of the data reveals that 35% of respondents systematically ranked options with reference to only one of the three criteria, referring to a second attribute only where particular pairs of options tied on the basis of the first criterion. This total can be broken down as follows.[7] About 21% of respondents seemed to be primarily concerned with the pollution (*P*) attribute, 13% with income (*Y*) and 1% with benefits (*B*). The most prevalent lexicographic ranking is by *P* then by *Y* and then by *B*. In all, 19% of respondents answered in this way. While, at first blush, these results appear to indicate that a proportion of respondents may have been using a lexicographic rule as a response algorithm, it can be shown that these rankings can be explained in terms of a standard utility function that accords a relatively high weight to the attribute concerned (Foster and Mourato, 2000). In the absence of a more sophisticated test to distinguish between these two hypotheses we argue that it is reasonable to speculate that respondents are willing to trade-off between competing principles. Clearly, however, it would be desirable to seek to reduce this ambiguity in future work.

7.4.2 Experiment 2

To the extent that there may be non-linearities in respondents' preferences the dummy variable model with only two levels of each attribute will not capture this. For example, either of the following two scenarios might be plausible. On one hand, we might speculate that individuals believe that the burden assigned should be increasing in the level of an attribute. On the other, the burden assigned might be a decreasing function of the attribute level as would be the case where respondents find it difficult to distinguish between relatively high levels of a given attribute. In order to evaluate how robust our initial conclusions are, Table 7.4 illustrates results from the continuous variable model, where each attribute was assigned three levels. As noted above, given that assigning 3 levels to 3 attributes would significantly complicate the ranking task that respondents have to perform, in our second experiment, we only asked them to consider two attributes at a time (i.e. *P*/*B* and *P*/*Y*) therefore holding the omitted attribute (respectively, *Y* and *B*) constant.

[7] Note that the probability of getting a (strong) lexicographic ranking amongst the 720 possible ranking combinations would be only 0.83% had the rankings been random.

Table 7.4 Continuous Variable Model Results (t-ratios in parenthesis)

Variable	Basic Model A	Quadratic Model A	Individual Interactions Model	Basic Model B	Quadratic Model B	Individual Interactions Model
Benefits	0.05683	0.25693	0.25835	—	—	—
	(16.315)	(12.338)	(12.351)			
Income	—	—	—	0.00860	0.01029	0.01003
				(52.922)	(9.733)	(6.533)
Pollutes	0.27220	0.80704	0.79688	0.22404	0.45553	0.46350
	(57.693)	(32.197)	(31.368)	(43.563)	(16.058)	(11.328)
$(\text{Benefits})^2$	—	-0.00813	-0.00812	—	—	—
		(-8.087)	(-8.042)			
$(\text{Income})^2$	—	—	—	—	-0.000005	-0.000003
					(-1.646)	(-0.622)
$(\text{Pollutes})^2$	—	-0.02804	-0.02822	—	-0.01445	-0.01504
		(-17.024)	(-17.088)		(-8.147)	(-5.944)
(Monthly petrol *Pollutes)	—	—	0.000001	—	—	0.000002
			(4.228)			(4.294)
(Health status *Benefits)	—	—	-0.01290	—	—	—
			(-0.668)			
(Monthly earnings *Income)	—	—	—	—	—	-0.000000003
						(-2.237)
N	232	232	231	226	226	117
Log-Likelihood	-1085.749	-936.5785	-929.6205	-1155.473	-1114.612	-550.3748

The presence of three levels for each attribute permits the estimation of quadratic specifications that allow for non-linear preferences for levels of an attribute:

$$U_{ij} = w_1 P_{ij} + w_2 B_{ij} + w_3 P^2_{ij} + w_4 B^2_{ij} \tag{7.9}$$

$$U_{ij} = z_1 P_{ij} + z_2 Y_{ij} + z_3 P^2_{ij} + z_4 Y^2_{ij} \tag{7.10}$$

The second column of results in Table 7.4 indicates that the coefficients on both the 'pollutes square' and 'benefits square' variables are negative and significant. Similarly, in column 5, the coefficients on both the 'pollutes square' and 'income square' variables are negative (with the latter only significant at the 10% level). This indicates, to some extent, the presence of non-linearities in respondents' preferences. In general, the results of the quadratic models A and B seem to confirm the most important conclusions drawn from previous specifications, therefore indicating a high degree of consistency between models.

Table 7.4 also contributes to our earlier discussion regarding selfish preferences. Results are illustrated in the individual interactions models in columns 3 and 6, corresponding to the following empirical utility specifications:

$$U_{ij} = w_1 P_{ij} + w_2 B_{ij} + w_3 P^2_{ij} + w_4 B^2_{ij} + w_5 \, Petrol_i{*}P_{ij} + w_6 \, Health_i{*}B_{ij} \qquad (7.11)$$

$$U_{ij} = z_1 P_{ij} + z_2 Y_{ij} + z_3 P^2_{ij} + z_4 Y^2_{ij} + z_5 \, Petrol_i{*}P_{ij} + z_6 \, Earnings_i{*}Y_{ij} \qquad (7.12)$$

Again, the effect of a respondent experiencing bad health appears to be an insignificant factor in assigning a burden. However, the coefficient on the $Petrol_i{*}P_{ij}$ variable is surprisingly positive and significant in both models. This would apparently suggest that respondents consider that they themselves should face a *higher* burden if they are responsible for pollution. However, in both models this effect is rather small in that the coefficient is relatively low (0.000001 and 0.000002 respectively). The interaction of the 'Income' attribute with respondents' earnings in the final column indicates a negative and significant coefficient as expected. This offers further evidence that respondents with high income were somewhat less inclined to assign burdens on the basis of the income attribute. Overall, considering both experimental treatments, the evidence of the impact of selfish preferences in burden allocations is mixed.

7.5 Discussion

It is interesting to ask how our results offer guidance regarding the design of burden-sharing rules. Although we did not frame our experiment so as to test a specific theory of distributive justice our results do cast light, albeit indirectly, on this debate. Most notably, our results offer evidence that rather than rely on a single theory or principle, it is a balance of competing theories or principles that more accurately reflect people's preferences for solving environmental burden sharing problems. For example, individuals may be concerned with 'wealth-' or 'distributional effects' associated with a particular allocation of burdens (or rights) between polluter and victim. Our results reveal an apparent willingness to trade-off between that principle guiding property right allocation and that for income distribution. Hence, although our results confirm that, by and large, it is the polluter that should pay, it can also be argued that if a beneficiary also has high income then that individual should pay something in return for receiving these benefits. Furthermore, our analysis of the responses regarding the award of a property right to either beneficiary or polluter, holding income constant, leads to a stronger proposition. If an individual receives a relatively large share of the benefits from an environmental programme then that person should pay something towards this programme. The relative weights that can be estimated from survey-based approaches such as in this chapter could be an important input into the design of burden sharing rules that quantify, in a meaningful way, the relative importance of competing principles.

A useful way to further examine the nature of these relationships is to calculate the indifference point between the allocation of burdens to distinct representative individuals. This is done in two steps. First, consider a certain baseline alternative. For example, in the continuous model B, we may take the baseline as being the situation where individuals with an income of 100,000 PTE and who pollutes 15,000 PTE pay for the environmental programme. Second, fix one of the attributes and raise the other until the linear utility score implied by the model is equal to that associated with the baseline option. For example, the level of pollution may be fixed at zero and the income raised until the utility score is the same as in the baseline alternative. This point of indifference indicates the threshold above which the average individual would no longer choose to allocate the burden to that particular alternative.

Table 7.5 Indifference Points: Example for Continuous Model B

Linear basic model		Quadratic model	
Income	Pollutes	Income	Pollutes
100	15	100	15
178	12	121	12
256	9	173	9
361	5	301	5
413	3	401	3
491	0	646	0

This example is described in Table 7.5 for the linear basic model B (column 4, Table 7.4). The results show that, on average, a zero polluting individual would have to have earn almost five times as much as the baseline individual to be accorded the same burden sharing priority as an individual in the baseline. An individual polluting 20% less than the baseline individual would have to earn less than twice baseline income for the implicit utility score (of 4.2206) to remain constant. The quadratic model (column 5, Table 7.4) indicates, retaining the same baseline as previously, that a zero polluting individual should earn about 6.5 times baseline income before being awarded the same status as someone in the baseline, in contrast to nearly 5 times baseline income in the linear basic model. It is by assessing trade-offs in this way that our results could be used to inform mixed criteria rules for burden sharing described towards the beginning of this chapter.

Finally, it is worth noting the wider relevance of these findings to the discussion of equity within the sustainable development debate. While much of that debate has been concerned with intergenerational equity and the welfare of future generations, two points are worth making in the current context. Firstly, to the extent that *intra*generational equity concerns can be considered part of the ambit of sustainable development then this chapter makes a direct contribution. In this respect, it is often forgotten that the original definition of sustainable development in the Brundtland Report (WCED, 1987) encompassed not only intergenerational equity but also intragenerational equity. That is, many would argue that a unequal or inequitable society is 'liveable' but not necessarily 'sustainable' in some sense. Secondly, the methodology that we have outlined could be used to address analogous questions regarding some of the diverse concepts of *inter*generational equity that have been proposed elsewhere.

7.6 Conclusions

If burden sharing rules are important in co-ordinating agreement as to 'who should pay' for the costs of an environmental programme then some means of balancing competing criteria for distributing these burdens needs to be sought. In this chapter, we have discussed candidate principles that are often related to environmental equity. It is well established in this literature that these principles often will be in conflict with one another. That is, different rules indicate that burdens are awarded in different ways. Rather than argue for one principle over another, we have proposed that mixed systems that attempt to combine principles offer greater potential for resolving this problem. If so, then it is important to understand and evaluate the nature of the trade-offs involved in the process of balancing distinct criteria. The experiment outlined in this chapter represents an attempt to provide a means of achieving this. We have illustrated this by assuming that people differ in three relevant attributes – responsibility for pollution (or damage caused), benefit received and income (or ability to pay) – and have outlined a model whereby preferences for an environmental programme's burden sharing allocations are influenced by these attributes. The data used in this chapter come from a series of contingent ranking experiments where respondents are required to rank a set of alternative options relating to combinations of these characteristics.

Three basic models were analysed. The first deals with all three attributes simultaneously. The second and third models deal with only two attributes at a time (respectively, responsibility vs. benefits and responsibility vs. income). The results appear to indicate a high degree of consistency between models. Responsibility for pollution receives the highest weight in all models as we might expect. Nevertheless, evidence of a trade-off between competing attributes, and thereby principles, is found. For example, both responsibility and ability to pay are found to be relevant and competing criteria for allocating the burdens of an environmental programme. The magnitude of this trade-off was explored further by quantifying points of indifference between different types of burden sharing rules that combine varying levels of these two attributes. Finally, while we found that preferences for different attributes appeared to be non-linear we did not find strong support for the proposition that respondents significantly allowed their own position to influence their ranking of different options.

Some directions for future avenues of research suggest themselves. Clearly, the investigation of preferences towards efficiency-equity trade-offs would enrich the analysis. In addition, if our approach is to contribute to the discussion of burden sharing and environmental policy at the wider level then we need to evaluate how 'robust' our results are if applied to other scenarios. Hence, it would be interesting to test our approach for its suitability with other analyses of burden sharing applied to other environmental issues such as climate change mitigation strategies or competing principles of intergenerational equity. Clearly, there are many more strategic concerns at stake in this larger arena than those questions relating purely to burden-sharing rules. However, we would argue that the approach we have outlined could make a significant contribution to the considerable discussion that has taken place in the context of the burden-sharing question.

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